STATE OF NEW MEXICO
WATER QUALITY CONTROL COMMISSION

IN THE MATTER OF: PROPOSED AMENDMENTS TO STANDARDS FOR INTERSTATE AND INTRASTATE SURFACE WATERS 20.6.4 NMAC **WQCC 20-51(R)**

DIRECT TECHNICAL TESTIMONY OF SHELLY LEMON

I. INTRODUCTION

My name is Shelly Lemon and I am the Bureau Chief of the New Mexico Environment Department ("NMED" or "Department") Surface Water Quality Bureau ("SWQB"). I present this direct written testimony (**NMED Exhibit 1**) on behalf of the SWQB concerning the SWQB's proposed amendments to the State of New Mexico's *Standards for Interstate and Intrastate Surface Waters* ("Standards"), codified as Title 20, Chapter 6, Part 4 of the New Mexico Administrative Code (20.6.4 NMAC). Section 303(c)(1) of the federal Clean Water Act ("CWA") (**EXHIBIT 11**) requires that the State hold public hearings at least once every three years to review and amend, as appropriate, its water quality standards. The Department proposes amendments to the Standards consistent with 20.6.4 NMAC, Section 303(c)(1) of the CWA, and 40 Code of Federal Regulations ("C.F.R.") Section 131.20 (**NMED Exhibit 21**). This process is known as the State's "Triennial Review."

The SWQB has five primary objectives for this Triennial Review:

1) update the Standards' objective to address climate change;
2) amend or add definitions for terms used in the Standards or to implement the Standards;
3) make the Standards more clear, informative, consistent, and accessible;
4) address segment-specific issues; and
5) update criteria applicable to existing, designated, or attainable uses (20.6.4.900

NMAC) to reflect new information and technical capabilities.

My testimony provides an overview of water quality standards; an outline of the regulatory requirements for adopting standards; information regarding this Triennial Review; background from prior Triennial Reviews along with a summary of the amendments proposed by the SWQB as part of this Triennial Review; supporting evidence and reasoning for the proposed climate change objective; and the basis and supporting evidence for amendments proposed in 20.6.4 NMAC associated with point source regulation and implementation.

In addition to my testimony, Kristopher Barrios, Program Manager for the Monitoring, Assessment and Standards Section of the SWQB, will provide testimony regarding proposed amendments to narrative and numeric criteria; amendments related to consistency and formatting; and elements not proposed as part of this Triennial Review. Jennifer Fullam, an Environmental Scientist/Specialist Supervisor and the Water Quality Standards Coordinator with the SWQB, will provide testimony regarding amendments proposed by the Department related to uses and criteria; proposed amendments for acute and chronic hardness-based metals criteria; demonstration and supporting evidence for proposed designated use amendments for three classified intermittent tributaries within Los Alamos National Laboratory; and testimony regarding the Department's efforts to ensure compliance with the rulemaking process. Finally, Diana Aranda, an Environmental Scientist/Specialist with the SWQB Standards, Planning and Reporting Team, will provide testimony focused on updates for language and criteria associated with definitions, the State's antidegradation policy, the process for Outstanding National Resource Waters, hardness-based metals reference table, the criteria for ammonia, and publication references. In addition, Ms. Aranda will present evidence for several designated use amendments for classified waters in 20.6.4 NMAC, based on two analyses, one being a Use Attainability Analysis for designating uses with

less stringent criteria, and the other an Existing Use Analysis for designating uses based on a water's ability to attain a designated use with more stringent criteria.

II. QUALIFICATIONS

I hold a Bachelor of Science degree in Biology from the University of Arizona, and a Master of Science degree in Hydrology from the University of Arizona. Before joining the Department, I was a Research Assistant for the Center for Sustainability of semi-Arid Hydrology and Riparian Areas, a National Science Foundation Science and Technology Center at the University of Arizona, and have also been a high school and middle school science teacher. I have been with the Surface Water Quality Bureau since 2004, serving first as a Total Maximum Daily Load ("TMDL") Scientist, and then the Nutrients and Lake Team Supervisor, Monitoring Team Supervisor and Program Manager of the Monitoring, Assessment, and Standards Section, as well as the Municipal Team Supervisor for the Point Source Regulation Section.

I have held the position of Bureau Chief of the Department's Surface Water Quality Bureau since March of 2017 and was the acting Bureau Chief prior to that for eight months. As Bureau Chief, I oversee the State program for surface water quality protection, including developing and revising water quality standards, monitoring and assessing state surface water quality, certifying federal permits issued under the CWA for point source discharges and dredge or fill discharges to surface waters, developing water quality planning documents to protect and restore water quality, and implementing watershed, river and wetland protection projects to maintain and improve water quality for present and future generations.

I have included a copy of my resume as **NMED Exhibit 5**. It is accurate and up-to-date.

III. WATER QUALITY STANDARDS

Section 74-6-4(D) of the State's Water Quality Act ("WQA") (NMSA 1978, §§ 74-6-1 to -17) provides that the Commission "shall adopt water quality standards for surface and ground waters of the state based on credible scientific data and other evidence appropriate under the WQA. The standards shall include narrative standards and, as appropriate, the designated uses of the waters and the water quality criteria necessary to protect such uses. The standards shall at a minimum protect the public health or welfare, enhance the quality of water and serve the purposes of the WQA." (**NMED Exhibit 14**). The CWA regulations provide similar direction: "States adopt water quality standards to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water Act." 40 C.F.R. § 131.2. The objective of the CWA, as found in Section 101(a) (33 U.S.C. § 1251) (**NMED Exhibit 10),** is to maintain and protect the physical, chemical, and biological integrity of the Nation's waters. Serving the purposes of the CWA, as defined in Sections 101(a)(2) and 303(c), means that "water quality standards should, wherever attainable, provide water quality for the protection and propagation of fish, shellfish, and wildlife and for recreation in and on the water" (also known as the "fishable/swimmable" goals). (**NMED Exhibit 10**). A water quality standard "defines the goals for a water body, or portion thereof, by designating the use or uses to be made of the water and by setting criteria that protect the designated uses." 40 C.F.R. § 131.2. The State of New Mexico *Standards for Interstate and Intrastate Surface Waters* (20.6.4 NMAC) ensure that all surface waters of the State, as defined in 20.6.4.7(S)(5) NMAC, have designated uses, criteria to protect those uses, and an antidegradation policy to ensure continued protection of those uses.

In accordance with 40 C.F.R. § 131.10 (**NMED Exhibit 22**), each state must specify appropriate water uses to be achieved and protected. The designated uses in New Mexico's Standards include:

- domestic water supply
- livestock watering
- irrigation and irrigation storage
- aquatic life (coldwater, coolwater, warmwater, and four other subcategories)
- recreational contact (i.e., primary and secondary contact)
- wildlife habitat
- fish culture
- public water supply
- industrial water supply

The Standards also establish water quality criteria that will protect the designated uses of a water body. The Standards contain narrative criteria that apply to all waters and all designated uses. An example of a narrative criterion is that for plant nutrients, which states, "Plant nutrients from other than natural causes shall not be present in concentrations that will produce undesirable aquatic life or result in a dominance of nuisance species in surface waters of the state." 20.6.4.13 NMAC. Further, the Standards also identify numeric criteria that are specific to designated uses. For example, a maximum temperature of 29 ºC (84 °F) applies to waters with the coolwater aquatic life use and the 200 micrograms per liter dissolved arsenic criterion applies to waters with the livestock watering use. 20.6.4.900 NMAC.

According to CWA regulations, water quality standards must also contain an antidegradation policy. (**EXHIBIT 23**). New Mexico's antidegradation policy is codified at 20.6.4.8 NMAC. The Commission has also adopted implementation measures specific to antidegradation in its Water Quality Management Plan and Continuing Planning Process ("WQMP/CPP"), specifically Appendix A: Antidegradation Policy Implementation Procedure.

Such measures are also subject to U.S. Environmental Protection Agency ("EPA") review and action consistent with § 303(c) of the CWA and with 40 C.F.R. § 131.12(a), which require states to identify methods for implementing their statewide antidegradation policy, and 40 C.F.R. § 130.5(b)(6), which requires that the state describe the process for establishing and assuring adequate implementation of new or revised standards in its WQMP/CPP. EPA approved New Mexico's current antidegradation policy implementation procedures on October 23, 2020.

The Department's proposed amendments include changes to designated uses and criteria, and only clarifying, non-substantive changes to the State's antidegradation policy.

In addition to setting water quality goals, 40 C.F.R. § 131.2 specifies that the Standards also serve "as the regulatory basis for the establishment of water-quality-based treatment controls and strategies beyond technology-based levels of treatment required by Sections 301(b) and 306 of the [Clean Water] Act." (**EXHIBIT 24**). Discharges from point sources or nonpoint sources are to be managed in such a manner that designated uses are protected. Point source discharges are regulated under National Pollutant Discharge Elimination System ("NPDES") permits issued by EPA under CWA Section 402, and the discharge of dredged or fill material requires a permit issued by the U.S. Army Corps of Engineers under CWA Section 404. In both cases, pursuant to Section 401 of the CWA, NMED must certify that the permitted activities will be conducted in a manner that will comply with applicable State water quality standards. (**EXHIBIT 30**). NMED also implements a Nonpoint Source Management Program that identifies non-regulatory strategies for controlling nonpoint sources of pollution to achieve the Standards. Finally, the WQA allows for direct enforcement of the Standards, which means that civil penalties may be assessed against a person violating a standard. (**EXHIBIT 16**).

IV. REGULATORY BASIS FOR ADOPTION OF STANDARDS

In accordance with NMSA 1978, § 74-6-3(E), of the New Mexico WQA (**NMED Exhibit 13**), "[t]he [Water Quality Control] Commission is the state water pollution control agency for this state for all purposes of the federal [Clean Water Act]" and, pursuant to NMSA 1978, § 74-6-4(D), the Commission must "adopt water quality standards for surface and ground waters of the state based on credible scientific data and other evidence appropriate under the WQA." (**NMED Exhibit 14**).

The Commission, as provided in NMSA 1978, § 74-6-4(F), "…shall assign responsibility for administering its regulations to constituent agencies to assure adequate coverage and prevent duplication of effort. To this end, the Commission may make such classification of waters and sources of water contaminants as will facilitate the assignment of administrative responsibilities to constituent agencies." (**NMED Exhibit 14**). Since the administrative authority for upholding the requirements under the federal CWA in the State of New Mexico is delegated to the Water Quality Control Commission, and the Commission has no technical staff of its own, many of the tasks associated with reviewing and proposing amendments to the State's water quality standards are delegated to the Department (**NMED Exhibit 31**). As such, the Department serves as the Petitioner for the Triennial Review.

V. TRIENNIAL REVIEW

In accordance with Section 303(c)(1) of the CWA (33 U.S.C. § 1313) (**NMED Exhibit 11**), the State is required to hold public hearings from time to time, but at least every three years, to review and, as appropriate, modify and adopt water quality standards. New or revised Standards adopted by the Commission must be submitted by the State to the EPA for approval in accordance with the CWA. In addition, the State must submit the methods used and analyses conducted to

support the water quality standard revisions and general information that aids EPA in determining the adequacy of the scientific basis of the amendments that do not include the fishable/swimmable uses specified in Section 101(a)(2) of the CWA (33 U.S.C. § 1251) (**NMED Exhibit 10**), as well as information on general policies applicable to the State's standards, which may affect their application and implementation (**NMED Exhibit 23**).

State and federal rulemaking regulations require an adequate opportunity for public participation. The public participation requirements can lead to unpredictable and lengthy hearings, but are necessary for upholding the intent of the process. Following the hearing, the Commission must take the time needed to deliberate, which may take over a year. A rule cannot be filed with the State or become effective until after the Commission has issued its Order and Statement of Reasons. The State's WQA provides additional time constraints with the effective date of a rule, where, in accordance with NMSA 1978, § 74-6-6(E) (**NMED Exhibit 15**), a regulation or water quality standard or amendment adopted by the Commission becomes effective 30 days after filing, in accordance with provisions of the State Rules Act (NMSA 1978, §§ 14-4-1 to -11). Pursuant to 40 C.F.R. § 131.6 (**NMED Exhibit 23**), the State is required to submit a Certification by the State Attorney General that the water quality standards were adopted pursuant to State law. EPA can only review the findings of the Triennial Review once the amendments are adopted into rule and effective for State purposes.

Based on the competing and sometimes contradictory circumstances beyond the Department's control, as described above, the State, since at least 2010, considers the "hearing," as termed in 40 C.F.R. § 131.20 (**NMED Exhibit 21**), as the "hearing process," which is initiated from the point of filing a request for hearing with the Commission to EPA's approval.

For this current Triennial Review, the Department satisfied the three-year review

requirements by initiating stakeholder outreach in February 2020, holding stakeholder discussions in July 2020, and filing the petition on August 19, 2020. EPA last approved such revisions to New Mexico's Standards in August 2017. Therefore this petition was timely filed. The Department filed a Statement of Reasons and the Proposed Amendments to the *New Mexico Standards for Interstate and Intrastate Surface Waters*, 20.6.4 NMAC in support of the petition. The Commission heard the Department's request for hearing on the petition at a regularly scheduled Commission meeting on October 13, 2020 and set a multiday public hearing commencing July 13, 2021. It is the purpose of this Triennial Review hearing to fulfill the requirements of the federal CWA as well as the WQA and other applicable federal and state regulations.

It should be noted that the State does not limit proposed amendments to the Standards to only the Triennial Review hearing. Rather, in accordance with 20.6.4.9 NMAC, 20.6.4.10 NMAC, and 20.6.4.15 NMAC, there are mechanisms by which the Commission may grant hearings for designating waters as Outstanding National Resource Waters; amending designated uses; adopting temporary standards; or adopting site-specific water quality criteria, as needed outside of the Triennial Review process. In fact, since the last Triennial Review hearing, the Standards have been amended three times – in 2018, 2019, and 2020.

VI. BACKGROUND OF PRIOR TRIENNIAL REVIEWS AND FOCUS FOR THIS TRIENNIAL REVIEW

Although the State, or other petitioners, may bring amendments before the Commission at any time, most amendments not directly related to particular uses for specific waterbodies are only proposed during the Triennial Review process. The State has undergone numerous Standards amendments to what is now codified as 20.6.4 NMAC, including those adopted in 1968, 1969, 1970, 1971, 1973, 1975, 1976, 1977, 1978, 1979, 1980, 1981, 1982, 1985, 1987, 1988, 1991, 1995,

2000, 2002, 2005, 2006, 2007, 2010, 2011, 2012, 2017, 2018, 2019 and most recently 2020. Although rare, there have been cases where there have been multiple amendments adopted and made effective for State purposes within a single calendar year. Those amendments adopted prior to the amendments to the CWA in 1972 and prior to the creation of the EPA were not subject to EPA approval. The Department has affirmed that it completed amendments made in 1985, 1991, 1995, 2000, 2005, 2010 and most recently in 2017 as formal Triennial Reviews. As can be noted by the dates, Triennial Reviews are not always adopted every three years as the name indicates. The additional time between Triennial Reviews is predominately due to the lengthy administrative process, which must provide adequate opportunity for public engagement.

The hearing for the last Triennial Review began in 2015; however, amendments were not adopted by the Commission and made effective for State implementation until March 2, 2017. In accordance with 40 C.F.R. § 131.20, following adoption, the Commission submitted the approved standards for review and approval by EPA (**NMED Exhibit 21**). EPA issued a technical support document in June 2017 in support of the amended Standards, however upon review, the Department brought to the Commission's attention that the Standards approved by EPA were not the legally effective rule adopted by the Commission. EPA addressed this oversight and the amended Standards became effective for CWA purposes on August 11, 2017 (**NMED Exhibit 32**).

As part of this Triennial Review, SWQB proposes several types of amendments to the Standards. SWQB considered all Triennial Review requirements detailed in 40 C.F.R. § 131.20 (**NMED Exhibit 21**) and proposed updates to the Standards accordingly. Proposed amendments include: updating numeric criteria, in accordance with 40 C.F.R. § 131.11 (**NMED Exhibit 25**); amending areas of the rule to which implementation of water quality standards may be affected;

updating references and citations as appropriate for adequate implementation of water quality standards; and amending designated uses that apply to multiple waters or larger geographical areas to ensure the State applies appropriate protections to classified waters.

VII. THE DEPARTMENT'S PROPOSED AMENDMENTS TO THE STANDARDS

A. 20.6.4.6 NMAC, Amend Objective

The State's water quality standards, codified in 20.6.4 NMAC, protect public health or welfare, enhance the quality of water, and serve the purposes of the state WQA and federal CWA, including protections for aquatic life and recreation. In order to implement and attain these goals, water quality standards must have an antidegradation policy, which at the very least protects for existing uses, in accordance with 40 C.F.R § 131.6 and 40 C.F.R § 131.12.

Science shows that anthropogenic activities threaten and harm water quality and quantity on a global scale. In addition, it is the premise of the CWA that states shall protect existing uses of a waterbody, even if those uses are not currently attained, and that those protections shall remain for use into the future. An existing use, as defined in 40 C.F.R. § 131.3 (**NMED Exhibit 26**) and 20.6.4.7(E)(3) NMAC, is a use attained by a waterbody at some point since November 28, 1975. The federal regulations protect existing uses through two mechanisms. First, in accordance with 40 C.F.R. § 131.10(i) (**NMED Exhibit 22**), a designated use may not be less stringent than an existing use. Second, in accordance with 40 C.F.R. § 131.10 (**NMED Exhibit 22**), a designated use may only be amended if it is not an existing use. The State's water quality standards protect, and have always protected, water quality from anthropogenic impacts by ensuring that the antidegradation policy maintains existing use protections and that designated use protections (goals) are attainable and not arbitrarily lowered without defensible investigation and

demonstration under state and federal regulations. These protections for the surface waters of the State inherently protect the State's water resources against all foreseen and unforeseen sources threatening surface water quality, including climate change.

NMSA 1978, Section 74-6-4(E)(7) (**NMED Exhibit 14**), indicates that "federal water quality requirements" are to be taken into consideration by the Commission when making regulations. Therefore, the state WQA directs the Commission to consider amendments to the water quality standards that originate from the federal CWA. Acknowledging the need to address the inherent threats to water quality resulting from climate change falls into that category.

Including language to clarify that one of the objectives of the water quality standards is, and has been, to plan for anticipated human-caused impacts and promote watershed resiliency due to climate change is explicitly clear in its intent and is beneficial for implementation of the standards. This addition updates the Standards to acknowledge that climate change is a threat to surface water quality and to explicitly recognize that an objective of the Standards is to protect against this threat.

Because understanding and mitigating the effects of climate change must be addressed at a global-scale, but the impacts of climate change are felt at the local watershed-scale, the Department proposes to add a definition for the term to coincide with its reference in the objectives. The proposed language is taken almost directly from EPA's definition of climate change, as provided on their website (**NMED Exhibit 33**).

Adoption of the new objective and the corresponding definition for "climate change" does not affect implementation, as the standards already accommodate for impacts to water quality (either local or global), but does clarify that the State's Water Quality Standards ensure protection of the waters of the state against the threats posed by climate change.

B. 20.6.4.7(A) NMAC, Add definition for "4Q3"

The Department proposes to move the definition for "4Q3" from 20.6.4.11(B)(2) NMAC to Definitions, 20.6.4.7(A) NMAC. It is more appropriate and consistent with other defined terms to relocate the definition to 20.6.4.7(A) NMAC. The "4Q3" is the critical low flow value used when developing NPDES permit requirements to meet criteria in 20.6.4.97 through 20.6.4.900 NMAC, with the exception of human health-organism only criteria. Although "4Q3" is referenced in the Standards only in relation to NPDES permits, other applications such as TMDL calculations and water quality modeling use the "4Q3" as a low-flow statistic. Moving the term to the definitions section provides an easy-to-reference location for those other applications. The proposed changes do not alter the meaning nor do they affect implementation.

C. 20.6.4.7(B) NMAC, Add definition for "Baseflow"

The Department proposes to add a definition for "baseflow" (20.6.4.7(B) NMAC) to provide reference to the term as it applies to flow condition and to clarify the word in the proposed definition of "effluent dominated." Although the use of the word "baseflow" is not referenced directly in 20.6.4 NMAC, it will assist in implementing water quality standards and other technical and guidance documents within the agency. Adding this definition will provide clear guidance in the implementation of water quality standards. The inclusion of the term "baseflow" in the proposed addition of the "effluent dominated" definition requires a definition to aid the implementation of Standards. The definition is borrowed, in part, from Price, 2011 (**NMED Exhibit 34**).

D. 20.6.4.7(E) NMAC, Add definition for "Effluent Dominated"

The Department proposes to add a definition for "effluent dominated" to the Definitions section, 20.6.4.7(E) NMAC. Although the term "effluent dominated" is not referenced directly in

20.6.4 NMAC, it is used in several procedural documents such as the state's WQMP/CPP and EPA's Technical Support Document for Water Quality Based Toxics Control regarding NPDES permits. Neither state statute nor regulation currently define this term. Having a regulatory reference for this term will aid in the implementation of the goals of the CWA. In addition, adding a definition for the term "effluent dominated" will be applicable should the State adopt a designated aquatic life use for "effluent dominated" waters. These waters may not be able to attain all the current applicable criteria (e.g., nutrients) and more environmental harm may be caused if the discharge ceases, which would eliminate a reliable source of baseflow for aquatic life and wildlife.

E. Relocate definition for "Harmonic Mean Flow" from 20.6.4.11 to 20.6.4.7 NMAC

The Department proposes to move the definition for "harmonic mean flow" from 20.6.4.11(B)(1) NMAC to 20.6.4.7 NMAC to provide a consistent location for definitions. The Department also proposes a minor expansion of the narrative equation to assist with implementing water quality standards. "Harmonic mean flow" is the critical low flow value used when developing NPDES permit requirements to meet human health-organism only criteria in 20.6.4.900 NMAC. The proposed changes do not alter the term's meaning nor do they affect implementation.

F. Amend 20.6.4.12 NMAC, Compliance with Water Quality Standards

The Department proposes an amendment to the description of compliance schedules in 20.6.4.12(G) NMAC to represent the process accurately. As currently written, the subsection implies that the Commission has a policy for granting compliance schedules. However, there is no known policy, nor has it been the Department's practice to bring compliance schedules before the Commission for consideration. Although the Commission has delegated some responsibilities to

the Department, it has not delegated compliance schedules. It is accurate to state that the Commission "may" approve compliance schedules on a case-by-case basis. This proposed change does not have an effect on permit implementation.

G. Amend 20.6.4.105 NMAC and 20.6.4.106 NMAC, Classified Waters within the Rio Grande Basin

Discharges from community sewerage systems to the Rio Grande basin must meet minimum effluent criteria, as described under 20.6.2.2102 NMAC. If the effluent does not meet the minimum requirements in 20.6.2.2102 NMAC then the discharge is not allowed. Since the waters described in 20.6.2.2102 NMAC are classified waters under sections 20.6.4.105 and 20.6.4.106 NMAC, the effluent criteria apply to these waters if the applicability conditions in 20.6.2.2100 NMAC are met. The Department proposes to add language to sections 20.6.4.105 NMAC and 20.6.4.106 NMAC to clarify that the criteria referenced in *Regulations for Surface and Ground Water* (20.6.2.2102 NMAC) for Rio Grande Basin-Community Sewerage Systems may also apply. The inclusion of these effluent criteria does not change or modify the current designated uses or related criteria in 20.6.4.105 NMAC and 20.6.4.106 NMAC but does add clarification regarding all potential applicable criteria.

VII. CONCLUSION

The Department recommends that the Commission adopt the proposed amendments to the Standards, filed as **NMED Exhibit 9**, based upon the testimony of the SWQB's witnesses. This concludes my direct testimony.

STATE OF NEW MEXICO
WATER QUALITY CONTROL COMMISSION

IN THE MATTER OF: PROPOSED AMENDMENTS TO STANDARDS FOR INTERSTATE AND INTRASTATE SURFACE WATERS 20.6.4 NMAC

WQCC 20-51(R)

DIRECT TECHNICAL TESTIMONY OF KRIS BARRIOS

I. INTRODUCTION

My name is Kris Barrios and I present this written testimony (**NMED Exhibit 2**) on behalf of the New Mexico Environment Department ("Department" or "NMED") Surface Water Quality Bureau ("SWQB") concerning the SWQB's proposed amendments to the State of New Mexico's Standards for Interstate and Intrastate Surface Waters ("Standards"), codified as Title 20, Chapter 6, Part 4 of the New Mexico Administrative Code (20.6.4 NMAC). Section 303(c)(1) of the federal Clean Water Act ("CWA") (33 U.S.C. § 1313) (**EXHIBIT 11**) requires each state to hold a public hearing at least once every three years to review and modify, as appropriate, its water quality standards, in a process known as the "Triennial Review" of the State's Standards. My testimony outlines the reasoning behind the following proposed changes:

- updated or new definitions for "Contaminants of Emerging Concern", "Persistent Toxic Pollutants", and "Unclassified Waters of the State" in 20.6.4.7 NMAC;
- an update to 20.6.4.13 NMAC, General Criteria that clarifies the substances considered under the narrative criterion for toxic pollutants;
- addition of cyanobacteria toxin criteria to the Primary Contact designated use, 20.6.4.900(D) NMAC;
- updates and additions to numeric criteria in 20.6.4.900(J)(1) NMAC; and
- reference and grammatical corrections to various sections.

My testimony also provides the reasoning for numeric criteria that were reviewed but not proposed as amendments in this Triennial Review.

II. QUALIFICATIONS

I am currently employed as the Program Manager for the Monitoring, Assessment, and Standards Section for the SWQB and have held this position since August 2017. I began work with the Department in October 2015 as the Monitoring Team Supervisor within the SWQB. Before employment with the Department, I supervised the water quality and hydrologic monitoring program for the Northwest Florida Water Management District ("NWFWMD"). In other capacities, I have served as a hydrogeologist responsible for ground water and surface water monitoring, a project geologist for petroleum storage tank investigations, an environmental scientist working on ground water contamination delineation, and a laboratory technician.

I have a Bachelor of Science degree in Geology from Florida State University with a minor in mathematics. I am also a licensed Professional Geologist (Florida License 2861). My publications include Barrios, K., 2011. *Nitrate Sources of Springs Discharging to Merritt's Mill Pond, Jackson County, FL*. NWFWMD TFR 2011-1, Barrios, K., 2006. *St. Marks River and Wakulla River Springs Inventory, Leon and Wakulla Counties, Florida*. NWFWMD WSR 06-03, Barrios, K. and DeFosset, K., 2005. *Ground Water Chemical Characterization of Jackson Blue Spring and Wakulla Springs, Florida*. NWFWMD WSR 05-01, among others. I have provided my updated resume as **NMED Exhibit 6**.

III. AMENDMENTS DESCRIBING WATERS

The Department proposes to move the definition for "unclassified waters of the state" from 20.6.4.11(H) NMAC to 20.6.4.7(U) NMAC to provide a consistent location for definitions. This change is intended to complement the definition of "classified water of the state", 20.6.4.7(C)(3)

NMAC. The State has used a definition for "classified water the state" since 1995; however, before adopting designated uses for the State's unclassified waters, there was no definition for the term "unclassified waters of the state". The term "non-classified" or "unclassified" was used only to describe the applicability of water quality standards for those waters that were not "classified". Since the State adopted designated uses for unclassified perennial and non-perennial waters, the term now serves a functional purpose as a definition. However, the State kept the language under the section entitled "Applicability of Water Quality Standards" (20.6.4.11 NMAC). It is more appropriate and consistent with other defined terms to relocate the definition to 20.6.4.7 NMAC. The proposed change does not alter the meaning nor affect the implementation of the term.

IV. AMENDMENTS ASSOCIATED WITH CRITERIA

A. General Criteria

1. Toxic Pollutants

The Department proposes the addition of "contaminants of emerging concern" and the toxic pollutants listed in 20.6.2.7(T)(2) NMAC to the general criteria for toxic pollutants, 20.6.4.13(F) NMAC. The definition for "toxic pollutants", located in 20.6.4.7(T) NMAC, refers to a pollutant or combination of pollutants that cause adverse impacts upon exposure to organisms or their offspring. The Department proposes adding "contaminants of emerging concern" to the general criterion for toxic pollutants. These compounds include pollutants that are known or suspected toxins but do not have numeric criteria. Similarly, the definition of "toxic pollutants" under the State's Regulations for Ground and Surface Water Protection (20.6.2.7(T)(2) NMAC) includes compounds that have numeric criteria in 20.6.4 NMAC, as well as those that do not. Since the State identifies these compounds as toxic pollutants, the Department proposes adding a reference to the toxic pollutants listed in 20.6.2.7(T)(2) NMAC to 20.6.4.13(F) NMAC. Adding

language to clarify that the general criterion for toxic pollutants in 20.6.4.13(F) NMAC includes contaminants of emerging concern and the toxic pollutants listed in 20.6.2.7(T)(2) NMAC will aid in implementing water quality standards and upholding the goals and objectives of the Clean Water Act.

2. Addition of a definition for "Contaminants of Emerging Concern"

The Department proposes to add a definition for "contaminants of emerging concern" or "CECs" to 20.6.4.7(C) NMAC to identify pollutants recognized as toxic to or have other harmful effects on aquatic life or other organisms. The Department bases the proposed definition on information provided at the U.S. Environmental Protection Agency's ("EPA's") website for contaminants of emerging concern (**EXHIBIT 35**).

The Standards include narrative criteria and numeric criteria. The narrative (i.e., "general") criteria are statements that describe the desired water quality goal, such as waters being "free from" pollutants like oil and scum, color and odor, and other substances that can harm people and fish. These criteria protect water bodies from contaminants for which numeric criteria are difficult to specify. Since "contaminants of emerging concern" is a proposed addition to the general criteria for toxic pollutants in 20.6.4.13(F)(1) NMAC, a definition is necessary to provide an attributable reference. Although EPA has not developed numeric criteria for CECs, clarification that NMED's general criterion for toxic pollutants regulates this group of pollutants provides greater clarity for implementing water quality standards.

3. Addition of a definition for "Persistent Toxic Pollutants"

The Department proposes to add a definition for "persistent toxic pollutants" to 20.6.4.7(P) NMAC to clarify its meaning since the term describes certain pollutants in 20.6.4.900(J)(1) NMAC. The term references those toxic pollutants, as defined in 20.6.4.7(T)(2) NMAC, that

persist in the environment or do not metabolize in a living organism and, as a result, bioaccumulate in organisms over time, causing harm or adverse impacts to human health and the environment. The designation of persistent toxic pollutants to human health-organism only criteria results in the application of that criterion to all tributaries of waters with designated, existing, or attainable aquatic life uses. Also, chronic criteria for persistent toxic pollutants are applicable for the limited aquatic life designated use. The addition of a definition for "persistent toxic pollutants" does not alter the implementation of water quality standards.

B. Numeric Criteria, 20.6.4.900 NMAC

40 Code of Federal Regulations ("C.F.R.") Section 131.20(a) (**NMED EXHIBIT 21**) requires states to review and, if appropriate, modify and adopt applicable water quality standards. States are required, pursuant to 40 C.F.R. § 131.11(b)(1) (**NMED Exhibit 25**), to adopt numeric water quality criteria that are either based on Section 304(a) of the CWA (33 U.S.C. § 1314), develop modified criteria from those in Section 304(a) of the CWA (33 U.S.C. § 1314) to reflect site-specific conditions, or develop criteria based on other scientifically defensible methods (**NMED Exhibit 12**). Based on EPA's published recommended criteria, the State proposes to adopt criteria for the primary contact recreational designated use and aquatic life designated use.

1. Recreational Use Primary Contact Numeric Criteria

In May 2019, EPA published its nationally recommended Human Health Recreational Ambient Water Quality Criteria for Microcystins and Cylindrospermopsin (**EXHIBIT 36**). The EPA based the new criteria on the latest scientific knowledge about the potential human exposure risk effects and the toxins' adverse effects to the liver and kidney, development, and the reproductive, respiratory, and digestive systems. These effects range from acute short-term to chronic long-term health effects. Under 40 C.F.R. § 131.20 (**NMED Exhibit 21**), and Section

304(a) of the CWA (33 U.S.C. § 1314) (**NMED Exhibit 12**), the Department proposes to adopt numeric criteria for the State's designated recreational primary contact use for toxins affiliated with harmful algal blooms, microcystins, and cylindrospermopsin.

The EPA includes magnitude, duration, and frequency components in its recommended criteria for microcystins and cylindrospermopsin (**EXHIBIT 36**). The recommendation for recreational water quality criteria is a maximum concentration for both microcystins and cylindrospermopsin with a duration of one day in a 10-day assessment period and a frequency of no more than three excursions per recreational season in more than one year. Based on the EPA's Draft Technical Support Document (**EXHIBIT 37**) for implementing the recommended recreational criteria, EPA is likely to provide states the flexibility to define the length of the recreational season and recurrence frequency for criteria associated with microcystins and cylindrospermopsin. Since the recreational season in New Mexico varies by region, elevation, and waterbody, the Department proposes to use a 12-month period instead of a defined recreational season. The Department also proposes a 12-month period for the frequency component of the criterion. Adding these criteria for waters with a primary contact designated use will enhance protections directly associated with human health. The Department may require entities with an National Pollutant Discharge Elimination System ("NPDES") permit to increase monitoring efforts to demonstrate compliance with microcystin and cylindrospermopsin permit limits.

2. Acute and Chronic Aquatic Life Numeric Criteria

The Department proposes the adoption of recommended EPA criteria in the Table of Numeric Criteria, 20.6.4.900(J)(1) NMAC. In accordance with 40 C.F.R. § 131.11 (**NMED Exhibit 25**), states must adopt those water quality criteria that protect the designated uses. States should base numeric criteria on either CWA Section 304(a) guidance, CWA Section 304(a)

guidance modified to reflect site-specific conditions, or other scientifically defensible methods. As part of the Triennial Review, and according to 40 C.F.R. § 131.20 (**NMED Exhibit 21**), if a State does not adopt new or revised criteria for parameters for which EPA has published new or updated CWA Section 304(a) criteria recommendations (**NMED Exhibit 12**), then the State shall provide an explanation when it submits the results of its Triennial Review to the Regional Administrator.

The State's water quality standards have a list of use-specific numeric criteria identified in 20.6.4.900(J)(1) NMAC. The table of use-specific criteria arranges the pollutant on the first column and the designated use numeric criterion in subsequent columns. Those columns (designated uses) that do not have a value do not have an associated numeric criterion for that pollutant.

There are three different types of criteria for the protection of aquatic life: those associated with acute exposure, those associated with chronic exposures, and those based on human consumption of an aquatic organism (human health-organism only). Although the human health-organism only exposure endpoint is the human consumption of an aquatic organism, these criteria are considered aquatic life protections, and the numeric criteria are, like the other criteria, based on concentrations in water, unless described otherwise.

The pollutants for human health-organism only are of particular concern because they are either persistent in the environment and bioaccumulate in the organism and/or they are carcinogenic, meaning they have been determined to cause cancer at a higher rate than what would be assumed normal for the general population. Because these endpoints impact both establishment of these numeric criteria and the implementation of the water quality standards, the last column

provides a delineation of the exposure endpoints for these pollutants. The State has 108 numeric criteria for human health-organism only pollutants, 60 of which have a carcinogenic endpoint.

Human health-organism only criteria were last updated in the 2010 Triennial Review. In 2015, EPA updated human health criteria for approximately 94 constituents. As part of this Triennial Review, the Department compared the State's numeric human health-organism only criteria to EPA's Section 304(a) criteria **(NMED Exhibit 38)**. The evaluation concluded that of the 108 pollutants with human health-organism only criteria listed in 20.6.4.900(J)(1) NMAC, 23 are equivalent to EPA Section 304(a) criteria and required no amendment, 60 pollutants have EPA Section 304(a) criteria more stringent than the State's, and 25 pollutants have EPA Section 304(a) criteria less stringent than the State's. In addition, 14 pollutants are listed on EPA Section 304(a) guidance but not adopted by the State. Adopting the proposed criteria into the State's water quality standards will result in 122 human health-organism only aquatic life criteria. For those criteria derived from a cancer-causing endpoint, the State has adjusted the numeric value by one order of magnitude to account for New Mexico's lifetime risk of more than one cancer per 100,000 exposed persons (20.6.4.13(F)(2)(a)) in comparison to EPA's lifetime risk of more than one cancer per 1,000,000 exposed persons.

For benzene, EPA's recommended criterion has a range of 16-58 micrograms per liter ("µg/L"), which is more stringent than the current 510 µg/L. Based on benzene's carcinogenic effects, EPA recommends the lower range of the criterion to protect human health (**EXHIBIT 39**). The Department proposes adopting the recommended lower range, increased by one order of to account for New Mexico's lifetime risk of more than one cancer per 100,000 exposed persons (20.6.4.13(F)(2)(a)20.6.2 NMAC) in comparison to EPA's lifetime risk of more than one cancer per 1,000,000 exposed persons, resulting in a proposed criterion of 160 µg/L.

The recommended criteria published by EPA, in accordance with Section 304(a) of the CWA (33 U.S.C. § 1314) (**NMED Exhibit 12**), includes criteria protecting acute and chronic aquatic life for 61 pollutants, of which 30 have narrative criteria only (**NMED Exhibit 40**). The Department compared these numeric aquatic life criteria to those criteria listed in 20.6.4.900(J)(1) NMAC (**NMED Exhibit 41**). Of the 31 pollutants listed in EPA's recommended criteria with an acute numeric EPA Section 304(a) criterion, six pollutants do not have numeric criteria under the State's water quality standards: chlorpyrifos, chloride, parathion, tributyltin, acrolein, and carbaryl. Fourteen pollutants have a chronic numeric aquatic life criterion listed in EPA's recommended criteria, but do not have numeric criteria under the State's water quality standards. These pollutants include those identified above for acute aquatic life as well as alkalinity, demeton, guthion, hydrogen sulfide, iron, malthion, methoxychlor and mirex. As part of the Triennial Review requirements, the Department proposes adopting the above noted EPA recommended criteria for acute and chronic aquatic life use.

Eight pollutants listed in EPA's recommended guidance for acute and chronic aquatic life criteria have hardness-based criteria under 20.6.4.900 NMAC. These constituents are evaluated and addressed in the testimony of Jennifer Fullam (**EXHIBIT 4**).

The Department proposes to take no action on the EPA's recommended aquatic life criteria for the following pollutants: aluminum, arsenic, manganese, and selenium. The Department provides its reasoning in section IV(B)(3) of this testimony.

There are no pollutants within 20.6.4.900(J)(1) NMAC with chronic numeric aquatic life criteria that are more stringent than EPA's recommended criteria. However, polychlorinated biphenyls ("PCBs") and selenium have acute criteria listed in 20.6.4.900(J)(1) NMAC but do not have associated acute criteria in EPA's recommended aquatic life criteria guidance. The

Department is not proposing a change in PCBs criteria; however, the Department proposes moving the criteria to fit alphabetically within organic pollutants in Table 20.6.4.900(J)(1) NMAC.

In addition to the proposed changes to the aquatic life criteria described above, the Department proposes spelling corrections or completion of missing chemical abstract service numbers for several pollutants.

3. Numeric Criteria Not Proposed for Adoption

a. EPA's Recommended Aluminum Criteria

The Department does not propose adopting the EPA's recommended acute and chronic aquatic life criteria for aluminum as a replacement of the current hardness-based water quality standard. In 2018, EPA published updated aquatic life criteria for aluminum, based on a multiple linear regression ("MLR") model that takes into account the effects of ambient water quality on the bioavailability of aluminum to freshwater aquatic life (**EXHIBIT 42**). The MLR is based on the observed interactions of aluminum, pH, hardness, and dissolved organic carbon ("DOC") in a compilation of toxicity tests consisting of *P. promelas* and *C. dubia.* The EPA found these three parameters have the most significant influence on the toxicity of aluminum. Development of the MLR model included a range of water quality conditions to capture the variability of ambient conditions: pH (6.0-8.7), hardness (9.8 to 428 mg/L), and DOC (0.08 to 12.3 mg/L). The EPA extrapolated the model to expand its applicability but cautions against using the MLR model for conditions outside the range of empirical testing, for pH in particular. The Department has concerns regarding EPA's linear regression extension of the model for pH ranges 5.0 to 6.0 and 8.7 to 10.5. Also of concern, the EPA MLR model guidance acknowledges temperature as a factor in aluminum solubility yet does not include temperature in the MLR model or explain why it did not use temperature.

Although the aluminum MLR model represents the best available science for calculating appropriate aluminum instantaneous water quality criteria ("IWQC") for freshwater aquatic life, the Department proposes retaining the current hardness-based standard. The Department cannot implement the MLR model effectively since the Department does not have a way to determine the MLR model input value of DOC with confidence. The New Mexico Department of Health Scientific Laboratory Division ("SLD") does not currently perform DOC analysis. SLD is building capacity for DOC analysis; however, the Department is uncertain of the implementation date. The Department has considered contract labs for DOC analysis but does not have the resources required for collection at every site. Recognizing that not all states or tribes can collect all required input parameters to the MLR model, the EPA implementation guidance (**EXHIBIT 43**) suggests using either default or ecoregional values for missing site-specific parameters. However, EPA cautions that the approach may be too general for areas of complex geology. The Department considers New Mexico geologically diverse. Default or ecoregional DOC values are unlikely to capture variability across the state or at a specific location under different flow conditions. The EPA provides ecoregional DOC values in its Draft Technical Support Document: Recommended Estimates for Missing Water Quality Parameters for Application in EPA's Biotic Ligand Model (Table 18, **EXHIBIT 44**) based on DOC results from EPA's National Rivers and Streams Assessment. However, the dataset for New Mexico consists of single site visits to relatively few waterbodies. For example, the entire eight-digit Hydrologic Unit Code ("HUC") representing the Pecos Headwaters watershed (13060001) contains four data points from 2008-2014 representing two ecoregions. The dataset does not represent many other areas of the state.

The Department compared criteria calculated from the MLR model and New Mexico's current hardness-based criteria (**EXHIBIT 45**). Overall, the MLR model results are more

conservative (criteria are lower) within the range of DOC values expected for New Mexico's surface waters. At very low hardness, approximately 50 mg/L or less, both the chronic and acute hardness-based criteria are lower than those from the MLR model. The Department completed an analysis of the difference between the current hardness-based criteria and the MRL model criteria for total recoverable aluminum results collected during the 2017-2018 Upper Rio Grande watershed survey. The Department divided each total recoverable aluminum result by the IWQC calculated from the required input parameters, resulting in an exceedance ratio for each sample. The Department used a DOC concentration of 0.7 mg/L, the average of the recommended DOC concentrations for Omernik Level III ecoregions 21 and 22, for the MLR model input value. Exceedance ratios greater than one indicate a sampling result higher than the applicable IWQC. **EXHIBIT 46** graphs the difference between the MLR model exceedance ratio and the hardness-based exceedance ratio. Values greater than zero indicate the MLR model criterion is more stringent than the hardness-based criterion. These results confirm that hardness-based criteria are more stringent than those of the MLR model at lower hardness concentrations. The largest exceedance ratio differences between the hardness-based calculation and MLR model also occur at low hardness. This analysis identified 42 acute and 111 chronic exceedances using the hardness-based calculation, and 59 acute and 110 chronic exceedances using the MLR model. Overall, the hardness-based calculation resulted in more exceedances at lower hardness values and the MLR model resulted in more exceedances at higher hardness values (**EXHIBIT 47**).

The implementation of the 2018 EPA aluminum ambient water quality criteria ("AWQC") is further complicated because the guidance does not address the distinction between the bioavailable species of aluminum and those forms that are geologically based and present in natural waters as suspended sediment. The EPA acknowledges this challenge in its Final Aquatic

Life Ambient Water Quality Criteria for Aluminum 2018 guidance (**EXHIBIT 42**): "…natural water samples may also contain other species of aluminum that are not biologically available (i.e., suspended particles, clays, and aluminosilicate minerals)…This creates uncertainty because the total recoverable aluminum concentrations measured in natural waters may overestimate the potential risks of toxicity to aquatic organisms." Further, the EPA states that new analytical methods are needed, and it expects ongoing research to improve accurate measurement of toxic aluminum. For total recoverable aluminum analyses, the Department currently filters high turbidity samples with a 10-micron filter to remove terrestrial sediment. However, the infiltration of clay and some silt can still occur since these particles may pass through the filter. Adopting the MLR model may require refinement of this process to better discriminate bioavailable aluminum to prevent unnecessary, and potentially costly, impairment listings in high turbidity areas.

The Department concludes that it does not have adequate information to implement the 2018 aluminum aquatic life criteria with confidence. The Department will continue to evaluate the adoption of the revised aluminum criteria and expects to begin sampling and analysis of DOC. The Department estimates an annual cost of 8,500 Work-Time Units ("WTUs") per year for DOC, which is approximately 5% of SWQB's fixed annual budget with SLD. This extra cost reduces the amount the Department can allocate to sampling for other pollutants. Costs may also increase for NPDES permittees to account for additional monitoring.

b. EPA Section 304(a) Arsenic Criteria

The Department does not propose the adoption of the 2002 EPA recommended human health criterion for arsenic. The State documented the reasoning behind the current human health-organism only criterion of 9.0 µg/L in the Statement of Reason from the 2005 Triennial Review (**EXHIBIT 48**). The State adopted a New Mexico-specific criterion using arsenic water column

and fish tissue concentration from the Rio Grande. The Department's analysis of surface water quality results for arsenic shows that undeveloped areas in New Mexico frequently exceed the EPA recommended concentration of 1.4 µg/L (increased by one order of magnitude to account for New Mexico's lifetime risk of more than one cancer per 100,000 exposed persons (20.6.2 NMAC)) (**EXHIBIT 49**). Since human health-organism only criteria cannot be modified for natural background (20.6.4.10(E) NMAC), adopting the more stringent criterion is not practicable.

c. EPA Section 304(a) Copper Criteria

The Department does not propose adopting the EPA's recommended aquatic life criteria for copper as a replacement of the current hardness-based water quality standard. In 2007, EPA introduced revised AWQC for copper using the Biotic Ligand Model ("BLM") (**EXHIBIT 50**) to take into account the various effects of ambient water quality on the toxicity of copper. Although the BLM provides a more accurate assessment of copper bioavailability than New Mexico's hardness-based criteria calculation, it requires the input of 11 coincident water quality parameters (some of which are not commonly available) for the calculation of an instantaneous water quality criterion. Recognizing the scarcity of data as a limitation of the BLM in its implementation guidance, the EPA recommends adopting the BLM for copper on a targeted basis while retaining hardness-based standards for all other waters (**EXHIBIT 51**). During the 2010 Triennial Review, the Commission adopted the provision described in 20.6.4.10(D)(4)(c) NMAC adding the BLM for copper as a scientifically defensible method for site-specific criteria development. The Department will continue to evaluate the implementation of the BLM for copper on a segment-specific basis.

d. EPA Section 304(a) Manganese Criteria

The Department does not propose the adoption of EPA's recommended water quality human health-organism only criterion for Manganese of 100 µg/L for human health. Manganese is a naturally occurring element commonly found in food and water and is a micronutrient required for cellular function. The EPA based its recommended human health criterion on manganese's organoleptic effects, including objectionable taste and laundry staining.

For application in New Mexico, as defined in 20.6.4.7(H)(2) NMAC, human health-organism only "means the health of humans who ingest fish or other organisms from waters that contain pollutants". Since the EPA criterion does not meet the definition of protecting human health, the Department does not support its adoption. Although there are numerous pollutants listed in 20.6.4.900(J)(1) NMAC with accompanying recommended organoleptic criteria in EPA's National Recommended Water Quality Criteria – Organoleptics (**EXHIBIT 52),** New Mexico has not adopted any numeric organoleptic criteria. However, the State does have a narrative criterion, provided in 20.6.4.13(D) NMAC, which protects against degradation of organoleptic quality from other than natural causes.

e. EPA Section 304(a) Selenium Criteria

In 2016, the EPA published a revised selenium criterion for freshwater aquatic life, available at https://www.epa.gov/wqc/final-aquatic-life-ambient-water-quality-criterion-selenium-freshwater-2016. Selenium is a naturally occurring element that is found usually in sedimentary rocks with high organic content, including coal-containing strata, and the soils derived from this lithology. Selenium also occurs in mineralized areas and is found in ores of copper, lead, and zinc. Deleterious concentrations of selenium in water may result from mining, petroleum extraction, or erosion of soils. Selenium bioaccumulates through the food web, primarily through

assimilation of dissolved selenium by microorganisms followed by particulate matter ingestion. According to the EPA's recommended criteria, selenium's most sensitive adverse effects are found in the reproductive effects in fish and are the basis for the updated chronic criterion. Due to the significant chronic effects, EPA did not develop an acute criterion for selenium. EPA's recommended chronic criterion consists of two media, fish tissue and water concentration. An exceedance in either medium is considered an excursion above the criterion.

The criterion expresses fish tissue concentration as either egg/ovary or fish whole body/muscle, and in either case, the criterion element is an instantaneous value not to exceed.

The water concentration element is a thirty-day average exposure value for rivers/streams and lakes (1.5 µg/L and 3.1 µg/L, respectively) to not exceed more than once in three years. In its guidance, the EPA also provides a formula for calculating allowable intermittent water concentration excursions above background during a thirty-day period.

Although the EPA published the updated selenium criterion in 2016, it has not provided implementation guidance to states or tribes. Given the complexity of implementation and the absence of implementation guidance from the EPA, the Department is reluctant to invest already constricted resources for collecting fish tissue or 30 consecutive daily waterbody samples for assessing a single site. Additional guidance is needed to translate the criterion to alternative assessment periods. The Department will further evaluate the revised selenium criterion once the EPA finalizes implementation guidance. Until that time, the Department proposes retaining the current total recoverable selenium criterion for aquatic life of 5.0 µg/L chronic and 20.0 µg/L acute.

V. SPELLING AND FORMATTING AMENDMENTS

A. Removal of Redundant Dash (multiple citations)

The Department proposes removing dashes following colons in the basin description for 97 classified sections in 20.6.4.100-899 NMAC. Removal of the dash is consistent with formatting throughout NMAC. According to State Records Center and Archives (**EXHIBIT 53**), the correct formatting includes the section name in all capital letters followed by a colon then two spaces. The State Record Center and Archives has clarified that grammatical corrections such as these do not require an amendment notation for the section (**NMED Exhibit 54**); therefore, the proposed amendment will not add an amendment notation.

B. Correction of Spelling "Canyon Largo" in 20.6.4.405 and 20.6.4.408 NMAC

The Department proposes to amend 20.6.4.405 and 20.6.4.408 NMAC to correct the spelling of "Canyon Largo" to "Cañon Largo" to be consistent with accepted geographical references for the waterbody. The United States Geological Survey ("USGS") 7.5-minute topographical map, Google Earth, and the Department's Surface Water Quality Information Database ("SQUID") all identify the waterbody as Cañon Largo (**EXHIBIT 55**). Amending the language to be consistent with common reference is critical for water quality standards implementation.

C. Removal of Hanging Period in 20.6.4.808 NMAC

The Department proposes removing a mistakenly placed period between the words "to" and "the" in the third line of the description for Section 20.6.4.808 NMAC. The State Record Center and Archives has clarified that grammatical corrections such as these do not require an amendment notation for the section (**NMED Exhibit 54**); therefore, the proposed amendment will not add an amendment notation.

VI. CONCLUSION

The Department recommends that the Commission adopt the proposed amendments to the Standards, filed as **NMED Exhibit 9**, based upon the testimony of the SWQB's witnesses.

This concludes my direct testimony.

STATE OF NEW MEXICO
WATER QUALITY CONTROL COMMISSION
IN THE MATTER OF: PROPOSED AMENDMENTS
TO STANDARDS FOR INTERSTATE AND INTRASTATE SURFACE WATERS
20.6.4 NMAC

WQCC 20-51(R)

DIRECT TECHNICAL TESTIMONY OF DIANA I. ARANDA

I. INTRODUCTION

My name is Diana Aranda, and I am presenting this written testimony (**NMED Exhibit 3**) on behalf of the New Mexico Environment Department ("Department" or "NMED") Surface Water Quality Bureau ("SWQB") concerning the Department's proposed amendments to the State of New Mexico's *Standards for Interstate and Intrastate Surface Waters* ("Standards"), codified as Title 20, Chapter 6, Part 4 of the New Mexico Administrative Code ("NMAC"). The Department is proposing these amendments (**NMED Exhibit 9**) in accordance with NMSA 1978, Section 74-6-6, 20.1.6 NMAC, 20.6.4.10(A) NMAC, and 40 Code of Federal Regulations ("C.F.R.") Section 131.20 (**NMED Exhibit 21**), in what is referred to as the State's "Triennial Review." My testimony describes the rationale and provides the supporting evidence for the following proposed changes:

- Adding a definition of the term "hardness" (20.6.4.7(H) NMAC);
- Amendments to the definition of "surface waters of the State" (20.6.4.7(S) NMAC);
- Amendments to the antidegradation policy and implementation plan (20.6.4.8 NMAC);
- Amendments to the procedures for nominating an Outstanding National Resource Water ("ONRW") (20.6.4.9(A) NMAC);

- Designated use amendments for classified waters with a secondary contact recreational use;
- Designated use amendments for selected classified non-perennial waters;
- Amendments to hardness-dependent metals criteria reference table (20.6.4.900(I)(3) NMAC);
- Amendments to the ammonia criteria for aquatic life (20.6.4.900(K), (L) and (M) NMAC); and
- Amendments to the publication references (20.6.4.901 NMAC)

II. QUALIFICATIONS

I have been employed with the Department's SWQB since February 2017. I worked with the Total Maximum Daily Loads Team for two years, and have been with the Standards, Planning, and Reporting Team ("Standards Team") since February 2019. I am an Environmental Scientist/Specialist-Advanced in the Standards Team where I am responsible for various aspects of developing water quality standards for New Mexico's surface waters in accordance with the State Water Quality Act and the federal Water Pollution Control Act or "Clean Water Act" ("CWA").

I hold a Bachelor of Science degree in Biology from the University of New Mexico, and a Master of Science degree in Coastal Zone Management from Nova Southeastern University. My master's work focused on recreational water quality assessment and my work was published in the Journal of Water & Health in 2016, with me as the principal author.

I have worked on water quality issues in various capacities, including: as a project manager in a consulting company; as a researcher for the National Oceanic and Atmospheric Administration ("NOAA") in collaborations with the U.S. Environmental Protection Agency ("EPA"), several

universities, and local agencies; as a marine biologist for Biscayne National Park; and as a researcher-in-training at University of Washington, Friday Harbor Laboratories. I have additional technical, research, laboratory, and teaching experience in other biology disciplines as well. A copy of my resume is included as **NMED Exhibit 7**. It is accurate and up to date.

III. AMENDMENTS ASSOCIATED WITH REFERENCING

A. Definition of Hardness

The Department proposes to add a definition of "hardness" to clarify the meaning of the term. As provided in the Department's proposed amendments (**NMED Exhibit 9**), this would become paragraph 20.6.4.7(H)(1) NMAC, while the existing paragraphs 20.6.4(H)(1) and (2) NMAC would be renumbered as paragraphs 20.6.4(H)(3) and (4) NMAC, respectively.

The word "hardness" is used several times throughout 20.6.4 NMAC. Adding a definition would clarify the term, providing consistency when implementing the State's hardness-based aquatic life use metals criteria (20.6.4.900(I) NMAC). This amendment is not intended to establish regulations nor implement any new processes and would not change the implementation of water quality standards.

B. Definition of Surface Waters of the State

The Department proposes to amend the format of the definition "surface water(s) of the State" to provide clarity, and aid with readability and referencing (**NMED Exhibit 9**). Although this definition would remain in paragraph 20.6.4.7(S)(5) NMAC, the Department proposes to reformat the definition to include subparagraphs. These subparagraphs include: the types of waters recognized as surface waters of the State; inclusions for surface waters of the State; and specific exclusions for waters of the State.

The amended format as proposed will clarify that ephemeral waters are surface waters of the State, as demonstrated through the adoption of designated uses for ephemeral waters and establishment of criteria that protect for those uses. This amendment would not change the definition of "surface water of the State" and would not alter the implementation of water quality standards.

C. Amendments to the State's Antidegradation Policy and Implementation Plan (20.6.4.8 NMAC)

The Department proposes three amendments to 20.6.4.8 NMAC, all of which provide clarification. For 20.6.4.8(A)(1) NMAC, the Department proposes to clarify that protections for existing uses apply to all surface waters of the State, not just streams. The Department is therefore proposing to remove the term "instream water." Although the federal antidegradation protections in 40 C.F.R. § 131.12 (**NMED Exhibit 27**) use the term "instream" to describe the water quality necessary to protect existing uses; in accordance with 40 C.F.R. § 131.12(a) (**NMED Exhibit 27**), the antidegradation policy is required to, at a minimum, protect and maintain existing uses for all waters of the United States, not just lotic waters. The State's antidegradation policy protects all surface waters of the State in the same manner.

Keeping the term "instream", as currently found in 20.6.4.8(A)(1) NMAC, may be misinterpreted to not to include protection of existing uses for lentic waterbodies. Amending the language as the Department proposes will clarify that existing uses are protected for all waters and reduce potential misapplication when implementing the antidegradation policy. As part of this clarification, the Department is also proposing to add language in this paragraph that references the "existing uses" definition.

For 20.6.4.8(A)(2) NMAC, the Department proposes to add language to clarify that high-quality waters exceeding the established levels of water quality are protected in accordance with 40 C.F.R. § 131.12(a)(2) (**NMED Exhibit 27**). High-quality waters are those waters in which either some or all water quality criteria exceed the goals set forth in Section 101(a)(2) of the CWA (33 U.S.C. § 1251) (**NMED Exhibit 10**).

In addition, the Department is proposing to correct the spelling of the acronym for Outstanding National Resource Waters ("ONRW") in 20.6.4.8(A)(3)(iv) and 20.6.4.8(A)(4)(iv) NMAC for consistency and accuracy.

D. Amendments to Procedures for nominating an ONRW (20.6.4.9(A) NMAC)

The Department proposes to amend language in the procedures for nominating an ONRW, as found in 20.6.4.9(A) NMAC, to reflect the appropriate procedural reference found in 20.1.6 NMAC. In March 2018, the former guidelines for Water Quality Control Commission ("WQCC" or "Commission") regulation hearings were superseded by the adoption of Rulemaking Procedures - Water Quality Control Commission (20.1.6 NMAC). The Department is proposing to amend this section to reflect the correct reference for rulemaking proceedings before the Commission.

E. Amendments to Publication References (20.6.4.901 NMAC)

The Department proposes to amend the publication references section to be consistent with the most current versions of these documents. Overall, some references needed updates, some were removed because they were no longer the correct reference and replaced with the correct references, and some were reworded for consistency within the reference section. These changes will facilitate appropriate implementation of the State's water quality standards.

IV. AMENDMENTS ASSOCIATED WITH DESIGNATED USES

A. Amendments to Selected Sections that Contain Secondary Contact Uses

Based on the findings of the Department's *Existing Use Analysis of Recreational Use for Classified Waters 20.6.4.101-20.6.4.899 NMAC* (**NMED Exhibit 56**), the Department proposes to amend language in 20.6.4.103, 20.6.4.112, 20.6.4.116, 20.6.4.204, 20.6.4.206, and 20.6.4.207 NMAC, and add a new section, 20.6.4.231 NMAC.

Note: the proposed language (**NMED Exhibit 9**) for section 20.6.4.206 NMAC also includes amended language discussed in Section IV-B of this testimony.

1. Background

Pursuant to 40 C.F.R. § 131.10(i) (**NMED Exhibit 22**), states are required to revise their standards to reflect the highest attainable use, which in turn is their existing use, whether or not it is currently being attained. An existing use, as defined in 40 C.F.R. § 131.3(e) (**NMED Exhibit 26**) and 20.6.4.7(E)(3) NMAC, is the use that is actually attained in the waterbody on or after November 28, 1975, whether or not they are included in the water quality standards. As part of the prior Triennial Review [WQCC 14-05(R)], the Department petitioned the WQCC to amend select waterbodies that have a secondary contact recreational designated use. The petition proposed to convert these waterbodies from secondary contact use to primary contact use under the premise that primary contact is likely attainable, see further details in Section III-A of the EUA (**NMED Exhibit 56**). Ultimately, the WQCC did not adopt the proposed amendments due to the lack of sufficient credible scientific evidence to establish the existing use as primary contact, see further details in Section III-A of the EUA (**NMED Exhibit 56**).

Given the historical context of this proposed amendment, this EUA provides a comprehensive investigation that demonstrates the existing recreational use attained by these waterbodies, through the evaluation of available data (**NMED Exhibit 56**).

2. Waterbodies evaluated

The Department reviewed all classified waters, 20.6.4.101-20.6.4.899 NMAC, to identify sections with a recreational designated use of secondary contact. Lakes, waterbodies with site-specific criteria, and other classified waters undergoing designated use investigations were excluded from the review. The EUA focused on classified waters which all have a secondary contact recreational use in: 20.6.4.103, 20.6.4.116, 20.6.4.204, 20.6.4.206, and 20.6.4.207 NMAC. Appendices B and C of the EUA (**NMED Exhibit 56**) detail the waterbodies evaluated along with their geographical locations.

The waterbodies evaluated as part of this EUA are located in the south-central, north-central and southeastern areas of the State. Their geographical areas also cover various ecological conditions that vary in elevation, geology, ecology, and weather conditions. Further details can be found in Sections VI-A and B of the EUA (**NMED Exhibit 56**).

Two of the waterbodies for which the Department proposed amendments, the Rio Salado and the Rio Chama, cross jurisdictional boundaries with tribal lands. The Department recognizes the importance of communication and collaboration with tribes to protect water quality across boundaries. Therefore, in accordance with the State-Tribal Collaboration Act, NMSA 1978, Section 11-18-3 (**NMED Exhibit 17**), Executive Order 2005-004, and the Department's Tribal Communication and Collaboration Policy[1]; the Department provided a specific notification regarding the existing recreational use analysis to all tribes on January 28, 2021 (**NMED Exhibit**

[1] Available at https://www.env.nm.gov/general/wp-content/uploads/sites/10/2016/08/2020-01-27-NMED-Tribal-Policy-2020-final-signed.pdf.

57). This communication was intended to ensure the proposed action, to amend the recreational designated use from secondary contact to primary contact, would not cause concern for tribes to which the waters crossed between State and Tribal jurisdiction. Due to this outreach effort, the Pueblo of Jemez met (virtually) with the Department on February 5, 2021, and as a result now has no outstanding concerns regarding the proposed amendment.

3. Antidegradation evaluation

As part of the State's antidegradation policy, any analysis evaluating designated uses must maintain the protection of all waters of the State. These protections include: protections for existing uses; protections for those waters where the quality of a surface water of the State exceeds levels necessary to support aquatic life, wildlife and recreational uses; and protections for waters with exceptional water quality designated as ONRWs. The antidegradation policy also requires an evaluation of downstream waters to ensure their protections are also sustained, should a designated use amendment be supported.

The Antidegradation Policy ensures the protection of the existing uses in waterbodies. Since the goal of the EUA is to evaluate the aquatic life designated use to reflect a more stringent existing use, then no degradation of a waterbody's existing use will occur.

The EUA also evaluated the selected waterbodies for existing ONRW designations. Of the evaluated waterbodies, only Las Animas Creek (20.6.4.103 NMAC), within the Aldo Leopold wilderness, was identified to have an ONRW designation. *See* 20.6.4.9(D)(3)(a)(i) NMAC. Given the proposed designated use amendment has more stringent criteria than the current designated recreational use, the protections afforded to Las Animas Creek within the Aldo Leopold wilderness would not cause degradation of water quality. The EUA concluded that these waters have more

stringent existing uses than the current designated uses; therefore, the proposed amendment would also protect downstream waters.

4. Threatened and endangered species review

In accordance with Section 7(a)(2) of the federal Endangered Species Act ("ESA"), the EPA is required to consult with the U.S. Fish and Wildlife Service ("USFWS") to ensure that any action authorized by the EPA is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of habitat of such species. In addition, this review also aids the State in identifying any possible threats to threatened and endangered species before proceeding with changes to the water quality standards.

To achieve the goals of both assisting EPA with their requirements (although not mandatory) and identifying threatened and endangered species for the State, the Department conducted a preliminary evaluation. The EUA evaluated threatened and endangered species according to the USFWS's Information for Planning and Consultation ("IPaC") project planning tool. The evaluation identified the geographical location of the waterbodies under review, and located listed species and/or critical habitat that overlapped those locations using the IPaC tool. The preliminary screening of listed threatened and endangered species and further details can be found in Section V and Appendix C of the EUA (**NMED Exhibit 56**).

The Department does not believe the proposed amendments to change secondary contact to primary contact will jeopardize the continued existence of any threatened or endangered species; nor result in the destruction or adverse modifications of critical habitat, because this evaluation considers designated uses that would be more stringent. This increased protection would presumably not negatively affect or degrade species habitat, but rather would provide enhanced water quality to the waterbodies and possibly further protect those species dependent on them.

5. National Pollutant Discharge Elimination System

The EUA contains a review of EPA's National Pollutant Discharge Elimination System ("NPDES") individual permits and stormwater permits (construction, industrial, and municipal) that are associated with each of the waterbodies being evaluated. Permitted point source discharges can affect water quality, yet, the permits identified as authorizing such discharges do not provide direct evidence or support towards establishing an existing use.

The Department evaluated NPDES permits associated with the waterbodies under review and identified five relevant permits. Four of the permits currently have *E. coli* discharge limits greater than the primary contact numeric criteria and may be affected by the proposed amendments. A summary table of the permits and other details can be found in Section VI-D of the EUA (**NMED Exhibit 56**).

The Department identified these permittees as stakeholders, since they may be directly affected by the proposed amendment. As such, the Department sent a letter on January 20, 2021, to inform the permittees of the proposed amendments (**NMED Exhibit** 58). The Department received a phone call from Mayor Louie Gallegos, on behalf of the Village of Fort Sumner Wastewater Treatment Plant, to get more information regarding the proposed changes to the recreational use designation. The Department did not receive any other communications from the permittees regarding the proposed amendment.

6. Recreational designated use criteria

The State's recreational designated use primary contact numeric criteria is based on the EPA's 2012 recommended *E. coli* single grab numeric criteria. According to 20.6.4.900(D) NMAC, to attain primary contact criteria, the pH must be within a range of 6.6 to 9.0 and *E. coli* must not exceed the monthly geometric mean of 126 colony forming units per 100 milliliters (126

cfu/100 mL) or exceed a single sample of 410 cfu/100 mL. According to 20.6.4.900(E) NMAC, to achieve secondary contact, *E. coli* must not exceed a monthly geometric mean of 548 cfu/100 mL or exceed a single sample of 2507 cfu/100 mL.

7. Data used for analysis

For this analysis, the Department extracted pH and *E. coli* data for the waterbodies under review from two different SWQB databases. A search for data collected before or during 2009 was conducted by querying validated SWQB Monitoring Team archived folders on the Department's shared server. Data collected after 2009 were acquired through the SWQB's Oracle-based Surface water Quality Information Database ("SQUID"). Archived and SQUID *E. coli* and pH data were collected for classified waters identified in 20.6.4.103, 20.6.4.116, 20.6.4.204, 20.6.4.206 and 20.6.4.207 NMAC. For data summary findings, see Appendix B and Section 7 of the EUA (**NMED Exhibit 56**).

8. Methods

Historical field data records were extracted and parsed to contain only the waterbodies under review with their associated *E. coli* and pH data. Any waterbodies that did not contain both pH and *E. coli* data were excluded from the analysis and were not included in the proposed recreational use designation change.

Then the Department determined which of the waterbodies meet the primary contact pH range of 6.6 to 9.0. Any waterbodies that were not within the appropriate pH range were excluded from the analysis and were not included in the proposed recreational use designation change.

Finally, the Department analyzed the *E. coli* data. For this analysis, the single grab criterion was utilized since the number of samples necessary to calculate a monthly geometric mean were not available. The analysis utilized the primary contact single grab *E. coli* criterion of 410 cfu/100

mL for the recreational designated existing use determination. If the waterbody segment contained at least one *E. coli* sample result equal to or less than 410 cfu/100 mL, then the existing use was determined to be at least primary contact. This single sample determination comes from the existing use definition in 20.6.4.7(E)(3) NMAC and 40 C.F.R. § 131.3 (**NMED Exhibit 26**); where an existing use equals the actual use that has been attained by a surface water. Meaning, even though the water could contain samples above 410 cfu/100 mL, if the water contains at least a single sample that was at or less than 410 cfu/100 mL, then it demonstrates that the water can actually attain that criterion. Therefore, if a segment under review achieves primary contact use designation once, then that is the appropriate designated use. However, if the waterbody segment single sample results for *E. coli* are all greater than 410 cfu/100 mL, then that waterbody segment's existing use was determined to be appropriately designated under secondary contact.

A summary table of the data findings is located in Tables VII-1 and VII-2 of the EUA (**NMED Exhibit 56**).

9. Findings

All waterbodies under review had available data records associated with recreational use criteria except for three waterbody segments, shown in Table VII-1 of the EUA as "No Data" (**NMED Exhibit 56**). The Department did not find an existing use dataset for three waterbody segments; therefore, the Department is not proposing a change in the designated recreational use for the following segments.

1. The Rio Felix (20.6.4.206 NMAC);
2. Perennial reaches of tributaries to the Rio Hondo downstream of Bonney canyon excluding North Spring river, which does have data (20.6.4.206 NMAC); and

3. Perennial reaches of tributaries to the Rio Grande in Sierra and Socorro counties excluding Rio Salado, Percha Creek, Alamosa Creek, Abo Arroyo, Las Animas Creek, and Palomas Creek (20.6.4.103 NMAC).

The remaining waterbody segments the Department analyzed were all within a pH range of 6.6 to 9.0 and all contained one or more *E. coli* samples at or less than the 410 cfu/100 mL primary contact criterion, see Table's VII-2 and VII-3, in the EUA (**NMED Exhibit 56**). The analysis also found exceedances of the *E. coli* criterion for primary contact above 410 cfu/100 mL within the waterbody segments. However, exceedances within the segment reach, are not an indication that primary contact use is not attainable. An existing use, by definition, states two supporting arguments towards a designated use change even if a segment reach contains exceedances of the criterion: first, that the existing use was actually reached, and second, this designation is appropriate whether or not that existing use is currently being attained.

10. Conclusion

Of the waterbodies with available data, all were within a pH range of 6.6 to 9.0 and at least one *E. coli* sample result less than or equal to 410 cfu/100 mL. These findings assert that the select listed waterbodies attain the criteria for primary contact recreational use. Therefore, in accordance with 40 C.F.R. § 131.10(i) (**NMED Exhibit 22**), the Department proposes amending, with some exceptions, the designated recreational use for classified waters in 20.6.4.103, 20.6.4.116, 20.6.4.204, 20.6.4.206, and 20.6.4.207 NMAC. The exceptions include: the Rio Felix (20.6.4.206 NMAC); the perennial reaches of tributaries to the Rio Hondo downstream of Bonney canyon excluding North Spring River (20.6.4.206 NMAC); and the perennial reaches of tributaries to the Rio Grande in Sierra and Socorro counties excluding Rio Salado, Percha Creek, Alamosa Creek, Abo Arroyo, Las Animas Creek and Palomas Creek (20.6.4.103 NMAC).

The Department proposes the division of 20.6.4.103 NMAC to remove the waterbodies identified as attaining a primary contact recreational use, and the incorporation of these waterbodies into a new section, 20.6.4.112 NMAC. The Department proposes the amendments of 20.6.4.116 and 20.6.4.204 NMAC to a primary contact recreational use. Finally, the Department proposes the division of 20.6.4.206 NMAC to remove the waterbodies identified as attaining a primary contact recreational use and incorporation of these waterbodies into a new section, 20.6.4.231 NMAC. Those waters for which the Department is not proposing amendments would remain in their established classified sections within 20.6.4 NMAC. The amended language, as proposed by the Department, can be found in **NMED Exhibit 9**.

B. Amendments to Selected Sections of non-perennial reaches

Based on the findings of the Department's *Use Attainability Analysis for Select Non-Perennial Reaches in Classified Waters 20.6.4.101-20.6.4.899 NMAC* ("non-perennial UAA") (**NMED Exhibit 59**), the Department proposes to amend language in 20.6.4.108, 20.6.4.115, 20.6.4.206, 20.6.4.208, 20.6.4.209, 20.6.4.215, 20.6.4.220, 20.6.4.307 and 20.6.4.309 NMAC.

Note: the proposed language (**NMED Exhibit 9**) for 20.6.4.206 NMAC, also includes amended language proposed and discussed in Section IV-A of this testimony.

1. Background

Pursuant to 40 C.F.R. § 131.10(g) (**NMED Exhibit 22**) and 20.6.4.15(A) NMAC, in order to amend a designated use (that is not an existing use) to one with less stringent criteria, a Use Attainability Analysis ("UAA") must be conducted. A UAA is a scientific study that demonstrates that the attainment of a use is not feasible due to one of the six factors listed in 40 C.F.R. § 131.10(g) (**NMED Exhibit 22**). The UAA is required to demonstrate the following: that the designated use is not the existing use; that one of the factors in 40 C.F.R. 131.10(g) (**NMED**

Exhibit 22) are preventing the attainment of the use; and finally, determine the highest attainable use as defined in 40 C.F.R. 131.3(m) (**NMED Exhibit 26**). According to these requirements, the non-perennial UAA has provided the appropriate demonstrations and proposes amending the designated uses for classified non-perennial waters to those designated uses associated with unclassified non-perennial waters in 20.6.4.98 NMAC.

This testimony will explain the rationale behind the non-perennial UAA, as well as provide an evaluation of protections afforded under the antidegradation policy; general site characteristics associated with these waters; the threatened and endangered species that the proposed amendments may effect; evidence of the factor preventing attainment of the current designated use; and evidence on how the Department established the highest attainable use.

2. Reasoning for evaluation

As part of the 2005 Triennial Review [WQCC 03-05(R)], the Commission adopted designated uses for unclassified, non-perennial waters of the state through the creation of a new section; 20.6.4.98 NMAC. As part of this amendment, and as established in the Commission's statement of reasons (**NMED Exhibit 60**), most of the classified non-perennial waters were unclassified and designated in those uses identified in 20.6.4.98 NMAC. However, for the present Triennial Review, the Department identified that some non-perennial waters are still classified in 20.6.4.101-20.6.4.899 NMAC. For details, see Section III-A, Appendix A and Appendix B in **NMED Exhibit 59**.

For the reasons described above, the non-perennial UAA intends to evaluate the remaining non-perennial waterbody portions that are still identified in 20.6.4.101-20.6.4.899 NMAC and amend the waterbodies, as appropriate.

3. Procedure

First, a UAA must demonstrate that a designated use is not attainable due to one of the factors identified in 40 C.F.R. § 131.10(g) (**NMED Exhibit 22**). The designated use, whether current or proposed, shall not be less stringent than the existing use. As defined in 20.6.4.7(E)(3) NMAC, existing uses are actually attained in a surface water of the State on or after November 28, 1975, whether or not it is a designated use. The existing use may or may not be the current water quality of any given waterbody. The UAA must also include an evaluation of antidegradation conditions and any other site details that could prevent the removal of a designated use.

Second, once it is demonstrated that the designated use is not attainable, the highest attainable use must be determined. Establishing the highest attainable use requires an evaluation of existing uses, biotic and abiotic conditions, anthropogenic influences, and the consideration of protected status.

4. Waterbodies and designated uses evaluated

The non-perennial UAA evaluated classified non-perennial waters in the following sections of 20.6.4 NMAC:

1. 20.6.4.108 NMAC, tributaries to the Jemez river above Soda dam and tributaries to the Guadalupe river;
2. 20.6.4.115 NMAC, tributaries to the Rio Vallecitos;
3. 20.6.4.206 NMAC, tributaries to the Rio Hondo downstream of Bonney canyon;
4. 20.6.4.208 NMAC, tributaries to the Rio Peñasco above state highway 24 near Dunken;
5. 20.6.4.209 NMAC, tributaries to the Rio Bonito upstream of state highway 48 (near Angus) and tributaries to the Rio Ruidoso upstream of the U.S. highway 70 bridge near Seeping Springs lakes, above and below the Mescalero Apache boundary;

6. 20.6.4.215 NMAC, tributaries to the Gallinas river upstream of the diversion for the Las Vegas municipal reservoir;
7. 20.6.4.220 NMAC, tributaries to the Gallinas river from its mouth upstream to the diversion for the Las Vegas municipal reservoir;
8. 20.6.4.307 NMAC, tributaries to Ocate creek downstream of Ocate; and
9. 20.6.4.309 NMAC, tributaries to Coyote creek and tributaries to Rayado creek above the Miami lake diversion.

The Department evaluated the designated uses for both aquatic life and recreational use for each of the listed waterbodies. The designated uses not directly associated with uses in Section 101(a)(2) of the CWA (33 U.S.C. § 1251) (**NMED Exhibit 10)** were evaluated for removal. These other uses include domestic water supply, public water supply, industrial water supply, irrigation, and fish culture.

5. Antidegradation evaluation

Any analysis evaluating designated uses must take into consideration and maintain the protection of all waters of the State under the state antidegradation policy. These include protections for existing uses; protections for those waters where the quality of a surface water of the State exceeds levels necessary to support aquatic life, wildlife, and recreational uses; and protections for waters with exceptional water quality designated as ONRWs. The antidegradation policy also requires an evaluation of downstream waters to ensure their protections are also sustained, should a designated use amendment be supported.

Since water quality in ONRWs cannot be degraded, the non-perennial UAA reviewed whether any waterbodies considered in the analysis are ONRWs. According to the list of ONRWs in 20.6.4.9(D) NMAC, none of the waterbody portions evaluated in the non-perennial UAA are

designated as ONRWs. Therefore, a change in criteria to one less stringent would not invoke antidegradation protections associated with ONRWs.

The Department searched readily available water quality data to identify the existing uses for each of the classified non-perennial waters considered for a designated use amendment. The Department's data search using SWQB's in-house database, SQUID, found no applicable data for these waterbodies. Consequently, existing uses could not be established. Since no existing uses were established, the implementation of this amendment will not result in the lowering of any known existing use.

6. Threatened and endangered species review

In accordance with Section 7(a)(2) of the ESA, the EPA is required to consult with the USFWS to ensure that any action authorized by the EPA is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of habitat of such species. In addition, this review also aids the Department in identifying any possible threats to threatened and endangered species before proceeding with changes to the water quality standards.

To achieve the goals of both assisting EPA with their requirements (although not mandatory), and identifying threatened and endangered species for the State, the Department conducted a preliminary evaluation. The UAA evaluated threatened and endangered species according to the USFWS's IPaC project planning tool. The evaluation identified the geographical location of the waterbodies under review, and located listed species and/or critical habitat that overlapped using the IPaC tool. The preliminary screening of listed threatened and endangered species and further details can be found in Section V and Appendix C of the non-perennial UAA (**NMED Exhibit 59**).

The proposed amendments to designated uses for non-perennial portions should not jeopardize the continued existence of any threatened and endangered species nor result in the destruction or adverse modification of critical habitat since it is not amending the natural conditions attainable by these waterbodies. Nor is it believed that these amendments would jeopardize natural communities of conservation concern (e.g., emergent wetland, riverine wetland, prairie, glade, fen, etc.) because the proposed amendment does not alter habitat, only the attainable water quality. In addition, the identification of the attainable uses for these classified waters could ensure comprehensive protections to the waterbodies.

7. Site condition assessment

Even though anthropogenic disturbances can affect water quality, the non-perennial UAA did not propose the removal of reaches that have low flow due to anthropogenic land use or dams and impoundments. Therefore, it was beyond the scope of the analysis to evaluate anthropogenic inputs and dam influences in any level of detail.

Non-perennial tributaries, regardless of anthropogenic influences, should be able to attain the established designated uses. Therefore, the potential anthropogenic impacts on water quality did not factor into the non-perennial UAA.

a) Surface water diversions

Surface water diversions may contain designated uses such as domestic water supply, fish culture and water supply, and irrigation. Therefore it is important to investigate if any of the waterbodies under review contained diversions that could affect the waterbodies natural hydrology.

The data for surface water diversion locations and designated uses, were obtained from the New Mexico Office of the State Engineer (“OSE”) "Point of Diversions" (“POD”) Open Data Site.

Details of the evaluation can be found in Section VI-C of the non-perennial UAA (**NMED Exhibit 59)**. This review identified surface declarations and permits associated with current designated uses for domestic water supply, fish culture and water supply, and irrigation. The Department recommends that the waterbodies with water rights permits remain in their current classified segments. These waterbodies include Calaveras Canyon (20.6.4.108 NMAC) and Cox Canyon (20.6.4.208 NMAC). A summary of the diversion evaluation can be found in Table VI-1 of the non-perennial UAA (**NMED Exhibit 59**).

b) National Pollutant Discharge Elimination System and Stormwater General Permit

The non-perennial UAA contains a review of EPA's NPDES individual permit and stormwater permits (construction, industrial, and municipal) associated with the waterbodies under review to identify any permits that would be affected by a change in designated use.

The NPDES individual Permit NM0030180 issued to Chevron Mining, Inc., Ancho, Gachupin, and Brackett Mines, regulates the discharge of mine drainage from active reclamation areas during precipitation events, see Table VI-2 in **NMED Exhibit 59**. One of the mines includes temporary impoundments and other measures to control sediment, which may affect flow conditions. Discharges under this permit are intermittent and do not provide a consistent source of water to offset the non-perennial nature of these tributaries.

8. Removal of a designated use.

The authority to remove a designated use falls under 40 C.F.R. § 131.10 (**NMED Exhibit 22**), in which states can remove a designated use if it is not an existing use, and if the rationale for removal of a designated use falls under one of the six factors in 40 C.F.R. § 131.10(g) (**NMED Exhibit 22**).

According to 40 C.F.R. § 131.10(g)(2) (**NMED Exhibit 22**)**,** a use may be removed if it is found unattainable based on "natural, ephemeral, intermittent or low flow conditions or water levels prevent the attainment of the use, unless these conditions may be compensated for by the discharge of sufficient volume of effluent discharges without violating State water." In addition to aquatic life use, the designated uses that would warrant removal from the selected waterbodies due to low flow conditions include: domestic water supply, fish culture and water supply, and irrigation. The non-perennial tributaries in one segment would see a higher attainable recreational use under 20.6.4.98 NMAC.

To further verify the non-perennial conditions for these waterbodies, the non-perennial UAA included an evaluation based on the National Hydrography Dataset ("NHD"). Details of this evaluation can be found in Section VII-C of the non-perennial UAA (**NMED Exhibit 59**). The NHD evaluation confirmed the presence of intermittent waterbody segments within the sections identified. Therefore, this evidence supports the removal of the aquatic life designated use based on low flow, ephemeral, or intermittent conditions.

9. Establishing the highest attainable use

The highest attainable aquatic life use was determined using the evidence presented by the USFWS in their testimony during the 2005 Triennial Review (see Section VIII-A of **NMED Exhibit 59**) and the Department's NHD map analysis. The 2005 Triennial Review USFWS testimony provided evidence that the aquatic life use under 20.6.4.98 NMAC was the highest attainable use for non-perennial intermittent waters. The NHD map analysis confirmed that the classified sections evaluated under this UAA contain non-perennial segments. Therefore, it is appropriate to designate the aquatic life use for non-perennial tributaries identified in this UAA as marginal warmwater as found under 20.6.4.98 NMAC. The listings for the selected waterbodies'

highest attainable designated uses would now be livestock watering, wildlife habitat, marginal warmwater aquatic life, and primary contact. A summary of the changes in designation for these non-perennial portions can be found in Table VIII-1, **NMED Exhibit 59**.

10. Conclusions

The non-perennial UAA presented evidence that the current designated uses for several classified non-perennial waterbodies are unattainable, with the exclusion of Calaveras Canyon (20.6.4.108 NMAC) and Cox Canyon (20.6.4.208 NMAC). This conclusion is based on three pieces of evidence: a designated use change will not result in the lowering of any known existing uses; the designated uses are not attainable due to factor 40 C.F.R. § 131.10(g)(2) (**NMED Exhibit 22**), where natural low-flow, intermittent, or ephemeral conditions prevent attainment of the current designated uses; and that the highest attainable use was determined to be consistent with 20.6.4.98 NMAC, as supported by both the NHD non-perennial water determination and the evidence presented by the USFWS testimony during the 2005 Triennial Review.

The highest attainable uses for unclassified non-perennial waters that are established in 20.6.4.98 NMAC include: livestock watering, wildlife habitat, marginal warmwater aquatic life, and primary contact.

The non-perennial UAA supports the removal of non-perennial reaches of tributaries from the listed classified sections, and establishes the highest attainable uses under 20.6.4.98 NMAC. As such, the Department is proposing amending the language to "perennial reaches of tributaries to" within the appropriate classified sections to clarify language and ensure appropriate application of the Standards **NMED Exhibit 9**. If amended, the non-perennial waters evaluated by the non-perennial UAA would fall under 20.6.4.98 NMAC (unclassified, non-perennial surface waters of

the state) with designated uses for: livestock watering, wildlife habitat, marginal warmwater aquatic life, and primary contact.

V. AMENDMENTS ASSOCIATED WITH CRITERIA

A. Amendments to Hardness-Dependent Criteria Table (20.6.4.900(I) NMAC)

The Department proposes to amend the values in the acute and chronic hardness-based metals criteria table located in 20.6.4.900(I)(3) NMAC consistent with the hardness-dependent equations for acute and chronic aquatic life criteria.

Aquatic life criteria for some metals are based on bioavailability, limited, in part, by hardness. The calculations for these criteria involve a standardized algorithm with variables specially derived for each metal.

The table in this section uses the EPA's national recommended aquatic life water quality criteria or, in some cases, criteria independently adopted by the State. The criterion for a hardness-based metal is specific to the water hardness (expressed as calcium carbonate, calculated from calcium and magnesium) at the time of sample collection. Because hardness varies over time, this leads to aquatic life metals criteria that can fluctuate. Due to these algorithms' complexity and the potentially fluctuating criteria concentrations based on varying hardness concentrations in water, the State's water quality standards include a table in 20.6.4.900(I)(3) NMAC depicting the graduating hardness levels with corresponding metals criteria to serve as a reference.

As part of the Triennial Review process, the Department verified the calculations for the hardness-based table values. This review determined that the table value criteria for chromium III and nickel had several errors. To fix these errors, the Department recalculated the equations using Microsoft Excel. These equations, along with the equation parameters from the State's water quality standards (20.6.4.900(I)(1) and 20.6.4.900(I)(2) NMAC), were recalculated to ensure that

the calculations were correct. In addition, staff ran the calculations manually to validate the Microsoft Excel computations. The amended table as proposed in **NMED Exhibit 9** reflects the correct calculated criteria values for varying water hardness concentrations.

Apart from cadmium, as provided in the direct written testimony of Jennifer Fullam (**NMED Exhibit 4**), the Department does not propose any change to the algorithms for hardness-based aquatic life; therefore, implementation of the hardness-based criteria will not be affected. Correcting the table value errors will provide consistency with calculated criteria and prevent the misapplication of water quality standards, particularly regarding the NPDES permitting program.

B. Amendments to the State's Ammonia Criteria (20.6.4.900(K), (L) and (M) NMAC)

For aquatic life ammonia criteria as found in 20.6.4.900(K), (L) and (M) NMAC, the Department proposes to update the criteria to be consistent with the federal aquatic life ambient water quality criteria for total ammonia, including acute and chronic aquatic life criteria for Total Ammonia Nitrogen ("TAN").

40 C.F.R. § 131.11 (**NMED Exhibit 25**) requires that states adopt the following: the EPA's recommended criteria based on Section 304(a) of the CWA (33 U.S.C. § 1314) (**NMED Exhibit 12**) guidance; a modification of the EPA's recommended criteria based on site-specific conditions; or any other defensible scientific reasoning. If a state does not adopt the EPA's recommended criteria, then the state must provide reasoning for this decision. Additionally, states are required to review new or updated recommended criteria during the triennial review of water quality standards, in partial fulfillment of 40 C.F.R. § 131.20 (**NMED Exhibit 21**).

In August 2013, the EPA published an update to its national recommended freshwater ammonia criteria for the protection of aquatic life (**NMED Exhibit 61**). Upon review of the EPA's

2013 recommended aquatic life criteria for ammonia, the Department is proposing to update the State's criteria accordingly.

The EPA based the updated ammonia criteria on the most recent scientific studies incorporating toxicity data for freshwater mussels (unionid) and freshwater gill-breathing snails. According to the EPA's 2013 recommended criteria, the toxicity studies demonstrate that these particular types of freshwater mussels and snails (including endangered species) are the most sensitive species to ammonia. As a result, the new criteria are more stringent than the previous aquatic life ammonia criteria (**NMED Exhibit 61**).

Although the EPA has updated the recommended criteria, the recommended exposure duration remains the same. The acute criterion duration has a one-hour average, and the chronic criterion has a 30-day average. The recommended acute criterion value for TAN is now 17 mg TAN per liter ("mg/L) at a pH of 7 and 20 degrees Celsius ("°C"), and the recommended chronic criterion for TAN is now 1.9 mg/L at a pH of 7 and 20°C, **NMED Exhibit 61**.

When water temperatures are greater than 15.7°C, the acute criterion value is determined primarily by effects on freshwater unionid mussels. At temperatures lower than 15.7°C, the acute criterion is based primarily on effects on *Oncorhynchus* spp. (a genus of fish in the family *Salmonidae*) (**NMED Exhibit 61**).

The Department is proposing the adoption of the EPA's recommendation to limit the highest four-day average within the 30-day averaging period to be no more than 2.5 times the criterion.

The new criteria, if adopted, would be more stringent and may impact permittees with ammonia effluent limits.

VI. CONCLUSION

The Department urges the Commission adopt the proposed amendments to the Standards, filed as **NMED Exhibit 9**, based upon the testimony of the Department's witnesses.

This concludes my direct testimony.

STATE OF NEW MEXICO
WATER QUALITY CONTROL COMMISSION

IN THE MATTER OF: PROPOSED AMENDMENTS
TO STANDARDS FOR INTERSTATE AND **WQCC 20-51(R)**
INTRASTATE SURFACE WATERS
20.6.4 NMAC

DIRECT TECHNICAL TESTIMONY OF JENNIFER FULLAM

I. INTRODUCTION

My name is Jennifer Fullam, and I am presenting this written direct testimony (**NMED Exhibit 4**) on behalf of the New Mexico Environment Department ("Department" or "NMED") in support of the Department's proposed amendments to the State of New Mexico's *Standards for Interstate and Intrastate Surface Waters* ("Standards"), codified as Title 20, Chapter 6, Part 4 of the New Mexico Administrative Code ("NMAC"). Section 303(c)(1) of the federal Clean Water Act ("CWA") (33 U.S.C. § 1313) (**EXHIBIT 11**) requires that the State hold public hearings at least once every three years to review and amend, as appropriate, its water quality standards. The Department is proposing these amendments consistent with NMSA 1978, Section 74-6-6 (**NMED Exhibit 15**), 20.1.6 NMAC, 20.6.4.10(A) NMAC, Section 303(c)(1) of the CWA (33 U.S.C. § 1313), and 40 Code of Federal Regulations ("C.F.R.") Section 131.20 (**NMED Exhibit 21**) in what is referred to as a "Triennial Review" of the State's Water Quality Standards. My testimony will provide supporting evidence for amendments being proposed by the Department that clarify uses and criteria; provide testimony regarding proposed amendments for hardness-based criteria; provide the supporting evidence for proposed designated use amendments for three classified intermittent tributaries within Los Alamos National Laboratory ("LANL"); and finally, provide evidence of the Department's efforts to ensure compliance with the regulatory process, including tribal outreach, stakeholder outreach and public engagement.

II. QUALIFICATIONS

I am currently employed as the Standards, Planning and Reporting Team Supervisor and serve as the Water Quality Standards Coordinator within the Department's Surface Water Quality Bureau ("SWQB") and have been in this position for four years as of March 27, 2021. Within the past four years I have served as an expert witness and provided testimony in three rulemaking proceedings before the Water Quality Control Commission ("Commission" or "WQCC"). These include designated aquatic life use amendments for Dog Canyon and Tecolote Creek; designated aquatic life use and recreational use amendments for San Isidro Arroyo and tributaries to San Isidro Arroyo within Lee Ranch Mine; and, adoption of the State's first temporary standard for the City of Raton's Wastewater Treatment Facility. As part of these rulemakings, and as the Standards Coordinator, I have filed amended rules with the State and the U.S. Environmental Protection Agency ("EPA") in accordance with state and federal regulations. In addition to rulemakings, I have also provided technical review of the EPA-recommended criteria for aluminum, provided review of work plans for potential Use Attainability Analyses ("UAAs") submitted by third-parties to the Department in accordance with 20.6.4.15(D) NMAC, overseen the filing and approval of the Triennial Review which became effective in 2017, and participated in the updates to the State's Water Quality Management Plan/Continuing Planning Process, which was approved by the Commission and EPA in 2020.

Overall, I have been with the Department for over 13 years. In addition to my current role, I have served as an Environmental Scientist for the Ground Water Quality Bureau Pollution Prevention Section and as the Delivery Prohibition Coordinator for the Petroleum Storage Tank Bureau. Prior to my service with the Department, I was the Environment Department Director for Tesuque Pueblo where, among other duties, I managed the surface and ground water quality

programs including conducting the Triennial Review for the Pueblo's surface water quality standards. I also served as a Graduate Research Assistant with LANL investigating polychlorinated biphenyl ("PCB") exposure pathways in surface water and as a biologist and Geographical Information System specialist with the Pueblo of Pojoaque.

I hold a Bachelor of Science degree from the University of New Mexico in Biology with a minor in geography (emphasis in remote sensing and geographical information systems) and a Master of Science degree from New Mexico Highlands University in Environmental Science and Management. My publications include Gonzales, G. and Montoya, J. (Fullam), 2005. *Polychlorinated biphenyls (PCBs) in the Rio Grande Sampled Using Semi-Permeable Membrane Devices* LA-14200, and Fullam, J., 2008. *Elk Habitat Utilization Within Lower Pinon Juniper Forests of Tesuque Pueblo*, New Mexico Highlands University Graduate Thesis. A copy of my resume is included as **NMED Exhibit 8**. It is accurate and up-to-date.

III. DEFINITIONS ASSOCIATED WITH USES AND CRITERIA

A. Definitions Relating to Use (20.6.4.7 NMAC)

1. Attainable Use

The Department proposes to add language to the definition for "attainable use" (20.6.4.7(A) NMAC) to aid in the implementation of Water Quality Standards. As written, the definition for "attainable" does not clarify that it refers specifically to a type of use for a surface water of the state. Other uses for surface waters of the state are "designated use" and "existing use", both of which include the word "use" in their definition to distinguish them from the common usage of the word.

Pursuant to 40 C.F.R. § 131.10(d) **(NMED Exhibit 22)**, at a minimum, uses are attainable if they can be achieved by the imposition of effluent limits required under Sections 301(b) and 306

of the CWA and cost-effective and reasonable best management practices for nonpoint source control. In conjunction with the established existing use, the attainable use provides the minimum level of water quality protection for a designated use. Amending the definition to include the word "use" is consistent with the definitions for the other "uses" in the WQS: "designated use" and "existing use". Also, the addition of "use" to the definition of "attainable" provides clarity since the term "attainable use" and the word "attainable" are referenced throughout 20.6.4 NMAC.

Since an attainable use is only the use that is achievable by the imposition of effluent limits, it may not be equivalent to the designated use, which is the goal for the waterbody. In addition to incorporating the word "use" into the definition, the Department is also proposing to add language clarifying that an "attainable use" may or may not be as stringent as the designated use.

This amendment does not establish regulations, implement new processes, or affect the implementation of the State's Water Quality Standards.

2. Limited Aquatic Life

The Department proposes to amend language in the definition for "limited aquatic life" (20.6.4.7(L) NMAC) to clarify that this designated aquatic life use is not limited in application only to ephemeral or intermittent waters.

As applied in the Standards, the definition for "limited aquatic life" is a subcategory of the designated aquatic life uses. The "limited aquatic life" definition states that the use supports aquatic life adapted to utilize a series of potentially limiting circumstances, including, but not limited to, hydrologically limiting environments such as those found in ephemeral and intermittent tributaries.

Some waters, based on one of the factors under 40 C.F.R. § 131.10(g) (**NMED Exhibit 22)**, may not be able to attain an aquatic life use with criteria more stringent than limited aquatic

life. This may or may not depend on the hydrologic regime. The language in this definition, as currently written, specifically identifies "ephemeral" or "intermittent" waters as limiting and is not entirely consistent with the language under 40 C.F.R. § 131.10(g) (**NMED Exhibit 22**), which describes, in part, "low-flow" conditions. The specification of "ephemeral or intermittent water" may preclude adequate protections for those perennial (i.e. – not ephemeral or intermittent) waters with limited aquatic life as the highest attainable use. The current language is also inconsistent with other classified waters with a designated limited aquatic life use, for example the perennial reaches of Sulphur Creek (20.6.4.124 NMAC).

Removing the reference to intermittent and ephemeral waters and replacing it with term "low-flow" prevents an implied exclusion of perennial waters. This amendment will aid in implementing and applying this aquatic life use, which is not based entirely on the hydrologic regime.

3. Marginal Coldwater

The Department proposes to amend language in the definition for "marginal coldwater" (20.6.4.7(M) NMAC) to clarify that this designated use is not limited to ephemeral or intermittent waters, and to include those conditions that distinguish it from a coldwater aquatic life use designation.

The reference to hydrologic condition in the definition for marginal coldwater originated when the language in the water quality standards migrated from "fishery" to "aquatic life" designated uses in 2005. What is currently referred to as "marginal coldwater" is equivalent to "marginal coldwater fishery." The definition of "marginal coldwater fishery", as found in 20.6.1.1007 NMAC prior to 2005, in part, "means a stream reach, lake or impoundment known to support a coldwater fish population during at least some portion of the year…". This designated

use was, in part, supportive of the National goal in Section 101(a)(2) of the CWA (33 U.S.C. § 1251) (**NMED Exhibit 10**) to provide for the protection and propagation of fish, shellfish and wildlife. However, the use did not protect aquatic organisms other than fish. The adoption of designated aquatic life uses in place of the designated fishery uses provided more comprehensive protection, as required by the federal CWA.

However, as part of the 2005 amendments, the reference to intermittent flow was added to reflect the "allowable seasonality" of these waters to support coldwater aquatic life. Many conditions may severely limit coldwater aquatic life populations other than intermittent or low flows. However, the definition as presently written could potentially be interpreted as applicable only to intermittent or low-flow waters when, in fact, it is the appropriate designated use for waters that can attain the numeric criteria, regardless of the hydrologic regime. Numerous classified perennial waters have a designated marginal coldwater aquatic life use, including those described in 20.6.4.103, 20.6.4.114, 20.6.4.216, 20.6.4.220, 20.6.4.307, 20.6.4.311, 20.6.4.401, 20.6.4.402, 20.6.4.408, 20.6.4.502, 20.6.4.601 and 20.6.4.805 NMAC. To clarify that marginal coldwater designated uses are not limited by hydrology, the Department is proposing to remove the phrase "intermittent or low flows, or other natural habitat".

The Department also proposes removing numeric temperature criteria from the definition. Definitions for aquatic life uses describe the use, not the criteria necessary to protect for that use. Each of the State's established designated uses have numeric criteria located under 20.6.4.900 NMAC in "Criteria Applicable to Existing, Designated or Attainable Uses…". The inclusion of all criteria in the definition for designated uses would render the definition lengthy and overly cumbersome for reference and implementation of water quality standards.

Since the Department is not proposing changes to the criteria for marginal coldwater, the implementation of the water quality criteria for this designated use would remain the same. Clarifying language in the definition aids in the implementation of the water quality standards by adding consistency between the definitions, and limiting numeric criteria to those identified in 20.6.4.900 NMAC.

B. Designated Uses and Numeric Criteria (20.6.4.900 NMAC)

1. Consistency for marginal warmwater designated use (20.6.4.900(H)(6) NMAC)

The Department proposes to clarify language for the designated marginal warmwater aquatic life use to be consistent with the definition in 20.6.4.7(M)(2) NMAC, which states that "…water temperature routinely exceeds 32.2°C (90°F)".

As currently written in 20.6.4.900(H)(6) NMAC, the numeric criteria for marginal warmwater aquatic life use are equivalent to the warmwater aquatic life use criteria. Although the definition for marginal warmwater provides a distinction from warmwater, it does not replace the established criteria in 20.6.4.900(H)(6) NMAC. The disparity between the marginal warmwater criteria and definition creates a conflict in applying the appropriate water quality standards for marginal warmwater aquatic life uses. The 2019 Comprehensive Assessment Listing Methodology ("CALM") for temperature illustrates this conflict where it states that although the definition states that historical water quality data may routinely exceed 32.2^{o} C, the associated temperature criterion in 20.6.4900(H)(6) NMAC is represented as a maximum temperature of 32.2^{o} C. As such, the CALM states that until such a time that the Department resolves the discrepancy between the temperature criterion and the definition, the listing methodology will assess against the numeric criteria in 20.6.4.900(H)(6) NMAC and identify any exceedances as

"Category 5B". Category 5B identifies that the available data indicate the designated use is not supported; however, a review of the water quality standard is required to verify the appropriate designated or existing use and/or criterion. Therefore, the Department cannot fully assess temperature in waters with a marginal warmwater aquatic life designated use.

Currently, there are 176 assessed waters with a designated marginal warmwater aquatic life use; 11 have been listed as impaired for temperature and classified as a Category 5B. But, based on the definition, these waters may not be impaired; however, until the temperature criterion is amended to be consistent with the attainable use described in the definition, these waters will remain listed.

Because the definition for "marginal warmwater" provides clarity in the differences between warmwater and marginal warmwater and describes the attainable temperature criterion for the aquatic life use, the Department is proposing to amend the temperature criterion for "marginal warmwater" in 20.6.4.900(H)(6) NMAC. Amending the numeric temperature criterion for the marginal warmwater aquatic life use to be consistent with the definition provides the ability to assess these waters appropriately.

Should this amendment be adopted, the Department will update water quality assessment procedures in the CALM, which undergoes a public comment period. The Department will then assess waters with a marginal warmwater temperature criterion against temperatures that may routinely exceed 32.2^{o} C.

This amendment would not change the designated use for any waters of the State but would provide for clearer implementation and assessment of the State's WQS for waters with a designated marginal warmwater aquatic life use.

2. Consistency with the term persistent toxic pollutants (20.6.4.900NMAC)

The current use of the terms "toxic pollutants", "persistent" and "persistent toxic pollutants" is varied throughout 20.6.4 NMAC, which could result in inconsistent application of water quality standards.

The term "toxic" (or some variation, including acute toxicity, chronic toxicity, persistent toxic pollutant) is found 31 times in 20.6.4 NMAC. The term "toxic pollutant" is found eight times in 20.6.4 NMAC, and the term "persistent" is found three times in 20.6.4 NMAC: once in 20.6.4.11(G) NMAC as "persistent toxic pollutant" (see above), once in 20.6.4.900(H)(7) NMAC as "persistent pollutant" and in 20.6.4.900(J)(2) NMAC as "persistent". In order to ensure the terminology is used consistently throughout 20.6.4 NMAC, the Department is proposing to clarify the language so that 20.6.4.900 NMAC properly references the defined terms.

This amendment will aid in the effectiveness as it pertains to implementing water quality standards. Failure to clarify the language could lead to inconsistent application of these standards, which could compromise protections of these waters as it applies to the federal CWA and state Water Quality Act.

IV. REGULATONS ASSOCIATED WITH AMENDING USES AND CRITERIA

A. Review of Standards; Need for Additional Studies (20.6.4.10 NMAC)

The Department proposes to amend the language pertaining to reviewing and amending standards (20.6.4.10 NMAC) to clarify when and how a designated use or criterion may be amended for a surface water of the State.

As established by the EPA, a water quality standard consists of designated uses for a water, the numeric criteria that protect those uses, and an antidegradation policy to ensure the protection

of those uses. The intent of 20.6.4.10 NMAC is to specify the regulatory process necessary for amending water quality standards.

In accordance with Section 303(c)(1) of the CWA (33 U.S.C. § 1313) (**NMED Exhibit 11**) and 40 C.F.R. § 131.20 (**NMED Exhibit 21**), state water quality standards are required to be reviewed at least once every three years.

Several mechanisms trigger an amendment of designated uses and the criteria that protect those uses. The first of these is when the existing use, as defined in 40 C.F.R. § 131.3 (**NMED Exhibit 26**) and 20.6.4.7(E)(3) NMAC, is determined to be more stringent than the designated use. In accordance with 40 C.F.R. § 131.10(i) (**NMED Exhibit 22)**, a state shall revise its standards to reflect the uses actually being attained where the existing water quality is greater than the designated use. Although this provision is currently included in 20.6.4.10(B) NMAC, it is embedded within other mechanisms for amending a standard and not explicitly referenced, which can cause inaccuracies with amending standards in accordance with federal regulations. The Department proposes to add a new subsection to clarify the required process for amending a designated use where the existing use is more stringent than the designated use. This additional detail will aid in the implementation and development of water quality standards in meeting the objective of Section 101(a)(2) of the CWA (33 U.S.C. § 1251) (**NMED Exhibit 10)**.

Contrary to those designated uses required to be amended to have criteria more stringent, there is also a mechanism to amend a designated use identified to be unattainable due to one of the six factors in 40 C.F.R. § 131.10(g) (**NMED Exhibit 22)**. This process requires the development of a UAA, the details of which are expanded on in greater detail in 20.6.4.15 NMAC. Since 20.6.4.10 NMAC is intended to describe the mechanisms for reviewing and amending standards, the Department proposes adding language to 20.6.4.10 NMAC referencing 20.6.4.15 NMAC. This

addition will ensure that 20.6.4.10 NMAC is inclusive of all conditions in which a review of standards would be appropriate and clarify regulatory mechanisms that drive water quality standard development and amendments.

The last type of amendment, the adoption of a site-specific criterion, is more specified and limited to particular conditions. A site-specific criterion is limited to instances in which the designated use is supported, but a particular criterion is unattainable due to localized conditions. In these cases, a site-specific criterion may be developed for particular waters with unique site conditions if it can be demonstrated the designated use is protected and the demonstration meets the conditions found in 20.6.4.10 NMAC.

Currently, the language in 20.6.4.10(B) NMAC uses terms such as "use designations" and "existing conditions" that are easily confused with other defined terms used throughout 20.6.4 NMAC; however, the Department proposes that 20.6.4.10(B) NMAC should read that, in some cases, the criterion for a particular designated use may not adequately reflect the local conditions or the aquatic communities adapted to localized conditions. In these cases, a water quality criterion for a designated use may be modified for a specific waterbody under a site-specific criteria demonstration so long as the designated use is still protected. Unlike with existing uses, where the designated use must be amended to be at least as stringent as the designated use, neither the CWA nor 20.6.4 NMAC require the adoption of site-specific criteria. The Department proposes to amend the language under this subsection to clarify this difference.

Amending the language and format in 20.6.4.10 NMAC to include subsections for each scenario in which a water quality standard may be amended provides clarity and transparency with respect to the regulatory process and therefore enhances water quality protection. These

amendments are intended to provide a defensible regulatory framework for water quality standard amendments.

As part of the proposed amendments to 20.6.4.10 NMAC (**NMED Exhibit 9**), including the addition of a new subsection, the Department proposes updates to references in 20.6.4.10 NMAC and 20.6.4.318 NMAC to reflect the renumbering of subsections.

B. Use Attainability Analysis (20.6.4.15 NMAC)

The Section pertaining to UAAs, 20.6.4.15 NMAC, focuses on a particular type of designated use amendment, as described in 20.6.4.10 NMAC, where a designated use is amended to one with less stringent criteria. In accordance with 40 C.F.R. § 131.10(g) (**NMED Exhibit 22)** a UAA is required for these types of designated use amendments. The process for a UAA to be consistent with the requirements of 40 C.F.R. § 131.10(g) (**NMED Exhibit 22)** is outlined in 20.6.4.15 NMAC.

In accordance with 40 C.F.R. § 131.10(g) (**NMED Exhibit 22)**, the UAA itself must demonstrate three critical components. First, the UAA must provide a demonstration that the designated use is not the existing use **(NMED Exhibit 62)** since the designated use may not have less stringent criteria than the existing use, as per 40 C.F.R. § 131.10(i) (**NMED Exhibit 22)**. Second, to remove the designated use, the UAA must demonstrate at least one of the six elements under 40 C.F.R. § 131.10(g) (**NMED Exhibit 22)** is applicable. Finally, the UAA must demonstrate the highest attainable use, which is the use that will be applied as the new designated use. Since 20.6.4 NMAC codifies designated uses, designated use amendments must undergo a rulemaking process before becoming effective for state purposes and and must be submitted to the Regional Administrator for EPA Region 6 in accordance with 40 C.F.R. § 131.20 (**NMED Exhibit 21)** for approval before becoming effective for purposes of the CWA.

Therefore, the Department is proposing to amend language in 20.6.4.15 NMAC to ensure that the state regulations, at a minimum, meet the federal regulations for amending a designated use to a use with less stringent criteria. The Department proposes several amendments that will ensure effective implementation of the UAA process.

The first proposed amendment organizes the subsections based on function. The Department proposes adding header language to each subsection based on the critical elements associated with conducting a UAA. These subsections include the three critical elements required under the federal regulations discussed above, the regulatory authority that provides for conducting a UAA, and the processes for amending a designated use both by the Department and by parties other than the Department.

Following a review of each subsection, the Department proposes amended language accordingly to reflect the application of that subsection.

The Department proposes an amendment to the first subsection, 20.6.4.15(A) NMAC, regarding the authority to remove a designated use, to include language consistent with 40 C.F.R. § 131.10(g) (**NMED Exhibit 22)** regarding antidegradation and the protection of existing uses. The Department also proposes language clarifying that, in accordance with 20.6.4.10 NMAC, those designated use amendments based on more stringent existing uses do not require a UAA. As part of the revisions to this subsection, language is proposed to clarify the federal CWA's regulatory references. This clarification is important as it relates to the implementation of WQS, since 20.6.4 NMAC cites various state and federal acts and regulations, and clarification of these citations will reduce confusion and provide consistency with referencing and implementation.

The second subsection, 20.6.4.15(B) NMAC, provides the regulatory requirements necessary to remove a designated use. For the most part, this subsection is comprehensive and

requires little change to reflect its intended function. The only substantial proposed amendment is to clarify that the Hydrology Protocol survey methodology, as described in the State's WQMP/CPP (**NMED Exhibit 63**), also provides for the determination of perennial waters.

The Department proposes a new subsection, 20.6.4.15(C) NMAC, "Determining the highest attainable use", which is not currently incorporated as part of the UAA process. This new language clarifies the requirements in 40 C.F.R. § 131.10(g) (**NMED Exhibit 22),** where a UAA must demonstrate that a designated use is not attainable, but is also required to demonstrate the highest attainable designated use. This additional subsection is critical towards ensuring the protection of surface waters and upholding the State's antidegradation policy. Based on the requirements of the antidegradation policy, the highest attainable use may not have criteria any less stringent than the existing use but based on the findings of the UAA, will not be as restrictive as the designated use determined unattainable. The Department proposes that the same methods used to demonstrate that a designated use is not attainable may also be utilized to demonstrate the highest attainable use.

The next subsection, 20.6.4.15(D) NMAC, describes the processes for amending a designated use based on a UAA. As currently worded in 20.6.4.15(C) NMAC, this subsection discusses an expedited UAA process but lacks any information regarding the standard (non-expedited) UAA process, which is generally pursued when amending a designated use with criteria less stringent than the current designated use. To remedy this, the Department is proposing to expand this subsection (renumbered as 20.6.4.15 (D) NMAC due to the insertion of proposed new 20.6.4.15(C) NMAC) to provide the necessary regulatory language for both the standard UAA process and the expedited UAA process.

The State's WQMP/CPP (**NMED Exhibit 64**) outlines the process to amend a designated use through a UAA; however, the WQMP/CPP is non-regulatory. To provide consistency and transparency in the regulatory process of amending a designated use with less stringent criteria, the Department proposes new language to be codified as 20.6.4(D)(1) NMAC.

There are particular provisions for specific designated use amendments where the aquatic and recreational designated uses are not attainable specifically due to ephemeral conditions. The lack of physical water in a waterbody preventing attainment of the designated use is a factor provided for in 40 C.F.R. § 131.10(g)(3) (**NMED Exhibit 22)**. These specific types of UAAs, may, if demonstrated through the State's approved Hydrology Protocol survey method, be amended through an "expedited" process. This process provides for the implementation of the designated uses, pending EPA's approval, to expeditiously apply the appropriate water quality criteria while avoiding inappropriate water quality impairment listings and imposing undue burdens on dischargers.

The language regarding this expedited process, as currently written, does not explicitly prohibit or limit the expedited process from being undertaken by parties other than the Department; however, in implementation, there is no mechanism for a third party to meet the requirements of this section. The proposed revised language clarifies that an expedited UAA process is limited to actions brought forward by the Department. The Commission identified, in its 2010 statement of reason regarding this section, that amending designated uses for ephemeral waters determined through the expedited Hydrology Protocol UAA process only applied to those conducted by the Department **(NMED Exhibit 65)**. To clarify that the expedited UAA process is limited to those matters being brought forth by the Department, new language is proposed for this paragraph.

The Department proposes amendments to the last subsection, 20.6.4.15(E) NMAC, to clarify the elements required for a party other than the Department to amend a designated use to a designated use with less stringent criteria.

A review of the language dating back to 1995, which required publication of a notice of intent to conduct a UAA, helps explain the subsection as currently written. The Commission later redacted the requirement to publish the notice of intent to conduct a UAA as it was burdensome to third parties (most of whom are permittees with limited resources). Instead of publishing the notice of intent to conduct a UAA, the language in 20.6.4.15 NMAC was revised to require the notice of intent be provided to NMED. This change then led to two separate notifications (the Notice of Intent and the work plan) to NMED with no known benefit to either party (NMED or the third-party). Since there is no administrative reasoning to have these requirements submitted independently, language was then amended to streamline the process such that the Notice of Intent and work plan be submitted to NMED concurrently to reduce temporal and financial resources for both NMED and the third party. The current language only requires the submittal of a work plan to NMED and EPA, with NMED's approval of the work plan, before conducting a UAA. However, the Department and third parties' experience is that the lack of descriptive work plan elements in 20.6.4.15(D) NMAC has caused confusion, inconsistency, and shortfalls as the UAA proceeds through the development and rulemaking process; proving to be resource-intensive for all involved. As a result, the Department is proposing several amendments for this subsection.

First, the Department proposes reformatting the subsection for ease in referencing and to be consistent with the other sections of 20.6.4 NMAC.

Second, the Department is proposing to amend language regarding the work plan elements and the responsibilities of public notice. In part, the Department proposes adding language to

include specific elements required in the work plan and clarification of the work plan process. The Department is also proposing language to clarify the public notice requirements for designated use amendments being brought forth by a party other than the Department.

There are several interconnected reasons for clarifying the public notice requirements in this subsection. First, the Department does not have the resources or the authority to be the gatekeeper for public notice of a rulemaking based on a UAA for which the Department is not the petitioner. The Department generally testifies independently of the petitioners in matters such as these, so to prevent a conflict of interest or confusion of responsibility, all public notice activities should be undertaken by the petitioner. The Department finds that since the work plan provides the petitioner's intended undertakings for a designated use amendment, a plan to meet the public notice requirements should be outlined in the work plan, which the Department approves. Standardizing requirements for elements in the work plan and the administrative process will reduce inconsistencies and resources expended by both NMED and third parties in the UAA process.

As part of the elements necessary for a functional work plan, clarification on the authority, application, and process for removing and applying the highest attainable use will help implement the WQS and proceed with defensible designated use amendments. In addition, this revised language provides consistency with the State's approved WQMP/CPP (**NMED Exhibit 64**) and clarity regarding the processes for a UAA conducted by an entity other than the Department. As a result of this proposed amendment, the Department also proposes updates to the reference in 20.6.4.97 NMAC (**NMED Exhibit 9**).

V. HARDNESS-BASED CRITERIA

In accordance with 40 C.F.R. § 131.11(**NMED Exhibit 25)**, states must adopt those water quality criteria that protect the designated uses. As part of this requirement, numeric criteria must either be the recommended criteria developed by EPA in accordance with Section 304(a) of the CWA (33 U.S.C. § 1314); numeric criteria based on a modified version of EPA's recommended criteria which reflects site-specific conditions; or numeric criteria based off of other scientifically defensible methods. As part of the Triennial Review, and pursuant to 40 C.F.R. § 131.20 (**NMED Exhibit 21)**, if a State does not adopt new or revised criteria for those constituents which EPA has published new or updated criteria recommendations in accordance with Section 304(a) of the CWA (33 U.S.C. § 1314) (**NMED Exhibit 40**), then the State shall provide an explanation when it submits the results of its Triennial Review to the Regional Administrator.

The State's water quality standards have a list of use-specific numeric criteria, including standard numeric criteria, ammonia criteria, and hardness-based metals criteria. As part of this Triennial Review, hardness-based criteria were evaluated based on EPA's most recent criteria recommendations. The Department determined four hardness-based aquatic life use criteria were not in alignment with the EPA's criteria recommendations. Although the State has criteria for aluminum, cadmium, selenium and zinc, they are not consistent with EPA's recommendations, as such the Department has investigated the reasons for this and is proposing updates to these criteria accordingly.

As it pertains to the acute and chronic hardness-based aquatic life criteria for aluminum and selenium, the State is not proposing to update the Standards at this time. The reasoning for this determination is presented in the direct written testimony of Kris Barrios (**NMED Exhibit 2**). In addition, the direct testimony of Diana Aranda (**NMED Exhibit 3**) discusses typographical

amendments to the hardness-based metals table in 20.6.4.900(I)(3) NMAC identified as having previously been miscalculated.

My testimony will describe the evaluation conducted by the Department for hardness-based cadmium and zinc criteria. In addition, and as part of the overall amendments proposed for Section 20.6.4.900(I) NMAC, my testimony will also provide the reasons for language amendments pertaining to EPA's prior disapproval of acute and chronic hardness-based aquatic life criteria for aluminum.

A. Aluminum

The Department is proposing to add language regarding the pH range for hardness-based criteria for total recoverable aluminum and to remove language regarding EPA's disapproval of hardness-based aluminum criteria (20.6.4.900(I)(1) and 20.6.4.900(I)(2) NMAC). The amendments being proposed as part of this Triennial Review regarding EPA's disapproval require some historical perspective.

In 1988, the EPA's National Recommended Aquatic Life Criteria for chronic and acute aluminum were 87 µg/L and 750 µg/L, respectively. The guidance that establishes these criteria addresses the applicability of these criteria for waters in which the pH ranges from 6.5 to 9.0 standard units ("SU") because the water quality criterion for pH states that a pH range of 6.5 to 9.0 SU appears to adequately protect freshwater fishes and bottom-dwelling invertebrate fish food organisms from the effects of the hydrogen ion (**NMED Exhibit 66**).

On or before 1991, the State adopted EPA's recommended chronic and acute aquatic life numeric criteria for aluminum. And although EPA's recommended criteria provided a pH range to which the criteria were applicable, the State's table for numeric criteria has never clarified or prescribed a pH range for numeric aluminum criteria. Therefore, as adopted by the State and

approved by EPA, the criteria applied to all waters with aquatic life uses, regardless of pH range. These adopted acute and chronic aluminum aquatic life criteria remained applicable until 2010 when the State adopted acute and chronic aquatic life hardness-based criteria for total recoverable aluminum. Similar to the numeric acute and chronic aluminum criteria, the language for the acute and chronic hardness-based aluminum criteria did not specify any pH range as it read "[f]or aluminum, the criteria are based on analysis of total recoverable aluminum in a sample that is filtered to minimize mineral phases as specified by the department."

As part of the State's adoption of the hardness-based total recoverable aluminum aquatic life criteria in 2010, it removed, in its entirety, the dissolved acute and chronic aquatic life aluminum numeric criteria. In the Commission's 2010 statement of reasons (**NMED Exhibit 67**), (emphasis added):

> The Commission adopts the proposal by Chevron Mining and Los Alamos National Laboratory/Department of Energy (CMI and LANS/DOE) <u>to replace the current acute and chronic aquatic life criteria for aluminum</u> in section 900.J with hardness-based criteria and to show total aluminum in this subsection to reflect findings of new toxicological studies...

These hardness-based aluminum criteria were developed independently from EPA's national guidance and, although the Commission adopted the criteria with no specified pH range, EPA only approved the hardness-based aluminum criteria for a pH range from 6.5 to 9.0 SU (**NMED Exhibit 68**). The pH limitation placed on the hardness-based criteria from EPA, along with the removal of the numeric criteria, left no protection for aquatic life against toxic levels of aluminum for waters with a pH less than 6.5 SU and greater than 9.0 SU.

As described in the statement of reasons for the 2015 Triennial Review [WQCC 14-05 (R)] (**NMED Exhibit 69**), EPA disapproved the State's hardness-based standards for waters with a pH less than 6.5 SU. The partial disapproval of the criteria left waters with a pH less than 6.5 SU without applicable chronic or acute aluminum criteria without justifiable reasoning. In that proceeding, the Commission added language to 20.6.4.900(I)(1) and 20.6.4.900(I)(2) NMAC clarifying EPA's conditional approval/disapproval of the hardness-based aluminum criteria. However, the numeric chronic and acute aluminum criteria were not incorporated back into the State's standards, perpetuating the disjunct between EPA's application of aluminum criteria for waters with a pH less than 6.5 and the State's.

Although EPA's responses and concerns focus on waters with a pH less than 6.5 SU, waters with a pH greater than 9.0 are equally left without aquatic life aluminum criteria. In order to remediate the lack of applicable criteria for aluminum in waters with a pH less than 6.5 and above 9.0, the Department is proposing to incorporate the previously approved chronic and acute dissolved aluminum criteria of 87 µg/L and 750 µg/L, respectively. These would be the applicable aluminum criteria for waters with a pH outside the applicable range for hardness-based aluminum criteria, for purposes of the federal CWA.

The Department is proposing several amendments to 20.6.4.900 NMAC. First, the range to which the hardness-based aluminum criteria are applicable should be added to 20.6.4.900(I)(1) and 20.6.4.900(I)(2) NMAC. Second, the addition of numeric chronic and acute criteria should be incorporated back into the numeric criteria table in 20.6.4.900(J)(1) NMAC. Third, a footnote in 20.6.4(J)(2)(i) NMAC should be added to explain that the numeric acute and chronic dissolved aluminum criteria are only applicable for waters with a pH outside the 6.5 to 9.0 SU range. All waters within the range are subject to the hardness-based acute and chronic aluminum criteria.

Finally, with the adoption of these amendments, EPA's partial disapproval of the hardness-based aluminum criteria as stated in 20.6.4.900(I)(1) and 20.6.4.900(I)(2) NMAC will no longer be necessary and should be removed, assuming approval of the other amendments. The amendments, as proposed, have been provided as **NMED Exhibit 9**.

As it pertains to the implementation of water quality standards, clarifying the pH range to which hardness-based aluminum criteria apply, in conjunction with re-establishing aluminum criteria for those conditions outside the hardness-based criteria, provides a comprehensive application of aquatic life criteria for aluminum. Without this clarification, there are inconsistent and potentially reduced protections for aquatic life.

B. Cadmium

In 2010, the State adopted acute and chronic hardness-based aquatic life use criteria for cadmium based on testimony presented by Los Alamos National Security, LLC, the operational contractor for LANL at the time (Triad National Security, LLC is the successor-in-interest). In particular, testimony presented by Dr. Robert Gensemer illustrated the reasoning for amending cadmium criteria based on the most relevant and current science regarding bioavailability and the toxicological effects of cadmium on aquatic organisms. The evidence was further supported by similar standards adopted by the State of Colorado, which had been approved for CWA purposes by EPA Region 8. The EPA had last updated the acute and chronic toxicity criteria in 2001, and new toxicological studies demonstrated more relevancy. The Commission adopted revised acute and chronic hardness-based aquatic life criteria for cadmium based on this new information. However, in 2016, EPA revised its recommendations for acute and chronic hardness-based aquatic life criteria (**NMED Exhibit 70**). As part of this Triennial Review, to comply with 40 C.F.R. §

131.11 (**NMED Exhibit 25)**, the Department reviewed the hardness-based criteria for cadmium for consideration in State standards.

EPA's peer-reviewed recommended criteria, "Aquatic Life Ambient Water Quality Criteria Cadmium 2016" (**NMED Exhibit 70**), identifies cadmium as a relatively rare metal, naturally occurring in low concentrations in the environment. Anthropogenic sources of cadmium are predominately from the manufacturing of batteries (currently representing 80% of the global cadmium consumption), pigments, plastic stabilizers, metal coatings, alloys, electronics, and nanoparticles used in semiconductors in photovoltaic devices. These anthropogenic sources are the primary source of toxic levels of bioavailable cadmium in the environment.

Cadmium is not an essential metal for aquatic organisms and, although it can bioaccumulate in tissues of animals, it is more likely to cause toxicological effects from direct exposure on an acute and chronic level than through bioaccumulation. Chronic exposure of cadmium to aquatic life, predominately through its competition with calcium, may adversely affect the function of internal organs in fish, such as the gill, liver, and kidneys. Cadmium can also impact growth and cause overall deleterious effects on reproduction, immune function, endocrine function, and aquatic life behavior. EPA's guidance states that, based on findings from recent studies, cadmium interferes with the potassium chloride channels impeding enzymatic function, and this may be the primary cause for adverse physiological effects.

EPA based its development of acute and chronic aquatic life criteria on the culmination of numerous scientifically defensible toxicological studies on cadmium. For acute freshwater cadmium toxicity, EPA evaluated studies that represented 101 species within 75 genera. Similarly, for chronic freshwater cadmium toxicity, EPA evaluated studies that represented 27 species within 20 genera. In both the acute and chronic studies, the available number of species posed a wide

range of taxa, some of which were not previously included in evaluations. The diversity of taxa in the acute toxicological studies, evaluated as part of this criteria development process, included 66 invertebrate species, 33 fish species, one salamander species, and one frog species.

EPA's evaluation also considered the parameters affecting cadmium toxicity, including hardness, pH, alkalinity, and organic carbon. Although EPA recognized that dissolved organic carbon ("DOC") is influential on the overall bioavailability of cadmium to aquatic life, the use of the biotic ligand model ("BLM") was believed to be insufficient to demonstrate toxicity since the literature suggests the pathway for toxicity is not across the gill (where the BLM focuses uptake) so much as it may be across the potassium chloride channels within the cells causing enzymatic disfunction. Therefore, EPA determined a BLM would not be an appropriate model for determining acute and chronic toxicity for cadmium at this time.

The Department then compared the acute and chronic hardness-based cadmium criteria currently found in 20.6.4.900(I) NMAC with EPA's recommended criteria (**NMED Exhibit 71**). The findings show that the EPA-recommended criteria are slightly more stringent, indicating more bioavailability for uptake by aquatic organisms than the current criteria.

The Department finds that EPA's revised criteria demonstrate the most available up-to-date toxicological demonstration for protection of aquatic life as it pertains to cadmium. The Department does not have additional resources or reasons to develop independent criteria based on scientific principles or site-specific criteria that would be different from those proposed by EPA. Therefore, in accordance with 40 C.F.R. § 131.11 (**NMED Exhibit 25**), the Department proposes the adoption of acute and chronic hardness-based cadmium criteria based on EPA's current recommended criteria.

C. Zinc

Based on comments received during the Public Comment Period on NMED's Proposed Amendments, the Department recognized the acute and chronic hardness-based aquatic life criteria for zinc were not reflective of EPA's recommended criteria. In response to this disparity, and in accordance with 40 C.F.R. § 131.11 (**NMED Exhibit 25**) as part of this Triennial Review, the Department reviewed the acute and chronic hardness-based aquatic life use criteria for zinc compared to EPA's recommended criteria. This review demonstrated that EPA's criteria posed more protective numeric limits (**NMED Exhibit 71**).

To provide an updated series of standards for interested parties to review, the Department subsequently proposed amendments based on the more stringent 1996 EPA-recommended criteria for the acute and chronic hardness-based aquatic life criteria for zinc.

The Department has since determined the history associated with the current acute and chronic hardness-based aquatic life criteria for zinc is similar to cadmium in the fact that the Commission adopted alternative criteria from EPA's recommended criteria based recent toxicological studies and Colorado's approved alternative acute and chronic hardness-based aquatic life criteria for zinc.

The Department's findings show that EPA's recommended aquatic life use criteria are more stringent when compared to the criteria adopted in 2010. However, the reasoning for the criteria adopted by the Commission and approved by EPA Region 6 still holds. The Department therefore proposes to retain the current acute and chronic hardness-based numeric aquatic life use criteria for zinc as reflected in the proposed language for 20.6.4.900(I) NMAC (**NMED Exhibit 9**).

VI. EXISTING USE ANALYSIS ("EUA") LANL WATERS

A. EUA Background

Water quality standards contain three key elements that are intricate to their regulatory function. These include establishing designated uses, criteria to protect for those uses, and an antidegradation policy. These requirements uphold the objective of Section 101 of the CWA (33 U.S.C. § 1251) (**NMED Exhibit 10**) to "restore and maintain the chemical, physical and biological integrity of the Nation's waters."

According to 40 C.F.R. § 131.3(e) (**NMED Exhibit 26**) and 20.6.4.7(E)(3) NMAC, existing uses "are those uses actually attained in the water body on or after November 28, 1975, whether or not they are included in the water quality standards." In accordance with 40 C.F.R. § 131.10(i) (**NMED Exhibit 22**) a state shall revise its standards to reflect the uses actually being attained if the designated use is less stringent than those attained and, in accordance with 40 C.F.R. § 131.20, if new information indicates that a use is attainable, such as the information used to determine an existing use, the state shall revise its standards accordingly. In short, a designated use may not be less stringent than an existing use (**NMED Exhibit 21**).

As part of the efforts made to determine the appropriate designated uses for classified ephemeral and intermittent waters within lands managed by the U.S. Department of Energy ("DOE") within LANL as identified in 20.6.4.128 NMAC, Amigos Bravos, DOE, Los Alamos National Security LLC, and NMED entered into a Joint Stipulation Regarding Proposed Changes to 20.6.4.128 NMAC ("Joint Stipulation") (**NMED Exhibit 72**) on October 9, 2015. The Joint Stipulation required the parties meet, share available data, and confer regarding the appropriate level of water quality protections for ephemeral and intermittent waters classified under 20.6.4.128 NMAC. The findings to which all parties could concur were to be petitioned for by the Department

on or before the next Triennial Review (i.e. – this proceeding). A lack of concurrence on these tributaries would not preclude any of the parties from filing independent petitions for amending the designated uses, as found appropriate, given the demonstration was prepared in accordance with 20.6.4.15 NMAC and filed with the Commission in accordance with 20.1.6 NMAC.

Following several years of discussions and data gathering, the three parties reached concurrence in December 2020 for increased aquatic life protections based predominately on hydrology. Since the analysis focused on determining if the waters have attained a use with more stringent criteria than the current designated use, a UAA is not required. Although a UAA is not required under federal regulations, other factors must be considered for any designated use amendment. For purposes of demonstration, the Department prepared an analysis referred to herein as an "Existing Use Analysis" or "EUA" (**NMED Exhibit 73**). The Department limited the scope of this EUA to evaluating only those tributaries on which the three parties were in concurrence. These tributaries include Effluent Canyon in its entirety from Mortandad Canyon to its origin in TA-16, Two-Mile Canyon from its confluence with Pajarito Canyon to its confluence with Upper Two-Mile Canyon, and S-Site Canyon from alluvial monitoring well MSC 16-06293 to Martin Spring. Based on the Joint Stipulation, this EUA did not evaluate the appropriate designated uses for the other ephemeral and intermittent waters within LANL, which warrants evaluation independently of this analysis.

B. History of LANL Waters

Dating back to at least 1991, perennial reaches of tributaries to the Rio Grande in Santa Fe County, unless included in other segments, were designated with a high quality coldwater aquatic life use as currently identified in 20.6.4.121 NMAC. The most recent amendment to 20.6.4.121

NMAC was adopted in 2013, but retained the description of water bodies and their designated aquatic life use.

Although 20.6.4.121 NMAC notes the segment was divided to include an additional section under 20.6.4.128 NMAC, the tributaries identified in 20.6.4.128 NMAC specifically describe the ephemeral and intermittent portions, not perennial reaches of tributaries to the Rio Grande, as found under 20.6.4.121 NMAC.

In the same Triennial Review that 20.6.4.128 NMAC was adopted, the Commission adopted language for 20.6.4.98 NMAC. The language that was initially proposed for 20.6.4.98 NMAC included both ephemeral and intermittent waters with designated uses for wildlife habitat, livestock watering, limited aquatic life and secondary contact, just as 20.6.4.128 NMAC was originally proposed. However, the approved version of 20.6.4.98 NMAC reflected the Commission's determination that intermittent waters were able to attain and support a more stringent designated aquatic life use than ephemeral waters, which applies, "…chronic criteria to intermittent waters because of the potential long-term exposure of aquatic life to pollutants." Based on the evidence presented, the Commission created not one but two sections in 20.6.4 NMAC for unclassified non-perennial waters, adopting separate designated uses for ephemeral and intermittent waters, 20.6.4.97 NMAC and 20.6.4.98 NMAC, respectively. The designated uses for unclassified ephemeral waters specifically listed under 20.6.4.97 NMAC were those uses initially proposed (limited aquatic life and secondary contact). However, based on testimony and evidence presented at the hearing, unclassified intermittent waters under 20.6.4.98 NMAC were determined to have a more stringent attainable aquatic life use as supported under Section 101(a)(2) of the CWA (33 U.S.C. § 1251) (**NMED Exhibit 10**).

As discussed in further detail in the EUA (**NMED Exhibit 73**), the Statement of Reasons for Amendment of Standards [WQCC 03-05(R)] (**NMED Exhibit 74**) provided the Commission's reasoning for the determination to adopt unique, aquatic life and recreational uses for unclassified ephemeral and intermittent waters as 20.6.4.97 NMAC and 20.6.4.98 NMAC, respectively. These designations applied to the unclassified ephemeral and intermittent waters on the Pajarito Plateau, outside of LANL, including tributaries upstream and downstream of those within LANL.

However, it appears that although the reasoning and evidence was the same for non-perennial waters within LANL as well as outside LANL, this determination was not carried over and reflected in the adoption of 20.6.4.128 NMAC.

The changes described above became effective under state law on May 23, 2005. However, when the proposed amendments were submitted to EPA for review and approval in accordance with 40 C.F.R. § 131.20 (**NMED Exhibit 21**), EPA determined these amendments did not meet the requirements of the CWA without the accompaniment of a UAA. The Department responded to this determination by submitting a UAA using the study by the U.S. Fish and Wildlife Service (Lusk and MacRae, 2002), as filed in the preceding Triennial Review, as supporting evidence for the designated uses. Although the UAA asserts the highest attainable life use for non-perennial waters is limited aquatic life, the U.S. Fish and Wildlife Service study which "investigated the biological, chemical, and physical characteristics of four intermittent streams on the Los Alamos National Laboratory in New Mexico…to identify suitable living space for fish and benthic macroinvertebrates" found that "aquatic life is an existing use of these intermittent streams that should be protected." Despite this apparent discrepancy between the UAA and the supporting evidence, EPA approved the designated limited aquatic life use for ephemeral and intermittent waters within LANL as classified under 20.6.4.128 NMAC on September 12, 2007.

In September 2014, the non-profit organization Amigos Bravos filed proposed amendments and a statement of basis to change the designated use for all waters in 20.6.4.128 NMAC from Limited Aquatic Life to Marginal Warmwater Aquatic Life, arguing these waters were under-protected. The basis for the proposed amendment, as presented by Amigos Bravos, was to maintain consistency in the application of aquatic life uses for intermittent waters, and only until such a time there is a demonstration (i.e., a UAA) that the waterbody is unable to attain that designated use should it be amended to a limited aquatic life designated use. Additionally, Amigos Bravos found fault in the approval of the 2005 rule change creating 20.6.4.128 NMAC since, as argued, the designated aquatic life use was made less stringent without supporting evidence for the change through the required UAA process. Amigos Bravos considered the submittal of a UAA post-facto (i.e., after the 2005 Triennial Review) for EPA approval to be a "…textbook example of arbitrary and capricious action." However, no party appealed the Commission's adoption of 20.6.4.97, 20.6.4.98, or 20.6.4.128 NMAC.

Amigos Bravos withdrew the proposed amendments to 20.6.4.128 NMAC in that proceeding and, in exchange, entered into the previously mentioned Joint Stipulation (**NMED Exhibit 72**). The parties recognized, as provided in the agreement itself, that additional data were needed to determine the appropriate water quality protections. In part, this required documenting the hydrologic regime of the tributaries.

C. Waters Being Evaluated

For purposes of this analysis, in fulfillment of the Joint Stipulation, the parties concurred on portions of three tributaries within LANL. These waterbodies include Effluent Canyon from its confluence with Mortandad Canyon to its headwaters; the upper portion of S-site Canyon from alluvial monitoring well MSC 16-06293 to Martin Spring; and Two-Mile Canyon from Pajarito

Canyon upstream to its confluence with upper Two-Mile Canyon. All three of these tributaries are currently classified waters in 20.6.4.128 NMAC.

The Department did not evaluate designated uses for livestock watering and wildlife habitat under this analysis since they are designated uses for all waters of the state and have been established as attainable. Therefore, only aquatic life and recreational uses were evaluated under this analysis.

The current designated aquatic life use for Effluent Canyon, S-Site Canyon, and Two-Mile Canyon is limited aquatic life, which has acute numeric criteria but does not have chronic numeric criteria or numeric criteria for pH, dissolved oxygen, temperature, or ammonia.

The current designated recreational use for Effluent Canyon, S-Site Canyon, and Two-Mile Canyon is secondary contact which has an *Escherichia coli* ("*E. coli*") bacteria monthly geometric mean of 548 colony forming untis ("cfu") per 100 milliliters ("mL") or most probable number ("MPN") per 100 mL and a single sample of 2,507 cfu per 100 mL or MPN per 100 mL.

D. General Site Characteristics

Effluent Canyon, S-Site Canyon, and Two-Mile Canyon are all located within lands managed by DOE within LANL, on the Pajarito Plateau within Los Alamos county. The geologic formation of the Plateau occurred through several volcanic flow events (**NMED Exhibit 75**), with wind and water and erosion causing steep canyons on the eastern side of the Plateau as it terminates with the Rio Grande. The area has an elevational range from approximately 7,300 feet with ecoregional characteristics associated with volcanic subalpine and mid-elevation forests on the upper reaches of the plateau to approximately 5,400 feet at the shrubland foothilled valley floor where these tributaries converge with the Rio Grande. Anthropogenic activities in and around the

Pajarito Plateau are predominately associated with activities from the Laboratory as well as the town of Los Alamos and White Rock.

As provided in the EUA (**NMED Exhibit 73**), the tributaries evaluated as part of this analysis, Effluent Canyon in its entirety, is approximately 0.50 miles long from its confluence with Mortandad Canyon to its origin in TA-16; S-Site Canyon is approximately two miles long from its confluence with Water Canyon to its origin at Martin Spring in TA-33; and Two-Mile Canyon is approximately 4.5 miles from its confluence with Pajarito Canyon to its origin west of State Road 501 and LANL. Due to the limited scope of the EUA, the Department did not evaluate existing uses for the full extent of S-Site Canyon or Two-Mile Canyon. Although these tributaries are all within LANL, to which there are associated stormwater and treated effluent permits administered by EPA under the National Pollutant Discharge Elimination System ("NPDES") program, none of the tributaries under this EUA traverse through the town of Los Alamos or through White Rock. The Department identified several points of diversion identified through the New Mexico Office of the State Engineer's Water Rights Reporting System; however, all but two were groundwater diversions and the two identified as surface water diversions are "inactive", and likely not impacting the determination of the existing uses.

These aspects were evaluated as part of the EUA to ensure that natural or anthropogenic conditions would not prevent the Department's ability to determine the existing uses. For this EUA, there were no underlying conditions that prevented proceeding with the determination.

E. Antidegradation

As part of the State's antidegradation policy, any analysis evaluating designated uses must ensure the protection of waters is maintained. This evaluation includes determining the tier of protection afforded to the water under the State's antidegradation policy (20.6.4.8 NMAC),

including enhanced protections for waters designated as Outstanding National Resource Waters ("ONRWs"); waters with exceptional water quality; or for all other surface waters of the State. The antidegradation policy also requires an evaluation of downstream waters to ensure their sustained protections, should a designated use amendment be supported.

The findings of this EUA demonstrated that none of the waters in this analysis were designated ONRWs and, should these waters be determined to have more stringent criteria for existing uses than the designated uses, the protection of downstream waters would also be protected.

F. Threatened and Endangered Species

In accordance with Section 7(a)(2) of the Endangered Species Act ("ESA"), EPA is required to consult with the U.S. Fish and Wildlife Service to ensure that any action authorized by the EPA is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of habitat of such species. This requirement is federal agency to federal agency; however, to assist EPA with evaluation, the EUA included a preliminary screening of listed threatened and endangered species within the larger LANL area. The Department conducted this screening using the U.S. Fish and Wildlife Service's Information for Planning and Consultation (IPaC) project planning tool, the same tool used by EPA.

The Department did find several federally listed threatened and endangered species as well as areas delineated as critical habitat. These findings are detailed in the EUA (**NMED Exhibit 73**). The Department therefore concludes that the proposed increased protections in WQS would not negatively affect or degrade habitat because, if anything, they provide enhanced protections to the waterbodies and their dependent species.

G. Existing Use Analysis

1. Existing Recreational Use

Recreational uses within 20.6.4 NMAC have criteria for maximum and geometric mean of *E. coli* bacteria, expressed in colony forming units ("cfu") or most probable number ("MPN"). *E. coli* is used as an indicator bacterium for measuring levels of fecal contamination which has been demonstrated to pose serious health risks to humans, if ingested. The Department conducted an existing use analysis on Effluent Canyon, S-Site Canyon and Two-Mile Canyon, for recreational uses through an evaluation of available *E. coli* data. For purposes of this analysis, the Department used readily available *E. coli* data from SWQB, the Department's DOE Oversight Bureau ("DOE-OB") and LANL, through the Surface Water Quality Information Database ("SQUID") (from 2009 to 2020) and Intellus database (from 1975 to 2020). However, no *E. coli* data were found for purposes of this analysis for Effluent Canyon, S-Site Canyon, and Two-Mile Canyon. Therefore, the existing recreational use, based on *E. coli*, was found to be indeterminate at this time based on insufficient evidence and no further analysis of recreational use was conducted. Until further data are available, the existing recreational use is assumed to be at least secondary contact.

2. Existing Aquatic Life Use.

Aquatic life uses were analyzed using multiple parameters, all of which have criteria associated with aquatic life protection. Water quality data used to determine the existing aquatic life uses were collected under ambient, baseflow conditions and not the result of a direct and immediate result of precipitation. The Department focused on hydrologic condition, since the presence of intermittent conditions or persistent flow (spanning more than 96 hours) would demonstrate that aquatic life have potential exposure to chronic conditions, warranting, at a minimum, a designated use with chronic criteria. In addition to the hydrologic condition, the

Department also evaluated maximum temperature, pH, dissolved oxygen, and benthic macroinvertebrates, all of which are valuable in providing a more thorough analysis of the existing aquatic life uses for these waters. Other numeric aquatic life criteria, as listed in 20.6.4.900(I) NMAC through 20.6.4.900(M) NMAC, were assumed to be attainable to establish the existing use unless determined that natural sources are preventing attainment, which is beyond the scope and resources for this analysis.

Each of the waters were evaluated independently, given the unique hydrogeological characteristics of the canyons within the Pajarito Plateau. The hydrologic condition was determined as a foundation to establish the minimal level of protection for aquatic life. The presence of water (spanning more than 96 hours) is a limiting factor for aquatic life. Should there be evidence of intermittent conditions or persistent flow, the existing use, at a minimum, is marginal warmwater aquatic life (i.e., the minimum level of protection against chronic exposure to pollutants.

a. Hydrologic Condition

The hydrologic condition was determined using two lines of evidence. The first was through the use of the Department's Hydrology Protocol survey method (**NMED Exhibit 63**) while the second was supporting evidence through stream gage and alluvial well data. The Hydrology Protocol (**NMED Exhibit 63)** is the only hydrologic condition survey method for surface waters approved, as a component of the State's WQMP/CPP, by both the Commission and the EPA. The Hydrology Protocol is a tool used to determine the hydrologic regime (ephemeral, intermittent, or perennial) of surface waters in New Mexico based on hydrological, geomorphic, and biological indicators related to the persistence of water. Information gathered and evaluated as part of a Hydrology Protocol survey include drought conditions, recent weather conditions,

modifications to the tributary (diversions and impoundments), discharges to the tributary, and photodocumentation. During the survey, the assessor evaluates different indicators related to the persistence of water. such as the presence of water, fish, benthic macroinvertebrates, filamentous algae or periphyton, hydric soils, seeps or springs, and iron oxidizing bacteria or fungi.

The assessor also evaluates the vegetation profile, channel morphology (e.g., sinuosity, flood-prone area), in-channel structure (riffle-pool sequences), and general stream substrate composition. The survey takes observational data and transcribes it to a numeric value. The sum of the numeric values asserts the hydrologic regime of the surveyed stream reach. A score of less than nine is considered ephemeral, a score of at least nine but less than or equal to 19 is considered intermittent, and a score greater than 19 is considered perennial. There are transitional ranges in the Hydrology Protocol to which scores near the thresholds may need more information to provide more confidence in the determination; however, they are assumed to attain a more persistent hydrologic condition until demonstrated otherwise.

The Hydrology Protocol survey method is defensible due to several built-in quality control mechanisms including having multiple cross-disciplined personnel trained on the Hydrology Protocol for each survey to reduce bias; a multi-parameter weighted survey which is designed to ensure no particular observation skews the overall determination, thus reducing bias; prescribed observational periods to ensure indicators are not overlooked, as well as increasing accuracy and repeatability; and the standardization of the process which increases accuracy and reduces bias. When implemented correctly, the Hydrology Protocol survey can be highly accurate and reliable for determining the long-term hydrologic conditions of surface waters.

In addition to the Hydrology Protocol survey method's built-in quality control mechanisms, the Department incorporated additional steps as part of the routine process for those surveys

conducted as part of this analysis. Representatives from LANL and the Department conducted the Level 1 surveys in their entirety. That is, all 14 indicators were evaluated even though the Hydrology Protocol provides "off ramps" for partial scores that clearly indicate ephemeral (or perennial) systems. Prior to recording on field sheets, all field observations were discussed and referenced until the survey team reached consensus. For a majority of the surveys, the survey team included two independent recorders to cross-reference data for accuracy. Call-and-response methodologies between the observers and recorders were also used to ensure data were recorded accurately and post-survey review of data was conducted as a verification method. Staff validated data entry into a Microsoft Excel spreadsheet and any data with apparent transcription errors or missing information were discussed with the survey team before making assumptions or changes on field sheets or in the data entry spreadsheets.

b. Water Quality

The Department queried readily available water quality data from two primary sources. The first being SQUID. Although surface water chemistry data collected by SWQB have met the sensitivity and accuracy requirements specified in SWQB Quality Assurance Project Plan for Water Quality Management Programs, the Department found no readily available data in SQUID for the waters evaluated as part of this analysis.

The second primary source used by the Department for this evaluation was LANL's "Intellus New Mexico" database. Intellus New Mexico is a publicly accessible database containing environmental monitoring data provided by LANL, DOE-OB, and other third-party providers (such as local and Tribal entities). As stated on the website itself (**NMED Exhibit 76**), Intellus was established to provide complete transparency into LANL's unclassified environmental monitoring and sampling data. Specifically, Intellus is intended to ensure that the public has real-

time access to the most recent data used by managers, analysts, and scientists to help guide environmental stewardship decisions. System data are updated nightly and all data are verified and validated before release. The database contains over 16 million records, including more than 32,000 locations and about 400,000 samples. These records include recent and historical information about samples and laboratory analyses for air, soil, sediment, biota, and water. In order to determine the available data pertinent to this analysis, the Department reviewed the data collection activities prescribed under the 2014 and 2020 Interim Facility-Wide Groundwater Monitoring Plans ("IFGMPs"). The Department determined, through its evaluation, that the available data for temperature, pH and dissolved oxygen obtained from Intellus, as collected by LANL according to the IFGMPs, were sufficient for evaluation in determining the existing aquatic life use.

The Department used stream gage data to determine the duration, intensity, and frequency of surface water flows for a waterbody at a discrete location over a specified period. Stream flow was captured through two sources. The first was stream flow data extracted from LANL's Intellus database. This data was collected by LANL through instantaneous measurements (usually with water quality data collection activities) or through continuous monitoring in both automated stream and alluvial well samplers, according to LANL's IFGMPs. The Department compared this data to historical weather data collected from the National Oceanic Atmospheric Administration ("NOAA") for Los Alamos county between September 1, 2005, and July 1, 2007. The data extracted from NOAA was used to confirm that streamflow was not in direct response to significant precipitation events. In addition to streamflow data, LANL provided hydrographs for several gage stations within S-Site Canyon (Martin Spring). A hydrograph is a graph or plot that shows stream flow in relation to time, given a specific point or cross section where the flow is

measured. Although the presence of streamflow data supports the persistence of water, the lack of streamflow data does not demonstrate, in and of itself, conditions that cannot support aquatic life. All hydrology and weather data used as part of the analysis are provided as an Appendix to the EUA (**NMED Exhibit 73**)

Ambient water temperature is one of the criteria necessary to determine the existing aquatic life use and can be a limiting factor based on species temperature tolerance ranges. Most of the State's criteria for designated aquatic life uses include both a maximum temperature criterion, which indicates an acute exposure, as well as a magnitude-duration temperature criterion (4T3 or 6T3), which represents a chronic exposure. In order to accurately capture a maximum temperature, 4T3 or 6T3, as defined in 20.6.4.7 NMAC, long term deployment of a temperature data logger is required. However, instantaneous measurements can provide information on what temperatures may not be attainable. The Department obtained water temperature data through an Intellus query for all sites spanning between 1975 and 2021. Data were collected by LANL using LANL's IFGMPs. All water temperature data used as part of the analysis are provided as an Appendix to the EUA (**NMED Exhibit 73**)

Whether anthropogenically influenced or natural, a waterbody's pH can be a limiting factor when it comes to that waterbody's ability to support aquatic life. It is also one of the criteria for the State's designated recreational primary contact use. Since there were no associated *E. coli* data for analysis of the recreational existing use, pH was only evaluated as it relates to aquatic life uses. For this analysis, data associated with the circumneutral range for pH was used to determine the existing aquatic life use for each of the canyons. All pH data used as part of the analysis are provided as an Appendix to the EUA (**NMED Exhibit 73**)

Ambient dissolved oxygen concentration is another water quality parameter used to evaluate the existing aquatic life use. The amount of oxygen that can be dissolved in water is temperature and atmospheric pressure-dependent. All dissolved oxygen data used as part of the analysis are provided as an Appendix to the EUA (**NMED Exhibit 73**)

3. Existing Use Findings

Effluent Canyon was surveyed once, below outfall 051, using the Hydrology Protocol survey method, resulting in a total score of 15.00. Although stream flow data were limited, as captured by automated stream gages (E1-W and E1-E), the data demonstrate a measurable seasonal flow in the fall and spring not directly related to precipitation events. The findings from the Hydrology Protocol survey and the stream flow data are supportive of each other, indicating Effluent Canyon has an intermittent hydrologic condition. As part of the Level 1 Hydrology Protocol survey, benthic macroinvertebrates are evaluated based on presence or absence. The survey team observed benthic macroinvertebrates during the Level 1 Hydrology Protocol surveys conducted with representatives from the Department and LANL. The presence of benthic macroinvertebrates was confirmed through a Level 2 survey conducted independently, and on a different date, by LANL. Due to sustained periods of water and the presence of benthic macroinvertebrates, the Department concludes that the existing aquatic life use for Effluent Canyon is, at a minimum, marginal warmwater, which includes protection for aquatic life against chronic exposure to pollutants.

The Department did not find available water chemistry data for Effluent Canyon. The closest representative data was limited to what could be extrapolated from Mortandad Canyon just below the confluence with Effluent Canyon.

Temperature data was limited in its usability to demonstrate existing use as there was no long-term continuous data necessary to determine the lowest instantaneous and sustained maximum temperatures. The discrete data points were analyzed and demonstrated Effluent Canyon currently does not support coldwater aquatic life use. Based on actual temperature data from Mortandad Canyon, Effluent Canyon has an existing aquatic life use of at least marginal warmwater.

Data obtained from Mortandad Canyon below its confluence with Effluent Canyon was used to infer the water quality for Effluent Canyon. Data demonstrated that 92% of the measurements obtained from 1997 to 2009 were within the circumneutral pH range supportive of aquatic life.

Dissolved oxygen data was more abstract in demonstrating the existing aquatic life use due to several issues with the data set. The data set includes what appear to be outliers, those data values that are outside an achievable range. The Department identified these data points and did not evaluate them as part of the overall dataset. The other challenge for using this dataset was the lack of long-term continuous monitoring. Continuous monitoring (hourly measurements over 3-14 days) is necessary to derive the diel variation of dissolved oxygen that can be used to evaluate the water's ability to support aquatic life. Overall, although dissolved oxygen data were available, the Department was unable to definitively use the data to support an existing use determination, other than to conclude that there was sufficient flow to collect instantaneous dissolved oxygen data.

S-Site Canyon was surveyed three times in three locations, using the Hydrology Protocol survey method. Two of these surveys were within the geographical area of this analysis while one was outside of the study area but plays relevance to the hydrologic condition of S-Site Canyon as

a whole. The two surveys within the study area were conducted on August 29, 2019, one at the monitoring well MSC-16-06293, which scored 8.00 (ephemeral), and one above the monitoring well, which scored 16.00 (intermittent). The third survey which is outside the geographical range of this analysis, is located just before its confluence with Water Canyon, approximately 2,700 meters downstream from monitoring well MSC-16-06293, and scored 9.00 (intermittent). Although the third survey site is beyond the geographical extent of this analysis, the survey scores, when evaluated collectively, provide evidence that overall S-Site Canyon is intermittent. Hydrographs of three alluvial well gages along S-Site Canyon provide further evidence of the presence of surface water. The alluvial well gages, MSC-16-06293, MSC-16-06294 and MSC-16-06295, are located approximately 340 meters (m), 1,000m and 1,300m downstream of Martin Spring, respectively. However, based on the limited scope of this EUA, data for S-Site Canyon was only evaluated from alluvial monitoring well MSC 16-06293 upstream to its origin at Martin Spring.

As it pertains to this EUA, the ground water elevation data for alluvial monitoring well MSC-16-06293 indicate seasonal fluctuation in groundwater depth to levels just around a foot below ground elevation. The monitoring well, located above and outside the stream channel, indicates that seasonal surface water flows are likely in the channel at this location, supporting the intermittent findings from the Hydrology Protocol surveys. Since groundwater is generally less responsive to direct precipitation, water levels shown in hydrographs are likely not the result of a specific precipitation event.

Data indicate that seasonal levels in groundwater fluctuate regularly throughout S-Site Canyon to elevations at or above the stream bed surface, likely resulting in an intermittent flow regime sufficient to support aquatic life at least from alluvial monitoring well MSC 16-06293 to

Martin Spring. Survey staff documented benthic macroinvertebrates during the Level 1 Hydrology Protocol survey at Martin Spring but not at alluvial monitoring well MSC 16-06293. However, a Level 2 survey conducted independently, and on a different date, by LANL at a site identified as "below Martin Spring" supports the presence of benthic macroinvertebrates within this reach. Due to sustained periods of water and the presence of benthic macroinvertebrates, the Department concludes that the existing aquatic life use for S-Site Canyon from alluvial monitoring well MSC 16-06293 to its origin at Marin Spring is, at a minimum, marginal warmwater, which protects aquatic life against chronic exposure to pollutants.

Temperature data for S-Site Canyon was relatively robust as it pertains to the number of data points but limited to results from Martin Spring. Similar to Effluent Canyon, this data did not provide the long-term continuous data necessary for determination of the lowest instantaneous maximum temperature and sustained maximum temperatures. Still, the data demonstrate S-Site Canyon does not support coldwater aquatic life use. S-Site Canyon, based on actual water temperatures, has an existing life use of at least marginal warmwater.

Sample measurements at S-Site Canyon at Martin Spring demonstrated that 89% of the 53 sampling measurements between 1995 and 2020 had a circumneutral pH range that supports aquatic life.

The dissolved oxygen data for S-Site Canyon was limited to the samples taken at Martin Spring between 2005 and 2020. The data, similar to that found in Mortandad Canyon, had outliers that the Department determined unusable for this analysis. Of the 41 useable data points, 90% had instantaneous dissolved oxygen measurements greater than 6.0 mg/L. However, as discussed for Effluent Canyon, the diel swing is unknown without a long-term continuous dataset. Based on the

available information, the Department determined that S-Site Canyon has a dissolved oxygen concentration that supports at least a marginal warmwater use.

Two-Mile Canyon was surveyed four times in four locations along the two-mile reach evaluated under this analysis, using the Hydrology Protocol survey method. The survey locations include Two-Mile above Pajarito at E244, approximately 200m from Two-Mile Canyon's confluence with Pajarito, representing the terminal end of Two-Mile Canyon. Three of the surveys along this section of Two-Mile Canyon scored intermittent (18.00, 19.00, and 10.50), and one (identified as below TA-59) scored perennial (20.50). Similar to the findings for S-Site Canyon, the data demonstrate variable flow patterns as the water travels downstream, gaining water in some reaches and losing waters in others. Overall, the Department finds the data indicate Two-Mile Canyon is intermittent from its confluence with Pajarito Canyon to its confluence with Upper Two-Mile Canyon. Representatives from the Department and LANL observed benthic macroinvertebrates during the Level 1 Hydrology Protocol surveys at three of the four survey sites. The only location at which no benthic macroinvertebrates were observed was at Two-Mile Canyon above Pajarito Canyon at E244. Due to sustained periods of water and the presence of benthic macroinvertebrates, the Department concludes that Two-Mile Canyon from its confluence with Pajarito Canyon to its confluence with Upper Two-Mile Canyon has, at a minimum, an existing aquatic life use of marginal warmwater, which protects aquatic life against chronic exposure to pollutants.

Stream flow data was limited to two instantaneous measurements, in April 2016, in Two-Mile Canyon (below TA-59). No hydrograph data was provided by LANL. Although streamflow data could have supported the EUA determination, it is not required and does not influence the

determination that Two-Mile Canyon from its confluence with Pajarito Canyon to its confluence with Upper Two-Mile Canyon has an intermittent hydrologic regime.

Temperature data for Two-Mile Canyon had two sampling locations located approximately 1.5 miles from each other. This data, again, did not provide the long-term continuous data necessary for the determination of the lowest instantaneous and sustained maximum temperatures. Still, it can provide additional support to conclude that Two-Mile Canyon is not supporting a coldwater aquatic life use. Based on actual water temperatures, Two-Mile Canyon has an existing aquatic life use of at least marginal warmwater.

Sample measurements in two discrete sampling locations within Two-Mile Canyon demonstrated that between 88% and 93% of the sampling measurements between 1998 and 2019 had a circumneutral pH range that supports aquatic life.

Similar to Mortandad Canyon and S-Site Canyon, Two-Mile Canyon had dissolved oxygen outliers in the dataset, which the Department could not use as part of this analysis. There were two sample locations, as with the other parameters, with dissolved oxygen data limited to discrete sampling events unable to provide the diel range. Overall, the dissolved oxygen for Two-Mile Canyon shows between 77% and 100% of the 22 useable samples attaining a dissolved oxygen concentration of 6 mg/L or greater. However, as discussed for Effluent Canyon, the diel swing is unknown without a long-term continuous dataset. Based on the available information, the Department determined that S-Site Canyon has a dissolved oxygen concentration that supports at least a marginal warmwater aquatic life use.

H. Determination

Based on this analysis, the Department finds that Effluent Canyon, S-Site Canyon from alluvial monitoring well MSC 06-016293 to its headwaters, and Two-Mile Canyon from its

confluence with Pajarito to its confluence with Upper Two-Mile Canyon support secondary contact recreation and marginal warmwater aquatic life. The secondary contact recreational use is the current designated recreational use and requires no amendment at this time. However, the existing marginal warmwater aquatic life use has criteria more stringent than the current designated limited aquatic life use and requires a designated use amendment.

I. Conclusion

The Department finds that recreational use did not have sufficient evidence, at this time, to determine an existing use more stringent than secondary contact. However, for the designated aquatic life use under 20.6.4 NMAC, the Department did find these waters support marginal warmwater aquatic life and should be amended in accordance with 40 C.F.R. § 131.10(i) (**NMED Exhibit 22**) to be reflective of the existing use. This amendment requires adding a new section to 20.6.4 NMAC, which the Department proposes as 20.6.4.140 NMAC (**NMED Exhibit 9**). Adoption of the proposed changes would also require an update to the reserved section 20.6.4.141-200 as proposed in **NMED Exhibit 9**. The Department recognizes that other tributaries within lands managed by DOE within LANL, outside the scope of this analysis, warrant further investigation to determine the appropriate designated uses. The Department intends to evaluate these waters, if warranted.

VII. OTHER WATERS WITHIN LANL

Most waters within lands managed by DOE within LANL are currently classified in 20.6.4.126 and 20.6.4.128 NMAC. Specified perennial waters, delineated with their upstream and downstream boundaries, are classified in 20.6.4.126 NMAC, while portions of ephemeral and intermittent waters within LANL are classified broadly in 20.6.4.128 NMAC.

The Department, with representatives from LANL and some extent Amigos Bravos, conducted 52 Hydrology Protocol surveys from November 2016 to October 2019. As a result of these surveys, the Department identified four previously unclassified perennial reaches of tributaries within LANL. Pursuant to 20.6.4.99 NMAC, unclassified perennial waters, including those identified within LANL, have designated uses protecting for primary contact, warmwater aquatic life, livestock watering, and wildlife habitat.

The reaches identified as perennial, with concurrence from the three parties, include Ancho Canyon from the Rio Grande to Ancho Springs; Pajarito Canyon from Starmers Gulch to Homestead Spring; Pajarito Canyon from 500 meters downstream of Arroyo de la Delfe to Arroyo de la Delfe, and DP Canyon from 100 meters downstream of grade control to 400 meters upstream of grade control.

The language in 20.6.4.99 NMAC is clear that all perennial surface waters of the State except those classified in 20.6.4.101-899 NMAC are protected under this Section. Therefore, the identification of these unclassified perennial waters does not require a rulemaking action since the designated uses in 20.6.4.99 NMAC have always applied, even though it was not known these waters were perennial prior to this point. This situation is addressed explicitly in the WQMP/CPP section entitled "Establishing or Revising a Designated Use using the Hydrology Protocol" (**NMED EXHIBIT 64**). The designated uses in 20.6.4.99 NMAC are applicable for these waters until the Commission adopts and EPA approves an amended designated use. The Commission may adopt an amended designated use based on either (1) an analysis that demonstrates the criteria for the existing use are more stringent than the designated use in 20.6.4.99 NMAC or (2) a UAA that demonstrates the designated use in 20.6.4.99 NMAC is not an existing use and is not attainable

due to one of the factors in 40 C.F.R. § 131.10(g) (**NMED Exhibit 22**), and the UAA establishes the highest attainable use for that water.

There may be other intermittent tributaries not represented in this triennial review that have existing aquatic life and recreational uses with criteria more stringent than the current designated uses in 20.6.4.128 NMAC. Therefore, the Department will continue to evaluate available water quality data to determine the existing uses, in accordance with 40 C.F.R. § 131.10(i) (**NMED Exhibit 22**), and make recommendations for designated use amendments before the Commission, accordingly.

VIII. ADMINISTRATIVE PROCESS

A. Tribal Engagement

Although the State's water quality regulations do not apply to tribal waters within the exterior boundaries of a tribe or those lands to which the tribe has incorporated into federal trust; many waters cross boundaries and jurisdictional protections, and there is a shared interest in the protection of water quality between tribes and the State of New Mexico. The State recognizes the importance of communication and collaboration with tribes to ensure water quality across boundaries. The State has memorialized this sentiment through the State-Tribal Collaboration Act, NMSA 1978, § 11-18-3 (**NMED Exhibit 17**), Executive Order 2005-004, and the Department's Tribal Communication and Collaboration Policy ("Policy")[1]. Through the Policy, NMED engages tribes during any Department action(s) that may impact the natural, cultural, or environmental resources of a tribe. Tribes are sovereign entities. Therefore, the State interacts with tribes in a government-to-government capacity. These interactions with tribes are independent of stakeholder and public outreach efforts.

[1] Available at https://www.env.nm.gov/general/wp-content/uploads/sites/10/2016/08/2020-01-27-NMED-Tribal-Policy-2020-final-signed.pdf.

As part of this endeavor, the Department solicited participation from tribes via an initial email notification sent to 36 individuals representing 24 tribes and one tribal consortium on January 30, 2020 (**NMED Exhibit 77**). Although there was no direct response, ongoing communication with tribes is vital to providing an opportunity to engage, as they find appropriate.

In addition to the initial notification, the Department provided notice of the proposed amendments and public comment opportunity to over 30 tribal representatives on November 2, 2020 (**NMED Exhibit 78**). On November 25, 2020, the Department notified the tribes of an extension to the public comment period (**NMED Exhibit 79**). Finally, on February 2, 2021, the Department notified tribes of the public hearing with information on how to participate, should they choose to do so (**NMED Exhibit 80**). The Department's Tribal Liaison conducted all communication through email. Approximately 36 to 38 individuals representing 22 tribes and one tribal consortium were notified during each stage in the Triennial Review Process. This included notifications to the Pueblo of Acoma, Cochiti Pueblo, Fort Sill Apache, Isleta Pueblo, Jemez Pueblo, Jicarilla Apache Nation, Nambe Pueblo, Navajo Nation, Ohkay Owingeh, Picuris Pueblo, Pueblo de San Ildefonso, Pueblo of Picuris, Pueblo of Pojoaque, Pueblo of Santa Ana, San Felipe Pueblo, Santa Ana Pueblo, Santa Clara Pueblo, Kewa (Santo Domingo) Pueblo, Taos Pueblo, Tesuque Pueblo, Ysleta del Sur Pueblo, Zia Pueblo and Eight Northern Indian Pueblos Council.

Specific notification regarding the existing recreational use analysis went out to all the tribes on January 28, 2021, (**NMED Exhibit 57**) to ensure the proposed action to amend the recreational designated use from secondary contact to primary contact would not cause concern to tribes for those water crossing between State and Tribal jurisdiction. Through this outreach effort, the Pueblo of Jemez reached out and met with the Department. The Pueblo of Jemez expressed no outstanding concerns regarding the proposed amendment at that time.

B. Stakeholder Outreach

As part of the Triennial Review, the opportunity for meaningful engagement is afforded to stakeholders at several steps prior to the hearing notice.

As part of the outreach efforts outlined in the Public Involvement Plan, and in line with the requirements in 40 C.F.R. § 25.3 (**NMED Exhibit 28**), the Department engaged individuals through the SWQB's listserv, requesting them to self-identify as stakeholders to the Triennial Review, as applicable. The Department sent an email to 1,746 subscribers in the SWQB's GovDelivery listserv on February 3, 2020 (**NMED Exhibit 81**). This endeavor resulted in 64 individuals self-identifying as stakeholders to the matter. In addition to self-identifying, the Department identified 58 individuals, predominately representing state and federal government agencies or soil and water conservation districts that may be affected by the Triennial Review's actions.

The Department then invited self-identified and Department-identified stakeholders to one of three virtually held stakeholder discussions on July 17, 22, and 24, 2020. At these discussions, the Department presented the Triennial Review Process, the general water quality standards sections being considered for amendment, the prioritization process for consideration of amendments, the tentative timeline for the Triennial Review process, and the additional opportunities to participate. The slides from the presentation are presented as **NMED Exhibit 82**. Over the course of the three discussions, approximately 38 individuals participated. Feedback from these discussions helped the Department formalize the proposed amendments filed as part of the petition to request a hearing beforethe Commission on August 19, 2020.

In addition to the general stakeholder outreach, two amendments proposed by the Department as part of this Triennial Review warranted individualized notification to Department-

identified stakeholders. The first was to individuals that may be directly impacted by the proposed designated use amendments associated with the EUA evaluating recreational designated uses for classified waters with secondary contact recreational use (**NMED Exhibit 56**). The Department provided notification via email regarding these amendments to the City of Roswell, the City of Truth or Consequences, the Village of Fort Sumner, and the City of Artesia on January 20, 2021 (**NMED Exhibit 58**). These outreach efforts resulted in dialogue between the Department and the Village of Fort Sumner regarding the proposed amendments and potential impacts to the Village.

The second amendment was part of the Joint Stipulation entered into between the Department, Amigos Bravos, DOE, and Los Alamos National Security, LLC, on October 9, 2015 (**NMED Exhibit 72**). As part of the Joint Stipulation, the Department regularly engaged with the parties to gather information and reach consensus on the appropriate designated uses for ephemeral and intermittent waters within LANL currently classified in 20.6.4.128 NMAC. In addition to the Department's direct outreach to the parties spanning five years through email, phone calls, meetings, field work, and written communication, the Department also provided regular updates to EPA Region 6, as they are considered stakeholders to all matters related to the CWA.

Overall engagement with stakeholders continued as needed throughout the process, predominately through phone calls or emails, addressing questions or concerns on the proposed amendments or the general process.

C. Public Outreach

In accordance with 40 C.F.R. § 25.3 (**NMED Exhibit 28**) states are required to provide for, encourage, and assist the participation of the public for activities being carried out under the CWA, such as a rulemaking. Public participation includes providing access to the decision-making process, seeking input from and conducting dialogue with the public, assimilating public

viewpoints and preferences, and demonstrating that those viewpoints and preferences are considered. Public agencies should encourage full presentation of issues at an early stage so that they can be resolved, and timely decisions can be made. The Department ensures adherence to the requirements in 40 C.F.R. § 25.3 (**NMED Exhibit 28**) through the application of public outreach actions prescribed in the State's approved WQMP/CPP.

In addition to stakeholder outreach, and in accordance with 40 C.F.R. § 25.3 (**NMED Exhibit 28**), the Department also provided a public comment period for proposed amendments. As part of the Department's efforts to provide the public opportunity for meaningful engagement, notification was distributed via several outlets, including posting the notification on the Bureau's main webpage, the Water Quality Standards webpage, and the 2020 Triennial Review webpage (**NMED Exhibit 83**). The Department provided notification in both English and Spanish (**NMED Exhibit 84)** through a GovDelivery email distribution to 1,805 individuals subscribed to the Bureau's distribution list (**NMED Exhibit 85**); notification to 37 individuals representing 24 tribes (**NMED Exhibit 78**); and 101 identified stakeholders (**NMED Exhibit 86**).

The public comment period was initially open from November 2, 2020, through December 2, 2020; however, due to public response, the Department extended the period for an additional 35 days through January 6, 2021. The Department conducted two virtual public meetings as a part of the public outreach efforts. The first was held on November 12, 2020, at 5:30 pm (**NMED Exhibit 87**), which 36 individuals attended, and the other on November 16, 2020, at 2:30 pm which approximately 17 individuals attended. The scheduled dates and times varied to provide more opportunity for public participation. In addition, and to provide additional opportunity for individuals to review the information presented, the Department posted recordings of the public meetings on the Bureau's dedicated Triennial Review webpage (https://www.env.nm.gov/surface-

water-quality/2020-triennial-review/). As a result of the public comment period and outreach discussions, the Department received approximately 200 comments regarding the proposed amendments. The Department received comments from EPA Region 6 on December 22, 2020 (**NMED Exhibit 88**); Environmental Defense Fund on December 9, 2020, and January 6, 2021 (**NMED Exhibit 89**); LANL-Triad National Security, LLC on January 6, 2021 (**NMED Exhibit 90**); New Mexico Mining Association on January 6, 2021 (**NMED Exhibit 91**); San Juan Water Commission on January 6, 2021 (**NMED Exhibit 92**); Chevron Mining, Inc. Questa Mine on January 5, 2021 (**NMED Exhibit 93**); Concerned Citizens for Nuclear Safety on January 6, 2021 (**NMED Exhibit 94**); and a collaborative submission from Amigos Bravos, Audubon New Mexico, Defenders of Wildlife, Flower Hill Institute, Gila Resources Information Project, Honor Our Pueblo Existence, Molino de la Isla Organics LLC, New Mexico Environmental Law Center, New Mexico Wild, Rio Grande Restoration, Rovers and birds, Rio Grande Chapter of the Sierra Club, Tewa Women United, Trout Unlimited, Upper Pecos Watershed Alliance, and Western Environmental Law Center on January 6, 2021 (**NMED Exhibit 95**). The Department posted the comments, as submitted, on the Department's Triennial Review webpage.

D. Hearing Notice

The Triennial Review is a rulemaking process governed by several federal and state laws. The New Mexico Legislature delegated authority to the Commission to "adopt water quality standards for surface and ground waters of the state based on credible scientific data and other evidence appropriate under the Water Quality Act." NMSA 1978, § 74-6-4(D) (**NMED Exhibit 14**). Amendments to the Standards must comply with NMSA 1978, § 74-6-6 (**NMED Exhibit 15**) and 20.1.6 NMAC, which set forth procedural requirements for rulemaking proceedings before the Commission.

In accordance with NMSA 1978, § 74-6-6 (**NMED Exhibit 15**), no amendment to a water quality standard may be adopted or amended until after a public hearing. In addition, and in accordance with 40 C.F.R. § 131.20 (**NMED Exhibit 21**), EPA requires states to have a "public hearing for the purpose of reviewing applicable water quality standards adopted pursuant to 40 C.F.R. §§ 131.10 through 131.15…". The EPA requires that these hearings are held according to State law and EPA's public participation regulation (40 C.F.R. § 25).

The EPA has promulgated regulations in 40 C.F.R. § 25.5 (**NMED Exhibit 28**) regarding a state's responsibilities to notify the public for hearings associated with the federal CWA, including hearings for Triennial Reviews of States Standards. In part, EPA requires that a notice of the hearing be well-publicized and mailed to interested and affected parties at least 45 days prior to the hearing date. The State's rulemaking regulations for the Commission are more prescriptive than the federal regulations. In accordance with 20.1.6.201 NMAC, the State must provide notice of the hearing in the State Register and in at least one newspaper of general circulation at least 60 days prior to the hearing (**NMED Exhibit 29**). The Procedural Order for this matter, issued by the hearing officer on November 9, 2020, required hearing notification to be provided by February 1, 2021, 162 days prior to the hearing, thus fulfilling both federal and state regulatory timelines for public hearing notification.

The Department, in fulfillment of both the federal and State requirements, as well as the Procedural Order, published the notice in the State Register on January 26, 2021, 168 days prior to the hearing (**NMED Exhibit 96**); provided hearing notice to identified stakeholders, those that may be directly or indirectly affected by the proposed amendments, via email on January 29, 2021, 165 days prior to the hearing (**NMED Exhibit 97**); published the notice of hearing in both the

Albuquerque Journal (**NMED Exhibit 98**) and the Santa Fe New Mexican (**NMED Exhibit 99**) on January 30, 2021, 164 days prior to the hearing.

In accordance with the State Rules Act NMSA 1978, § 14-4-5.2 (**NMED Exhibit 19**), notice of a proposed rulemaking is required to be "provide[d] to the public" at least 30 days prior to the hearing. The term "provide to the public", as defined in NMSA 1978, § 14-4-2 (**NMED Exhibit 18**), means posting the notice on the agency website, posting on the sunshine portal, making it available in the agency's district, field and regional offices, sending notice to persons who have been requested to be notified of such announcements and providing it to the New Mexico Legislative Council for distribution to appropriate interim and standing legislative committees. In fulfillment of this requirement, the Department provided notification to the agency's District Managers by SWQB's Bureau Chief, Shelly Lemon, on January 28, 2021 (**NMED Exhibit 100**); notice of hearing was posted on the Surface Water Quality Bureau's website (**NMED Exhibit 101**) on January 29, 2021; notification was distributed, via email to identified stakeholders on January 29, 2021 (**NMED Exhibit 97**); notification was distributed, as discussed above, to the Surface Water Quality Bureau's listserv subscribers on January 29, 2021 (**NMED Exhibit 102**); notification was posted on the Sunshine Portal on March 10, 2021 (**NMED Exhibit 103**); and notification was provided to the Legislative Council Service on March 31, 2021 (**NMED Exhibit 104**).

Finally, in accordance with the Small Business Regulatory Relief Act NMSA 1978, § 14-4A-4 (**NMED Exhibit 20**), the Department provided notification of the proposed amendments to the Small Business Regulatory Advisory Commission **(NMED Exhibit 105**) on March 29, 2021**.**

Pursuant to 40 C.F.R. § 25.5 (**NMED Exhibit 28**) the public hearing notification must identify the matters to be discussed at the hearing and the agency's tentative determination on

major issues. Reports, documents and data relevant to the discussion at the public hearing shall be available to the public at least 30 days before the hearing. In addition, the agency shall schedule witnesses in advance to ensure maximum participation and allotment of adequate time for all speakers. In accordance with 20.1.6.202 NMAC (**NMED Exhibit 29**), the Notice of Intent to Present Technical Testimony along with the supporting exhibits must be filed at least 20 days prior to the hearing or in accordance with the procedural order. As prescribed in the Procedural Order issued on November 9, 2020, the Department's Notice of Intent to Provide Technical Testimony, along with the supporting evidence for the proposed amendments, was required to be filed with the Commission on May 3, 2021, 71 days prior to the hearing, fulfilling the requirements of both 40 C.F.R. § 25.5 (**NMED Exhibit 28**) and 20.1.6.202 NMAC. In accordance with 40 C.F.R. § 25.5 (**NMED Exhibit 28**), the hearing, according to the Order for Hearing and Appointment of Hearing Officer, provided adequate time to accommodate all witnesses with ample opportunity to participate.

E. Administrative Filings

The Department filed a petition for a public hearing to amend the *Standards for Interstate and Intrastate Surface Waters* (20.6.4 NMAC) with the Commission on August 19, 2020, consistent with 20.6.4.10(A) NMAC and pursuant to 20.1.6.200 NMAC, Section 303(c)(1) of the CWA (33 U.S.C. § 1313) (**NMED Exhibit 11**) and related federal regulation 40 C.F.R. § 131.20 (**NMED Exhibit 21**). The petition included the proposed amendments and a preliminary statement of reasons for the amendments. The petition was docketed as WQCC 20-51(R) and on October 13, 2020, the Commission granted a multi-day hearing to commence July 13, 2021. The Commission issued a Scheduling Order and Appointment of a Hearing Officer on October 19, 2020.

Several entities submitted Entry of Appearances into the matter, including Triad and DOE October 23, 2020; Western Environmental Law Center on March 15, 2021; San Juan Water Commission on March 23, 2021; New Mexico Mining Association on March 25, 2021; and Buckman Direct Diversion Board on April 20, 2021.

Following the public comment period, the Department filed an amended petition with the Commission on March 12, 2021. This amended petition included revised language for proposed amendments to 20.6.4 NMAC along with an index of those changes to aid with referencing the petition filed in August 2020 and the new proposed amendments.

In accordance with the amended Procedural Order's Table of Relevant Deadlines issued by the Hearing officer on April 1, 2021, the Department filed a Notice of Intent to Provide technical Testimony on May 3, 2021.

IX. PROPOSED AMENDMENTS

In conclusion, the Department's testimony and supporting demonstrations support the proposed amendments. The Department thereby requests the Commission approve the proposed amended language to the State's *Standards for Interstate and Intrastate Surface Waters* (20.6.4 NMAC) as submitted, (**NMED Exhibit 9**).

This concludes my direct testimony.

Shelly Lemon

Education

M.S. HYDROLOGY | UNIVERSITY OF ARIZONA, TUSCON, AZ

B.S. BIOLOGY | UNIVERSITY OF ARIZONA, TUCSON, AZ

Experience

BUREAU CHIEF | NMED – SURFACE WATER QUALITY BUREAU (SWQB), SANTA FE, NM
08/2016 – PRESENT

- Manage the Surface Water Quality Bureau (SWQB) of the New Mexico Environment Department (NMED) by planning, setting, and achieving goals set forth in the NMED Strategic Plan, EPA approved work plans, and program planning documents.
- Work with the public, industry and decision makers (legislators, Governor's staff, etc.) to ensure that the goals of the bureau are achieved.
- Oversee an operating budget of $6.4 million dollars that requires administration of general funds, special revenue funds, interagency transfers, and federal grants.
- Directly or indirectly supervise 35 technical and administrative staff including hiring, work performance evaluations, and discipline, if needed. Ensure performance goals are met and activities are conducted in accordance with applicable statutes, policies, rules, permits, orders, and grant commitments.
- Develop and respond to legislative proposals and develop regulatory initiatives; e.g., assist the NMED's legislative tracking office during various legislative sessions, including bill analysis and being a lobbyist and expert witness; participate in the development and revision of state surface water quality standards and regulations, and present technical testimony during hearings.
- Participate in meetings and strategy discussions to refine technical processes and work products and to ensure technical work is of high quality and defensible.
- Work with staff to identify future trends and opportunities to develop strategies that improve the program, bureau, and agency.
- Facilitate coordination between EPA, and other public/private agencies/entities involved in surface water quality protection, management and regulation.
- Develop policies, guidelines and templates to facilitate successful completion of projects and ensure efficient implementation of all programs.
- Ensure that information requests are responded to in a timely and professional manner.
- Oversee short-term investigations in response to citizen complaints, accidental spills, and other emergencies (e.g., Gold King Mine, Cimarron River tanker spill, etc.).
- Work with the Department's webmaster to create, update, and maintain webpages, resources, and links associated with activities of the Bureau.

PROGRAM MANAGER MONITORING, ASSESSMENT & STANDARDS | NMED-SWQB, SANTA FE, NM
06/2015 – 10/2016

- Managed the Surface Water Quality Bureau's monitoring, assessment, and standards program including writing, submitting, and managing the Clean Water Act (CWA) Section 106 grant and the CWA Section 106 Supplemental grant on an annual basis with semi-annual updates.
- Oversaw and evaluated the performance of 14 staff.
- Participated in the development and revision of state surface water quality standards and regulations including the 2013 Triennial Review presented during the Water Quality Control Commission's October 2015 hearing and subsequent deliberations.

- Planned water quality surveys throughout New Mexico that met budgetary constraints and data quality objectives.
- Reviewed, integrated, and assessed data for use in Clean Water Act required activities.
- Prepared water quality reports (e.g. watershed survey summaries, use attainability analyses, TMDLs, etc.) for the public and as a deliverable to EPA.
- Reviewed, updated, and developed protocols to standardize tasks including sample collection, data assessment, and report writing.
- Represented the Bureau at meetings, professional conferences, workshops, and Water Quality Control Commission meetings.
- Conducted short-term investigations in response to citizen complaints, accidental spills, and other emergencies including NMED's response to the Gold King Mine release.
- Worked with the Bureau's webmaster to create, update, and maintain webpages, resources, and links associated with activities of the Section and Bureau.

MUNICIPAL TEAM LEADER | NMED-SWQB, SANTA FE, NM
03/2014 – 05/2015

- Reviewed and evaluated the performance of the Municipal Team by providing meaningful, frequent, and ongoing input on work performance and prioritization of workloads.
- Cooperated with and supported the efforts of other SWQB sections. Facilitated positive working relationships with other state and federal agencies, stakeholders, and cooperators involved in NPDES permitting activities.
- Reviewed, analyzed data, and prepared comments on NPDES discharge permits submitted to the Bureau for certification under Section 401 of the Federal Clean Water Act. Ensured consistency in NPDES permit certifications.
- Investigated regulated facilities for compliance/non-compliance with applicable state and federal surface water quality laws, standards and regulations, and prepared and submitted comprehensive inspection reports that documented the status of the facilities regarding the federal NPDES permit program and regulations.
- Collected accurate and detailed information and useable evidence during site investigations to supplement information contained in NPDES permits, to evaluate violations of state surface water quality standards and regulations, and to assist in enforcement.
- Reviewed, analyzed, and prepared well-written, clear, concise, and factual comments on proposed or new amended federal and state agency policies and procedures, regulations, and technical recommendations.
- Developed standard operating procedures for wastewater sampling and compliance sampling. Evaluated and acquired sampling equipment necessary for monitoring NPDES permitted facilities.
- Reviewed, analyzed data, and prepared comments relevant to regulatory requirements and surface water quality studies and findings on Environmental Assessments (EA) and Environmental Impact Statements (EIS) submitted to SWQB for review.

ACTING MANAGER MONITORING, ASSESSMENT & STANDARDS | NMED-SWQB, SANTA FE, NM
07/2012 – 07/2013

- Managed the Surface Water Quality Bureau's monitoring, assessment, and standards program including writing, submitting, and managing the Clean Water Act Section 106 Supplemental grant on an annual basis with semi-annual updates.
- Oversaw and evaluated the performance of 15 staff.
- Participated in the development and revision of state surface water quality standards and regulations.
- Planned water quality surveys throughout New Mexico that met budgetary constraints and data quality objectives.
- Reviewed, integrated, and assessed data for use in Clean Water Act required activities.
- Prepared water quality reports (e.g. watershed survey summaries, use attainability analyses, TMDLs, etc.) for the public and as a deliverable to EPA.
- Developed protocols to standardize tasks including sample collection, data assessment, and report writing.

- Represented the Bureau at meetings, professional conferences, workshops, and Water Quality Control Commission meetings.
- Conducted short-term investigations in response to citizen complaints, accidental spills, and other emergencies.
- Maintained analytical results in the SWQB water quality database, prepared retrievals of stored data as requested, and scheduled uploads of data to the EPA's national database.
- Worked with the Bureau's webmaster to create, update, and maintain webpages, resources, and links associated with activities of the Section and Bureau.

MONITORING TEAM LEADER | NMED-SWQB, SANTA FE, NM
04/2011 – 07/2013

- Managed the statewide ambient monitoring program for the Bureau. The Monitoring Team is responsible for collecting water quality data and associated flow measurements in surface waters of the state. Data collected by the Monitoring Team is used to determine if the water body meets water quality standards and supports designated uses.
- Oversaw and evaluated the performance of 5 staff.
- Planned water quality surveys throughout New Mexico that met budgetary constraints and data quality objectives.
- Ensured adequate and appropriate data were collected to support a variety of Clean Water Act required activities (e.g., WQS changes, TMDL development, NPDES permits, NPS monitoring effectiveness, etc.).
- Prepared watershed survey summaries for the public and as a deliverable to EPA.
- Developed protocols to standardize tasks including sample collection, data assessment, and report writing; specifically, responsible for developing, updating, and revising the *Field Sampling Plan* and *Physical Habitat* standard operating procedures.
- Maintained analytical results in the SWQB water quality database, prepared retrievals of stored data as requested, and scheduled uploads of data to the EPA's national database.
- Conducted short-term investigations in response to citizen complaints, accidental spills, and other emergencies.
- Worked with the Bureau's webmaster to create, update, and maintain monitoring webpages, resources, and links.
- Represented the Bureau at meetings, professional conferences, and workshops.

NUTRIENTS AND LAKES TEAM LEADER & TMDL WRITER | NMED-SWQB, SANTA FE, NM
08/2004 – 04/2011

- Prepared watershed planning documents (TMDLs) to improve water quality and conducted public meetings to address stakeholder comments and concerns.
- Presented the final draft documents to the NM Water Quality Control Commission (WQCC) for inclusion and adoption into the State's Water Quality Management Plan.
- Oversaw the nutrient criteria development program for NM's streams, rivers, and lakes.
- Headed efforts in hydrology and monitoring design to develop a *Hydrology Protocol* that distinguishes between ephemeral, intermittent, and perennial waters in New Mexico and to create a practical yet thorough 10-year monitoring and assessment strategy for the Bureau.
- Managed and evaluated the performance of 3 technical staff.

GRADUATE RESEARCH ASSISTANT | SUSTAINABILITY OF SEMI-ARID HYDROLOGY AND RIPARIAN AREAS (SAHRA) – UNIVERSITY OF ARIZONA, TUCSON, AZ
01/2002 – 01/2004

- Designed, coordinated, and implemented a hydrologic research project to determine the influence of land use and regional hydrology on surface water quality in a semi-arid stream.
- Organized and prepared an objective, scientifically sound thesis describing the methods, results, conclusions, and management implications of this research.

- Co-authored the journal article, "Spatial variability in dissolved organic matter and inorganic nitrogen concentrations in a semiarid stream, San Pedro River, Arizona" for the *Journal of Geophysical Research* Volume: 112, Issue: G3.

GRADUATE TEACHING ASSISTANT | UNIVERSITY OF ARIZONA, TUCSON, AZ
01/2002 – 12/2003

- Assisted in the instruction of an "Introduction to Global Change" class for undergraduates and a "Fundamentals of Water Quality" class for graduates.
- Developed hands-on activities to enhance global awareness and environmental stewardship.
- Designed and facilitated a final project to encourage critical analysis and informed decision-making.

Other Experience

Middle School Science Teacher | Academy of Technology and the Classics, Santa Fe, NM

High School Science Teacher | Chino Valley High School, Chino Valley, AZ

Teacher Fellow | Earth Watch Institute – Forest Birds Project, Bellavista Preserve, Ecuador

Science Instructor | Nizhoni - Upward Bound Summer Academy, Flagstaff, AZ

Naturalist | San Joaquin Outdoor Education, La Honda, CA

Science Instructor & Dive Master | Catalina Island Marine Institute, Avalon, CA

Kristopher F. Barrios

Education

B.S. Geology, Florida State University – August 1998

Work Experience

New Mexico Environment Department – Santa Fe, NM

Program Manager – Monitoring, Assessment, and Standards Section
Surface Water Quality Bureau
August 2017 – Present

- Manage the Monitoring, Assessment, and Standards Section, including planning and budget development
- Manage and evaluate the performance of 13 staff
- Serve as the grant manager for the Clean Water Act Section 106 Monitoring Initiative grant
- Manage and coordinate activities between the Monitoring, TMDL and Assessment, and Standards, Planning, and Reporting teams
- Review and prepare water quality survey reports, TMDLs, and water quality standards documents
- Develop, update, and review water data collection Standard Operating Procedures
- Perform data assessment and assessment validation for Clean Water Act reporting
- Guide development of monitoring plans and strategies, including chemical, continuous and biological components
- Oversee development and enhancement of the SWQB water quality database
- Create and maintain automated data management solutions
- Represent the section during Water Quality Control Commission meetings
- Perform fieldwork in accordance with Field Sampling Plans and Standard Operating Procedures
- Respond to public complaints, spills, and illicit discharges

Monitoring Team Supervisor
Surface Water Quality Bureau
October 2015 – August 2017

- Managed the team responsible for ambient surface water quality monitoring for the SWQB
- Managed and evaluated the performance of 5 staff
- Planned watershed surveys and led the development of Field Sampling Plans that met SWQB programmatic needs
- Prepared water quality survey reports
- Performed fieldwork following Field Sampling Plans and Standard Operating Procedures
- Developed and updated water data collection Standard Operating Procedures
- Created and maintained automated data management solutions
- Responded to public complaints, spills, and illicit discharges

Northwest Florida Water Management District (NWFWMD) – Midway, FL

Bureau Chief – Bureau of Hydrology and Engineering
Division of Resource Management
January 2015 – October 2015

- Managed the operation, budget, and planning for the Bureau of Hydrology and Engineering
- Supervised a team of 12 hydrologists, engineers, and hydrogeologists in the completion of Bureau responsibilities
- Evaluated hydrologic information required for resource management planning and assessment and plan responsive Bureau activities following policy directives
- Coordinated multi-agency water resource evaluations and restoration projects
- Pursued grant and funding opportunities at the state and federal level
- Supervised the District's participation as a Cooperating Technical Partner with the Federal Emergency Management Administration (FEMA) Flood Mapping Program
- Provided assistance as a subject matter expert in watershed advisory groups
- Served as the lead of a multi-disciplinary team developing and implementing tools to improve access and public understanding of hydrologic information

Manager – Hydrologic Data Section
Surface Water Bureau/Division of Resource Management
September 2006 – January 2015

- Conducted the hydrologic and water quality monitoring program for the Division of Resource Management
- Supervised eight full-time positions plus one student intern
- Developed and administered the annual budget for the Hydrologic Data Section
- Developed and implemented the NWFWMD Hydrologic Monitoring Plan
- Trained section personnel in the operation, maintenance, and troubleshooting of monitoring equipment
- Negotiated and managed water resource monitoring contracts with federal, state, and local governments
- Operated and maintained a network of over 150 continuous hydrologic data collection stations
- Prepared data reports and water resource publications
- Developed and administered relational databases for water quality, level, and flow
- Reviewed, verified, and published hydrologic data
- Designed and constructed hydrologic and water quality monitoring platforms
- Provided technical assistance to other divisions of the NWFWMD
- Represented the NWFWMD in watershed advisory groups

Hydrogeologist
Ground Water Bureau/Division of Resource Management
April 2004 – September 2006

- Served as the NWFWMD project manager for the Integrated Water Resources Monitoring (IWRM) Status and Groundwater Temporal Variability (GWTV) Contract and Springs Initiative Contract
- Developed and implemented projects and submitted contract deliverables as data reports and finished water resource publications

- Assisted the Ground Water Bureau Chief with personnel management and the development of budgetary estimates
- Conducted quality assurance review of data collected by Ground Water Bureau staff
- Performed additional water supply and quality assessment projects for the District's Ground Water Bureau

Hydrogeologist
Ground Water Bureau/Division of Resource Management
September 2002 – April 2004

- Facilitated the District's implementation of the IWRM Status and GWTV Contract projects following the Florida Department of Environmental Protection (FDEP) requirements, including preparation of quarterly progress and quality assurance reports, submission of budget estimates and contract proposals, and oversight and coordination of field activities
- Assisted the District's FDEP Springs Initiative project manager in the completion of field activities and data management associated with the Springs Initiative contract
- Purchased and maintain equipment and materials used to perform field tasks
- Performed additional water supply and quality assessment projects for the District's Ground Water Bureau
- Reviewed field and lab data
- Conducted internal field quality assurance audits

Advanced Environmental Technologies, LLC (AET) – Tallahassee, FL
Field Geologist
July 2001 – September 2002

- Supervised field activities under the Florida Department of Environmental Protection, Bureau of Petroleum Storage Systems (BPSS) Petroleum Cleanup Pre-approval Program
- Conducted monitor well installation, soil borings, soil and water sampling, and site surveys
- Created groundwater elevation contour maps and stratigraphic cross-sections from historical and field-collected data
- Performed and analyzed pump and slug tests
- Assisted in the preparation of proposals, change orders, and assessment reports submitted to BPSS personnel

Florida Department of Environmental Protection (FDEP) – Tallahassee, FL
Ground Water Protection Section
Environmental Scientist I
September 2000 – July 2001

- Managed the Florida Unique Well ID Program and Water Well Contractor Violation Clearinghouse
- Assisted in the administration of the Groundwater Contamination Delineation Program
- Developed and maintained interactive relational databases
- Created maps of ground water contamination using geographic information systems (GIS) software

- Sampled potable water wells and collected locational data using differentially corrected GPS equipment
- Participated in FDEP rulemaking for water well construction and water well contractor regulation

Bureau of Laboratories
Laboratory Technician
April 2000 – September 2000

- Prepared field sampling kits for the Chemistry Lab
- Logged returned sample information into the laboratory database and checked samples for proper preservation
- Inspected and maintained deionized water system
- Cleaned used glass sample containers for reuse
- Monitored laboratory gas pressures
- Disposed of laboratory wastes

Other Experience

- Office Manager – Parker Services, Inc., Tallahassee, FL, 1995-2000
- HVAC Technician – Fagan and Parker Heating and Air Conditioning, Tallahassee, FL, 1992-1995
- HVAC Technician – E & B Heating and Air Conditioning, Tallahassee, FL, 1991

Publications

- NWFWMD Water Resource Assessment 11-01, Jackson Blue Spring, FL, 2011
- NWFWMD Technical File Report 2011-01, Nitrogen Sources of Floridan Aquifer Springs, Merritt's Mill Pond, FL, 2011
- NWFWMD Technical File Report 07-01, Merritt's Mill Pond Springs Characterization, 2007
- NWFWMD Technical File Report 06-01, Wakulla Spring Discharge Rating, 2006
- NWFWMD Water Resources Special Report 06-01, Wakulla and St. Marks Rivers Springs Inventory, 2006
- NWFWMD Water Resources Special Report 06-02, St. Marks River Rise Chemical Characterization, 2006
- NWFWMD Water Resources Special Report 06-03, Morrison Spring Chemical Characterization, 2006
- NWFWMD Water Resources Special Report 05-02, Chemical Characterization of Wakulla and Jackson Blue Springs, FL, 2005
- NWFWMD Water Resources Special Report 05-01, Choctawhatchee River Springs Inventory, 2005
- NWFWMD Water Resources Special Report 04-01, Chipola River Springs Inventory, 2004
- NWFWMD Water Resources Special Report 04-02, Econfina Creek Springs Inventory, 2004

Licenses/Certifications

- Florida Licensed Professional Geologist #2861
- Geostatistical Short Course Certificate – University of Florida, August 2005
- FDEP Stream Condition Index and Habitat Assessment Certification, December 2005 to October 2015
- FDOT Work Zone Traffic Control Certification, May 2002

- FDEP Sample Collection Certification, IWRM Status and TV, December 2002
- OSHA Hazardous Materials Certification, September 2001 and June 2002
- ESRI ArcView I, January 2001

DIANA IXCHEL ARANDA
1190 St. Francis Drive, Suite N4050 Santa Fe, New Mexico 87505
Diana.Aranda@state.nm.us (505) 946-8666

Education

Nova Southeastern University Oceanographic Center - Ft. Lauderdale, FL. 4/2013
Master of Science in Coastal Zone Management

University of New Mexico - Albuquerque, NM. 5/2005
Bachelor of Science in Biology

Publication

Diana Ixchel Aranda, Jose V. Lopez, Helena M. Solo-Gabriele, and Jay Fleisher. 2016. Using Probabilities of Enterococci Exceedance and Logistic Regression to Evaluate Long Term Weekly Beach Monitoring Data. Journal of Water & Health, (1) : 81 -89.

Certification

Secondary Teacher Certification(7-12) 1/2017.

Current Employment

Environmental Scientist and Specialist Advanced. N.M. Department of Environment. 1/2017– Present.

ESS-A position in the Standards, Planning and Reporting Team. 2/2018-present.
Generates regulatory documentations that are scientifically defensible for the development and revisions to the State of New Mexico surface water quality standards in accordance with the federal Clean Water Act and NM Water Quality Act. These documents become public and undergo the Water Quality Control Commission and U.S. Environmental Protection Agencies approval process. Researches and reviews federal and state requirements, guidance's, public comment and historical information to guide document creation and decision making for standards. Investigates relevant scientific publications and data to aid in the development of standard regulatory documents. Collects relevant datasets from internal and external sources and conducts complex analysis of these data to aid in regulatory determinations. Creates maps and databases utilizing ArcGIS that can be included in public publication. Identifies key stakeholders and conducts the appropriate outreach. Presents findings in public and if appropriate, testify as an expert in the topic. Organize meetings, produce public notice documentation and adhere to regulatory deadlines. Advise third party constituents in the creation of documents pertaining the State's standards. Edit and consult internal departmental documents regarding standards.

Past Employment

Environmental Scientist TMDL Writer, N.M. Department of Environment. 2/2017-1/2019
Generated scientifically defendable department reports called, Total Maximum Daily Loads (TMDL) that establish the pollutant loading for specific surface waters in the State in according to The Clean Water Act. Assessed and analyzed water quality field data for the implementations of TMDLs. Participated in field work for gathering water quality data and habitat assessments following department standard operating procedures (SOP). Contributed in the evaluation of water quality impairments. Creates maps and databases utilizing ArcGIS that can be included in public publication. Conducted outreach with the public, state, federal, tribal, and municipal agencies to address stakeholders and constituent needs with individual groups and in public meetings, the State's Water Quality Control Commission and the Federal Environmental Protection Agency.

Biology High School Teacher, Santa Fe Public Schools. 8/2015 – 1/2017
Actively engaged students in academic learning with a Biology focus. Developed an exciting curriculum based on Common Core State Standards and an environment favorable to learning and personal growth. Instituted clear objectives for all lessons, assignments, units and projects in accordance with curriculum goals and communicates

objectives to students. Established effective rapport with students and motivated them to develop skills attitudes and knowledge needed to obtain a good foundation for continuous achievement growth and development in education. Cooperated with staff and support personnel in assessing and helping students with health, attitude, learning and behavioral problems. Utilized new and innovative ideas and technology in the classroom.

Surveillance Technician, Florid Keys Mosquito Control District. 8/2014 – 3/2015
Conduct mosquito surveillance for program operations designed to monitor and identify mosquito pest & disease vector species that affect the Florida Keys. Act as support to ongoing research projects such as the genetically modified mosquito project. Duties included; trap setup and retrieval, mosquito identification and data entry. Acted as public liaison for a door-to-door campaign and town hall meetings to educate the community about the release of genetically modified mosquito and assisted with public questions.

Project Manager, SWC Consultants. 8/2013-8/2014
Managed projects involving ecological, land use, and public involvement services for both public and private sector clients. Conducted Environmental Site Assessments (ESA) site visits, collected field and historical research data to produced technical reports for commercial real estate transaction due diligence under CERCLA. Collected historical and field data for several environmentally sensitive projects, conducted data analysis, created reports under an oversight of compliance timetables and budgets. Collaborated in the production of the Port Everglades Master Plan publication, and contributed with data collection and analysis, document review for existing conditions and impacts resulting from changes and expansion of the Port. Utilized ArcGIS software to create maps for reports and communications assignments.

Biologist I, Miami-Dade County, Coastal Resources Division. 8/2012- 4/2013
Processed Class I permits for coastal construction which included: assessment of benthic resources, plan review and report and permit writing. Reviewed and determine pre-construction conditions and created a report. Identified and notified of any code violations. Conducted quantitative underwater site transects and visual estimates of natural resource percent cover and evaluated on site benthic organisms, including sediment type, macroalgae, seagrass and corals.

Biological Technician, U.S. National Park Service - Biscayne National Park, 7/2011-6/2012
Assisted in the management and coordination of Biscayne National Park's water monitoring program within the park's marine waters. Conducted the deployment, data extraction, maintenance, and calibration of over 50 YSI instruments that our team managed. Planned and executed field work which involved: field safety planning, small boat operations, diving and maintenance of sites and equipment. Administer routine maintenance for the boat, the gear, the instruments and the lab. Prepared, collected, processed and analyzes data. Facilitated projects with partners and contractors.

Research Associate, National Oceanic and Atmospheric Administration (NOAA), University of Miami Cooperative Institute for Marine and Atmospheric Studies (CIMAS). 7/2008-6/2011

Research Associate II, (NOAA/AOML/PHOD) Ship of Opportunity Program, 9/2010-6/2011
Assisted in the management and coordination of the Ship of Opportunity Program Oceanographic program. Organized logistics of transatlantic sample transects which included and were not limited to updating cruise plans, monitoring the sampling progress in real time, trouble shooting and reporting any problems, participate in ship recruitment, and process completed cruise reports. Responded to request of information of monitoring transects by providing computer-generated maps. Replied to any of our constituent's request for information or troubleshooting. Maintained up-to date inventories within the program and its collaborators. Managed billing and payments of satellite transmissions and instrument upkeep. Maintained the programs web page maintenance.

Research Associate I, (NOAA)-Microbiology Laboratory, 7/2008 – 8/2010
Assisted in management, coordination, water quality sample processing, microbial plate and PCR molecular analysis, data management, report findings and execution of several projects for the Microbiology lab. Conducted water quality sampling and sediment sampling for the detection and enumeration of microbial contamination. Conducted qPCR-based microbial source tracking methods. Managed, processed and analyzed project data. Participated in various interagency collaborative research projects (USEPA, UM, FDEP, DOH, NOAA) that focused on the efficacy and development of microbial source tracking as well as surveillance to inform and improve water quality on recreational waters. The collaborative projects included: EPA 'STREAMS' (in collaboration with multi-

satellite projects and stakeholders to aid in science that will contribute to aquatic microbial field tracking; Miami-Dade, Department of Health's "Healthy Beaches" program; Marathon Key, "Little Venice" coastal construction technologies infrastructure improvement microbial source tracking assessment project. Prepared technical reports and communicated findings in the 2009 American Society for Microbiology, and the 2010 Ocean Sciences Meeting. Participated in oceanographic research cruises: Nancy Foster, (10 days); Walton Smith, (5 days).

Field Technician, Broward County Sea Turtle Conservation Program. 5-6/2008
Conducted beach surveillance for sea turtle tracks in order to locate and mark nests. Surveillance included: check existing nests for hatching and relocate nest as necessary, as well as record any pertinent field information such as false crawls and other observations.

Chemical analyst, Florida-Spectrum Environmental 4-6/2008
Analyze and report soil and water samples in the inorganic/wet chemistry department for; percent solids, total suspended solids, total dissolved solids, sulfides, MBAS, pH and Chemical oxygen demand, using EPA and SM methods. Upkeep and maintain the wet chemistry department. Assisted in other analysis assessments as needed and in other departments.

Teacher Assistant, Nova Southeastern University- Microbiology Division, 9/2007 – 5/ 2008
Facilitated the Microbiology laboratory. Prepared microbiology experiments for the undergraduate students. Instructed and trained students in specific laboratory techniques and laboratory protocols. Taught laboratory safety, protocol and microbiology methods such as: sample staining, enumeration, selective media, identification of unknown bacteria, RFLP, and transformation of plasmids. Coordinate and aid in management of a university Microbiology Laboratory. Prepared and conducted all necessary experiments for the day. Graded and provided comment to student's laboratory reports. Evaluated reports and prepared the class for the days experiment.

Program Intern, Arthur Marshall Foundation,5-7/2007
Traveled to different youth summer programs and educated them about the Everglades restoration efforts. Lead the rehabilitation of Palm Beach Science Museum wetlands exhibition. Educated the general public about wetlands and the protection of the Everglades Traveled the length of the Greater Everglades, viewed various restoration projects and learned about environmental management and mitigation processes. Managed and executed the rehabilitation/curation of a wetland in Palm Beach Science Museum to educate visitors about the ecology and protection of the Everglades. Assessed best management practices and delivered public comment to distinct audiences in government and public meetings.

Program Coordinator, AmeriCorps VISTA-Southwest Youth Services. 2/2006-2/2007
Supervised, designed and launched dynamic and positive strategies to sustain program development and implementation. Worked with diverse groups, established partnerships and collaborations with organizations to subsidized employees, saving the organization thousands of dollars. Managed and recruited staff and volunteers for our programs. Created and maintained the organizations website. Worked on advertisement products for the program using Photoshop and Publisher. Coordinated, developed and organized the annual Gala fundraiser for the program. Developed and implemented health curriculum to children ages 5-18. Mentored youth on leadership and tutored math and reading. Coordinated, planned, designed and launched dynamic and positive strategies to sustain program development and implementation for the non-profit. Built from the ground up a positive partnership that provided subsidized employees to implement our services within the communities we serve and saved the non-profit thousands of dollars. Supported the hiring of the subsidized employees and their management of up to four staff. Scheduled, monitored, communicated and coordinated staff's and volunteers (ranged from 2-20 volunteers) work-loads and hours. Developed, coached and implemented soccer and health curriculum for children ages 5-18. As well as traveled to the different communities to deliver our services. Managed communications through media relations, created and upkeeped the organizations website and worked on marketing and design using Photoshop. Coordinated, developed and organized the annual Gala fundraiser for our program.

Research Apprentice, Friday Harbor Laboratories-University of Washington. 9-12/2005
Investigated the physical-biological coupling of oceanographic processes and biota in the San Juan Archipelago. Conducted independent research with a final presentation and written report on the "Spatial and temporal variations of chlorophyll in the San Juan Islands, WA, in the Fall of 2005". Investigated the physical-biological coupling of oceanographic processes and biota in the San Juan Archipelago. I conducted independent research with a final oral presentation and written report: "Spatial and temporal variations of chlorophyll in the San Juan Islands, WA".

Oceanographic Cruises: Research Vessel Centennial, San Juan Archipelago, Washington. University of Washington (Sept. - Dec. 2005 six one-day cruises). Awarded the Apprenticeship to study the Pelagic Ecosystem Function in the San Juan Archipelago.

Teachers Assistant, Upward Bound Program, UNM. 6-8/2005
Instructed and tutored in Math and reading comprehension to high school students of various levels in an intensive summer session. Mentored students in academic achievement and college preparation. Prepared class, graded, disciplined the classroom, tutored, college prep counseled and chaperoned.

Research Assistant, Cancer Research and Treatment Center, University of New Mexico (UNM). 6– 8/2004
Conducted microbiology and genetic research on Myeloid cell regulation to better understand onset of Leukemia. Generated new and publishable data on the Homeobox Protein Hex and the regulation of the C-Kit Promoter. Implemented molecular techniques such as Northern and Western Blots, DNA purification, PCR amplification, cell transformations and transfections for the experiments. Presented final findings in the Minority Biomedical Research program's symposium.

Research Assistant, Mosquito Ecology and West Nile Virus Surveillance, UNM. 5/2003-5/2004
Operated and executed experimental field sampling of larval and adult mosquitoes. Collected blood samples from sentinel species. Processed data pertaining to an Environmental Health Project for the Environmental Health Department and Center for Disease Control (CDC) in the Rio Grande Valley, NM. Performed data analysis, public presentations and education, and wrote and co-authored our findings in a university research periodical.

Research Assistant, High-Performance Computational Biology Laboratory, UNM. 5/2002- 5/2004
Performed research on Phylogenetic tree reconstruction and molecular sequencing database tools. Programmed, using PERL, the reconstruction of complex evolutionary histories through computational modeling. Researched BLAST sequences and utilized sequences for tree reconstruction.

Lab Technician, Reproductive Ecology Laboratory, UNM. 5/2001-5/2002
Conducted botanical experiments of cross-pollinations. Microscopy analysis of pollen competition and gel electrophoresis. Collected and managed data.

Lab Technician, Yeast Genomic Research Laboratory, UNM. 9/2000 – 1/2001
Prepared molecular and chemical experiment setup, data collection and lab maintenance.

Curriculum Vitae (CV)
Jennifer T. Fullam

WORK HISTORY

March 2017- Present
State of New Mexico Environment Department, Santa Fe, New Mexico
Surface Water Quality Bureau
Standards, Planning and Reporting (SPR) Team Supervisor

- Serve as the coordinator for New Mexico's surface water quality standards which includes but is not limited to applying the procedures established for adopting changes to the surface water quality standards, petitioning for a hearing to the Water Quality Control Commission (WQCC), preparing and advertising public notices, providing written and oral testimony for a hearing before the WQCC, preparing for cross examination, understanding and applying hearing guidelines, assisting with the development of post-hearing submittals and filing rule changes to State Records and Archives in accordance with 20.1.24.10 NMAC.
- Maintain knowledge of State and Federal statutory requirements that affect surface water quality standards and standards development.
- Coordinate with the United States Environmental Protection Agency (EPA) on actions pertaining to the State's Water Quality Standards and the Federal Clean Water Act. This includes submitting surface water quality standards (new and revised) to EPA Region 6 for review and action (approval or disapproval).
- Conduct and review use attainability analysis and hydrology protocol surveys which propose to revise, remove or add segment specific water quality standards to 20.6.4 NMAC.
- Responsible for the daily management and oversight of work conducted by the Standards, Planning and Reporting Team which oversees the implementation of the Bureau's Quality Assurance requirements, technical and educational outreach activities and development of regulatory and rulemaking actions.
- Review and revise the Water Quality Management Plan and Continuing Planning Process as required under the Clean Water Act.
- Coordinate and provide guidance and appropriate training for staff on program procedures.
- Ensure that all written work products from the SPR team are of high quality, reflect the professionalism of the Bureau and Department, and support New Mexico Environment Department's (NMED's) role as the lead agency for surface water quality protection in New Mexico.
- Conduct employee performance reviews of staff under the SPR Team.
- Conduct recruitment, disciplinary and hiring actions in accordance with State Personnel and Human Resources policies and procedures.
- Conduct technical and educational public outreach for proposed rulemaking actions to the surface water quality standards. This includes coordinating public notices through the website, listserv, newspapers, media releases and public meetings, providing technical and regulatory information from members of the public and recognized stakeholders.
- Collaborate and facilitate dialogue with Tribes on water quality standard issues. Reviewing Tribal Water Quality Standards and providing input, as applicable.
- Participate on national issues pertaining to water quality standards such as variances, proposed rules on Waters of the United States (WOTUS) and proposed guidelines for standards.

- Facilitate positive working relationships with other state and federal agencies, stakeholders and cooperators involved in surface water quality standards activities.
- Oversee the development of quality assurance guidance documents such as the Quality Management Plan (QMP) and Quality Assurance Project Plan (QAPP), Standard Operating Procedures and Field Sampling Plans
- Oversee the Quality Assurance Manager responsible for quality assurance activities pertaining to surface water data collection both within the Bureau and with outside entities seeking to submit water quality data for assessment purposes.

March 2014-March 2017
State of New Mexico Environment Department, Santa Fe, New Mexico
Petroleum Storage Tank Bureau
Compliance Assistance Coordinator/Environmental Scientist Specialist-A

- Responsible for the implementation and daily management of the Delivery Prohibition enforcement program.
- Development and implementation of strategic data management processes.
- Create and maintain tracking tools to assist in data collection and case management.
- Effectively track specific violations and enforcement actions for approximately 300 new cases (1300 individual violations) per year in a consistent, objective and timely manner.
- Compile information, through active data mining within these internal tracking applications, the Department's database and facility owner's files, to be able to provide compliance and enforcement statistics to meet the Federal Environmental Protection Agency's mandates and State reporting requirements.
- Effectively communicate, both verbally as well as in writing to various audiences including peers, management, regulated community and legal counsel.
- Review and clarify observations documented by the inspectors in the field and prepare a legally defensible enforcement case.
- In the event enforcement actions are appealed to the Secretary, assist in preparing testimony for a hearing.
- Apply knowledge of Federal (specifically 40 CFR §280) and State regulations (20.5 NMAC) with technical and legal writing skills experience to draft and edit enforcement documents.
- Involved in the development of new regulations to meet 40 CFR § 280.
- Regularly coordinate with the Bureau Chief and Program Managers within the Bureau
- Seek input and collaborate with staff from other Bureaus as it applies towards implementation of State and Federal Regulations.
- Network with other States and Tribes on processes and regulatory implementation.
- Provide written and verbal notification to facility owners of upcoming enforcement actions and offer assistance on actions required to obtain compliance.
- Maintain open communication with inspectors to assemble the chronological histories of ongoing outreach with owners and operators facing enforcement actions.
- Gather, collaborate and discuss ongoing applicability of the regulations and disseminate this information to inspectors to ensure continuity within the delivery prohibition program.
- Manage and delegate tasks to technical and administrative staff assisting with the delivery prohibition program.
- Serve as a Bureau-wide web author, updating the Bureau's website as necessary using cloud-based programs and Adobe Contribute.
- Assist with additional projects such as with the development of standard operating procedures for the Prevention Inspection Program and database development and management.

- Assist the Bureau's Prevention Inspection Program by contributing to the ongoing program development and conducting compliance inspections at facilities around the State; which requires knowledge of the technical aspects of both UST and AST systems.

July 2007- March 2014
State of New Mexico Environment Department, Santa Fe, New Mexico
Ground Water Quality Bureau
Pollution Prevention Section
Environmental Scientist-O

- Ensuring the protection of ground water throughout the State of New Mexico by regulatory management for over 70 ground water discharge permits. The diversity of sites range from large federal industrial facilities, large domestic wastewater treatment plants and small septic tank/leachfield systems.
- Administering regulatory functions as they pertain to permitted and un-permitted facilities. Actions include but are not limited to management of records subject to the public information act, data entry of facility monitoring reports, database management for assigned facilities, ground water and wastewater sampling, response to unauthorized releases and enforcement actions.
- Successfully worked with Permittees and the general public in achieving voluntary compliance through non-enforcement actions. Refined experience in assessing potentially volatile situations and diffusing with effective and clear communication. Ground water protection has also been achieved through promoting cost-effective and source control mechanisms to reduce potential contaminants from reaching ground water. Discharge Permits are designed to address protection of ground water and human health while working towards long-term sustainability of small businesses.
- Maintain and continuously enhancing an already robust understanding of Federal Regulations such as the Clean Water Act, Resource Conservation and Recovery Act, Biosolids Standards the State of New Mexico's Clean Water act, Water Quality Control Commission regulations which have been used for evaluating site specific conditions and development of priority actions.
- Ongoing coordination and collaboration with Tribal entities including but not limited to compiling the annual Tribal Collaboration report for the Ground Water Quality Bureau, participation in the Annual Tribal Summit, planning and serving as a mentor and instructor for the annual Tribal Youth Environmental Science Camp.
- Effectively facilitate dialogue among a diverse group of individuals, with varying backgrounds and expertise, in order to develop and strategize a productive approach in resolving complex issues. Ongoing work includes facilitation of discussions between the GWQB, Hazardous Waste Bureau, Department of Energy Oversight Bureau, Surface Water Bureau and Los Alamos National Laboratory in order to maintain regulatory compliance and cross-departmental communications for the management of the facility.
- Provide assistance to legal counsel on litigation cases. These have included involvement with a federal negotiation case with the Hazardous Waste Bureau and Los Alamos National Laboratory as well as a Chapter 11 Bankruptcy Lawsuit against Mark IV Industries for the continued remediation of a contaminated ground water site in east Albuquerque which resulted in an Order of Consent.
- Actively participated and spearheaded discussions in various workgroups within the section to enhance the regulatory process and streamline the efficiency of the program to ensure protection of the State's resources as well as promoting economic development for rural communities. These have included the development of Best Management Practices for RV

Parks, Tribal Consultation Policy, Grease Trap Management Practices and Domestic Wastewater Discharge Permit Template.

- Development of sound investigative skills to truth information submitted to NMED through remote sensing technologies, ground-truthing or through various technological resources.
- Development of internal mechanisms and processes to effectively manage and increase efficiency in the management of regulatory processes.
- Serve as a Quality Assurance Manager (QAM) for the Radiation Control Bureau's Quality Assurance Project Plan (QAPP) ensuring all data collection activities are collected in a consistent and defensible manner.
- Applying federal laws and regulations, effective approaches to gain voluntary compliance and general management tools and resources to increase efficacy in job performance.
- General program administration functions to include preparing timesheets, vehicle logs, travel requests, along with submitting quarterly and annual reports to management as assigned.

April 2003-July 2007
Pueblo of Tesuque, Santa Fe, New Mexico
Environment Department
Biologist/Director

- Responsible for overseeing the management and execution of activities associated with the protection of environmental resources. The program included surface water quality, water rights, ground water, planning and development, forest restoration, wildlife habitat management, wildland fire response, emergency response as it pertained to the community and potential environmental impacts, general community assistance, education (pre-k through college) and outreach.
- Reported directly to the Tribal Administrator, Governor and Tribal Council on the department's activities.
- Supervised up to 11 individuals on routine and special projects undertaken by the department which included but were not limited to surface water, forest restoration, WUI fire suppression projects, wildlife surveys and habitat assessments, economic development projects, Aamodt water rights settlement committee, community activities, educational outreach (kindergarten through college), assistance with organic farm program, community assistance as requested.
- Worked and collaborated with numerous federal, state and local government agencies such as the Environmental Protection Agency (EPA), Army Corp of Engineers, Bureau of Indian Affairs, Indian Health Services, State of New Mexico Environment Department, Santa Fe County, City of Santa Fe, and various Tribal governments.
- Responsible for writing and managing over $1,000,000.00 in State and Federal grants through the U.S. Forest Service, Administration for Native Americans, Environmental Protection Agency, U.S. Fish and Wildlife, and New Mexico Clean and Beautiful, submitting quarterly and annual reports on a regular basis as well as auditing expenses to ensure allocation of funds was completed and reported appropriately.
- Served as a member on the Tribal Emergency Planning committee and Land Use Planning Committee, Board member of Inter-Tribal Bison Cooperative, Inter-Tribal Resource Advisory Committee, and Water Rights Committee and as a voting member for EPA Region 6 Regional Tribal Operations Committee.
- Responsible for writing and implementing Quality Assurance documents and the department's annual Quality Management Plan, Quality Assurance Project Plans for water quality monitoring, GIS/GPS, and the Elk Demonstration Project.
- Prepared and conducted the triennial review of Tesuque Pueblo's Water Quality Standards.

- Actively engaged with community members to better understand the needs and priorities of the Tribe in order to effectively target financial mechanisms and internal resources which could be utilized to achieve long-term goals.
- Designed and implemented a multi-parameter study to assess movement and habitat utilization of elk herds within lower pinon/juniper forests of Tesuque Pueblo. Field work consisted of off-road driving and heavy lifting of equipment and supplies, remote sensing and data management.
- Use of various field equipment for work pertaining to water quality monitoring (ground and surface water), riparian ecosystem rehabilitation projects, wildlife habitat and behavior. Data correction and management of files.

January 2002- April 2003
Los Alamos National Laboratory, Los Alamos, New Mexico
Contaminant Monitoring Team
Graduate Research Assistant

- Provided technical research support for the Ecology Group including compiling, writing and editing portions of the published technical reports as well as the annual Environmental Surveillance Report.
- Collected and processed field samples from remote areas with a wide array of equipment.
- Analyzed data in MS Excel for risk assessment of contaminant such as high explosives, radionuclide, polychlorinated biphenyls (PCBs), dioxins, furans and pesticides such as DDT.
- Actively participated in a cooperative group with the New Mexico Environment Department, Los Alamos County and Tribal entities to designed and implement a contaminant mobilization study in the Rio Grande to assess the possible PCB risk levels that may be associated with LANL's historic PCB releases and the potential of mobilization after the Cerro Grande fire using semi-permeable membrane devises (LANL Publication Gonzales and Montoya 2005).

EDUCATION

2002-2008 New Mexico Highlands University, Las Vegas, New Mexico
Master of Science (May 2008)

- Thesis on the unique characteristics of elk movement and habitat utilization within the pinon/juniper forests of Tesuque Pueblo
- Other studies included toxicology, environmental assessment, surface hydrology, dendrology, wildlife habitat assessment and research methods
- Research on the use of semi-permeable membrane devices to assess the effects of pulse flooding events on PCB concentrations in the Rio Grande river near Los Alamos
- Cumulative GPA 4.0

1999-2002 University of New Mexico, Albuquerque, New Mexico
Bachelor of Science, Biology with minor in Geography (May 2001)

- Studies in riparian ecology, conservation biology, animal behavior, zoology and physiology.
- Formal studies and research in Geographical Information Systems.
- Graduated Cum Laude.

1994-1997 Northern New Mexico Community College, Espanola, New Mexico
Associate of Science in Science

- Graduated with Honors

APPLICABLE CERTIFICATIONS/TRAININGS

Quality Assurance

- EPA QMP/QAPP Training, Santa Fe, NM(Certificate)
- EPA Training to Quality Assurance Management, Data Quality Objectives, Santa Fe, NM (Certificate)

Water and Wastewater

- EPA Tribal Water Quality Standards Academy Intermediate level, Kalispell Montana (Certificate)
- EPA Water Quality Standards Academy, Washington D.C. (Certificate)
- NMSU WTAP Advanced Secondary Treatment (certificate)
- National Onsite Wastewater Recycling Association A to Z Course (certificate)
- YSI Training on 6920 Multi-parameter water quality monitoring unit, Yellow Springs, OH
- Stream Habitat Assessment Training, Taos Pueblo, NM
- Biological Assessment Training, Santa Ana Pueblo, NM
- Fundamentals of Drilling (certificate)

Emergency Response/Safety

- FEMA National Incident management System (IS-700) Tesuque Pueblo, NM (Certificate)
- BIA Northern Pueblos Wildland Firefighter Training (S-110, S-133, S-134, I-100, L-180, S-130, S-190), Ohkay Owingeh, NM (Red Card Certification)
- Pandemic Flu, Train the Trainer, Albuquerque, NM (Certificate)
- Zoonotic Disease Training, Los Alamos National Laboratory, Los Alamos, NM
- HAZWOPER certified (2007-2017)
- AHMP Essentials of Hazardous Materials Management, Albuquerque, NM (Certificate)
- U.S. Dept of Transportation Awareness for Initial Response to Haz-Mat Incidents Course (Certificate)
- National Safety Council Defensive Driving Course (Certificate)
- Swiftwater Rescue for River Professionals Training; Level II NFPA-compliant 1670 “Operations” (Certificate)

Inspection and Enforcement

- Western States Project NMED Environmental Enforcement Procedure Training (certificate)
- UST Inspector Training (Certificate)
- State of NM HR and OGC Inspector Training (certificate)

Lawmaking and Regulations

- State of NM State Rulemaking Training
- State of NM Records and Information Management Training

Management

- EdX Online Audit Course Best Practices for Project Management Success

PUBLICATIONS/PROFESSIONAL ORGANIZATIONS

- Gonzales, G. and Montoya, J., 2005. Polychlorinated biphenyls (PCBs) in the Rio Grande Sampled Using Semi-permeable Membrane Devices. LA-14200.
- Fullam, J., 2008. Elk Habitat Utilization Within Lower Pinon Juniper Forests of Tesuque Pueblo, New Mexico Highlands University Graduate Thesis.
- Golden Key National Honor Society (2001-Present)
- Native American Fish and Wildlife Society (2003-2007)
- The Quivera Coalition (2003-2007)
- Ecological Society of America (2016)
- The Wildlife Society (2011-2017)
- Society of Environmental Toxicology and Applied Chemistry (2002-2007; 2017-Present)

TITLE 20 ENVIRONMENTAL PROTECTION
CHAPTER 6 WATER QUALITY
PART 4 STANDARDS FOR INTERSTATE AND INTRASTATE SURFACE WATERS

20.6.4.1 ISSUING AGENCY: Water Quality Control commission.
[20.6.4.1 NMAC - Rp 20 NMAC 6.1.1001, 10/12/2000]

20.6.4.2 SCOPE: Except as otherwise provided by statute or regulation of the water quality control commission, this part governs all surface waters of the state of New Mexico, which are subject to the New Mexico Water Quality Act, Sections 74-6-1 through 74-6-17 NMSA 1978.
[20.6.4.2 NMAC - Rp 20 NMAC 6.1.1002, 10/12/2000; A, 5/23/2005]

20.6.4.3 STATUTORY AUTHORITY: This part is adopted by the water quality control commission pursuant to Subsection C of Section 74-6-4 NMSA 1978.
[20.6.4.3 NMAC - Rp 20 NMAC 6.1.1003, 10/12/2000]

20.6.4.4 DURATION: Permanent.
[20.6.4.4 NMAC - Rp 20 NMAC 6.1.1004, 10/12/2000]

20.6.4.5 EFFECTIVE DATE: October 12, 2000, unless a later date is indicated in the history note at the end of a section.
[20.6.4.5 NMAC - Rp 20 NMAC 6.1.1005, 10/12/2000]

20.6.4.6 OBJECTIVE:

A. The purpose of this part is to establish water quality standards that consist of the designated use or uses of surface waters of the state, the water quality criteria necessary to protect the use or uses and an antidegradation policy.

B. The state of New Mexico is required under the New Mexico Water Quality Act (Subsection C of Section 74-6-4 NMSA 1978) and the federal Clean Water Act, as amended (33 U.S.C. Section 1251 *et seq*.) to adopt water quality standards that protect the public health or welfare, enhance the quality of water and are consistent with and serve the purposes of the New Mexico Water Quality Act and the federal Clean Water Act. It is the objective of the federal Clean Water Act to restore and maintain the chemical, physical and biological integrity of the nation's waters, including those in New Mexico. This part is consistent with Section 101(a)(2) of the federal Clean Water Act, which declares that it is the national goal that wherever attainable, an interim goal of water quality that provides for the protection and propagation of fish, shellfish and wildlife and provides for recreation in and on the water be achieved by July 1, 1983. Agricultural, municipal, domestic and industrial water supply are other essential uses of New Mexico's surface water; however, water contaminants resulting from these activities will not be permitted to lower the quality of surface waters of the state below that required for protection and propagation of fish, shellfish and wildlife and recreation in and on the water, where practicable.

C. Pursuant to Subsection A of Section 74-6-12 NMSA 1978, this part does not grant to the water quality control commission or to any other entity the power to take away or modify property rights in water.

D. These surface water quality standards serve to address the inherent threats to water quality due to climate change.

[20.6.4.6 NMAC - Rp 20 NMAC 6.1.1006, 10/12/2000; A, 5/23/2005; A, XX/XX/XXXX]

20.6.4.7 DEFINITIONS: Terms defined in the New Mexico Water Quality Act, but not defined in this part will have the meaning given in the Water Quality Act.

A. Terms beginning with numerals or the letter "A," and abbreviations for units.

(1) "4Q3" means the critical low flow as determined by the minimum average flow over four consecutive days that occurs with a frequency of once in three years.

[~~(1)~~](2) "4T3 temperature" means the temperature not to be exceeded for four or more consecutive hours in a 24-hour period on more than three consecutive days.

[~~(2)~~](3) "6T3 temperature" means the temperature not to be exceeded for six or more consecutive hours in a 24-hour period on more than three consecutive days.

[~~(3)~~](4) Abbreviations used to indicate units are defined as follows:

(a) **"cfu/100 mL"** means colony-forming units per 100 milliliters; the results for *E. coli* may be reported as either colony forming units (CFU) or the most probable number (MPN), depending on the analytical method used;

(b) **"cfs"** means cubic feet per second;

(c) **"µg/L"** means micrograms per liter, equivalent to parts per billion when the specific gravity of the solution equals 1.0;

(d) **"µS/cm"** means microsiemens per centimeter; one µS/cm is equal to one µmho/cm;

(e) **"mg/kg"** means milligrams per kilogram, equivalent to parts per million;

(f) **"mg/L"** means milligrams per liter, equivalent to parts per million when the specific gravity of the solution equals 1.0;

(g) **"MPN/100 mL"** means most probable number per 100 milliliters; the results for *E. coli* may be reported as either CFU or MPN, depending on the analytical method used;

(h) **"NTU"** means nephelometric turbidity unit;

(i) **"pCi/L"** means picocuries per liter;

(j) **"pH"** means the measure of the acidity or alkalinity and is expressed in standard units (su).

[~~(4)~~](5) **"Acute toxicity"** means toxicity involving a stimulus severe enough to induce a response in 96 hours of exposure or less. Acute toxicity is not always measured in terms of lethality, but may include other toxic effects that occur within a short time period.

[~~(5)~~](6) "**Adjusted gross alpha**" means the total radioactivity due to alpha particle emission as inferred from measurements on a dry sample, including radium-226, but excluding radon-222 and uranium. Also excluded are source, special nuclear and by-product material as defined by the Atomic Energy Act of 1954.

[~~(6)~~](7) **"Aquatic life"** means any plant or animal life that uses surface water as primary habitat for at least a portion of its life cycle, but does not include avian or mammalian species.

[~~(7)~~](8) "**Attainable Use"** means a use that is achievable by the imposition of effluent limits required under sections 301(b) and 306 of the federal Clean Water Act and implementation of cost-effective and reasonable best management practices for nonpoint source control. An attainable use may or may not be as stringent as the designated use.

B. Terms beginning with the letter "B".

(1) **"Baseflow"** refers to the sustained flow volume of a stream or river. In natural systems, baseflow is comprised from regional groundwater inflow and local shallow subsurface inflow that is temporarily stored in the watershed during snowmelt and rain events and slowly released to the stream or river over time. In effluent dominated systems, baseflow is comprised predominantly from effluent with limited subsurface contributions. Baseflow in both scenarios is critical for sustaining flow in streams and rivers over seasonal and longer timeframes.

[~~(1)~~](2) "Best management practices" or "BMPs":

(a) for national pollutant discharge elimination system (NPDES) permitting purposes means schedules of activities, prohibitions of practices, maintenance procedures and other management practices to prevent or reduce the pollution of "waters of the United States;" BMPs also include treatment requirements, operating procedures and practices to control plant site runoff, spillage or leaks, sludge or waste disposal or drainage from raw material storage; or

(b) for nonpoint source pollution control purposes means methods, measures or practices selected by an agency to meet its nonpoint source control needs; BMPs include but are not limited to structural and nonstructural controls and operation and maintenance procedures; BMPS can be applied before, during and after pollution-producing activities to reduce or eliminate the introduction of pollutants into receiving waters; BMPs for nonpoint source pollution control purposes shall not be mandatory except as required by state or federal law.

[~~(2)~~](3) **"Bioaccumulation"** refers to the uptake and retention of a substance by an organism from its surrounding medium and food.

[~~(3)~~](4) **"Bioaccumulation factor"** is the ratio of a substance's concentration in tissue versus its concentration in ambient water, in situations where the organism and the food chain are exposed.

[~~(4)~~](5) **"Biomonitoring"** means the use of living organisms to test the suitability of effluents for discharge into receiving waters or to test the quality of surface waters of the state.

C. Terms beginning with the letter "C".

(1) "**CAS number**" means an assigned number by chemical abstract service (CAS) to identify a substance. CAS numbers index information published in chemical abstracts by the American chemical society.

(2) **"Chronic toxicity"** means toxicity involving a stimulus that lingers or continues for a relatively long period relative to the life span of an organism. Chronic effects include, but are not limited to, lethality, growth impairment, behavioral modifications, disease and reduced reproduction.

(3) **"Classified water of the state"** means a surface water of the state, or reach of a surface water of the state, for which the commission has adopted a segment description and has designated a use or uses and applicable water quality criteria in 20.6.4.101 through 20.6.4.899 NMAC.

(4) "Climate change" refers to any significant change in the measures of climate lasting for an extended period of time, typically decades or longer, and includes major changes in temperature, precipitation, wind patterns or other weather-related effects. Climate change may be due to natural processes or human-caused changes of the atmosphere, or a combination of the two.

[~~(4)~~](5) **"Closed basin"** is a basin where topography prevents the surface outflow of water and water escapes by evapotranspiration or percolation.

[~~(5)~~](6) **"Coldwater"** in reference to an aquatic life use means a surface water of the state where the water temperature and other characteristics are suitable for the support or propagation or both of coldwater aquatic life.

(7) "Contaminants of emerging concern" or "CECs" refer to water contaminants including, but not limited to, pharmaceuticals and personal care products that may cause significant ecological or human health effects at low concentrations. CECs are generally chemical compounds that, although suspected to potentially have impacts, may not have regulatory standards, and the concentrations to which negative impacts are observed have not been fully studied.

[~~(6)~~](8) **"Coolwater"** in reference to an aquatic life use means the water temperature and other characteristics are suitable for the support or propagation of aquatic life whose physiological tolerances are intermediate between and may overlap those of warm and coldwater aquatic life.

[~~(7)~~](9) **"Commission"** means the New Mexico water quality control commission.

[~~(8)~~](10) **"Criteria"** are elements of state water quality standards, expressed as constituent concentrations, levels or narrative statements, representing a quality of water that supports a use. When criteria are met, water quality will protect the designated use.

D. Terms beginning with the letter "D".

(1) "**DDT and derivatives**" means 4,4'-DDT (CAS number 50293), 4,4'-DDE (CAS number 72559) and 4,4'-DDD (CAS number 72548).

(2) **"Department"** means the New Mexico environment department.

(3) **"Designated use"** means a use specified in 20.6.4.97 through 20.6.4.899 NMAC for a surface water of the state whether or not it is being attained.

(4) **"Dissolved"** refers to the fraction of a constituent of a water sample that passes through a 0.45-micrometer pore-size filter. The "dissolved" fraction is also termed "filterable residue."

(5) **"Domestic water supply"** means a surface water of the state that could be used for drinking or culinary purposes after disinfection.

E. Terms beginning with the letter "E".

(1) **"E. coli"** means the bacteria Escherichia coli.

(2) "Effluent dominated" refers to a water that has, over a 12-month average, more than three-quarters of its baseflow attributed to discharges from a permitted effluent discharge. Waters that are effluent dominated are of significant value by providing aquatic life and wildlife habitat.

[~~(2)~~](3) **"Ephemeral"** when used to describe a surface water of the state means the water body contains water briefly only in direct response to precipitation; its bed is always above the water table of the adjacent region.

[~~(3)~~](4) **"Existing use"** means a use actually attained in a surface water of the state on or after November 28, 1975, whether or not it is a designated use.

F. Terms beginning with the letter "F".

(1) **"Fish culture"** means production of coldwater or warmwater fishes in a hatchery or rearing station.

(2) "**Fish early life stages**" means the egg and larval stages of development of fish ending when the fish has its full complement of fin rays and loses larval characteristics.

G. Terms beginning with the letter "G". [RESERVED]

H. Terms beginning with the letter "H".

(1) "Hardness" means the measure of dissolved calcium and magnesium salts in water expressed as dissolved calcium carbonate (CaCO3) unless otherwise noted.

(2) "Harmonic mean flow" is the number of daily flow measurements divided by the sum of the reciprocals of the flows; that is, it is the reciprocal of the arithmetic mean of reciprocal daily flow measurements consistent with the equations in Paragraph (1) of Subsection B of 20.6.4.11 NMAC.

[~~(1)~~](3) "High quality coldwater" in reference to an aquatic life use means a perennial surface water of the state in a minimally disturbed condition with considerable aesthetic value and superior coldwater aquatic life habitat. A surface water of the state to be so categorized must have water quality, stream bed characteristics and other attributes of habitat sufficient to protect and maintain a propagating coldwater aquatic life population.

[~~(2)~~](4) "Human health-organism only" means the health of humans who ingest fish or other aquatic organisms from waters that contain pollutants.

I. Terms beginning with the letter "I".

(1) "Industrial water supply" means the use or storage of water by a facility for process operations unless the water is supplied by a public water system. Industrial water supply does not include irrigation or other agricultural uses.

(2) "Intermittent" when used to describe a surface water of the state means the water body contains water for extended periods only at certain times of the year, such as when it receives seasonal flow from springs or melting snow.

(3) "Interstate waters" means all surface waters of the state that cross or form a part of the border between states.

(4) "Intrastate waters" means all surface waters of the state that are not interstate waters.

(5) "Irrigation" means application of water to land areas to supply the water needs of beneficial plants.

(6) "Irrigation storage" means storage of water to supply the needs of beneficial plants.

J. Terms beginning with the letter "J". [RESERVED]

K. Terms beginning with the letter "K". [RESERVED]

L. Terms beginning with the letter "L".

(1) "LC-50" means the concentration of a substance that is lethal to fifty percent of the test organisms within a defined time period. The length of the time period, which may vary from 24 hours to one week or more, depends on the test method selected to yield the information desired.

(2) "**Limited aquatic life**" as a designated use, means the surface water is capable of supporting only a limited community of aquatic life. This subcategory includes surface waters that support aquatic species selectively adapted to take advantage of naturally occurring rapid environmental changes, [~~ephemeral or intermittent water,~~]low-flow, high turbidity, fluctuating temperature, low dissolved oxygen content or unique chemical characteristics.

(3) "Livestock watering" means the use of a surface water of the state as a supply of water for consumption by livestock.

M. Terms beginning with the letter "M".

(1) "Marginal coldwater" in reference to an aquatic life use means that natural [~~intermittent or low flows, or other natural~~]habitat conditions severely limit maintenance of a coldwater aquatic life population during at least some portion of the year or historical data indicate that the temperature [~~in~~] of the surface water of the state may exceed that which could continually support aquatic life adapted to coldwater[~~25°C (77°F)~~].

(2) "Marginal warmwater" in reference to an aquatic life use means natural intermittent or low flow or other natural habitat conditions severely limit the ability of the surface water of the state to sustain a natural aquatic life population on a continuous annual basis; or historical data indicate that natural water temperature routinely exceeds 32.2°C (90°F).

(3) "Maximum temperature" means the instantaneous temperature not to be exceeded at any time.

(4) "Minimum quantification level" means the minimum quantification level for a constituent determined by official published documents of the United States environmental protection agency.

N. Terms beginning with the letter "N".

(1) "Natural background" means that portion of a pollutant load in a surface water resulting only from non-anthropogenic sources. Natural background does not include impacts resulting from historic or existing human activities.

(2) "Natural causes" means those causal agents that would affect water quality and the effect is not caused by human activity but is due to naturally occurring conditions.

(3) "Nonpoint source" means any source of pollutants not regulated as a point source that degrades the quality or adversely affects the biological, chemical or physical integrity of surface waters of the state.

O. Terms beginning with the letter "O".

(1) "**Organoleptic**" means the capability to produce a detectable sensory stimulus such as odor or taste.

(2) "Oversight agency" means a state or federal agency, such as the United States department of agriculture forest service, that is responsible for land use or water quality management decisions affecting nonpoint source discharges where an outstanding national resource water is located.

P. Terms beginning with the letter "P".

(1) "**Playa**" means a shallow closed basin lake typically found in the high plains and deserts.

(2) "Perennial" when used to describe a surface water of the state means the water body typically contains water throughout the year and rarely experiences dry periods.

(3) "Persistent toxic pollutants" means pollutants, generally organic, that are resistant to environmental degradation through chemical, biological and photolytic processes and can bioaccumulate in organisms, causing adverse impacts on human health and aquatic life.

[~~(3)~~](4) "Point source" means any discernible, confined and discrete conveyance from which pollutants are or may be discharged into a surface water of the state, but does not include return flows from irrigated agriculture.

[~~(4)~~](5) "**Practicable**" means that which may be done, practiced or accomplished; that which is performable, feasible, possible.

[~~(5)~~](6) "Primary contact" means any recreational or other water use in which there is prolonged and intimate human contact with the water, such as swimming and water skiing, involving considerable risk of ingesting water in quantities sufficient to pose a significant health hazard. Primary contact also means any use of surface waters of the state for cultural, religious or ceremonial purposes in which there is intimate human contact with the water, including but not limited to ingestion or immersion, that could pose a significant health hazard.

[~~(6)~~](7) "Public water supply" means the use or storage of water to supply a public water system as defined by New Mexico's Drinking Water Regulations, 20.7.10 NMAC. Water provided by a public water system may need to undergo treatment to achieve drinking water quality.

Q. Terms beginning with the letter "Q". [RESERVED]

R. Terms beginning with the letter "R". [RESERVED]

S. Terms beginning with the letter "S".

(1) "Secondary contact" means any recreational or other water use in which human contact with the water may occur and in which the probability of ingesting appreciable quantities of water is minimal, such as fishing, wading, commercial and recreational boating and any limited seasonal contact.

(2) "Segment" means a classified water of the state described in 20.6.4.101 through 20.6.4.899 NMAC. The water within a segment should have the same uses, similar hydrologic characteristics or flow regimes, and natural physical, chemical and biological characteristics and exhibit similar reactions to external stresses, such as the discharge of pollutants.

(3) "**Specific conductance**" is a measure of the ability of a water solution to conduct an electrical current.

(4) "State" means the state of New Mexico.

(5) "Surface water(s) of the state"

(i) means all surface waters situated wholly or partly within or bordering upon the state, including the following:

(1) lakes[~~,~~];
(2) rivers[~~,~~];
(3) streams (including intermittent and ephemeral streams) [~~,~~];
(4) mudflats[~~,~~];
(5) sandflats[~~,~~];
(6) wetlands[~~,~~];
(7) sloughs[~~,~~];
(8) prairie potholes [~~,~~];
(9) wet meadows[~~,~~];

(10) playa lakes[,];
(11) reservoirs[,]; [~~or~~]and
(12) natural ponds.

(ii) [~~Surface waters of the state~~]also means all tributaries of such waters, including adjacent wetlands, any manmade bodies of water that were originally created in surface waters of the state or resulted in the impoundment of surface waters of the state, and any "waters of the United States" as defined under the Clean Water Act that are not included in the preceding description.

(iii) [~~Surface waters of the state~~]does not include private waters that do not combine with other surface or subsurface water or any water under tribal regulatory jurisdiction pursuant to Section 518 of the Clean Water Act. Waste treatment systems, including treatment ponds or lagoons designed and actively used to meet requirements of the Clean Water Act (other than cooling ponds as defined in 40 CFR Part 423.11(m) that also meet the criteria of this definition), are not surface waters of the state, unless they were originally created in surface waters of the state or resulted in the impoundment of surface waters of the state.

T. Terms beginning with the letter "T".

(1) "TDS" means total dissolved solids, also termed "total filterable residue."

(2) "Toxic pollutant" means those pollutants, or combination of pollutants, including disease-causing agents, that after discharge and upon exposure, ingestion, inhalation or assimilation into any organism, either directly from the environment or indirectly by ingestion through food chains, will cause death, shortened life spans, disease, adverse behavioral changes, reproductive or physiological impairment or physical deformations in such organisms or their offspring.

(3) "**Tributary**" means a perennial, intermittent or ephemeral waterbody that flows into a larger waterbody, and includes a tributary of a tributary.

(4) "Turbidity" is an expression of the optical property in water that causes incident light to be scattered or absorbed rather than transmitted in straight lines.

U. Terms beginning with the letter "U". [~~RESERVED~~]

(1) "Unclassified waters of the state" means those surface waters of the state not identified in 20.6.4.101 through 20.6.4.899 NMAC. An unclassified surface water of the state is presumed to support the uses specified in Section 101(a)(2) of the federal Clean Water Act. As such, it is subject to 20.6.4.98 NMAC if nonperennial, or 20.6.4.99 NMAC if perennial. The commission may include an ephemeral unclassified surface water of the state in 20.6.4.97 NMAC only if a use attainability analysis demonstrates, pursuant to 20.6.4.15 NMAC, that attainment of Section 101(a)(2) uses is not feasible.

V. Terms beginning with the letter "V". [RESERVED]

W. Terms beginning with the letter "W".

(1) "Warmwater" with reference to an aquatic life use means that water temperature and other characteristics are suitable for the support or propagation or both of warmwater aquatic life.

(2) "Water contaminant" means any substance that could alter if discharged or spilled the physical, chemical, biological or radiological qualities of water. "Water contaminant" does not mean source, special nuclear or by-product material as defined by the Atomic Energy Act of 1954, but may include all other radioactive materials, including but not limited to radium and accelerator-produced isotopes.

(3) "Water pollutant" means a water contaminant in such quantity and of such duration as may with reasonable probability injure human health, animal or plant life or property, or to unreasonably interfere with the public welfare or the use of property.

(4) "Wetlands" means those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions in New Mexico. Wetlands that are constructed outside of a surface water of the state for the purpose of providing wastewater treatment and that do not impound a surface water of the state are not included in this definition.

(5) "Wildlife habitat" means a surface water of the state used by plants and animals not considered as pathogens, vectors for pathogens or intermediate hosts for pathogens for humans or domesticated livestock and plants.

X. Terms beginning with the letters "X" through "Z". [RESERVED]

[20.6.4.7 NMAC - Rp 20 NMAC 6.1.1007, 10/12/2000; A, 7/19/2001; A, 5/23/2005; A, 7/17/2005; A, 8/1/2007; A, 12/1/2010; A, 1/14/2011; A, 3/2/2017; A, XX/XX/XXXX]

20.6.4.8 ANTIDEGRADATION POLICY AND IMPLEMENTATION PLAN:

A. Antidegradation Policy: This antidegradation policy applies to all surface waters of the state.

(1) Existing [~~instream water~~]uses, as defined in Paragraph (4) of Subsection E of 20.6.4.7 NMAC, and the level of water quality necessary to protect the existing uses shall be maintained and protected in all surface waters of the state.

(2) Where the quality of a surface water of the state exceeds the established levels necessary to support the propagation of fish, shellfish, and wildlife, and recreation in and on the water, that level of quality shall be maintained and protected unless the commission finds, after full satisfaction of the intergovernmental coordination and public participation provisions of the state's continuing planning process, that allowing lower water quality is necessary to accommodate important economic and social development in the area in which the water is located. In allowing such degradation or lower water quality, the state shall assure water quality adequate to protect existing uses fully. Further, the state shall assure that there shall be achieved the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable BMPs for nonpoint source control. Additionally, the state shall encourage the use of watershed planning as a further means to protect surface waters of the state.

(3) No degradation shall be allowed in waters designated by the commission as outstanding national resource waters (ONRWs), except as provided in Subparagraphs (a) through (e) of this paragraph and in Paragraph (4) of this Subsection A.

(a) After providing a minimum 30-day public review and comment period, the commission determines that allowing temporary and short-term degradation of water quality is necessary to accommodate public health or safety activities in the area in which the ONRW is located. Examples of public health or safety activities include but are not limited to replacement or repair of a water or sewer pipeline or a roadway bridge. In making its decision, the commission shall consider whether the activity will interfere with activities implemented to restore or maintain the chemical, physical or biological integrity of the water. In approving the activity, the commission shall require that:

(i) the degradation shall be limited to the shortest possible time and shall not exceed six months;

(ii) the degradation shall be minimized and controlled by best management practices or in accordance with permit requirements as appropriate; all practical means of minimizing the duration, magnitude, frequency and cumulative effects of such degradation shall be utilized;

(iii) the degradation shall not result in water quality lower than necessary to protect any existing use in the ONRW; and

(iv) the degradation shall not alter the essential character or special use that makes the water an [~~ORNW~~]ONRW.

(b) Prior to the commission making a determination, the department or appropriate oversight agency shall provide a written recommendation to the commission. If the commission approves the activity, the department or appropriate oversight agency shall oversee implementation of the activity.

(c) Where an emergency response action that may result in temporary and short-term degradation to an ONRW is necessary to mitigate an immediate threat to public health or safety, the emergency response action may proceed prior to providing notification required by Subparagraph (a) of this paragraph in accordance with the following:

(i) only actions that mitigate an immediate threat to public health or safety may be undertaken pursuant to this provision; non-emergency portions of the action shall comply with the requirements of Subparagraph (a) of this paragraph;

(ii) the discharger shall make best efforts to comply with requirements (i) through (iv) of Subparagraph (a) of this paragraph;

(iii) the discharger shall notify the department of the emergency response action in writing within seven days of initiation of the action;

(iv) within 30 days of initiation of the emergency response action, the discharger shall provide a summary of the action taken, including all actions taken to comply with requirements (i) through (iv) of Subparagraph (a) of this paragraph.

(d) Preexisting land-use activities, including grazing, allowed by federal or state law prior to designation as an ONRW, and controlled by best management practices (BMPs), shall be allowed to continue so long as there are no new or increased discharges resulting from the activity after designation of the ONRW.

(e) Acequia operation, maintenance, and repairs are not subject to new requirements because of ONRW designation. However, the use of BMPs to minimize or eliminate the introduction of pollutants into receiving waters is strongly encouraged.

(4) This antidegradation policy does not prohibit activities that may result in degradation in surface waters of the state when such activities will result in restoration or maintenance of the chemical, physical or biological integrity of the water.

(a) For ONRWs, the department or appropriate oversight agency shall review on a case-by-case basis discharges that may result in degradation from restoration or maintenance activities, and may approve such activities in accordance with the following:

(i) the degradation shall be limited to the shortest possible time;

(ii) the degradation shall be minimized and controlled by best management practices or in accordance with permit requirements as appropriate, and all practical means of minimizing the duration, magnitude, frequency and cumulative effects of such degradation shall be utilized;

(iii) the degradation shall not result in water quality lower than necessary to protect any existing use of the surface water; and

(iv) the degradation shall not alter the essential character or special use that makes the water an [~~ORNW~~]ONRW.

(b) For surface waters of the state other than ONRWs, the department shall review on a case-by-case basis discharges that may result in degradation from restoration or maintenance activities, and may approve such activities in accordance with the following:

(i) the degradation shall be limited to the shortest possible time;

(ii) the degradation shall be minimized and controlled by best management practices or in accordance with permit requirements as appropriate, and all practical means of minimizing the duration, magnitude, frequency and cumulative effects of such degradation shall be utilized; and

(iii) the degradation shall not result in water quality lower than necessary to protect any existing use of the surface water.

(5) In those cases where potential water quality impairment associated with a thermal discharge is involved, this antidegradation policy and implementing method shall be consistent with Section 316 of the federal Clean Water Act.

(6) In implementing this section, the commission through the appropriate regional offices of the United States environmental protection agency will keep the administrator advised and provided with such information concerning the surface waters of the state as he or she will need to discharge his or her responsibilities under the federal Clean Water Act.

B. Implementation Plan: The department, acting under authority delegated by the commission, implements the water quality standards, including the antidegradation policy, by describing specific methods and procedures in the continuing planning process and by establishing and maintaining controls on the discharge of pollutants to surface waters of the state. The steps summarized in the following paragraphs, which may not all be applicable in every water pollution control action, list the implementation activities of the department. These implementation activities are supplemented by detailed antidegradation review procedures developed under the state's continuing planning process. The department:

(1) obtains information pertinent to the impact of the effluent on the receiving water and advises the prospective discharger of requirements for obtaining a permit to discharge;

(2) reviews the adequacy of existing data and conducts a water quality survey of the receiving water in accordance with an annually reviewed, ranked priority list of surface waters of the state requiring total maximum daily loads pursuant to Section 303(d) of the federal Clean Water Act;

(3) assesses the probable impact of the effluent on the receiving water relative to its attainable or designated uses and numeric and narrative criteria;

(4) requires the highest and best degree of wastewater treatment practicable and commensurate with protecting and maintaining the designated uses and existing water quality of surface waters of the state;

(5) develops water quality based effluent limitations and comments on technology based effluent limitations, as appropriate, for inclusion in any federal permit issued to a discharger pursuant to Section 402 of the federal Clean Water Act;

(6) requires that these effluent limitations be included in any such permit as a condition for state certification pursuant to Section 401 of the federal Clean Water Act;

(7) coordinates its water pollution control activities with other constituent agencies of the commission, and with local, state and federal agencies, as appropriate;

(8) develops and pursues inspection and enforcement programs to ensure that dischargers comply with state regulations and standards, and complements EPA's enforcement of federal permits;

(9) ensures that the provisions for public participation required by the New Mexico Water Quality Act and the federal Clean Water Act are followed;

(10) provides continuing technical training for wastewater treatment facility operators through the utility operators training and certification programs;

(11) provides funds to assist the construction of publicly owned wastewater treatment facilities through the wastewater construction program authorized by Section 601 of the federal Clean Water Act, and through funds appropriated by the New Mexico legislature;

(12) conducts water quality surveillance of the surface waters of the state to assess the effectiveness of water pollution controls, determines whether water quality standards are being attained, and proposes amendments to improve water quality standards;

(13) encourages, in conjunction with other state agencies, implementation of the best management practices set forth in the New Mexico statewide water quality management plan and the nonpoint source management program, such implementation shall not be mandatory except as provided by federal or state law;

(14) evaluates the effectiveness of BMPs selected to prevent, reduce or abate sources of water pollutants;

(15) develops procedures for assessing use attainment as required by 20.6.4.15 NMAC and establishing site-specific standards; and

(**16)** develops list of surface waters of the state not attaining designated uses, pursuant to Sections 305(b) and 303(d) of the federal Clean Water Act.

[20.6.4.8 NMAC - Rp 20 NMAC 6.1.1101, 10/12/2000; A, 5/23/2005; A, 8/1/2007; A, 1/14/2011; A, XX/XX/XXXX]

20.6.4.9 OUTSTANDING NATIONAL RESOURCE WATERS:

A. Procedures for nominating an ONRW: Any person may nominate a surface water of the state for designation as an ONRW by filing a petition with the commission pursuant to [~~the guidelines for water quality control commission regulation hearings~~]20.1.6 NMAC, Rulemaking Procedures - Water Quality Control Commission. A petition to designate a surface water of the state as an ONRW shall include:

(1) a map of the surface water of the state, including the location and proposed upstream and downstream boundaries;

(2) a written statement and evidence based on scientific principles in support of the nomination, including specific reference to one or more of the applicable ONRW criteria listed in Subsection B of this section;

(3) water quality data including chemical, physical or biological parameters, if available, to establish a baseline condition for the proposed ONRW;

(4) a discussion of activities that might contribute to the reduction of water quality in the proposed ONRW;

(5) any additional evidence to substantiate such a designation, including a discussion of the economic impact of the designation on the local and regional economy within the state of New Mexico and the benefit to the state; and

(6) affidavit of publication of notice of the petition in a newspaper of general circulation in the affected counties and in a newspaper of general statewide circulation.

B. Criteria for ONRWs: A surface water of the state, or a portion of a surface water of the state, may be designated as an ONRW where the commission determines that the designation is beneficial to the state of New Mexico, and:

(1) the water is a significant attribute of a state special trout water, national or state park, national or state monument, national or state wildlife refuge or designated wilderness area, or is part of a designated wild river under the federal Wild and Scenic Rivers Act; or

(2) the water has exceptional recreational or ecological significance; or

(3) the existing water quality is equal to or better than the numeric criteria for protection of aquatic life and contact uses and the human health-organism only criteria, and the water has not been significantly modified by human activities in a manner that substantially detracts from its value as a natural resource.

C. Pursuant to a petition filed under Subsection A of this section, the commission may classify a surface water of the state or a portion of a surface water of the state as an ONRW if the criteria set out in Subsection B of this section are met.

D. Waters classified as ONRWs: The following waters are classified as ONRWs:

(1) Rio Santa Barbara, including the west, middle and east forks from their headwaters downstream to the boundary of the Pecos Wilderness; and

(2) the waters within the United States forest service Valle Vidal special management unit including:

(a) Rio Costilla, including Comanche, La Cueva, Fernandez, Chuckwagon, Little Costilla, Powderhouse, Holman, Gold, Grassy, LaBelle and Vidal creeks, from their headwaters downstream to the boundary of the United States forest service Valle Vidal special management unit;

(b) Middle Ponil creek, including the waters of Greenwood Canyon, from their headwaters downstream to the boundary of the Elliott S. Barker wildlife management area;

(c) Shuree lakes;

(d) North Ponil creek, including McCrystal and Seally Canyon creeks, from their headwaters downstream to the boundary of the United States forest service Valle Vidal special management unit; and

(e) Leandro creek from its headwaters downstream to the boundary of the United States forest service Valle Vidal special management unit.

(3) the named perennial surface waters of the state, identified in Subparagraph (a) below, located within United States department of agriculture forest service wilderness. Wilderness are those lands designated by the United States congress as wilderness pursuant to the Wilderness Act. Wilderness areas included in this designation are the Aldo Leopold wilderness, Apache Kid wilderness, Blue Range wilderness, Chama River Canyon wilderness, Cruces Basin wilderness, Dome wilderness, Gila wilderness, Latir Peak wilderness, Pecos wilderness, San Pedro Parks wilderness, Wheeler Peak wilderness, and White Mountain wilderness.

(a) The following waters are designated in the Rio Grande basin:

(i) in the Aldo Leopold wilderness: Byers Run, Circle Seven creek, Flower canyon, Holden Prong, Indian canyon, Las Animas creek, Mud Spring canyon, North Fork Palomas creek, North Seco creek, Pretty canyon, Sids Prong, South Animas canyon, Victorio Park canyon, Water canyon;

(ii) in the Apache Kid wilderness Indian creek and Smith canyon;

(iii) in the Chama River Canyon wilderness: Chavez canyon, Ojitos canyon, Rio Chama;

(iv) in the Cruces Basin wilderness: Beaver creek, Cruces creek, Diablo creek, Escondido creek, Lobo creek, Osha creek;

(v) in the Dome wilderness: Capulin creek, Medio creek, Sanchez canyon/creek;

(vi) in the Latir Peak wilderness: Bull creek, Bull Creek lake, Heart lake, Lagunitas Fork, Lake Fork creek, Rito del Medio, Rito Primero, West Latir creek;

(vii) in the Pecos wilderness: Agua Sarca, Hidden lake, Horseshoe lake (Alamitos), Jose Vigil lake, Nambe lake, Nat lake IV, No Fish lake, North Fork Rio Quemado, Rinconada, Rio Capulin, Rio de las Trampas (Trampas creek), Rio de Truchas, Rio Frijoles, Rio Medio, Rio Molino, Rio Nambe, Rio San Leonardo, Rito con Agua, Rito Gallina, Rito Jaroso, Rito Quemado, San Leonardo lake, Santa Fe lake, Santa Fe river, Serpent lake, South Fork Rio Quemado, Trampas lake (East), Trampas lake (West);

(viii) in the San Pedro Parks wilderness: Agua Sarca, Cañon Madera, Cave creek, Cecilia Canyon creek, Clear creek (North SPP), Clear creek (South SPP), Corralitos creek, Dove creek, Jose Miguel creek, La Jara creek, Oso creek, Rio Capulin, Rio de las Vacas, Rio Gallina, Rio Puerco de Chama, Rito Anastacio East, Rito Anastacio West, Rito de las Palomas, Rito de las Perchas, Rito de los Pinos, Rito de los Utes, Rito Leche, Rito Redondo, Rito Resumidero, San Gregorio lake;

(ix) in the Wheeler Peak wilderness: Black Copper canyon, East Fork Red river, Elk lake, Horseshoe lake, Lost lake, Sawmill creek, South Fork lake, South Fork Rio Hondo, Williams lake.

(b) The following waters are designated in the Pecos River basin:

(i) in the Pecos wilderness: Albright creek, Bear creek, Beatty creek, Beaver creek, Carpenter creek, Cascade canyon, Cave creek, El Porvenir creek, Hollinger creek, Holy Ghost creek, Horsethief creek, Jack's creek, Jarosa canyon/creek, Johnson lake, Lake Katherine, Lost Bear lake, Noisy brook, Panchuela creek, Pecos Baldy lake, Pecos river, Rio Mora, Rio Valdez, Rito Azul, Rito de los Chimayosos, Rito de los Esteros, Rito del Oso, Rito del Padre, Rito las Trampas, Rito Maestas, Rito Oscuro, Rito Perro, Rito Sebadilloses, South Fork Bear creek, South Fork Rito Azul, Spirit lake, Stewart lake, Truchas lake (North), Truchas lake (South), Winsor creek;

(ii) in the White Mountain wilderness: Argentina creek, Aspen creek, Bonito creek, Little Bonito creek, Mills canyon/creek, Rodamaker creek, South Fork Rio Bonito, Turkey

canyon/creek.

(c) The following waters are designated in the Gila River basin:

(i) in the Aldo Leopold wilderness: Aspen canyon, Black Canyon creek, Bonner canyon, Burnt canyon, Diamond creek, Falls canyon, Fisherman canyon, Running Water canyon, South Diamond creek;

(ii) in the Gila wilderness: Apache creek, Black Canyon creek, Brush canyon, Canyon creek, Chicken Coop canyon, Clear creek, Cooper canyon, Cow creek, Cub creek, Diamond creek, East Fork Gila river, Gila river, Gilita creek, Indian creek, Iron creek, Langstroth canyon, Lilley canyon, Little creek, Little Turkey creek, Lookout canyon, McKenna creek, Middle Fork Gila river, Miller Spring canyon, Mogollon creek, Panther canyon, Prior creek, Rain creek, Raw Meat creek, Rocky canyon, Sacaton creek, Sapillo creek, Sheep Corral canyon, Skeleton canyon, Squaw creek, Sycamore canyon, Trail canyon, Trail creek, Trout creek, Turkey creek, Turkey Feather creek, Turnbo canyon, West Fork Gila river, West Fork Mogollon creek, White creek, Willow creek, Woodrow canyon.

(d) The following waters are designated in the Canadian River basin: in the Pecos wilderness Daily creek, Johns canyon, Middle Fork Lake of Rio de la Casa, Middle Fork Rio de la Casa, North Fork Lake of Rio de la Casa, Rito de Gascon, Rito San Jose, Sapello river, South Fork Rio de la Casa, Sparks creek (Manuelitas creek).

(e) The following waters are designated in the San Francisco River basin:

(i) in the Blue Range wilderness: Pueblo creek;

(ii) in the Gila wilderness: Big Dry creek, Lipsey canyon, Little Dry creek, Little Whitewater creek, South Fork Whitewater creek, Spider creek, Spruce creek, Whitewater creek.

(f) The following waters are designated in the Mimbres Closed basin: in the Aldo Leopold wilderness Corral canyon, Mimbres river, North Fork Mimbres river, South Fork Mimbres river.

(g) The following waters are designated in the Tularosa Closed basin: in the White Mountain wilderness Indian creek, Nogal Arroyo, Three Rivers.

(h) The wetlands designated are identified on the *Maps and List of Wetlands Within United States Forest Service Wilderness Areas Designated as Outstanding National Resource Waters* published at the New Mexico state library and available on the department's website.

[20.6.4.9 NMAC - Rn, Subsections B, C and D of 20.6.4.8 NMAC, 5/23/2005; A, 5/23/2005; A, 7/17/2005; A, 2/16/2006; A, 12/1/2010; A, 1/14/2011; A, XX/XX/XXXX]

20.6.4.10 REVIEW OF STANDARDS; NEED FOR ADDITIONAL STUDIES:

A. Section 303(c)(1) of the federal Clean Water Act requires that the state hold public hearings at least once every three years for the purpose of reviewing water quality standards and proposing, as appropriate, necessary revisions to water quality standards.

B. In accordance with 40 CFR 131.10(i), when an existing use, as defined under 20.6.4.7 NMAC, is higher quality water than prescribed by the designated use and supporting evidence demonstrates the presence of that use, the designated use shall be amended accordingly to be no less stringent than the existing use.

[~~B.~~] C. It is recognized that, in some cases, numeric criteria [~~have been adopted that reflect use designations rather than existing conditions of surface waters of the state.~~]for a particular designated use may not adequately reflect the local conditions or the aquatic communities adapted to those localized conditions. In these cases, a water quality criterion may be modified to reflect the natural condition of a specific waterbody. The modification of the criterion does not change the designated use; the modification only changes the criterion for that specific waterbody. [~~Narrative criteria are required for many constituents because accurate data on background levels are lacking. More intensive water quality monitoring may identify surface waters of the state where existing quality is considerably better than the established criteria.~~]When justified by sufficient data and information, a numeric [~~the~~] water quality [~~criteria~~]criterion [~~will~~]may be adopted or modified in accordance with 20.6.4.10(F) and 20.6.4.10(G) NMAC, to protect the attainable uses of the waterbody.

D. The removal or amendment of a designated use to a designated use with less stringent criteria can only be done through a use attainability analysis in accordance with 20.6.4.15 NMAC.

[~~C.~~] E. It is also recognized that contributions of water contaminants by diffuse nonpoint sources of water pollution may make attainment of certain criteria difficult. Revision of these criteria may be necessary as new information is obtained on nonpoint sources and other problems unique to semi-arid regions.

[~~D.~~] F. Site-specific criteria.

(1) The commission may adopt site-specific numeric criteria applicable to all or part of a surface water of the state based on relevant site-specific conditions such as:

(a) actual species at a site are more or less sensitive than those used in the national criteria data set;

(b) physical or chemical characteristics at a site such as pH or hardness alter the biological availability and/or toxicity of the chemical;

(c) physical, biological or chemical factors alter the bioaccumulation potential of a chemical;

(d) the concentration resulting from natural background exceeds numeric criteria for aquatic life, wildlife habitat or other uses if consistent with Subsection [~~E~~]G of 20.6.4.10 NMAC; or

(e) other factors or combination of factors that upon review of the commission may warrant modification of the default criteria, subject to EPA review and approval.

(2) Site-specific criteria must fully protect the designated use to which they apply. In the case of human health-organism only criteria, site-specific criteria must fully protect human health when organisms are consumed from waters containing pollutants.

(3) Any person may petition the commission to adopt site-specific criteria. A petition for the adoption of site-specific criteria shall:

(a) identify the specific waters to which the site-specific criteria would apply;

(b) explain the rationale for proposing the site-specific criteria;

(c) describe the methods used to notify and solicit input from potential stakeholders and from the general public in the affected area, and present and respond to the public input received;

(d) present and justify the derivation of the proposed criteria.

(4) A derivation of site-specific criteria shall rely on a scientifically defensible method, such as one of the following:

(a) the recalculation procedure, the water-effect ratio for metals procedure or the resident species procedure as described in the water quality standards handbook (EPA-823-B-94-005a, 2nd edition, August 1994);

(b) the streamlined water-effect ratio procedure for discharges of copper (EPA-822-R-01-005, March 2001);

(c) the biotic ligand model as described in aquatic life ambient freshwater quality criteria - copper (EPA-822-R-07-001, February 2007);

(d) the methodology for deriving ambient water quality criteria for the protection of human health (EPA-822-B-00-004, October 2000) and associated technical support documents; or

(e) a determination of the natural background of the water body as described in Subsection [~~E~~]G of 20.6.4.10 NMAC.

[~~E.~~]G. Site-specific criteria based on natural background. The commission may adopt site-specific criteria equal to the concentration resulting from natural background where that concentration protects the designated use. The concentration resulting from natural background supports the level of aquatic life and wildlife habitat expected to occur naturally at the site absent any interference by humans. Domestic water supply, primary or secondary contact, or human health-organism only criteria shall not be modified based on natural background. A determination of natural background shall:

(1) consider natural spatial and seasonal to interannual variability as appropriate;

(2) document the presence of natural sources of the pollutant;

(3) document the absence of human sources of the pollutant or quantify the human contribution; and

(4) rely on analytical, statistical or modeling methodologies to quantify the natural background.

[~~F.~~]H. Temporary standards[~~:~~].

(1) Any person may petition the commission to adopt a temporary standard applicable to all or part of a surface water of the state as provided for in this section and applicable sections in 40 CFR Part 131, Water Quality Standards; specifically, Section 131.14. The commission may adopt a proposed temporary standard if the petitioner demonstrates that:

(a) attainment of the associated designated use may not be feasible in the short term due to one or more of the factors listed in 40 CFR 131.10(g), or due to the implementation of actions necessary to facilitate restoration such as through dam removal or other significant wetland or water body reconfiguration activities as demonstrated by the petition and supporting work plan requirements in Paragraphs (4) and (5) of Subsection [~~F~~]H of 20.6.4.10 NMAC;

(b) the proposed temporary standard represents the highest degree of protection feasible in the short term, limits the degradation of water quality to the minimum necessary to achieve the original standard by the expiration date of the temporary standard, and adoption will not cause the further impairment or loss of an existing use;

(c) for point sources, existing or proposed discharge control technologies will comply with applicable technology-based limitations and feasible technological controls and other management alternatives, such as a pollution prevention program; and

(d) for restoration activities, nonpoint source or other control technologies shall limit downstream impacts, and if applicable, existing or proposed discharge control technologies shall be in place consistent with Subparagraph (c) of Paragraph (1) of Subsection [~~F~~]H of 20.6.4.10 NMAC.

(2) A temporary standard shall apply to specific designated use(s), pollutant(s), or permittee(s), and to specific water body segment(s). The adoption of a temporary standard does not exempt dischargers from complying with all other applicable water quality standards or control technologies.

(3) Designated use attainment as reported in the federal Clean Water Act, Section 305(b)/303(d) Integrated Report shall be based on the original standard and not on a temporary standard.

(4) A petition for a temporary standard shall:

(a) identify the currently applicable standard(s), the proposed temporary standard for the specific pollutant(s), the permittee(s), and the specific surface water body segment(s) of the state to which the temporary standard would apply;

(b) include the basis for any factor(s) specific to the applicability of the temporary standard (for example critical flow under Subsection B of 20.6.4.11 NMAC);

(c) demonstrate that the proposed temporary standard meets the requirements in this subsection;

(d) present a work plan with timetable of proposed actions for achieving compliance with the original standard in accordance with Paragraph (5) of Subsection [~~F~~]H of 20.6.4.10 NMAC;

(e) include any other information necessary to support the petition.

(5) As a condition of a petition for a temporary standard, in addition to meeting the requirements in this Subsection, the petitioner shall prepare a work plan in accordance with Paragraph (4) of Subsection [~~F~~]H of 20.6.4.10 NMAC and submit the work plan to the department for review and comment. The work plan shall identify the factor(s) listed in 40 CFR 131.10(g) or Subparagraph (a) of Paragraph (1) of Subsection [~~F~~]H of 20.6.4.10 NMAC affecting attainment of the standard that will be analyzed and the timeline for proposed actions to be taken to achieve the uses attainable over the term of the temporary standard, including baseline water quality, and any investigations, projects, facility modifications, monitoring, or other measures necessary to achieve compliance with the original standard. The work plan shall include provisions for review of progress in accordance with Paragraph (8) of Subsection [~~F~~]H of 20.6.4.10 NMAC, public notice and consultation with appropriate state, tribal, local and federal agencies.

(6) The commission may condition the approval of a temporary standard by requiring additional monitoring, relevant analyses, the completion of specified projects, submittal of information, or any other actions.

(7) Temporary standards may be implemented only after a public hearing before the commission, commission approval and adoption pursuant to Subsection [~~F~~]H of 20.6.4.10 NMAC for all state purposes, and the federal Clean Water Act Section 303 (c) approval for any federal action.

(8) All temporary standards are subject to a required review during each succeeding review of water quality standards conducted in accordance with Subsection A of 20.6.4.10 NMAC. The petitioner shall provide a written report to the commission documenting the progress of proposed actions, pursuant to a reporting schedule stipulated in the approved temporary standard. The purpose of the review is to determine progress consistent with the original conditions of the petition for the duration of the temporary standard. If the petitioner cannot demonstrate that sufficient progress has been made the commission may revoke approval of the temporary standard or provide additional conditions to the approval of the temporary standard.

(9) The commission may consider a petition to extend a temporary standard. The effective period of a temporary standard shall be extended only if demonstrated to the commission that the factors precluding attainment of the underlying standard still apply, that the petitioner is meeting the conditions required for approval of the temporary standard, and that reasonable progress towards meeting the underlying standard is being achieved.

(10) A temporary standard shall expire no later than the date specified in the approval of the temporary standard. Upon expiration of a temporary standard, the original standard becomes applicable.

(11) Temporary standards shall be identified in 20.6.4.97-899 NMAC as appropriate for the surface water affected.

(12) "Temporary standard" means a time-limited designated use and criterion for a specific pollutant(s) or water quality parameter(s) that reflect the highest attainable condition during the term of the temporary standard.

[20.6.4.10 NMAC - Rp 20 NMAC 6.1.1102, 10/12/2000; Rn, 20.6.4.9 NMAC, 5/23/2005; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017; A, XX/XX/XXXX]

20.6.4.11 APPLICABILITY OF WATER QUALITY STANDARDS:

A. [RESERVED]

B. Critical low flow: The critical low flow of a stream at a particular site shall be used in developing point source discharge permit requirements to meet numeric criteria set in 20.6.4.97 through 20.6.4.900 NMAC and Subsection F of 20.6.4.13 NMAC.

(1) For human health-organism only criteria, the critical low flow is the harmonic mean flow[; ~~"harmonic mean flow" is the number of daily flow measurements divided by the sum of the reciprocals of the flows; that is, it is the reciprocal of the mean of reciprocals~~]. For ephemeral waters the calculation shall be based upon the nonzero flow intervals and modified by including a factor to adjust for the proportion of intervals with zero flow. The equations are as follows:

$$\text{Harmonic Mean} = \frac{n}{\sum 1/Q}$$

where n = number of flow values
and Q = flow value

$$\text{Modified Harmonic Mean} = \left[\frac{\sum_{i=1}^{Nt-N0} \frac{1}{Qi}}{Nt - N0}\right]^{-1} x \left[\frac{Nt - N0}{Nt}\right]$$

where Qi = nonzero flow
Nt = total number of flow values
and N_0 = number of zero flow values

(2) For all other narrative and numeric criteria, the critical low flow is the minimum average four consecutive day flow that occurs with a frequency of once in three years (4Q3). The critical low flow may be determined on an annual, a seasonal or a monthly basis, as appropriate, after due consideration of site-specific conditions.

C. Guaranteed minimum flow: The commission may allow the use of a contractually guaranteed minimum streamflow in lieu of a critical low flow determined under Subsection B of this section on a case-by-case basis and upon consultation with the interstate stream commission. Should drought, litigation or any other reason interrupt or interfere with minimum flows under a guaranteed minimum flow contract for a period of at least 30 consecutive days, such permission, at the sole discretion of the commission, may then be revoked. Any minimum flow specified under such revoked permission shall be superseded by a critical low flow determined under Subsection B of this section. A public notice of the request for a guaranteed minimum flow shall be published in a newspaper of general circulation by the department at least 30 days prior to scheduled action by the commission. These water quality standards do not grant to the commission or any other entity the power to create, take away or modify property rights in water.

D. Mixing zones: A limited mixing zone, contiguous to a point source wastewater discharge, may be allowed in any stream receiving such a discharge. Mixing zones serve as regions of initial dilution that allow the application of a dilution factor in calculations of effluent limitations. Effluent limitations shall be developed that will protect the most sensitive existing, designated or attainable use of the receiving water.

E. Mixing zone limitations: Wastewater mixing zones, in which the numeric criteria set under Subsection F of 20.6.4.13 NMAC, 20.6.4.97 through 20.6.4.899 NMAC or 20.6.4.900 NMAC may be exceeded, shall be subject to the following limitations:

(1) Mixing zones are not allowed for discharges to lakes, reservoirs, or playas; these effluents shall meet all applicable criteria set under Subsection F of 20.6.4.13 NMAC, 20.6.4.97 through 20.6.4.899 NMAC and 20.6.4.900 NMAC at the point of discharge.

(2) The acute aquatic life criteria, as set out in Subsection I, Subsection J, and Subsection K of 20.6.4.900 NMAC, shall be attained at the point of discharge for any discharge to a surface water of the state with a designated aquatic life use.

(3) The general criteria set out in Subsections A, B, C, D, E, G, H and J of 20.6.4.13 NMAC, and the provision set out in Subsection D of 20.6.4.14 NMAC are applicable within mixing zones.

(4) The areal extent and concentration isopleths of a particular mixing zone will depend on site-specific conditions including, but not limited to, wastewater flow, receiving water critical low flow, outfall design, channel characteristics and climatic conditions and, if needed, shall be determined on a case-by-case basis. When the physical boundaries or other characteristics of a particular mixing zone must be known, the methods presented in Section 4.4.5, "Ambient-induced mixing," in "Technical support document for water quality-based toxics control" (March 1991, EPA/505/2-90-001) shall be used.

(5) All applicable water quality criteria set under Subsection F of 20.6.4.13 NMAC, 20.6.4.97 through 20.6.4.899 NMAC and 20.6.4.900 NMAC shall be attained at the boundaries of mixing zones. A continuous zone of passage through or around the mixing zone shall be maintained in which the water quality meets all applicable criteria and allows the migration of aquatic life presently common in surface waters of the state with no effect on their populations.

F. Multiple uses: When a surface water of the state has more than a single designated use, the applicable numeric criteria shall be the most stringent of those established for such water.

G. Human health-organism only criteria in Subsection J of 20.6.4.900 NMAC apply to those waters with a designated, existing or attainable aquatic life use. When limited aquatic life is a designated use, the human health-organism only criteria apply only if adopted on a segment-specific basis. The human health-organism only criteria for persistent toxic pollutants, as identified in Subsection J of 20.6.4.900 NMAC, also apply to all tributaries of waters with a designated, existing or attainable aquatic life use.

[~~**H. Unclassified waters of the state:** Unclassified waters of the state are those surface waters of the state not identified in 20.6.4.101 through 20.6.4.899 NMAC. An unclassified surface water of the state is presumed to support the uses specified in Section 101(a)(2) of the federal Clean Water Act. As such, it is subject to 20.6.4.98 NMAC if nonperennial or subject to 20.6.4.99 NMAC if perennial. The commission may include an ephemeral unclassified surface water of the state under 20.6.4.97 NMAC only if a use attainability analysis demonstrates pursuant to 20.6.4.15 NMAC that attainment of Section 101(a)(2) uses is not feasible.~~]

[~~**I.** ~~]**H. Exceptions:** Numeric criteria for temperature, dissolved solids, dissolved oxygen, sediment or turbidity adopted under the Water Quality Act do not apply when changes in temperature, dissolved solids, dissolved oxygen, sediment or turbidity in a surface water of the state are attributable to:

(1) natural causes (discharges from municipal separate storm sewers are not covered by this exception.); or

(2) the reasonable operation of irrigation and flood control facilities that are not subject to federal or state water pollution control permitting; major reconstruction of storage dams or diversion dams except for emergency actions necessary to protect health and safety of the public are not covered by this exception.

[20.6.4.11 NMAC - Rp 20 NMAC 6.1.1103, 10/12/2000; A, 10/11/2002; Rn, 20.6.4.10 NMAC, 5/23/2005; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.12 COMPLIANCE WITH WATER QUALITY STANDARDS: The following provisions apply to determining compliance for enforcement purposes; they do not apply for purposes of determining attainment of uses. The department has developed assessment protocols for the purpose of determining attainment of uses that are available for review from the department's surface water quality bureau.

A. Compliance with acute water quality criteria shall be determined from the analytical results of a single grab sample. Acute criteria shall not be exceeded.

B. Compliance with chronic water quality criteria shall be determined from the arithmetic mean of the analytical results of samples collected using applicable protocols. Chronic criteria shall not be exceeded more than once every three years.

C. Compliance with water quality standards for total ammonia shall be determined by performing the biomonitoring procedures set out in Subsections D and E of 20.6.4.14 NMAC, or by attainment of applicable ammonia criteria set out in Subsections K, L and M of 20.6.4.900 NMAC.

D. Compliance with the human health-organism only criteria shall be determined from the analytical results of representative grab samples, as defined in the water quality management plan. Human health-organism only criteria shall not be exceeded.

E. The commission may establish a numeric water quality criterion at a concentration that is below the minimum quantification level. In such cases, the water quality standard is enforceable at the minimum quantification level.

F. For compliance with hardness-dependent numeric criteria, dissolved hardness (as mg $CaCO_3$/L) shall be determined from a sample taken at the same time that the sample for the contaminant is taken.

G. **Compliance schedules**: [~~It shall be the policy of the commission to allow on a case-by-case basis t~~]The commission may allow the inclusion of a schedule of compliance in a NPDES permit issued to an existing facility on a case-by-case basis. Such schedule of compliance will be for the purpose of providing a permittee with adequate time to make treatment facility modifications necessary to comply with water quality based permit limitations determined to be necessary to implement new or revised water quality standards or wasteload allocation. Compliance schedules may be included in NPDES permits at the time of permit renewal or modification and shall be written to require compliance at the earliest practicable time. Compliance schedules shall also specify milestone dates so as to measure progress towards final project completion (e.g., design completion, construction start, construction completion, date of compliance).

H. It is a policy of the commission to allow a temporary standard approved and adopted pursuant to Subsection [~~F~~]H of 20.6.4.10 NMAC to be included in the applicable federal Clean Water Act permit as enforceable limits and conditions. The temporary standard and any schedule of actions may be included at the earliest practicable time, and shall specify milestone dates so as to measure progress towards meeting the original standard. [20.6.4.12 NMAC - Rp 20 NMAC 6.1.1104, 10/12/2000; A, 10/11/2002; Rn, 20.6.4.11 NMAC, 5/23/2005; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017; A, XX/XX/XXXX]

20.6.4.13 GENERAL CRITERIA: General criteria are established to sustain and protect existing or attainable uses of surface waters of the state. These general criteria apply to all surface waters of the state at all times, unless a specified criterion is provided elsewhere in this part. Surface waters of the state shall be free of any water contaminant in such quantity and of such duration as may with reasonable probability injure human health, animal or plant life or property, or unreasonably interfere with the public welfare or the use of property.

A. **Bottom deposits and suspended or settleable solids:**

(1) Surface waters of the state shall be free of water contaminants including fine sediment particles (less than two millimeters in diameter), precipitates or organic or inorganic solids from other than natural causes that have settled to form layers on or fill the interstices of the natural or dominant substrate in quantities that damage or impair the normal growth, function or reproduction of aquatic life or significantly alter the physical or chemical properties of the bottom.

(2) Suspended or settleable solids from other than natural causes shall not be present in surface waters of the state in quantities that damage or impair the normal growth, function or reproduction of aquatic life or adversely affect other designated uses.

B. **Floating solids, oil and grease:** Surface waters of the state shall be free of oils, scum, grease and other floating materials resulting from other than natural causes that would cause the formation of a visible sheen or visible deposits on the bottom or shoreline, or would damage or impair the normal growth, function or reproduction of human, animal, plant or aquatic life.

C. **Color:** Color-producing materials resulting from other than natural causes shall not create an aesthetically undesirable condition nor shall color impair the use of the water by desirable aquatic life presently common in surface waters of the state.

D. **Organoleptic quality:**

(1) **Flavor of fish:** Water contaminants from other than natural causes shall be limited to concentrations that will not impart unpalatable flavor to fish.

(2) **Odor and taste of water:** Water contaminants from other than natural causes shall be limited to concentrations that will not result in offensive odor or taste arising in a surface water of the state or otherwise interfere with the reasonable use of the water.

E. Plant nutrients: Plant nutrients from other than natural causes shall not be present in concentrations that will produce undesirable aquatic life or result in a dominance of nuisance species in surface waters of the state.

F. Toxic pollutants:

(1) Except as provided in 20.6.4.16 NMAC, surface waters of the state shall be free of toxic pollutants, including but not limited to contaminants of emerging concern and those toxic pollutants listed in 20.6.2 NMAC, from other than natural causes in amounts, concentrations or combinations that affect the propagation of fish or that are toxic to humans, livestock or other animals, fish or other aquatic organisms, wildlife using aquatic environments for habitation or aquatic organisms for food, or that will or can reasonably be expected to bioaccumulate in tissues of fish, shellfish and other aquatic organisms to levels that will impair the health of aquatic organisms or wildlife or result in unacceptable tastes, odors or health risks to human consumers of aquatic organisms.

(2) Pursuant to this section, the human health-organism only criteria shall be as set out in 20.6.4.900 NMAC. When a human health-organism only criterion is not listed in 20.6.4.900 NMAC, the following provisions shall be applied in accordance with 20.6.4.11, 20.6.4.12 and 20.6.4.14 NMAC.

(a) The human health-organism only criterion shall be the recommended human health criterion for "consumption of organisms only" published by the U.S. environmental protection agency pursuant to Section 304(a) of the federal Clean Water Act. In determining such criterion for a cancer-causing toxic pollutant, a cancer risk of 10^{-5} (one cancer per 100,000 exposed persons) shall be used.

(b) When a numeric criterion for the protection of human health for the consumption of organism only has not been published by the U.S. environmental protection agency, a quantifiable criterion may be derived from data available in the U.S. environmental protection agency's Integrated Risk Information System (IRIS) using the appropriate formula specified in *Methodology For Deriving Ambient Water Quality Criteria For The Protection Of Human Health (2000)*, EPA-822-B-00-004.

(3) Pursuant to this section, the chronic aquatic life criteria shall be as set out in 20.6.4.900 NMAC. When a chronic aquatic life criterion is not listed in 20.6.4.900 NMAC, the following provisions shall be applied in sequential order in accordance with 20.6.4.11, 20.6.4.12 and 20.6.4.14 NMAC.

(a) The chronic aquatic life criterion shall be the "freshwater criterion continuous concentration" published by the U.S. environmental protection agency pursuant to Section 304(a) of the federal Clean Water Act;

(b) If the U.S. environmental protection agency has not published a chronic aquatic life criterion, a geometric mean LC-50 value shall be calculated for the particular species, genus or group that is representative of the form of life to be preserved, using the results of toxicological studies published in scientific journals.

(i) The chronic aquatic life criterion for a toxic pollutant that does not bioaccumulate shall be ten percent of the calculated geometric mean LC-50 value; and

(ii) The chronic aquatic life criterion for a toxic pollutant that does bioaccumulate shall be: the calculated geometric mean LC-50 adjusted by a bioaccumulation factor for the particular species, genus or group representative of the form of life to be preserved, but when such bioaccumulation factor has not been published, the criterion shall be one percent of the calculated geometric mean LC-50 value.

(4) Pursuant to this section, the acute aquatic life criteria shall be as set out in 20.6.4.900 NMAC. When an acute aquatic life criterion is not listed in 20.6.4.900 NMAC, the acute aquatic life criterion shall be the "freshwater criterion maximum concentration" published by the U.S. environmental protection agency pursuant to Section 304(a) of the federal Clean Water Act.

(5) Within 90 days of the issuance of a final NPDES permit containing a numeric criterion selected or calculated pursuant to Paragraph (2), Paragraph (3) or Paragraph (4) of Subsection F of this section, the department shall petition the commission to adopt such criterion into these standards.

G. Radioactivity: The radioactivity of surface waters of the state shall be maintained at the lowest practical level and shall in no case exceed the criteria set forth in the New Mexico Radiation Protection Regulations, 20.3.1 and 20.3.4 NMAC.

H. Pathogens: Surface waters of the state shall be free of pathogens from other than natural causes in sufficient quantity to impair public health or the designated, existing or attainable uses of a surface water of the state.

I. Temperature: Maximum temperatures for surface waters of the state have been specified in 20.6.4.97 through 20.6.4.900 NMAC. However, the introduction of heat by other than natural causes shall not increase the temperature, as measured from above the point of introduction, by more than 2.7°C (5°F) in a stream, or

more than 1.7°C (3°F) in a lake or reservoir. In no case will the introduction of heat be permitted when the maximum temperature specified for the reach would thereby be exceeded. These temperature criteria shall not apply to impoundments constructed offstream for the purpose of heat disposal. High water temperatures caused by unusually high ambient air temperatures are not violations of these criteria.

J. Turbidity: Turbidity attributable to other than natural causes shall not reduce light transmission to the point that the normal growth, function or reproduction of aquatic life is impaired or that will cause substantial visible contrast with the natural appearance of the water. Activities or discharges shall not cause turbidity to increase more than 10 NTU over background turbidity when the background turbidity, measured at a point immediately upstream of the activity, is 50 NTU or less, nor to increase more than twenty percent when the background turbidity is more than 50 NTU. However, limited-duration turbidity increases caused by dredging, construction or other similar activities may be allowed provided all practicable turbidity control techniques have been applied and all appropriate permits, certifications and approvals have been obtained.

K. Total dissolved solids (TDS): TDS attributable to other than natural causes shall not damage or impair the normal growth, function or reproduction of animal, plant or aquatic life. TDS shall be measured by either the "calculation method" (sum of constituents) or the filterable residue method. Approved test procedures for these determinations are set forth in 20.6.4.14 NMAC.

L. Dissolved gases: Surface waters of the state shall be free of nitrogen and other dissolved gases at levels above one hundred ten percent saturation when this supersaturation is attributable to municipal, industrial or other discharges.

M. Biological integrity: Surface waters of the state shall support and maintain a balanced and integrated community of aquatic organisms with species composition, diversity and functional organization comparable to those of natural or minimally impacted water bodies of a similar type and region.

[20.6.4.13 NMAC - Rp 20 NMAC 6.1.1105, 10/12/2000; A, 10/11/2002; Rn, 20.6.4.12 NMAC, 5/23/2005; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.14 SAMPLING AND ANALYSIS:

A. Sampling and analytical techniques shall conform with methods described in the following references unless otherwise specified by the commission pursuant to a petition to amend these standards:

(1) "*Guidelines Establishing Test Procedures For The Analysis Of Pollutants Under The Clean Water Act*," 40 CFR Part 136 or any test procedure approved or accepted by EPA using procedures provided in 40 CFR Parts 136.3(d), 136.4, and 136.5;

(2) *Standard Methods For The Examination Of Water And Wastewater*, latest edition, American public health association;

(3) *Methods For Chemical Analysis Of Water And Waste*, and other methods published by EPA office of research and development or office of water;

(4) *Techniques Of Water Resource Investigations Of The U.S. Geological Survey*;

(5) *Annual Book Of ASTM Standards*: volumes 11.01 and 11.02, water (I) and (II), latest edition, ASTM international;

(6) *Federal Register*, latest methods published for monitoring pursuant to Resource Conservation and Recovery Act regulations;

(7) *National Handbook Of Recommended Methods For Water-Data Acquisition*, latest edition, prepared cooperatively by agencies of the United States government under the sponsorship of the U.S. geological survey; or

(8) *Federal Register*, latest methods published for monitoring pursuant to the Safe Drinking Water Act regulations.

B. Bacteriological Surveys: The monthly geometric mean shall be used in assessing attainment of criteria when a minimum of five samples is collected in a 30-day period.

C. Sampling Procedures:

(1) Streams: Stream monitoring stations below discharges shall be located a sufficient distance downstream to ensure adequate vertical and lateral mixing.

(2) Lakes: Sampling stations in lakes shall be located at least 250 feet from a discharge.

(3) Lakes: Except for the restriction specified in Paragraph (2) of this subsection, lake sampling stations shall be located at any site where the attainment of a water quality criterion is to be assessed. Water quality measurements taken at intervals in the entire water column at a sampling station shall be averaged for the epilimnion, or in the absence of an epilimnion, for the upper one-third of the water column of the lake to determine attainment of criteria, except that attainment of criteria for toxic pollutants shall be assessed during

periods of complete vertical mixing, e.g., during spring or fall turnover, or by taking depth-integrated composite samples of the water column.

D. Acute toxicity of effluent to aquatic life shall be determined using the procedures specified in U.S. environmental protection agency "*Methods For Measuring The Acute Toxicity Of Effluents And Receiving Waters To Freshwater And Marine Organisms*" (5th Ed., 2002, EPA 821-R-02-012), or latest edition thereof if adopted by EPA at 40 CFR Part 136, which is incorporated herein by reference. Acute toxicities of substances shall be determined using at least two species tested in whole effluent and a series of effluent dilutions. Acute toxicity due to discharges shall not occur within the wastewater mixing zone in any surface water of the state with an existing or designated aquatic life use.

E. Chronic toxicity of effluent or ambient surface waters of the state to aquatic life shall be determined using the procedures specified in U.S. environmental protection agency "*Short-Term Methods For Estimating The Chronic Toxicity Of Effluents And Receiving Waters To Freshwater Organisms*" (4th Ed., 2002, EPA 821-R-02-013), or latest edition thereof if adopted by EPA at 40 CFR Part 136, which is incorporated herein by reference. Chronic toxicities of substances shall be determined using at least two species tested in ambient surface water or whole effluent and a series of effluent dilutions. Chronic toxicity due to discharges shall not occur at the critical low flow, or any flow greater than the critical low flow, in any surface water of the state with an existing or designated aquatic life use more than once every three years.

[20.6.4.14 NMAC - Rp 20 NMAC 6.1.1106, 10/12/2000; Rn, 20.6.4.13 NMAC, 5/23/2005 & A, 5/23/2005; A, 12/1/2010]

20.6.4.15 USE ATTAINABILITY ANALYSIS:

A. Authority to remove a designated use. A use attainability analysis is a scientific study conducted for the purpose of assessing the factors affecting the attainment of a use. Whenever a use attainability analysis is conducted, it shall be subject to the requirements and limitations set forth in 40 CFR Part 131, Water Quality Standards; specifically, Subsections 131.3(g), 131.10(g), 131.10(h) and 131.10(j) shall be applicable. In accordance with 40 CFR 131.10(i), and 20.6.4.10 NMAC, the amendment of a designated use, based on a more stringent existing use, does not require a use attainability analysis.

(1) The commission may remove a designated use, that is not an existing use, specified in Section 101(a)(2) of the federal Clean Water Act or adopt subcategories of a use in Section 101(a)(2) of the federal Clean Water Act[~~use~~] requiring less stringent criteria only if a use attainability analysis demonstrates that attaining the use is not feasible because of a factor listed in 40 CFR 131.10(g). Uses in Section 101(a)(2) of the federal Clean Water Act[~~uses~~], which refer to the protection and propagation of fish, shellfish and wildlife and recreation in and on the water, are also specified in Subsection B of 20.6.4.6 NMAC.

(2) A designated use cannot be removed if it is an existing use unless a use requiring more stringent criteria is designated.

B. The mechanism to remove a designated use. A use attainability analysis shall assess the physical, chemical, biological, economic or other factors affecting the attainment of a use. The analysis shall rely on scientifically defensible methods such as the methods described in the following documents:

(1) *Technical Support Manual: Waterbody Surveys And Assessments For Conducting Use Attainability Analyses*, volume I (November 1983) and volume III (November 1984) or latest editions, United States environmental protection agency, office of water, regulations and standards, Washington, D.C., for the evaluation of aquatic life or wildlife uses;

(2) the department's *Hydrology Protocol*, latest edition, approved by the commission, for identifying ephemeral, [~~and~~] intermittent, and perennial waters; or

(3) *Interim Economic Guidance For Water Quality Standards - Workbook*, March 1995, United States environmental protection agency, office of water, Washington, D.C. for evaluating economic impacts.

C. Determining the highest attainable use. If the use attainability analysis determines that the designated use is not attainable based on one of the factors in 40 CFR 131.10(g), the use attainability analysis shall then determine the highest attainable use for the protection and propagation of fish, shellfish and wildlife and recreation in and on the water based on methods described in Subsection B of this section.

D. Process to amend a designated use through a use attainability analysis.

(1) The process for developing a use attainability analysis and petitioning the commission for removing a designated use and establishing the highest attainable use shall be done in accordance with the State's current *Water Quality Management Plan/Continuing Planning Process*.

[~~C.~~](2) If the findings of a use attainability analysis, conducted by the department, [~~based on~~] in accordance with the department's *Hydrology Protocol* (latest edition)[~~, approved by the commission,~~] demonstrates

[~~to the satisfaction of the department~~]that federal Clean Water Act Section 101(a)(2) uses, that are not existing uses, are not feasible in an ephemeral water body due to the factor in 40 CFR 131.10(g)(2), the department may consider proceeding with the expedited use attainability analysis process in accordance with the State's current *Water Quality Management Plan/Continuing Planning Process*. The following elements must be met for the expedited use attainability analysis process to be authorized and implemented:

(a) The department is the primary investigator of the use attainability analysis;

(b) The use attainability analysis determined, through the application of the *Hydrology Protocol*, that the water being investigated is ephemeral and has no effluent discharges of sufficient volume that could compensate for the low-flow;

(c) The use attainability analysis determined that the existing uses of the water being investigated are not more stringent than those in 20.6.4.97 NMAC;

(d) The designated uses in 20.6.4.97 NMAC have been determined to be the highest attainable uses for the water being analyzed;

(e) The department [~~shall~~]posted the use attainability analysis on its water quality standards website and [~~notify~~] notified its interested parties list of a 30-day public comment period~~.~~ ;

(f) [~~After reviewing~~]The department reviewed and responded to any comments received during the 30-day public comment period~~,~~; and

(g) The department [~~may proceed by submitting~~]submitted the use attainability analysis and response to comments to region 6 EPA for technical approval.

If EPA approves the revision under section 303(c) of the Clean Water Act [~~technical approval is granted~~], the water shall be subject to 20.6.4.97 NMAC for federal Clean Water Act purposes. The use attainability analysis, the technical support document,[~~- approval,~~] and the applicability of 20.6.4.97 NMAC to the water shall be posted on the department's water quality standards website. The department shall periodically petition the commission to list ephemeral waters under Subsection C of 20.6.4.97 NMAC and to incorporate changes to classified segments as appropriate.

[~~D.~~]E. Use attainability analysis conducted by an entity other than the department. Any person may submit notice to the department stating their intent to conduct a use attainability analysis.

(1) The proponent shall provide such notice along with [~~develop~~]a work plan supporting [~~to conduct~~]the development of a use attainability analysis[~~. and shall submit the work plan~~] to the department and region 6 EPA for review and comment.

(2) Upon approval of the work plan by the department, the proponent shall conduct the use attainability analysis and implement public noticing in accordance with the approved work plan.

(3) Work plan elements. The work plan shall identify, at a minimum:

(a) the waterbody of concern and the reasoning for conducting a use attainability analysis;

(b) the [~~scope~~]source and validity of data [~~currently available and the scope of data to be gathered~~]to be used to demonstrate whether the current designated use is not attainable;[~~,~~]

(c) the factors in 40 CFR 131.10(g) affecting [~~use~~]the attainment of that use;

(d) [~~that will be analyzed~~] a description of the data being proposed to be used to demonstrate the highest attainable use;

(e) [~~and~~]the provisions for consultation with appropriate state and federal agencies;

(f) a description of how stakeholders and potentially affected tribes will be identified and engaged;

(g) a description of the public notice mechanisms to be employed; and [~~consultation with appropriate state and federal agencies~~]

(h) the expected timelines outlining the administrative actions to be taken for a rulemaking petition, pending the outcome of the use attainability analysis.

(4) [~~Upon approval of the work plan by the department, the proponent shall conduct the use attainability analysis in accordance with the approved work plan. The cost of such analysis shall be the responsibility of the proponent.~~] Upon completion of the use attainability analysis, the proponent shall submit the data, findings and conclusions to the department, and provide public notice of the use attainability analysis in accordance with the approved work plan.

(5) Pending the conclusions of the use attainability analysis and as described in the approved work plan, [~~T~~]the department or the proponent may petition the commission to modify the designated use [~~if the conclusions of the analysis support such action~~]. The cost of such use attainability analysis shall be the responsibility

of the proponent. Subsequent costs associated with the administrative rulemaking process shall be the responsibility of the petitioner.

[20.6.4.15 NMAC - Rp 20 NMAC 6.1.1107, 10/12/2000; Rn, 20.6.4.14 NMAC, 5/23/2005; A, 5/23/2005; A, 7/17/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.16 PLANNED USE OF A PISCICIDE: The use of a piscicide registered under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. Section 136 *et seq*., and under the New Mexico Pesticide Control Act (NMPCA), Section 76-4-1 *et seq*. NMSA 1978 (1973) in a surface water of the state, shall not be a violation of Subsection F of 20.6.4.13 NMAC when such use is covered by a federal national pollutant discharge elimination system (NPDES) permit or has been approved by the commission under procedures provided in this section. The use of a piscicide which is covered by a NPDES permit shall require no further review by the commission and the person whose application is covered by the NPDES permit shall meet the additional notification and monitoring requirements outlined in Subsection G of 20.6.4.16 NMAC. The commission may approve the reasonable use of a piscicide under this section if the proposed use is not covered by a NPDES permit to further a Clean Water Act objective to restore and maintain the physical or biological integrity of surface waters of the state, including restoration of native species.

A. Any person seeking commission approval of the use of a piscicide not covered by a NPDES permit shall file a written petition concurrently with the commission and the surface water bureau of the department. The petition shall contain, at a minimum, the following information:

(1) petitioner's name and address;

(2) identity of the piscicide and the period of time (not to exceed five years) or number of applications for which approval is requested;

(3) documentation of registration under FIFRA and NMPCA and certification that the petitioner intends to use the piscicide according to the label directions, for its intended function;

(4) target and potential non-target species in the treated waters and adjacent riparian area, including threatened or endangered species;

(5) potential environmental consequences to the treated waters and the adjacent riparian area, and protocols for limiting such impacts;

(6) surface water of the state proposed for treatment;

(7) results of pre-treatment survey;

(8) evaluation of available alternatives and justification for selecting piscicide use;

(9) documentation of notice requesting public comment on the proposed use within a 30-day period, including information as described in Paragraphs (1), (2) and (6) of Subsection A of 20.6.4.16 NMAC, provided to:

(a) local political subdivisions;

(b) local water planning entities;

(c) local conservancy and irrigation districts; and

(d) local media outlets, except that the petitioner shall only be required to publish notice in a newspaper of circulation in the locality affected by the proposed use.

(10) copies of public comments received in response to the publication of notice and the petitioner's responses to public comments received;

(11) post-treatment assessment monitoring protocol; and

(12) any other information required by the commission.

B. Within 30 days of receipt of the petition, the department shall review the petition and file a recommendation with the commission to grant, grant with conditions or deny the petition. The recommendation shall include reasons, and a copy shall be sent to the petitioner by certified mail.

C. The commission shall review the petition, the public comments received under Paragraphs (9) and (10) of Subsection A of 20.6.4.16 NMAC, the petitioner's responses to public comments and the department's technical recommendations for the petition. A public hearing shall be held if the commission determines there is substantial public interest. The commission shall notify the petitioner and those commenting on the petition of the decision whether to hold a hearing and the reasons therefore in writing.

D. If the commission determines there is substantial public interest a public hearing shall be held within 90 days of receipt of the department's recommendation in the locality affected by the proposed use in accordance with 20.1.3 NMAC, Adjudicatory Procedures - Water Quality Control Commission. Notice of the hearing shall be given in writing by the petitioner to individuals listed under Subsection A of 20.6.4.16 NMAC as well as to individuals who provided public comment under that subsection at least 30 days prior to the hearing.

E. In a hearing provided for in this section or, if no hearing is held, in a commission meeting, the registration of a piscicide under FIFRA and NMPCA shall provide a rebuttable presumption that the determinations of the EPA Administrator in registering the piscicide, as outlined in 7 U.S.C. Section 136a(c)(5), are valid. For purposes of this Section the rebuttable presumptions regarding the piscicide include:

(1) Its composition is such as to warrant the proposed claims for it;

(2) Its labeling and other material submitted for registration comply with the requirements of FIFRA and NMPCA;

(3) It will perform its intended function without unreasonable adverse effects on the environment; and

(4) When used in accordance with all FIFRA label requirements it will not generally cause unreasonable adverse effects on the environment.

(5) "Unreasonable adverse effects on the environment" has the meaning provided in FIFRA, 7 U.S.C. Section 136(bb): "any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of any pesticide."

F. After a public hearing, or commission meeting if no hearing is held, the commission may grant the petition in whole or in part, may grant the petition subject to conditions, or may deny the petition. In granting any petition in whole or part or subject to conditions, the commission shall require the petitioner to implement post-treatment assessment monitoring and provide notice to the public in the immediate and near downstream vicinity of the application prior to and during the application.

G. Any person whose application is covered by a NPDES permit shall provide written notice to local entities as described in Subsection A of 20.6.4.16 NMAC and implement post-treatment assessment monitoring within the application area as described in Subsection F of 20.6.4.16 NMAC.

[20.6.4.16 NMAC - Rn, Paragraph (6) of Subsection F of 20.6.4.12 NMAC, 5/23/2005; A, 5/23/2005; A, 3/2/2017]

20.6.4.17 - 20.6.4.49 [RESERVED]

20.6.4.50 BASINWIDE PROVISIONS - Special provisions arising from interstate compacts, international treaties or court decrees or that otherwise apply to a basin are contained in 20.6.4.51 through 20.6.4.59 NMAC.

[20.6.4.50 NMAC - N, 5/23/2005]

20.6.4.51 [RESERVED]

20.6.4.52 PECOS RIVER BASIN - In order to protect existing and designated uses, it is a goal of the state of New Mexico to prevent increases in TDS in the Pecos river above the following benchmark values, which are expressed as flow-weighted, annual average concentrations, at three USGS gaging stations: at Santa Rosa 500 mg/L; near Artesia 2,700 mg/L; and near Malaga 3,600 mg/L. The benchmark values serve to guide state action. They are adopted pursuant to the New Mexico Water Quality Act, not the Clean Water Act.

[20.6.4.52 NMAC - N, 12/1/2010]

20.6.4.53 [RESERVED]

20.6.4.54 COLORADO RIVER BASIN - For the tributaries of the Colorado river system, the state of New Mexico will cooperate with the Colorado river basin states and the federal government to support and implement the salinity policy and program outlined in the most current "review, water quality standards for salinity, Colorado river system" or equivalent report by the Colorado river salinity control forum.

A. Numeric criteria expressed as the flow-weighted annual average concentration for salinity are established at three points in the Colorado river basin as follows: below Hoover dam, 723 mg/L; below Parker dam, 747 mg/L; and at Imperial dam, 879 mg/L.

B. As a part of the program, objectives for New Mexico shall include the elimination of discharges of water containing solids in solution as a result of the use of water to control or convey fly ash from coal-fired electric generators, wherever practicable.

[20.6.4.54 NMAC - Rn, Paragraphs (1) through (3) of Subsection K of 20.6.4.12 NMAC, 5/23/2005; A, 5/23/2005]

20.6.4.55 - 20.6.4.96 [RESERVED]

20.6.4.97 EPHEMERAL WATERS: Ephemeral surface waters of the state as identified below and additional ephemeral waters as identified on the department's water quality standards website pursuant to Paragraph (2) of Subsection [~~C~~]D of 20.6.4.15 NMAC are subject to the designated uses and criteria as specified in this section. Ephemeral waters classified in 20.6.4.101-899 NMAC are subject to the designated uses and criteria as specified in those sections.

A. Designated uses: livestock watering, wildlife habitat, limited aquatic life and secondary contact.

B. Criteria: the use-specific criteria in 20.6.4.900 NMAC are applicable to the designated uses.

C. Waters:

(1) the following waters are designated in the Rio Grande basin:

(a) Cunningham gulch from Santa Fe county road 55 upstream 1.4 miles to a point upstream of the Lac minerals mine, identified as Ortiz mine on U.S. geological survey topographic maps;

(b) an unnamed tributary from Arroyo Hondo upstream 0.4 miles to the Village of Oshara water reclamation facility outfall;

(c) an unnamed tributary from San Pedro creek upstream 0.8 miles to the PAA-KO community sewer outfall;

(d) Inditos draw from the crossing of an unnamed road along a power line one-quarter mile west of McKinley county road 19 upstream to New Mexico highway 509;

(e) an unnamed tributary from the diversion channel connecting Blue canyon and Socorro canyon upstream 0.6 miles to the New Mexico firefighters academy treatment facility outfall;

(f) an unnamed tributary from the Albuquerque metropolitan arroyo flood control authority (AMAFCA) Rio Grande south channel upstream of the crossing of New Mexico highway 47 upstream to I-25;

(g) the south fork of Cañon del Piojo from [~~Canon~~]Cañon del Piojo upstream 1.2 miles to an unnamed tributary;

(h) an unnamed tributary from the south fork of Cañon del Piojo upstream 1 mile to the Resurrection mine outfall;

(i) Arroyo del Puerto from San Mateo creek upstream 6.8 miles to the Ambrosia Lake mine entrance road;

(j) an unnamed tributary from San Mateo creek upstream 1.5 miles to the Roca Honda mine facility outfall;

(k) San Isidro arroyo, including unnamed tributaries to San Isidro arroyo, from Arroyo Chico upstream to its headwaters;

(l) Arroyo Tinaja, including unnamed tributaries to Arroyo Tinaja, from San Isidro arroyo upstream to 2 miles northeast of the Cibola national forest boundary;

(m) Mulatto canyon from Arroyo Tinaja upstream to 1 mile northeast of the Cibola national forest boundary; and

(n) Doctor arroyo, including unnamed tributaries to Doctor arroyo, from San Isidro arroyo upstream to its headwaters, and excluding Doctor Spring and Doctor arroyo from the spring to its confluence with the unnamed tributary approximately one-half mile downstream of the spring.

(2) the following waters are designated in the Pecos river basin:

(a) an unnamed tributary from Hart canyon upstream 1 mile to South Union road;

(b) Aqua Chiquita from Rio Peñasco upstream to McEwan canyon; and

(c) Grindstone canyon upstream of Grindstone reservoir.

(3) the following waters are designated in the Canadian river basin:

(a) Bracket canyon upstream of the Vermejo river;

(b) an unnamed tributary from Bracket canyon upstream 2 miles to the Ancho mine; and

(c) Gachupin canyon from the Vermejo river upstream 2.9 miles to an unnamed west tributary near the Ancho mine outfall.

(4) in the San Juan river basin an unnamed tributary of Kim-me-ni-oli wash upstream of the mine outfall.

(5) the following waters are designated in the Little Colorado river basin:

(a) Defiance draw from County Road 1 to upstream of West Defiance Road; and

(b) an unnamed tributary of Defiance draw from McKinley county road 1 upstream to New Mexico highway 264.

(6) the following waters are designated in the closed basins:

(a) in the Tularosa river closed basin San Andres canyon downstream of South San Andres canyon; and

(b) in the Mimbres river closed basin San Vicente arroyo from the Mimbres river upstream to Maudes canyon.

[20.6.4.97 NMAC - N, 5/23/2005; A, 12/1/2010; A, 3/2/2017; A, 12/17/2019; A, XX/XX/XXXX]

20.6.4.98 INTERMITTENT WATERS: All non-perennial surface waters of the state, except those ephemeral waters included under section 20.6.4.97 NMAC or classified in 20.6.4.101-899 NMAC.

A. Designated uses: livestock watering, wildlife habitat, marginal warmwater aquatic life and primary contact.

B. Criteria: the use-specific criteria in 20.6.4.900 NMAC are applicable to the designated uses, except that the following site-specific criteria apply: the monthly geometric mean of E. coli bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

[20.6.4.98 NMAC - N, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.99 PERENNIAL WATERS: All perennial surface waters of the state except those classified in 20.6.4.101-899 NMAC.

A. Designated uses: Warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: The use-specific criteria in 20.6.4.900 NMAC are applicable to the designated uses, except that the following site-specific criteria apply: the monthly geometric mean of E. coli bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

[20.6.4.99 NMAC - N, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.100 [RESERVED]

20.6.4.101 RIO GRANDE BASIN: The main stem of the Rio Grande from the international boundary with Mexico upstream to one mile downstream of Percha dam.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses except that the following segment-specific criterion applies: temperature 34°C (93.2°F) or less.

(2) At mean monthly flows above 350 cfs, the monthly average concentration for: TDS 2,000 mg/L or less, sulfate 500 mg/L or less and chloride 400 mg/L or less.

C. Remarks: sustained flow in the Rio Grande below Caballo reservoir is dependent on release from Caballo reservoir during the irrigation season; at other times of the year, there may be little or no flow.

[20.6.4.101 NMAC - Rp 20 NMAC 6.1.2101, 10/12/2010; A, 12/15/2001; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.102 RIO GRANDE BASIN: The main stem of the Rio Grande from one mile downstream of Percha dam upstream to Caballo dam.

A. Designated uses: irrigation, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

C. Remarks: sustained flow in the Rio Grande downstream of Caballo reservoir is dependent on release from Caballo reservoir during the irrigation season; at other times of the year, there may be little or no flow.

[20.6.4.102 NMAC - Rp 20 NMAC 6.1.2102, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.103 RIO GRANDE BASIN: [-] [~~The main stem of the Rio Grande from the headwaters of Caballo reservoir upstream to Elephant Butte dam and~~ p]Perennial reaches of tributaries to the Rio Grande in Sierra and Socorro counties not specifically identified under other sections of 20.6.4 NMAC, excluding waters on tribal lands.

A. Designated uses: irrigation, livestock watering, wildlife habitat, marginal coldwater aquatic life, secondary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

~~**[C. Remarks:** flow in this reach of the Rio Grande main stem is dependent upon release from Elephant Butte dam.]~~

[20.6.4.103 NMAC - Rp 20 NMAC 6.1.2103, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

[NOTE: This segment was divided effective XX/XX/XXXX. The standards for the main stem of the Rio Grande from the headwaters of Caballo reservoir upstream to Elephant Butte dam, perennial reaches of Palomas creek, perennial reaches of Rio Salado, perennial reaches of Percha creek, perennial reaches of Alamosa creek, and perennial reaches of Abo arroyo are under 20.6.4.112 NMAC.]

20.6.4.104 RIO GRANDE BASIN: - Caballo and Elephant Butte reservoir.

A. Designated uses: irrigation storage, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.104 NMAC - Rp 20 NMAC 6.1.2104, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.105 RIO GRANDE BASIN: [-] The main stem of the Rio Grande from the headwaters of Elephant Butte reservoir upstream to Alameda bridge (Corrales bridge), excluding waters on Isleta pueblo.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, public water supply, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) At mean monthly flows above 100 cfs, the monthly average concentration for: TDS l,500 mg/L or less, sulfate 500 mg/L or less and chloride 250 mg/L or less.

(3) Criteria referenced in 20.6.2.2102 NMAC, Rio Grande basin-community sewerage systems, apply if the applicability conditions in 20.6.2.2100 NMAC are met.

[20.6.4.105 NMAC - Rp 20 NMAC 6.1.2105, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.106 RIO GRANDE BASIN: [-] The main stem of the Rio Grande from Alameda bridge (Corrales bridge) upstream to the Angostura diversion works, excluding waters on Santa Ana pueblo, and intermittent water in the Jemez river below the Jemez pueblo boundary, excluding waters on Santa Ana and Zia pueblos, that enters the main stem of the Rio Grande. Portions of the Rio Grande in this segment are under the joint jurisdiction of the state and Sandia pueblo.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and primary contact; and public water supply on the Rio Grande.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) At mean monthly flows above 100 cfs, the monthly average concentration for: TDS 1,500 mg/L or less, sulfate 500 mg/L or less and chloride 250 mg/L or less.

(3) Criteria referenced in 20.6.2.2102, NMAC Rio Grande basin-community sewerage systems, apply if the applicability conditions in 20.6.2.2100 NMAC are met.

[20.6.4.106 NMAC - Rp 20 NMAC 6.1.2105.1, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.107 RIO GRANDE BASIN: [-] The Jemez river from the Jemez pueblo boundary upstream to Soda dam near the town of Jemez Springs and perennial reaches of Vallecito creek.

A. Designated uses: coldwater aquatic life, primary contact, irrigation, livestock watering and wildlife habitat; and public water supply on Vallecito creek.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F).

[20.6.4.107 NMAC - Rp 20 NMAC 6.1.2105.5, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.108 RIO GRANDE BASIN: [-] Perennial reaches of the Jemez river upstream of Soda dam near the town of Jemez Springs and [all its]perennial reaches of tributaries to the Jemez river except those not specifically identified under other sections of 20.6.4 NMAC [above Soda dam near the town of Jemez Springs, except San Gregorio lake and Sulphur creek above its confluence with Redondo creek], and perennial reaches of the Guadalupe river and perennial reaches of [all its] tributaries to the Guadalupe river, and Calaveras canyon.

A. Designated uses: domestic water supply, fish culture, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 400 µS/cm or less (800 µS/cm or less on Sulphur creek); the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less; and pH within the range of 2.0 to 8.8 on Sulphur creek.

[20.6.4.108 NMAC - Rp 20 NMAC 6.1.2106, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012; A, XX/XX/XXXX]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional segment are under 20.6.4.124 NMAC. The standards for San Gregorio lake are in 20.6.4.134 NMAC, effective 7/10/2012]

20.6.4.109 RIO GRANDE BASIN: [-] Perennial reaches of Bluewater creek excluding Bluewater lake and waters on tribal lands, Rio Moquino upstream of Laguna pueblo, Seboyeta creek, Rio Paguate upstream of Laguna pueblo, the Rio Puerco upstream of the northern boundary of Cuba, and all other perennial reaches of tributaries to the Rio Puerco, including the Rio San Jose in Cibola county from the USGS gaging station at Correo upstream to Horace springs excluding waters on tribal lands.

A. Designated uses: coldwater aquatic life, domestic water supply, fish culture, irrigation, livestock watering, wildlife habitat and primary contact; and public water supply on La Jara creek.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: phosphorus (unfiltered sample) 0.1 mg/L or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.109 NMAC - Rp 20 NMAC 6.1.2107, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012]

[**NOTE:** The standards for Bluewater lake are in 20.6.4.135 NMAC, effective 7/10/2012]

20.6.4.110 RIO GRANDE BASIN: The main stem of the Rio Grande from Angostura diversion works upstream to Cochiti dam, excluding the reaches on San Felipe, Kewa and Cochiti pueblos.

A. Designated uses: irrigation, livestock watering, wildlife habitat, primary contact, coldwater aquatic life and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: pH within the range of 6.6 to 9.0 and temperature 25°C (77°F) or less.

[20.6.4.110 NMAC - Rp 20 NMAC 6.1.2108, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.111 RIO GRANDE BASIN: [-] Perennial reaches of Las Huertas creek from the San Felipe pueblo boundary to the headwaters.

A. Designated uses: high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.111 NMAC - Rp 20 NMAC 6.1.2108.5, 10/12/2000; A, 7/25/2001; A, 5/23/2005; A-12/1/2010]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional segment are under 20.6.4.125 NMAC.]

20.6.4.112 [[RESERVED]] RIO GRANDE BASIN: - The main stem of the Rio Grande from the headwaters of Caballo reservoir upstream to Elephant Butte dam, perennial reaches of Palomas creek, perennial reaches of Rio Salado, perennial reaches of Percha creek, perennial reaches of Alamosa creek, and perennial reaches of Abo arroyo.

A. Designated uses: irrigation, livestock watering, wildlife habitat, marginal coldwater aquatic life, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

C. Remarks: flow in this reach of the Rio Grande main stem is dependent upon release from Elephant Butte dam.

[20.6.4.112 NMAC - Rp 20 NMAC 6.1.2109, 10/12/2000; A, 5/23/2005; Repealed, 12/1/2010; A, XX/XX/XXXX]

20.6.4.113 RIO GRANDE BASIN: [-] The Santa Fe river and perennial reaches of its tributaries from the Cochiti pueblo boundary upstream to the outfall of the Santa Fe wastewater treatment facility.

A. Designated uses: irrigation, livestock watering, wildlife habitat, primary contact and coolwater aquatic life.

B. Criteria: The use-specific criteria in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 30°C (86°F) or less.

[20.6.4.113 NMAC - Rp 20 NMAC 6.1.2110, 10/12/2000; A, 10/11/2002; A, 5/23/2005; A, 12/1/2010; A, 2/14/2013]

20.6.4.114 RIO GRANDE BASIN: [-] The main stem of the Rio Grande from the Cochiti pueblo boundary upstream to Rio Pueblo de Taos excluding waters on San Ildefonso, Santa Clara and Ohkay Owingeh pueblos, Embudo creek from its mouth on the Rio Grande upstream to the Picuris Pueblo boundary, the Santa Cruz river from the Santa Clara pueblo boundary upstream to the Santa Cruz dam, the Rio Tesuque except waters on the Tesuque and Pojoaque pueblos, and the Pojoaque river from the San Ildefonso pueblo boundary upstream to the Pojoaque pueblo boundary. Some Rio Grande waters in this segment are under the joint jurisdiction of the state and San Ildefonso pueblo.

A. Designated uses: irrigation, livestock watering, wildlife habitat, marginal coldwater aquatic life, primary contact and warmwater aquatic life; and public water supply on the main stem Rio Grande.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: 6T3 temperature 22°C (71.6°F) and maximum temperature 25°C (78.8°F). In addition, the following criteria based on a 12-month rolling average are applicable to the public water supply use for monitoring and public disclosure purposes only:

Radionuclide	pCi/L
Americium-241	1.9
Cesium-137	6.4
Plutonium-238	1.5
Plutonium-239/240	1.5
Strontium-90	3.5
Tritium	4,000

(2) At mean monthly flows above 100 cfs, the monthly average concentration for: TDS 500 mg/L or less, sulfate 150 mg/L or less and chloride 25 mg/L or less.

[20.6.4.114 NMAC - Rp 20 NMAC 6.1.2111, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.115 RIO GRANDE BASIN: [-] The perennial reaches of Rio Vallecitos, [~~and its~~]perennial reaches of tributaries to Rio Vallecitos except Hopewell lake, and perennial reaches of Rio del Oso and perennial reaches of El Rito creek above the town of El Rito.

A. Designated uses: domestic water supply, irrigation, high quality coldwater aquatic life, livestock watering, wildlife habitat and primary contact; public water supply on the Rio Vallecitos and El Rito creek.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.115 NMAC - Rp 20 NMAC 6.1.2112, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012; A, XX/XX/XXXX]

[**NOTE:** The standards for Hopewell lake are in 20.6.4.134 NMAC, effective 7/10/2012]

20.6.4.116 RIO GRANDE BASIN: The Rio Chama from its mouth on the Rio Grande upstream to Abiquiu reservoir, perennial reaches of the Rio Tusas, perennial reaches of the Rio Ojo Caliente, perennial reaches of Abiquiu creek and perennial reaches of El Rito creek downstream of the town of El Rito.

A. Designated uses: irrigation, livestock watering, wildlife habitat, coldwater aquatic life, warmwater aquatic life and [~~secondary~~]primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 31°C (87.8°F) or less.

[20.6.4.116 NMAC - Rp 20 NMAC 6.1.2113, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017; A, XX/XX/XXXX]

20.6.4.117 RIO GRANDE BASIN: [-] Abiquiu reservoir.

A. Designated uses: irrigation storage, livestock watering, wildlife habitat, primary contact, coldwater aquatic life and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.117 NMAC - Rp 20 NMAC 6.1.2114, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.118 RIO GRANDE BASIN: [-] The Rio Chama from the headwaters of Abiquiu reservoir upstream to El Vado reservoir and perennial reaches of the Rio Gallina and Rio Puerco de Chama north of state highway 96. Some Rio Chama waters in this segment are under the joint jurisdiction of the state and the Jicarilla Apache tribe.

A. Designated uses: irrigation, livestock watering, wildlife habitat, coldwater aquatic life, warmwater aquatic life and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 26°C (78.8°F) or less.

[20.6.4.118 NMAC - Rp 20 NMAC 6.1.2115, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.119 RIO GRANDE BASIN: [-] All perennial reaches of tributaries to the Rio Chama above Abiquiu dam, except Canjilon lakes a, c, e and f and the Rio Gallina and Rio Puerco de Chama north of state highway 96 and excluding waters on Jicarilla Apache reservation, and the main stem of the Rio Chama from the headwaters of El Vado reservoir upstream to the New Mexico-Colorado line. Some Cañones creek and Rio Chama waters in this segment are under the joint jurisdiction of the state and the Jicarilla Apache tribe.

A. Designated uses: domestic water supply, fish culture, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact; and public water supply on the Rio Brazos and Rio Chama.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 500 µS/cm or less (1,000 µS or less for Coyote creek); the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.119 NMAC - Rp 20 NMAC 6.1.2116, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012]

[**NOTE:** The standards for Canjilon lakes a, c, e and f are in 20.6.4.134 NMAC, effective 7/10/2012]

20.6.4.120 RIO GRANDE BASIN: [-] El Vado and Heron reservoirs.

A. Designated uses: irrigation storage, livestock watering, wildlife habitat, public water supply, primary contact and coldwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.120 NMAC - Rp 20 NMAC 6.1.2117, 10/12/2000; A. 5/23/2005; A, 12/1/2010]

20.6.4.121 RIO GRANDE BASIN: [-] Perennial tributaries to the Rio Grande in Bandelier national monument and their headwaters in Sandoval county and all perennial reaches of tributaries to the Rio Grande in Santa Fe county unless included in other segments and excluding waters on tribal lands.

A. Designated uses: domestic water supply, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact; and public water supply on Little Tesuque creek, the Rio en Medio, and the Santa Fe river.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.121 NMAC - Rp 20 NMAC 6.1.2118, 10/12/2000; A. 5/23/2005; A, 12/1/2010; A, 2/14/2013]

[NOTE: The segment covered by this section was divided effective 5/23/2005. The standards for the additional segments are under 20.6.4.126, 20.6.4.127 and 20.6.4.128 NMAC.]

20.6.4.122 RIO GRANDE BASIN: [-] The main stem of the Rio Grande from Rio Pueblo de Taos upstream to the New Mexico-Colorado line, the Red river from its mouth on the Rio Grande upstream to the mouth of Placer creek, and the Rio Pueblo de Taos from its mouth on the Rio Grande upstream to the mouth of the Rio Grande del Rancho. Some Rio Grande and Rio Pueblo de Taos waters in this segment are under the joint jurisdiction of the state and Taos pueblo.

A. Designated uses: coldwater aquatic life, fish culture, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.122 NMAC - Rp 20 NMAC 6.1.2119, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.123 RIO GRANDE BASIN: [-] Perennial reaches of the Red river upstream of the mouth of Placer creek, all perennial reaches of tributaries to the Red river, and all other perennial reaches of tributaries to the Rio Grande in Taos and Rio Arriba counties unless included in other segments and excluding waters on Santa Clara, Ohkay Owingeh, Picuris and Taos pueblos.

A. Designated uses: domestic water supply, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact; and public water supply on the Rio Pueblo and Rio Fernando de Taos.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 400 µS/cm or less (500 µS/cm or less for the Rio Fernando de Taos); the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less; and phosphorus (unfiltered sample) less than 0.1 mg/L for the Red river.

[20.6.4.123 NMAC - Rp 20 NMAC 6.1.2120, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional segment are under 20.6.4.129 NMAC.]

20.6.4.124 RIO GRANDE BASIN: Perennial reaches of Sulphur creek from its confluence with Redondo creek upstream to its headwaters.

A. Designated uses: limited aquatic life, wildlife habitat, livestock watering and secondary contact.

B. Criteria: the use-specific criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: pH within the range of 2.0 to 9.0, maximum temperature 30ºC (86ºF), and the chronic aquatic life criteria of Subsections I and J of 20.6.4.900 NMAC.

[20.6.4.124 NMAC - N, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.125 RIO GRANDE BASIN: [-] Perennial reaches of San Pedro creek from the San Felipe pueblo boundary to the headwaters.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.125 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.126 RIO GRANDE BASIN: [-] Perennial portions of Cañon de Valle from Los Alamos national laboratory (LANL) stream gage E256 upstream to Burning Ground spring, Sandia canyon from Sigma

canyon upstream to LANL NPDES outfall 001, Pajarito canyon from Arroyo de La Delfe upstream into Starmers gulch and Starmers spring and Water canyon from Area-A canyon upstream to State Route 501.

A. Designated uses: coldwater aquatic life, livestock watering, wildlife habitat and secondary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.126 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.127 RIO GRANDE BASIN: [-] Perennial portions of Los Alamos canyon upstream from Los Alamos reservoir and Los Alamos reservoir.

A. Designated uses: coldwater aquatic life, livestock watering, wildlife habitat, irrigation and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.127 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.128 RIO GRANDE BASIN: [-] Ephemeral and intermittent [~~watercourses~~]waters within lands managed by U.S. department of energy (DOE) within LANL[-], including but not limited to: Mortandad canyon, Cañada del Buey, Ancho canyon, Chaquehui canyon, Indio canyon, Fence canyon, Potrillo canyon, and portions of Cañon de Valley, Los Alamos canyon, Sandia canyon, Pajarito canyon and Water canyon not specifically identified in 20.6.4.126 NMAC or 20.6.4.140 NMAC. (Surface waters within lands scheduled for transfer from DOE to tribal, state or local authorities are specifically excluded.)

A. Designated uses: livestock watering, wildlife habitat, limited aquatic life and secondary contact.

B. Criteria: the use-specific criteria in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the acute total ammonia criteria set forth in Subsection [~~K~~]L of 20.6.4.900 NMAC ([~~salmonids~~]*Oncorhynchus* spp. absent).

[20.6.4.128 NMAC - N, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

[**NOTE:** This section was divided effective XX/XX/XXXX. The standards for some intermittent waters within LANL are in 20.6.4.140 NMAC.]

20.6.4.129 RIO GRANDE BASIN: [-] Perennial reaches of the Rio Hondo.

A. Designated uses: domestic water supply, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact**.**

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 400 µS/cm or less and phosphorus (unfiltered sample) less than 0.1 mg/L.

[20.6.4.129 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.130 RIO GRANDE BASIN: [-] The Rio Puerco from the Rio Grande upstream to Arroyo Chijuilla, excluding the reaches on Isleta, Laguna and Cañoncito Navajo pueblos. Some waters in this segment are under the joint jurisdiction of the state and Isleta, Laguna or Cañoncito Navajo pueblos.

A. Designated uses: irrigation, warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) At mean monthly flows above 100 cfs, the monthly average concentration for: TDS l,500 mg/L or less, sulfate 500 mg/L or less and chloride 250 mg/L or less.

[20.6.4.130 NMAC - N, 12/1/2010]

20.6.4.131 RIO GRANDE BASIN: [-] The Rio Puerco from the confluence of Arroyo Chijuilla upstream to the northern boundary of Cuba.

A. Designated uses: warmwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.131 NMAC - N, 12/1/2010]

20.6.4.132 RIO GRANDE BASIN: [-] Rio Grande (Klauer) spring

A. Designated uses: domestic water supply, wildlife habitat, livestock watering, coldwater aquatic life use and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.132 NMAC - N, 12/1/2010]

20.6.4.133 RIO GRANDE BASIN: [-] Bull Creek lake, Cow lake, Elk lake, Goose lake, Heart lake, Hidden lake (Lake Hazel), Horseshoe lake, Horseshoe (Alamitos) lake, Jose Vigil lake, Lost lake, Middle Fork lake, Nambe lake, Nat II lake, Nat IV lake, No Fish lake, Pioneer lake, San Leonardo lake, Santa Fe lake, Serpent lake, South Fork lake, Trampas lakes (east and west) and Williams lake.

A. Designated uses: high quality coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.133 NMAC - N, 7/10/2012]

20.6.4.134 RIO GRANDE BASIN: [-] Cabresto lake, Canjilon lakes a, c, e and f, Fawn lakes (east and west), Hopewell lake and San Gregorio lake.

A. Designated uses: high quality coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.134 NMAC - N, 7/10/2012]

20.6.4.135 RIO GRANDE BASIN: [-] Bluewater lake.

A. Designated uses: coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses except that the following segment-specific criteria apply: phosphorus (unfiltered sample) 0.1 mg/L or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.135 NMAC - N, 7/10/2012]

20.6.4.136 RIO GRANDE BASIN: [-] The Santa Fe river from the outfall of the Santa Fe wastewater treatment facility to Guadalupe street.

A. Designated uses: limited aquatic life, wildlife habitat, primary contact, livestock watering, and irrigation.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.136 NMAC - N, 2/14/2013]

20.6.4.137 RIO GRANDE BASIN: [-] The Santa Fe river from Guadalupe street to Nichols reservoir.

A. Designated uses: coolwater aquatic life, wildlife habitat, primary contact, livestock watering, and irrigation.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.137 NMAC - N, 2/14/2013]

20.6.4.138 RIO GRANDE BASIN: [-] Nichols and McClure reservoirs.

A. Designated uses: high quality coldwater aquatic life, wildlife habitat, primary contact, public water supply and irrigation.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.
[20.6.4.138 NMAC - N, 2/14/2013]

20.6.4.139 RIO GRANDE BASIN: [-] Perennial reaches of Galisteo creek and perennial reaches of its tributaries from Kewa pueblo upstream to 2.2 miles upstream of Lamy.

A. Designated uses: coolwater aquatic life, primary contact, irrigation, livestock watering, domestic water supply and wildlife habitat; and public water supply on Cerrillos reservoir.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.
[20.6.4.139 NMAC - N, 2/14/2013]

20.6.4.140 RIO GRANDE BASIN: Effluent canyon from Mortandad canyon to its headwaters, intermittent portions of S-Site canyon from monitoring well MSC 16-06293 to Martin spring, and intermittent portions of Two-Mile canyon from its confluence with Pajarito canyon to Upper Two-Mile canyon. (Surface waters within lands scheduled for transfer from DOE to tribal, state or local authorities are specifically excluded.)

A. Designated uses: livestock watering, wildlife habitat, marginal warmwater aquatic life and secondary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.
[20.6.4.140 NMAC - N, XX/XX/XXXX]

20.6.4.~~[140]~~141 - 20.6.4.200 [RESERVED]

20.6.4.201 PECOS RIVER BASIN: [-] The main stem of the Pecos river from the New Mexico-Texas line upstream to the mouth of the Black river (near Loving).

A. Designated uses: irrigation, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria:

(l) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: dissolved boron for irrigation use 2,000 µg/L or less.

(2) At all flows above 50 cfs: TDS 20,000 mg/L or less, sulfate 3,000 mg/L or less and chloride 10,000 mg/L or less.
[20.6.4.201 NMAC - Rp 20 NMAC 6.1.2201, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.202 PECOS RIVER BASIN: [-] The main stem of the Pecos river from the mouth of the Black river upstream to lower Tansil dam, including perennial reaches of the Black river, the Delaware river and Blue spring.

A. Designated uses: industrial water supply, irrigation, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria:

(l) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 34°C (93.2°F) or less.

(2) At all flows above 50 cfs: TDS 8,500 mg/L or less, sulfate 2,500 mg/L or less and chloride 3,500 mg/L or less.

C. Remarks: diversion for irrigation frequently limits summer flow in this reach of the main stem Pecos river to that contributed by springs along the watercourse.
[20.6.4.202 NMAC - Rp 20 NMAC 6.1.2202, 10/12/2000; A, 5/23/2005; A, 12/1/2010]
[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for Lower Tansil Lake and Lake Carlsbad are under 20.6.4.218 NMAC.]

20.6.4.203 PECOS RIVER BASIN: [-] The main stem of the Pecos river from the headwaters of Lake Carlsbad upstream to Avalon dam.

A. Designated uses: industrial water supply, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: temperature 34°C (93.2°F) or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.203 NMAC - Rp 20 NMAC 6.1.2203, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for Lower Tansil Lake and Lake Carlsbad are under 20.6.4.218 and for Avalon Reservoir are under 20.6.4.219 NMAC.]

20.6.4.204 PECOS RIVER BASIN: [-] The main stem of the Pecos river from the headwaters of Avalon reservoir upstream to Brantley dam.

A. Designated uses: irrigation, livestock watering, wildlife habitat, [~~secondary~~]primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.204 NMAC - Rp 20 NMAC 6.1.2204, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for Avalon Reservoir are under 20.6.4.219 NMAC.]

20.6.4.205 PECOS RIVER BASIN: [-] Brantley reservoir.

A. Designated uses: irrigation storage, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.205 NMAC - Rp 20 NMAC 6.1.2205, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.206 PECOS RIVER BASIN: [~~The main stem of the Pecos river from the headwaters of Brantley reservoir upstream to Salt creek (near Acme), perennial reaches of the Rio Peñasco downstream from state highway 24 near Dunken, perennial reaches of the Rio Hondo and its~~]Perennial reaches of the Rio Felix and perennial reaches of tributaries to the Rio Hondo downstream of Bonney canyon, excluding North Spring river[~~, and perennial reaches of the Rio Felix~~].

A. Designated uses: irrigation, livestock watering, wildlife habitat, secondary contact and warmwater aquatic life.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) At all flows above 50 cfs: TDS 14,000 mg/L or less, sulfate 3,000 mg/L or less and chloride 6,000 mg/L or less.

[20.6.4.206 NMAC - Rp 20 NMAC 6.1.2206, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017; A, XX/XX/XXXX]

[**NOTE:** This segment was divided effective XX/XX/XXXX. The standards for the main stem of the Pecos river from the headwaters of Brantley reservoir upstream to Salt creek (near Acme), perennial reaches of the Rio Peñasco downstream from state highway 24 near Dunken, and perennial reaches of the Rio Hondo are under 20.6.4.231 NMAC.]

20.6.4.207 PECOS RIVER BASIN: [-] The main stem of the Pecos river from Salt creek (near Acme) upstream to Sumner dam.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and [~~secondary~~]primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) At all flows above 50 cfs: TDS 8,000 mg/L or less, sulfate 2,500 mg/L or less and chloride 4,000 mg/L or less.

[20.6.4.207 NMAC - Rp 20 NMAC 6.1.2207, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.208 PECOS RIVER BASIN: [-] Perennial reaches of the Rio Peñasco above state highway 24 near Dunken, [and its]perennial reaches of tributaries to the Rio Peñasco above state highway 24 near Dunken, perennial reaches of Cox canyon, perennial reaches of the Rio Bonito downstream from state highway 48 (near Angus), the Rio Ruidoso downstream of the U.S. highway 70 bridge near Seeping Springs lakes, perennial reaches of the Rio Hondo upstream from Bonney canyon and perennial reaches of Agua Chiquita.

A. Designated uses: fish culture, irrigation, livestock watering, wildlife habitat, coldwater aquatic life and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: temperature 30°C (86°F) or less, and phosphorus (unfiltered sample) less than 0.1 mg/L.

[20.6.4.208 NMAC - Rp 20 NMAC 6.1.2208, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.209 PECOS RIVER BASIN: [-] Perennial reaches of Eagle creek upstream of Alto dam to the Mescalero Apache boundary, perennial reaches of the Rio Bonito upstream of state highway 48 (near Angus) excluding Bonito lake, [and its]perennial reaches of tributaries to the Rio Bonito upstream of state highway 48 (near Angus)[-], [and] perennial reaches of the Rio Ruidoso upstream of the U.S. highway 70 bridge near Seeping Springs lakes[,] above and below the Mescalero Apache boundary and [its]perennial reaches of tributaries to the Rio Ruidoso upstream of the U.S. highway 70 bridge near Seeping Springs lakes[,] above and below the Mescalero Apache boundary.

A. Designated uses: domestic water supply, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat, public water supply and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 600 µS/cm or less in Eagle creek, 1,100 µS/cm or less in Bonito creek and 1,500 µS/cm or less in the Rio Ruidoso; phosphorus (unfiltered sample) less than 0.1 mg/L; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.209 NMAC - Rp 20 NMAC 6.1.2209, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012; A, XX/XX/XXXX]

[**NOTE:** The standards for Bonito lake are in 20.6.4.223 NMAC, effective 7/10/2012]

20.6.4.210 PECOS RIVER BASIN: [-] Sumner reservoir.

A. Designated uses: irrigation storage, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.210 NMAC - Rp 20 NMAC 6.1.2210, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.211 PECOS RIVER BASIN: [-] The main stem of the Pecos river from the headwaters of Sumner reservoir upstream to Tecolote creek excluding Santa Rosa reservoir.

A. Designated uses: fish culture, irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) At all flows above 50 cfs: TDS 3,000 mg/L or less, sulfate 2,000 mg/L or less and chloride 400 mg/L or less.

[20.6.4.211 NMAC - Rp 20 NMAC 6.1.2211, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012]

[**NOTE:** The standards for Santa Rosa reservoir are in 20.6.4.225 NMAC, effective 7/10/2012]

20.6.4.212 PECOS RIVER BASIN: [-] Perennial tributaries to the main stem of the Pecos river from the headwaters of Sumner reservoir upstream to Santa Rosa dam.

A. Designated uses: irrigation, coldwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.212 NMAC - Rp 20 NMAC 6.1.2211.1, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.213 PECOS RIVER BASIN: [-] McAllister lake.

A. Designated uses: coldwater aquatic life, secondary contact, livestock watering and wildlife habitat.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.213 NMAC - Rp 20 NMAC 6.1.2211.3, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.214 PECOS RIVER BASIN: [-] Storrie lake.

A. Designated uses: coldwater aquatic life, warmwater aquatic life, primary contact, livestock watering, wildlife habitat, public water supply and irrigation storage.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.214 NMAC - Rp 20 NMAC 6.1.2211.5, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.215 PECOS RIVER BASIN: [-] Perennial reaches of the Gallinas river upstream of the diversion for the Las Vegas municipal reservoir, [~~and all its~~]perennial reaches of tributaries to the Gallinas river upstream of the diversion for the Las Vegas municipal reservoir, perennial reaches of Tecolote creek upstream of Blue creek[~~,~~] and all perennial reaches of tributaries [~~of~~]to Tecolote creek upstream of Blue creek.

A. Designated uses: domestic water supply, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat, industrial water supply and primary contact; and public water supply on the Gallinas river.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less (450 µS/cm or less in Wright Canyon creek); the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.215 NMAC - Rp 20 NMAC 6.1.2212, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 2/13/2018; A, XX/XX/XXXX]

[**NOTE:** This segment was divided effective 2/13/2018. The standards for Tecolote creek from I-25 to Blue creek are under 20.6.4.230 NMAC.]

20.6.4.216 PECOS RIVER BASIN: [-] The main stem of the Pecos river from Tecolote creek upstream to Cañon de Manzanita.

A. Designated uses: irrigation, livestock watering, wildlife habitat, marginal coldwater aquatic life and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 30°C (86°F) or less.

(2) At all flows above 10 cfs: TDS 250 mg/L or less, sulfate 25 mg/L or less and chloride 5 mg/L or less.

[20.6.4.216 NMAC - Rp 20 NMAC 6.1.2213, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.217 PECOS RIVER BASIN: [-] Perennial reaches of Cow creek and all perennial reaches of its tributaries and the main stem of the Pecos river from Cañon de Manzanita upstream to its headwaters, including perennial reaches of all tributaries thereto except lakes identified in 20.6.4.222 NMAC.

A. Designated uses: domestic water supply, fish culture, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact; and public water supply on the main stem of the Pecos river.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.
[20.6.4.217 NMAC - Rp 20 NMAC 6.1.2214, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012]
[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional segments are under 20.6.4.220 and 20.6.4.221 NMAC.]

20.6.4.218 PECOS RIVER BASIN: [-] Lower Tansil lake and Lake Carlsbad.

A. Designated uses: industrial water supply, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 34°C (93.2°F) or less.
[20.6.4.218 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.219 PECOS RIVER BASIN: [-] Avalon reservoir.

A. Designated uses: irrigation storage, livestock watering, wildlife habitat, secondary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.
[20.6.4.219 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.220 PECOS RIVER BASIN: [-] Perennial reaches of the Gallinas river and ~~[its]~~perennial reaches of tributaries to the Gallinas river from its mouth upstream to the diversion for the Las Vegas municipal reservoir, except Pecos Arroyo.

A. Designated uses: irrigation, livestock watering, wildlife habitat, marginal coldwater aquatic life and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 30°C (86°F) or less.
[20.6.4.220 NMAC - N, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.221 PECOS RIVER BASIN: [-] Pecos Arroyo.

A. Designated uses: livestock watering, wildlife habitat, warmwater aquatic life and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 206 cfu/100 mL, single sample 940 cfu/100 mL.
[20.6.4.221 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.222 PECOS RIVER BASIN: [-] Johnson lake, Katherine lake, Lost Bear lake, Pecos Baldy lake, Spirit lake, Stewart lake and Truchas lakes (north and south).

A. Designated uses: high quality coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.
[20.6.4.222 NMAC - N, 7/10/2012]

20.6.4.223 PECOS RIVER BASIN: [-] Bonito lake.

A. Designated uses: high quality coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering, wildlife habitat and public water supply.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses except that the following segment-specific criteria apply: specific conductance 1100 µS/cm or less; phosphorus (unfiltered sample) less than 0.1 mg/L; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.
[20.6.4.223 NMAC - N, 7/10/2012]

20.6.4.224 PECOS RIVER BASIN: [-] Monastery lake.

A. Designated uses: coolwater aquatic life, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

[20.6.4.224 NMAC - N, 7/10/2012]

20.6.4.225 PECOS RIVER BASIN: [-] Santa Rosa reservoir.

A. Designated uses: coolwater aquatic life, irrigation, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.225 NMAC - N, 7/10/2012]

20.6.4.226 PECOS RIVER BASIN: [-] Perch lake.

A. Designated uses: coolwater aquatic life, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.226 NMAC - N, 7/10/2012]

20.6.4.227 PECOS RIVER BASIN: [-] Lea lake.

A. Designated uses: warmwater aquatic life, primary contact and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.227 NMAC - N, 7/10/2012]

20.6.4.228 PECOS RIVER BASIN: [-] Cottonwood lake and Devil's Inkwell.

A. Designated uses: coolwater aquatic life, primary contact and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

[20.6.4.228 NMAC - N, 7/10/2012]

20.6.4.229 PECOS RIVER BASIN: [-] Mirror lake.

A. Designated uses: warmwater aquatic life, primary contact and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

[20.6.4.229 NMAC - N, 7/10/2012]

20.6.4.230 PECOS RIVER BASIN: [-] Perennial reaches of Tecolote creek from I-25 to Blue creek.

A. Designated uses: domestic water supply, coolwater aquatic life, irrigation, livestock watering, wildlife habitat, and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.230 NMAC - N, 2/13/2018]

20.6.4.231 PECOS RIVER BASIN: The main stem of the Pecos river from the headwaters of Brantley reservoir upstream to Salt creek (near Acme), perennial reaches of the Rio Peñasco downstream from state highway 24 near Dunken, perennial reaches of North Spring river and perennial reaches of the Rio Hondo downstream of Bonney canyon.

A. Designated uses: irrigation, livestock watering, wildlife habitat, primary contact and warmwater aquatic life.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) At all flows above 50 cfs: TDS 14,000 mg/L or less, sulfate 3,000 mg/L or less and chloride 6,000 mg/L or less.

[N, XX/XX/XXXX]

[~~20.6.4.231~~]20.6.4.232 - 20.6.4.300 [RESERVED]

20.6.4.301 CANADIAN RIVER BASIN: [-] The main stem of the Canadian river from the New Mexico-Texas line upstream to Ute dam, and any flow that enters the main stem from Revuelto creek.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) TDS 6,500 mg/L or less at flows above 25 cfs.

[20.6.4.301 NMAC - Rp 20 NMAC 6.1.2301, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.302 CANADIAN RIVER BASIN: [-] Ute reservoir.

A. Designated uses: livestock watering, wildlife habitat, public water supply, industrial water supply, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.302 NMAC - Rp 20 NMAC 6.1.2302, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.303 CANADIAN RIVER BASIN: [-] The main stem of the Canadian river from the headwaters of Ute reservoir upstream to Conchas dam, the perennial reaches of Pajarito and Ute creeks and their perennial tributaries.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.303 NMAC - Rp 20 NMAC 6.1.2303, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.304 CANADIAN RIVER BASIN: [-] Conchas reservoir.

A. Designated uses: irrigation storage, livestock watering, wildlife habitat, public water supply, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.304 NMAC - Rp 20 NMAC 6.1.2304, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.305 CANADIAN RIVER BASIN: The main stem of the Canadian river from the headwaters of Conchas reservoir upstream to the New Mexico-Colorado line, perennial reaches of the Conchas river, the Mora river downstream from the USGS gaging station near Shoemaker, the Vermejo river downstream from Rail canyon and perennial reaches of Raton, Chicorica (except Lake Maloya and Lake Alice) and Uña de Gato creeks.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) TDS 3,500 mg/L or less at flows above 10 cfs.

[20.6.4.305 NMAC - Rp 20 NMAC 6.1.2305, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

[**NOTE:** This segment was divided effective 12/1/2010. The standards for Lake Alice and Lake Maloya are under 20.6.4.311 and 20.6.4.312 NMAC, respectively.]

20.6.4.306 CANADIAN RIVER BASIN: [-] The Cimarron river downstream from state highway 21 in Cimarron to the Canadian river and all perennial reaches of tributaries to the Cimarron river downstream from state highway 21 in Cimarron.

A. Designated uses: irrigation, warmwater aquatic life, livestock watering, wildlife habitat and primary contact; and public water supply on Cimarroncito creek.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

(2) TDS 3,500 mg/L or less at flows above 10 cfs.

[20.6.4.306 NMAC - Rp 20 NMAC 6.1.2305.1, 10/12/2000; A, 7/19/2001; A, 5/23/2005; A, 12/1/2010]

20.6.4.307 CANADIAN RIVER BASIN: [-] Perennial reaches of the Mora river from the USGS gaging station near Shoemaker upstream to the state highway 434 bridge in Mora, all perennial reaches of tributaries to the Mora river downstream from the USGS gaging station at La Cueva in San Miguel and Mora counties except lakes identified in 20.6.4.313 NMAC, perennial reaches of Ocate creek downstream of Ocate, [~~and its~~]perennial reaches of tributaries to Ocate creek downstream of Ocate, and perennial reaches of Rayado creek downstream of Miami lake diversion in Colfax county.

A. Designated uses: marginal coldwater aquatic life, warmwater aquatic life, primary contact, irrigation, livestock watering and wildlife habitat.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.307 NMAC - Rp 20 NMAC 6.1.2305.3, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012; A, XX/XX/XXXX]

20.6.4.308 CANADIAN RIVER BASIN: [-] Charette lakes.

A. Designated uses: coldwater aquatic life, warmwater aquatic life, secondary contact, livestock watering and wildlife habitat.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.308 NMAC - Rp 20 NMAC 6.1.2305.5, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.309 CANADIAN RIVER BASIN: [-] The Mora river and perennial reaches of its tributaries upstream from the state highway 434 bridge in Mora except lakes identified in 20.6.4.313 NMAC, all perennial reaches of tributaries to the Mora river upstream from the USGS gaging station at La Cueva, perennial reaches of Coyote creek, [~~and its~~]perennial reaches of tributaries to Coyote creek, the Cimarron river above state highway 21 in Cimarron, [~~and its~~]perennial reaches of tributaries to the Cimarron river above state highway 21 in Cimarron except Eagle Nest lake, all perennial reaches of tributaries to the Cimarron river north and northwest of highway 64 except north and south Shuree ponds, perennial reaches of Rayado creek above Miami lake diversion, [~~and its~~]perennial reaches of tributaries to Rayado creek above Miami lake diversion, Ocate creek and perennial reaches of its tributaries upstream of Ocate, perennial reaches of the Vermejo river upstream from Rail canyon and all other perennial reaches of tributaries to the Canadian river northwest and north of U.S. highway 64 in Colfax county unless included in other segments.

A. Designated uses: domestic water supply, irrigation, high quality coldwater aquatic life, livestock watering, wildlife habitat, and primary contact; and public water supply on the Cimarron river upstream from Cimarron, [~~and~~]on perennial reaches of Rayado creek and on perennial reaches of [~~its~~]tributaries to Rayado creek.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 500 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.309 NMAC - Rp 20 NMAC 6.1.2306, 10/12/2000; A, 7/19/2001; A, 5/23/2005; A, 12/1/2010; A, 7/10/2012; A, XX/XX/XXXX]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional segment are under 20.6.4.310 NMAC. The standards for Shuree ponds are in 20.6.4.314 NMAC and the standards for Eagle Nest lake are in 20.6.4.315 NMAC, effective 7/10/2012]

20.6.4.310 CANADIAN RIVER BASIN: [-] Perennial reaches of Corrumpa creek.

A. Designated uses: livestock watering, wildlife habitat, irrigation, primary contact and coldwater aquatic life.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: temperature 25°C (77°F) or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

(2) TDS 1,200 mg/L or less, sulfate 600 mg/L or less, chloride 40 mg/L or less.

[20.6.4.310 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.311 CANADIAN RIVER BASIN: Lake Alice.

A. Designated uses: marginal coldwater aquatic life, irrigation, livestock watering, wildlife habitat, primary contact and public water supply.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.311 NMAC - N, 12/1/2010; A, XX/XX/XXXX]

20.6.4.312 CANADIAN RIVER BASIN: Lake Maloya.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, wildlife habitat, primary contact and public water supply.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.312 NMAC - N, 12/1/2010; A, XX/XX/XXXX]

20.6.4.313 CANADIAN RIVER BASIN: [-] Encantada lake, Maestas lake, Middle Fork lake of Rio de la Casa, North Fork lake of Rio de la Casa and Pacheco lake.

A. Designated uses: high quality coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.313 NMAC - N, 7/10/2012]

20.6.4.314 CANADIAN RIVER BASIN: [-] Shuree ponds (north and south).

A. Designated uses: high quality coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses except that the following segment-specific criteria apply: specific conductance 500 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.314 NMAC - N, 7/10/2012]

20.6.4.315 CANADIAN RIVER BASIN: [-] Eagle Nest lake.

A. Designated uses: high quality coldwater aquatic life, irrigation, domestic water supply, primary contact, livestock watering, wildlife habitat and public water supply.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses except that the following segment-specific criteria apply: specific conductance 500 µS/cm or less; the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.315 NMAC - N, 7/10/2012]

20.6.4.316 CANADIAN RIVER BASIN: [-] Clayton lake.

A. Designated uses: coolwater aquatic life, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

[20.6.4.316 NMAC - N, 7/10/2012]

20.6.4.317 CANADIAN RIVER BASIN: Springer lake.

A. Designated uses: coolwater aquatic life, irrigation, primary contact, livestock watering, wildlife habitat, and public water supply.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.317 NMAC - N, 07-10-2012; A, 3/2/2017]

20.6.4.318 CANADIAN RIVER BASIN: Doggett creek.

A. Designated uses: Warm water aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: The use-specific criteria in 20.6.4.900 NMAC are applicable to the designated uses, except that the following site-specific criteria apply: the monthly geometric mean of E. coli bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

C. Discharger-specific temporary standard:

(1) Discharger: City of Raton wastewater treatment plant

(2) NPDES permit number: NM0020273, Outfall 001

(3) Receiving waterbody: Doggett creek, 20.6.4.318 NMAC

(4) Discharge latitude/longitude: 36° 52' 13.91" N / 104° 25' 39.18" W

(5) Pollutant(s): nutrients; total nitrogen and total phosphorus

(6) Factor of issuance: substantial and widespread economic and social impacts (40 CFR 131.10(g)(6))

(7) Highest attainable condition: interim effluent condition of 8.0 mg/L total nitrogen and 1.6 mg/L total phosphorus as 30-day averages. The highest attainable condition shall be either the highest attainable condition identified at the time of the adoption, or any higher attainable condition later identified during any reevaluation, whichever is more stringent (40 CFR 131.14(b)(1)(iii)).

(8) Effective date of temporary standard: This temporary standard becomes effective for Clean Water Act purposes on the date of EPA approval.

(9) Expiration date of temporary standard: no later than 20 years from the effective date.

(10) Reevaluation period: at each succeeding review of water quality standards and at least once every five years from the effective date of the temporary standard (Paragraph (8) of Subsection H of 20.6.4.10[~~.F (8)~~] NMAC, 40 CFR 131.14(b)(1)(v)). If the discharger cannot demonstrate that sufficient progress has been made the commission may revoke approval of the temporary standard or provide additional conditions to the approval of the temporary standard. If the reevaluation is not completed at the frequency specified or the Department does not submit the reevaluation to EPA within 30 days of completion, the underlying designated use and criterion will be the applicable water quality standard for Clean Water Act purposes until the Department completes and submits the reevaluation to EPA. Public input on the reevaluation will be invited during NPDES permit renewals or triennial reviews, as applicable, in accordance with the State's most current approved water quality management plan and continuing planning process.

(11) Timeline for proposed actions. Tasks and target completion dates are listed in the most recent, WQCC-approved version of the New Mexico Environment Department, Surface Water Quality Bureau's "Nutrient Temporary Standards for City of Raton Wastewater Treatment Plant, NPDES No. NM0020273 to Doggett Creek."

[20.6.4.318 NMAC - N, 05/22/2020; A, XX/XX/XXXX]

20.6.4.319 - 20.6.4.400 [RESERVED]

20.6.4.401 SAN JUAN RIVER BASIN: [-] The main stem of the San Juan river from the Navajo Nation boundary at the Hogback upstream to its confluence with the Animas river. Some waters in this segment are under the joint jurisdiction of the state and the Navajo Nation.

A. Designated uses: public water supply, industrial water supply, irrigation, livestock watering, wildlife habitat, primary contact, marginal coldwater aquatic life and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 32.2°C (90°F) or less.

[20.6.4.401 NMAC - Rp 20 NMAC 6.1.2401, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional

segment are under 20.6.4.408 NMAC.]

20.6.4.402 SAN JUAN RIVER BASIN: [-] La Plata river from its confluence with the San Juan river upstream to the New Mexico-Colorado line.

A. Designated uses: irrigation, marginal warmwater aquatic life, marginal coldwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 32.2°C (90°F) or less.

[20.6.4.402 NMAC - Rp 20 NMAC 6.1.2402, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.403 SAN JUAN RIVER BASIN: The Animas river from its confluence with the San Juan river upstream to Estes arroyo.

A. Designated uses: Public water supply, industrial water supply, irrigation, livestock watering, wildlife habitat, coolwater aquatic life, and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 29°C (84.2°F) or less.

[20.6.4.403 NMAC - Rp 20 NMAC 6.1.2403, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.404 SAN JUAN RIVER BASIN: The Animas river from Estes arroyo upstream to the Southern Ute Indian tribal boundary.

A. Designated uses: Coolwater aquatic life, irrigation, livestock watering, wildlife habitat, public water supply, industrial water supply and primary contact.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: phosphorus (unfiltered sample) 0.l mg/L or less.

[20.6.4.404 NMAC - Rp 20 NMAC 6.1.2404, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.405 SAN JUAN RIVER BASIN: [-] The main stem of the San Juan river from [~~Canyon~~]Cañon Largo upstream to the Navajo dam.

A. Designated uses: high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat, public water supply, industrial water supply and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 400 µS/cm or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.405 NMAC - Rp 20 NMAC 6.1.2405, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.406 SAN JUAN RIVER BASIN: [-] Navajo reservoir in New Mexico.

A. Designated uses: coldwater aquatic life, warmwater aquatic life, irrigation storage, livestock watering, wildlife habitat, public water supply, industrial water supply and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: phosphorus (unfiltered sample) 0.1 mg/L or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.406 NMAC - Rp 20 NMAC 6.1.2406, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.407 SAN JUAN RIVER BASIN: [-] Perennial reaches of the Navajo river from the Jicarilla Apache reservation boundary to the Colorado border and perennial reaches of Los Pinos river in New Mexico.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, public water supply, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: phosphorus (unfiltered sample) 0.1 mg/L or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.407 NMAC - Rp 20 NMAC 6.1.2407, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.408 SAN JUAN RIVER BASIN: [-] The main stem of the San Juan river from its confluence with the Animas river upstream to its confluence with [~~Canyon~~]Cañon Largo.

A. Designated uses: public water supply, industrial water supply, irrigation, livestock watering, wildlife habitat, primary contact, marginal coldwater aquatic life and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 32.2°C (90°F) or less.

[20.6.4.408 NMAC - N, 5/23/2005; A, 12/1/2010; A, XX/XX/XXXX]

20.6.4.409 SAN JUAN RIVER BASIN: [-] Lake Farmington.

A. Designated uses: public water supply, wildlife habitat, livestock watering, primary contact, coldwater aquatic life and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.409 NMAC - N, 12/1/2010]

20.6.4.410 SAN JUAN RIVER BASIN: [-] Jackson lake.

A. Designated uses: coolwater aquatic life, irrigation, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 206 cfu/100 mL or less, single sample 940 cfu/100 mL or less.

[20.6.4.410 NMAC - N, 7/10/2012]

20.6.4.411 - 20.6.4.450: [RESERVED]

20.6.4.451 LITTLE COLORADO RIVER BASIN: [-] The Rio Nutria upstream of the Zuni pueblo boundary, Tampico draw, Agua Remora, Tampico springs.

A. Designated uses: coolwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.451 NMAC - N, 12/1/2010]

20.6.4.452 LITTLE COLORADO RIVER BASIN: [-] Ramah lake.

A. Designated uses: coldwater aquatic life, warmwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.452 NMAC - N, 12/1/2010]

20.6.4.453 LITTLE COLORADO RIVER BASIN: [-] Quemado lake.

A. Designated uses: coolwater aquatic life, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.453 NMAC - N, 7/10/2012]

20.6.4.454 - 20.6.4.500 [RESERVED]

20.6.4.501 GILA RIVER BASIN: [-] The main stem of the Gila river from the New Mexico-Arizona line upstream to Redrock canyon and perennial reaches of streams in Hidalgo county.

A. Designated uses: irrigation, marginal warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.501 NMAC - Rp 20 NMAC 6.1.2501, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.502 GILA RIVER BASIN: The main stem of the Gila river from Redrock canyon upstream to the confluence of the West Fork Gila river and East Fork Gila river and perennial reaches of tributaries to the Gila river downstream of Mogollon creek.

A. Designated uses: industrial water supply, irrigation, livestock watering, wildlife habitat, marginal coldwater aquatic life, primary contact and warmwater aquatic life.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: 28°C (82.4°F) or less.

[20.6.4.502 NMAC - Rp 20 NMAC 6.1.2502, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.503 GILA RIVER BASIN: All perennial tributaries to the Gila river upstream of and including Mogollon creek.

A. Designated uses: domestic water supply, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance of 400 µS/cm or less for all perennial tributaries except West Fork Gila and tributaries thereto, specific conductance of 300 µS/cm or less; 32.2°C (90°F) or less in the east fork of the Gila river and Sapillo creek downstream of Lake Roberts; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.503 NMAC - Rp 20 NMAC 6.1.2503, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.504 GILA RIVER BASIN: [-] Wall lake, Lake Roberts and Snow lake.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: specific conductance 300 µS/cm or less.

[20.6.4.504 NMAC - Rp 20 NMAC 6.1.2504, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional segment are under 20.6.4.806 NMAC.]

20.6.4.505 GILA RIVER BASIN: [-] Bill Evans lake.

A. Designated uses: coolwater aquatic life, primary contact, livestock watering and wildlife habitat.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.505 NMAC - N, 7/10/2012]

20.6.4.506 - 20.6.4.600 [RESERVED]

20.6.4.601 SAN FRANCISCO RIVER BASIN: [-] The main stem of the San Francisco river from the New Mexico-Arizona line upstream to state highway 12 at Reserve and perennial reaches of Mule creek.

A. Designated uses: irrigation, marginal warmwater and marginal coldwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.601 NMAC - Rp 20 NMAC 6.1.2601, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.602 SAN FRANCISCO RIVER BASIN: [-] The main stem of the San Francisco river from state highway 12 at Reserve upstream to the New Mexico-Arizona line.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: temperature 25°C (77°F) or less.

[20.6.4.602 NMAC - Rp 20 NMAC 6.1.2602, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.603 SAN FRANCISCO RIVER BASIN: [-] All perennial reaches of tributaries to the San Francisco river above the confluence of Whitewater creek and including Whitewater creek.

A. Designated uses: domestic water supply, fish culture, high quality coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 400 µS/cm or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less; and temperature 25°C (77°F) or less in Tularosa creek.

[20.6.4.603 NMAC - Rp 20 NMAC 6.1.2603, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.604 - 20.6.4.700 [RESERVED]

20.6.4.701 DRY CIMARRON RIVER: [-] Perennial portions of the Dry Cimarron river above Oak creek and perennial reaches of Oak creek.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: temperature 25°C (77°F) or less, the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

(2) TDS 1,200 mg/L or less, sulfate 600 mg/L or less and chloride 40 mg/L or less.

[20.6.4.701 NMAC - Rp 20 NMAC 6.1.2701, 10/12/2000; A, 5/23/2005 A, 12/1/2010]

[**NOTE:** The segment covered by this section was divided effective 5/23/2005. The standards for the additional segment are under 20.6.4.702 NMAC.]

20.6.4.702 DRY CIMARRON RIVER: [-] Perennial portions of the Dry Cimarron river below Oak creek, and perennial portions of Long canyon and Carrizozo creeks.

A. Designated uses: coolwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria:

(1) The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of *E. coli* bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

(2) TDS 1,200 mg/L or less, sulfate 600 mg/L or less and chloride 40 mg/L or less.

[20.6.4.702 NMAC - N, 5/23/2005; A, 12/1/2010; A, 7/10/2012]

20.6.4.703 - 20.6.4.800 [RESERVED]

20.6.4.801 CLOSED BASINS: [-] Rio Tularosa upstream of the old U.S. highway 70 bridge crossing east of Tularosa and all perennial tributaries to the Tularosa basin except Three Rivers and Dog Canyon creek, and excluding waters on the Mescalero tribal lands.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, wildlife habitat, public water supply and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.801 NMAC - Rp 20 NMAC 6.1.2801, 10/12/2000; A, 5/23/2005; A, 12/1/2010; A, 2/13/2018]

[**NOTE:** This segment was divided effective 2/13/2018. The standards for Dog Canyon creek are under 20.6.4.810 NMAC.]

20.6.4.802 CLOSED BASINS: [-] Perennial reaches of Three Rivers.

A. Designated uses: irrigation, domestic water supply, high quality coldwater aquatic life, primary contact, livestock watering and wildlife habitat.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 500 µS/cm or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.802 NMAC - Rp 20 NMAC 6.1.2802, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.803 CLOSED BASINS: Perennial reaches of the Mimbres river downstream of the confluence with Allie canyon and all perennial reaches of tributaries thereto.

A. Designated uses: Coolwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less and temperature of 30°C (86°F) or less.

[20.6.4.803 NMAC - Rp 20 NMAC 6.1.2803, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017]

20.6.4.804 CLOSED BASINS: Perennial reaches of the Mimbres river upstream of the confluence with Allie canyon to Cooney canyon, and all perennial reaches of East Fork Mimbres (McKnight canyon) downstream of the fish barrier, and all perennial reaches thereto.

A. Designated uses: Irrigation, domestic water supply, coldwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.804 NMAC - Rp 20 NMAC 6.1.2804, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 02-28-2018; A, 3/2/2017]

[**NOTE:** The segment covered by this section was divided effective 3/2/2017. The standards for the additional segment are covered under 20.6.4.807 NMAC.]

20.6.4.805 CLOSED BASINS: [-] Perennial reaches of the Sacramento river (Sacramento-Salt Flat closed basin) and all perennial tributaries thereto.

A. Designated uses: domestic water supply, livestock watering, wildlife habitat, marginal coldwater aquatic life and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses.

[20.6.4.805 NMAC - Rp 20 NMAC 6.1.2805, 10/12/2000; A, 5/23/2005; A, 12/1/2010]

20.6.4.806 CLOSED BASINS: [-] Bear canyon reservoir.

A. Designated uses: coldwater aquatic life, irrigation, livestock watering, wildlife habitat and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criterion applies: specific conductance 300 µS/cm or less.

[20.6.4.806 NMAC - N, 5/23/2005; A, 12/1/2010]

20.6.4.807 CLOSED BASINS: Perennial reaches of the Mimbres river upstream of Cooney canyon and all perennial reaches thereto, including perennial reaches of East Fork Mimbres river (McKnight canyon) upstream of the fish barrier.

A. Designated uses: Irrigation, domestic water supply, high quality coldwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: specific conductance 300 µS/cm or less; the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.807 NMAC - N, 3/2/2017]

20.6.4.808 CLOSED BASINS: Perennial and intermittent watercourses within Smelter Tailing Soils Investigation Unit lands at the Chino mines company, excluding those ephemeral waters listed in 20.6.4.809 NMAC and including, but not limited to[-] the mainstem of Lampbright draw, beginning at the confluence of Lampbright Draw with Rustler canyon, all tributaries that originate west of Lampbright draw to the intersection of Lampbright draw with U.S. 180, and all tributaries of Whitewater creek that originate east of Whitewater creek from the confluence of Whitewater creek with Bayard canyon downstream to the intersection of Whitewater creek with U.S. 180.

A. Designated uses: Warmwater aquatic life, livestock watering, wildlife habitat and primary contact.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the acute and chronic aquatic life criteria for copper set forth in Subsection I of 20.6.4.900 NMAC shall be determined by multiplying that criteria by the water effect ratio ("WER") adjustment expressed by the following equation:

$$WER = \frac{[10^{0.588+(0.703 \times \log DOC)+(0.395 \times \log Alkalinity)}] \times (\frac{100}{Hardness})^{0.9422}}{19.31}$$

For purposes of this section, dissolved organic carbon (DOC) is expressed in units of milligrams carbon per liter or mg C/L; alkalinity is expressed in units of mg/L as $CaCO_3$, and hardness is expressed in units of mg/L as $CaCO_3$. In waters that contain alkalinity concentrations greater than 250 mg/L, a value of 250 mg/L shall be used in the equation. In waters that contain DOC concentrations greater than 16 mg C/L, a value of 16 mg C/L shall be used in the equation. In waters that contain hardness concentrations greater than 400 mg/L, a value of 400 mg/L shall be used in the equation. The alkalinity, hardness and DOC concentrations used to calculate the WER value are those measured in the subject water sample.

[20.6.4.808 NMAC - N, 3/2/2017]

20.6.4.809 CLOSED BASINS: Ephemeral watercourses within smelter tailing soils investigation unit lands at the Chino mines company, limited to Chino mines property subwatershed drainage A and tributaries thereof, Chino mines property subwatershed drainage B and tributaries thereof (excluding the northwest tributary containing Ash spring and the Chiricahua leopard frog critical habitat transect); Chino mines property subwatershed drainage C and tributaries thereof (excluding reaches containing Bolton spring, the Chiricahua leopard frog critical habitat transect and all reaches in subwatershed C that are upstream of the Chiricahua leopard frog critical habitat); subwatershed drainage D and tributaries thereof (drainages D-1, D-2 and D-3, excluding the southeast tributary in drainage D1 that contains Brown spring) and subwatershed drainage E and all tributaries thereof (drainages E-1, E-2 and E-3).

A. Designated uses: Limited aquatic life, livestock watering, wildlife habitat and secondary contact.

B. Criteria: The use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the acute aquatic life criteria for copper set forth in Subsection I of 20.6.4.900 NMAC shall be determined by multiplying that criteria by the water effect ratio ("WER") adjustment expressed by the following equation:

$$WER = \frac{[10^{0.588+(0.703 \times \log DOC)+(0.395 \times \log Alkalinity)}] \times (\frac{100}{Hardness})^{0.9422}}{19.31}$$

For purposes of this section, dissolved organic carbon (DOC) is expressed in units of milligrams carbon per liter or mg C/L; alkalinity is expressed in units of mg/L as $CaCO_3$, and hardness is expressed in units of mg/L as $CaCO_3$. In waters that contain alkalinity concentrations greater than 250 mg/L, a value of 250 mg/L shall be used in the equation. In waters that contain DOC concentrations greater than 16 mg C/L, a value of 16 mg C/L shall be used in the equation. In waters that contain hardness concentrations greater than 400 mg/L, a value of 400 mg/L shall be used in the equation. The alkalinity, hardness and DOC concentrations used to calculate the WER value are those measured in the subject water sample.

[20.6.4.809 NMAC - N, 3/2/2017]

20.6.4.810 CLOSED BASINS: Perennial reaches of Dog Canyon creek.

A. Designated uses: coolwater aquatic life, irrigation, livestock watering, wildlife habitat, public water supply, and primary contact.

B. Criteria: the use-specific numeric criteria set forth in 20.6.4.900 NMAC are applicable to the designated uses, except that the following segment-specific criteria apply: the monthly geometric mean of E. coli bacteria 126 cfu/100 mL or less, single sample 235 cfu/100 mL or less.

[20.6.4.810 NMAC - N, 2/13/2018]

20.6.4.811 - 20.6.4.899 [RESERVED]

20.6.4.900 CRITERIA APPLICABLE TO EXISTING, DESIGNATED OR ATTAINABLE USES UNLESS OTHERWISE SPECIFIED IN 20.6.4.97 THROUGH 20.6.4.899 NMAC:

A. Fish culture and water supply: Fish culture, public water supply and industrial water supply are designated uses in particular classified waters of the state where these uses are actually being realized. However, no numeric criteria apply uniquely to these uses. Water quality adequate for these uses is ensured by the general criteria and numeric criteria for bacterial quality, pH and temperature.

B. Domestic water supply: Surface waters of the state designated for use as domestic water supplies shall not contain substances in concentrations that create a lifetime cancer risk of more than one cancer per 100,000 exposed persons. Those criteria listed under domestic water supply in Subsection J of this section apply to this use.

C. Irrigation and irrigation storage: the following numeric criteria and those criteria listed under irrigation in Subsection J of this section apply to this use:

(1)	dissolved selenium	0.13 mg/L
(2)	dissolved selenium in presence of >500 mg/L SO_4	0.25 mg/L.

D. Primary contact: The monthly geometric mean of E. coli bacteria of 126 cfu/100 mL or MPN/100 ml, [~~and~~]a single sample of E. coli bacteria of 410 cfu/100 mL or MPN/100 mL, a single sample of total microcystins of 8 µg/L with no more than three exceedances within a 12-month period and a single sample of cylindrospermopsin of 15 µg/L with no more than three exceedances within a 12-month period, and pH within the range of 6.6 to 9.0 apply to this use. The results for *E. coli* may be reported as either colony forming units (CFU) or the most probable number (MPN) depending on the analytical method used.

E. Secondary contact: The monthly geometric mean of E. coli bacteria of 548 cfu/100 mL or MPN/100 mL and single sample of 2507 cfu/100 mL or MPN/100 mL apply to this use. The results for *E. coli* may be reported as either colony forming units (CFU) or the most probable number (MPN), depending on the analytical method used.

F. Livestock watering: the criteria listed in Subsection J of this section for livestock watering apply to this use.

G. Wildlife habitat: Wildlife habitat shall be free from any substances at concentrations that are toxic to or will adversely affect plants and animals that use these environments for feeding, drinking, habitat or propagation; can bioaccumulate; or might impair the community of animals in a watershed or the ecological integrity of surface waters of the state. The numeric criteria listed in Subsection J for wildlife habitat apply to this use.

H. Aquatic life: Surface waters of the state with a designated, existing or attainable use of aquatic life shall be free from any substances at concentrations that can impair the community of plants and animals in or the ecological integrity of surface waters of the state. Except as provided in Paragraph (7) of this subsection, the acute and chronic aquatic life criteria set out in Subsections I, J, K and L of this section and the human health-organism only criteria set out in Subsection J of this section are applicable to all aquatic life use subcategories. In addition, the specific criteria for aquatic life subcategories in the following paragraphs apply to waters classified under the respective designations.

(1) High quality coldwater: dissolved oxygen 6.0 mg/L or more, 4T3 temperature 20°C (68°F), maximum temperature 23°C (73°F), pH within the range of 6.6 to 8.8 and specific conductance a segment-specific limit between 300 µS/cm and 1,500 µS/cm depending on the natural background in the particular surface water of the state (the intent of this criterion is to prevent excessive increases in dissolved solids which would result in changes in community structure). Where a single segment-specific temperature criterion is indicated in 20.6.4.101-899 NMAC, it is the maximum temperature and no 4T3 temperature applies.

(2) Coldwater: dissolved oxygen 6.0 mg/L or more, 6T3 temperature 20°C (68°F), maximum temperature 24°C (75°F) and pH within the range of 6.6 to 8.8. Where a single segment-specific temperature criterion is indicated in 20.6.4.101-899 NMAC, it is the maximum temperature and no 6T3 temperature applies.

(3) Marginal coldwater: dissolved oxygen 6 mg/L or more, 6T3 temperature 25°C (77°F), maximum temperature 29°C (84°F) and pH within the range from 6.6 to 9.0. Where a single segment-specific temperature criterion is indicated in 20.6.4.101-899 NMAC, it is the maximum temperature and no 6T3 temperature applies.

(4) Coolwater: dissolved oxygen 5.0 mg/L or more, maximum temperature 29°C (84°F) and pH within the range of 6.6 to 9.0.

(5) Warmwater: dissolved oxygen 5 mg/L or more, maximum temperature 32.2°C (90°F) and pH within the range of 6.6 to 9.0. Where a segment-specific temperature criterion is indicated in 20.6.4.101-899 NMAC, it is the maximum temperature.

(6) Marginal warmwater: dissolved oxygen 5 mg/L or more, pH within the range of 6.6 to 9.0 and [~~maximum~~]temperatures that may routinely exceed 32.2°C (90°F). Where a segment-specific temperature criterion is indicated in 20.6.4.101-899 NMAC, it is the maximum temperature.

(7) Limited aquatic life: The acute aquatic life criteria of Subsections I and J of this section apply to this subcategory. Chronic aquatic life criteria do not apply unless adopted on a segment-specific basis. Human health-organism only criteria apply only for persistent toxic pollutants unless adopted on a segment-specific basis.

I. Hardness-dependent acute and chronic aquatic life criteria for metals are calculated using the following equations. The criteria are expressed as a function of dissolved hardness (as mg $CaCO_3$/L). With the exception of aluminum, the equations are valid only for dissolved hardness concentrations of 0-400 mg/L. For dissolved hardness concentrations above 400 mg/L, the criteria for 400 mg/L apply. For aluminum the equations are valid only for dissolved hardness concentrations of 0-220 mg/L. For dissolved hardness concentrations above 220 mg/L, the aluminum criteria for 220 mg/L apply.

(1) Acute aquatic life criteria for metals: The equation to calculate acute criteria in µg/L is $\exp(m_A[\ln(\text{hardness})] + b_A)(CF)$. Except for aluminum, the criteria are based on analysis of dissolved metal. For aluminum, the criteria are based on analysis of total recoverable aluminum in a sample that has a pH between 6.5 and 9.0 and is filtered to minimize mineral phases as specified by the department. [~~The EPA has disapproved the hardness-based equation for total recoverable aluminum in waters where the pH is less than 6.5 in the receiving stream for federal purposes of the Clean Water Act.~~]The equation parameters are as follows:

Metal	**m_A**	**b_A**	**Conversion factor (CF)**
Aluminum (Al)	1.3695	1.8308	
Cadmium (Cd)	[~~0.8968~~]0.9789	[~~-3.5699~~]-3.866	1.136672-[(ln hardness)(0.041838)]
Chromium (Cr) III	0.8190	3.7256	0.316
Copper (Cu)	0.9422	-1.700	0.960
Lead (Pb)	1.273	-1.460	1.46203-[(ln hardness)(0.145712)]
Manganese (Mn)	0.3331	6.4676	
Nickel (Ni)	0.8460	2.255	0.998
Silver (Ag)	1.72	-6.59	0.85
Zinc (Zn)	0.9094	0.9095	0.978

(2) Chronic aquatic life criteria for metals: The equation to calculate chronic criteria in µg/L is $\exp(m_C[\ln(\text{hardness})] + b_C)(CF)$. Except for aluminum, the criteria are based on analysis of dissolved metal. For aluminum, the criteria are based on analysis of total recoverable aluminum in a sample that has a pH between 6.5 and 9.0 and is filtered to minimize mineral phases as specified by the department. [~~The EPA has disapproved the hardness-based equation for total recoverable aluminum in waters where the pH is less than 6.5 in the receiving stream for federal purposes of the Clean Water Act.~~]The equation parameters are as follows:

Metal	**m_C**	**b_C**	**Conversion factor (CF)**
Aluminum (Al)	1.3695	0.9161	
Cadmium (Cd)	[~~0.7647~~]0.7977	[~~-4.2180~~]-3.909	1.101672-[(ln hardness)(0.041838)]
Chromium (Cr) III	0.8190	0.6848	0.860
Copper (Cu)	0.8545	-1.702	0.960
Lead (Pb)	1.273	-4.705	1.46203-[(ln hardness)(0.145712)]
Manganese (Mn)	0.3331	5.8743	
Nickel (Ni)	0.8460	0.0584	0.997
Zinc (Zn)	0.9094	0.6235	0.986

(3) Selected values of calculated acute and chronic criteria (µg/L).

Hardness as $CaCO_3$, dissolved (mg/L)		**Al**	**Cd**	**Cr III**	**Cu**	**Pb**	**Mn**	**Ni**	**Ag**	**Zn**
25	Acute	512	[~~0.51~~]0.49	~~180~~[183]	4	14	1,881	~~140~~ [145]	0.3	45
	Chronic	205	[~~0.17~~]0.25	24	3	1	1,040	16		34
30	Acute	658	[~~0.59~~]0.58	[~~210~~]213	4	17	1,999	[~~170~~]169	0.4	54

Hardness as $CaCO_3$, dissolved (mg/L)		Al	Cd	Cr III	Cu	Pb	Mn	Ni	Ag	Zn
	Chronic	263	[~~0.19~~] 0.29	28	3	1	1,105	19		41
40	Acute	975	0.76	[~~270~~] 269	6	24	2,200	[~~220~~] 216	0.7	70
	Chronic	391	[~~0.23~~] 0.36	35	4	1	1,216	24		53
50	Acute	1,324	[~~0.91~~] 0.94	[~~320~~] 323	7	30	2,370	260	1.0	85
	Chronic	530	[~~0.28~~] 0.43	42	5	1	1,309	29		65
60	Acute	1,699	[~~1.07~~] 1.11	[~~370~~] 375	8	37	2,519	[~~300~~] 304	1.3	101
	Chronic	681	[~~0.31~~] 0.49	49	6	1	1,391	34		76
70	Acute	2,099	[~~1.22~~] 1.29	[~~430~~] 425	10	44	2,651	[~~350~~] 346	1.7	116
	Chronic	841	[~~0.35~~] 0.55	55	7	2	1,465	38		88
80	Acute	2,520	[~~1.37~~] 1.46	[~~470~~] 475	11	51	2,772	[~~390~~] 388	2.2	131
	Chronic	1,010	[~~0.39~~] 0.61	62	7	2	1,531	43		99
90	Acute	2,961	[~~1.51~~] 1.63	[~~520~~] 523	12	58	2,883	[~~430~~] 428	2.7	145
	Chronic	1,186	[~~0.42~~] 0.66	68	8	2	1,593	48		110
100	Acute	3,421	[~~1.65~~] 1.79	570	13	65	2,986	[~~470~~] 468	3.2	160
	Chronic	1,370	[~~0.45~~] 0.72	74	9	3	1,650	52		121
200	Acute	8,838	[~~2.98~~] 3.43	[~~1,010~~] 1,005	26	[~~140~~] 136	3,761	[~~840~~] 842	[~~11~~] 10.6	301
	Chronic	3,541	[~~0.75~~] 1.21	[~~130~~] 131	16	5	2,078	[~~90~~] 93		228
220	Acute	10,071	[~~3.23~~] 3.75	1,087	28	151	3,882	912	[~~13~~] 12.5	328
	Chronic	4,035	[~~0.80~~] 1.30	141	18	6	2,145	101		248
300	Acute		[~~4.21~~] 5.00	[~~1,400~~] 1,401	38	[~~210~~] 209	4,305	[~~1190~~] 1,186	[~~21~~] 21.3	435
	Chronic		[~~1.00~~] 1.64	[~~180~~] 182	23	8	2,379	[~~130~~] 132		329
400 and above	Acute		[~~5.38~~] 6.54	[~~1,770~~] 1,773	50	[~~280~~] 281	4,738	[~~1510~~] 1,513	[~~35~~] 34.9	564
	Chronic		[~~1.22~~] 2.03	[~~230~~] 231	29	11	2,618	[~~170~~] 168		428

J. Use-specific numeric criteria.

(1) Table of numeric criteria: The following table sets forth the numeric criteria applicable to existing, designated and attainable uses. For metals, criteria represent the total sample fraction unless otherwise

specified in the table. Additional criteria that are not compatible with this table are found in Subsections A through I, K and L of this section.

Pollutant	CAS Number	DWS	Irr/Irr storage	LW	WH	Aquatic Life			Type
						Acute	Chronic	HH-OO	
Aluminum, dissolved	7429-90-5		5,000			750 i	87 i		
Aluminum, total recoverable	7429-90-5					a	a		
Antimony, dissolved	7440-36-0	6						640	P
Arsenic, dissolved	7440-38-2	10	100	200		340	150	9.0	C,P
Asbestos	1332-21-4	7,000,000 fibers/L							
Barium, dissolved	7440-39-3	2,000							
Beryllium, dissolved	7440-41-7	4							
Boron, dissolved	7440-42-8		750	5,000					
Cadmium, dissolved	7440-43-9	5	10	50		a	a		
Chloride	1688-70-06					860,000	230,000		
Chlorine residual	7782-50-5				11	19	11		
Chromium III, dissolved	16065-83-1					a	a		
Chromium VI, dissolved	18540-29-9					16	11		
Chromium, dissolved	7440-47-3	100	100	1,000					
Cobalt, dissolved	7440-48-4		50	1,000					
Copper, dissolved	7440-50-8	1300	200	500		a	a		
Cyanide, total recoverable	57-12-5	200			5.2	22.0	5.2	[~~140~~] 400	
Iron	7439-89-6						1,000		
Lead, dissolved	7439-92-1	15	5,000	100		a	a		
Manganese, dissolved	7439-96-5					a	a	___	
Mercury	7439-97-6	2		10	0.77				
Mercury, dissolved	7439-97-6					1.4	0.77		
Methylmercury	22967-92-6							0.3 mg/kg in fish tissue	P
Molybdenum, dissolved	7439-98-7		1,000						
Molybdenum, total recoverable	7439-98-7					7,920	1,895		
Nickel, dissolved	7440-02-0	700				a	a	4,600	P
Nitrate as N		10 mg/L							
Nitrite + Nitrate				132 mg/L					
Selenium, dissolved	7782-49-2	50	b	50				4,200	P
Selenium, total recoverable	7782-49-2				5.0	20.0	5.0		
Silver, dissolved	7440-22-4					a			
Thallium, dissolved	7440-28-0	2						0.47	P
Uranium, dissolved	7440-61-1	30							
Vanadium, dissolved	7440-62-2		100	100					
Zinc, dissolved	7440-66-6	10,500	2,000	25,000		a	a	26,000	P
Adjusted gross alpha		15 pCi/L		15 pCi/L					
Radium 226 + Radium 228		5 pCi/L		30.0 pCi/L					
Strontium 90		8 pCi/L							

Pollutant	CAS Number	DWS	Irr/Irr storage	LW	WH	Aquatic Life			Type
						Acute	Chronic	HH-OO	
Tritium		20,000 pCi/L		20,000 pCi/L					
Acenaphthene	83-32-9	2,100						[990] 90	
Acrolein	107-02-8	18				3.0	3.0	[9] 400	
Acrylonitrile	107-13-1	0.65						[2.5] 70	C
Aldrin	309-00-2	0.021				3.0		[0.00050] 0.0000077	C,P
Anthracene	120-12-7	10,500						[40,000] 400	
Benzene	71-43-2	5						[510] 160	C
Benzidine	92-87-5	0.0015						[0.0020] 0.11	C
Benzo(a)anthracene	56-55-3	0.048						[0.18] 0.013	C
Benzo(a)pyrene	50-32-8	0.2						[0.18] 0.0013	C,P
Benzo(b)fluoranthene	205-99-2	0.048						[0.18] 0.013	C
Benzo(k)fluoranthene	207-08-9	0.048						[0.18] 0.13	C
alpha-BHC	319-84-6	0.056						[0.049] 0.0039	C
beta-BHC	319-85-7	0.091						[0.17] 0.14	C
[C]gamma-BHC (Lindane)	58-89-9	0.20				0.95		[1.8] 4.4	
Bis(2-chloroethyl) ether	111-44-4	0.30						[5.3] 22	C
Bis([2-chloroisopropyl] 2-chloro-1-methylethyl) ether	108-60-1	1,400						[65,000] 4,000	
Bis(2-ethylhexyl) phthalate	117-81-7	6						[22] 3.7	C
Bis(chloromethyl) ether	542-88-1							0.17	C
Bromoform	75-25-2	44						[1,400] 1,200	C
Butylbenzyl phthalate	85-68-7	7,000						[1,900] 1	C
Carbaryl	63-25-2					2.1	2.1		
Carbon tetrachloride	56-23-5	5						[16] 50	C
Chlordane	57-74-9	2				2.4	0.0043	[0.0081] 0.0032	C,P
Chlorobenzene	108-90-7	100						[1,600] 800	
Chlorodibromomethane	124-48-1	4.2						[130] 210	C
Chloroform	67-66-3	57						[4,700] 2,000	[C]
Chlorpyrifos	2921-88-2					0.083	0.041		
2-Chloronaphthalene	91-58-7	2,800						[1,600] 1,000	
2-Chlorophenol	95-57-8	175						[150] 800	
Chrysene	218-01-9	0.048						[0.18] 1.3	C
Demeton	8065-48-3						0.1		

Pollutant	CAS Number	DWS	Irr/Irr storage	LW	WH	Aquatic Life			Type
						Acute	Chronic	HH-OO	
Diazinon	333-41-5					0.17	0.17		
2,4-Dichlorophenoxyacetic acid	94-75-7							12,000	
Dichlorodiphenyldichloroethane (DDD)	72-54-8							0.0012	C
Dichlorodiphenyldichloroethylene (DDE)	72-55-9							0.00018	C
Dichlorodiphenyltrichloroethane (DDT)	50-29-3							0.0003	C,P
4,4'-DDT and derivatives		1.0			0.001	1.1	0.001	[0.0022]	[C,P]
Dibenzo(a,h)anthracene	53-70-3	0.048						[0.18] 0.0013	C
Dibutyl phthalate	84-74-2	3,500						[4,500] 30	
1,2-Dichlorobenzene	95-50-1	600						[1,300] 3,000	
1,3-Dichlorobenzene	541-73-1	469						[960] 10	
1,4-Dichlorobenzene	106-46-7	75						[190] 900	
3,3'-Dichlorobenzidine	91-94-1	0.78						[0.28] 1.5	C
Dichlorobromomethane	75-27-4	5.6						[170] 270	C
1,2-Dichloroethane	107-06-2	5						[370] 6,500	C
1,1-Dichloroethylene	75-35-4	7						[7,100] 20,000	[C]
2,4-Dichlorophenol	120-83-2	105						[290] 60	
1,2-Dichloropropane	78-87-5	5.0						[150] 310	C
1,3-Dichloropropene	542-75-6	3.5						[210] 120	C
Dieldrin	60-57-1	0.022				0.24	0.056	[0.00054] 0.000012	C,P
Diethyl phthalate	84-66-2	28,000						[44,000] 600	
Dimethyl phthalate	131-11-3	350,000						[1,100,000] 2,000	
2,4-Dimethylphenol	105-67-9	700						[850] 3,000	
Dinitrophenols	25550-58-7							1,000	
2,4-Dinitrophenol	51-28-5	70						[5,300] 300	
2,4-Dinitrotoluene	121-14-2	1.1						[34] 17	C
Dioxin	1746-01-6	3.0E-05						5.1E-08	C,P
1,2-Diphenylhydrazine	122-66-7	0.44						2.0	C
alpha-Endosulfan	959-98-8	62				0.22	0.056	[89] 30	
beta-Endosulfan	33213-65-9	62				0.22	0.056	[89] 40	
Endosulfan sulfate	1031-07-8	62						[89] 40	
Endrin	72-20-8	2				0.086	0.036	[0.060] 0.03	
Endrin aldehyde	7421-93-4	10.5						[0.30] 1	
Ethylbenzene	100-41-4	700						[2,100] 130	
Fluoranthene	206-44-0	1,400						[140] 20	
Fluorene	86-73-7	1,400						[5,300] 70	

Pollutant	CAS Number	DWS	Irr/Irr storage	LW	WH	Aquatic Life			Type
						Acute	Chronic	HH-OO	
Guthion	86-50-0						0.01		
Heptachlor	76-44-8	0.40				0.52	0.0038	[0.00079] 0.000059	C
Heptachlor epoxide	1024-57-3	0.20				0.52	0.0038	[0.00039] 0.00032	C
Hexachlorobenzene	118-74-1	1						[0.0029] 0.00079	C,P
Hexachlorobutadiene	87-68-3	4.5						[180] 0.1	C
Hexachlorocyclohexane (HCH)-Technical	608-73-1							0.1	C
Hexachlorocyclopen-tadiene	77-47-4	50						[1,100] 4	
Hexachloroethane	67-72-1	25						[33] 1	C
Ideno(1,2,3-cd)pyrene	193-39-5	0.048						[0.18] 0.013	C
Isophorone	78-59-1	368						[9,600] 18,000	C
Malathion	121-75-5						0.1		
Methoxychlor	72-43-5						0.03	0.02	
Methyl bromide	74-83-9	49						[1,500] 10,000	
3-Methyl-4-chlorophenol	59-50-7							2,000	
2-Methyl-4,6-dinitrophenol	534-52-1	14						[280] 30	
Methylene chloride	75-09-2	5						[5,900] 10,000	C
Mirex	2385-85-5						0.001		
Nitrobenzene	98-95-3	18						[690] 600	
Nitrosamines	Various							12.4	C
Nitrosodibutylamine	924-16-3							2.2	C
Nitrosodiethylamine	55-18-5							12.4	C
N-Nitrosodimethylamine	62-75-9	0.0069						30	C
N-Nitrosodi-n-propylamine	621-64-7	0.050						5.1	C
N-Nitrosodiphenylamine	86-30-6	71						60	C
N-Nitrosopyrrolidine	930-55-2							340	C
Nonylphenol	84852-15-3					28	6.6		
Parathion	56-38-2					0.065	0.013		
[Polychlorinated Biphenyls (PCBs)]	[1336-36-3]	[0.50]			[0.014]	[2]	[0.014]	[0.00064]	[C,P]
Pentachlorobenzene	608-93-5							0.1	
Pentachlorophenol	87-86-5	1.0				19	15	[30] 0.4	C
Phenol	108-95-2	10,500						[860,000] 300,000	
Polychlorinated Biphenyls (PCBs)	1336-36-3	0.50			0.014	2	0.014	0.00064	C,P
Pyrene	129-00-0	1,050						[4,000] 30	
1,2,4,5-Tetrachlorobenzene	95-94-3							0.03	

Pollutant	CAS Number	DWS	Irr/Irr storage	LW	WH	Aquatic Life			Type
						Acute	Chronic	HH-OO	
1,1,2,2-Tetrachloroethane	79-34-5	1.8						[~~40~~] 30	C
Tetrachloroethylene	127-18-4	5						[~~33~~] 290	C,P
Toluene	108-88-3	1,000						[~~15,000~~] 520	
Toxaphene	8001-35-2	3				0.73	0.0002	[~~0.0028~~] 0.0071	C
1,2-Trans-dichloroethylene	156-60-5	100						[~~10,000~~] 4,000	
Tributyltin (TBT)	Various					0.46	0.072		
1,2,4-Trichlorobenzene	120-82-1	70						[~~70~~] 0.76	C
1,1,1-Trichloroethane	71-55-6	200						200,000	
1,1,2-Trichloroethane	79-00-5	5						[~~160~~] 89	C
Trichloroethylene	79-01-6	5						[~~300~~] 70	C
2,4,5-Trichlorophenol	95-95-4							600	
2,4,6-Trichlorophenol	88-06-2	32						[~~24~~] 28	C
2-(2,4,5-Trichlorophenoxy)propionic acid (Silvex)	93-72-1							400	
Vinyl chloride	75-01-4	2						[~~24~~] 16	C

(2) Notes applicable to the table of numeric criteria in Paragraph (1) of this subsection.

(a) Where the letter "a" is indicated in a cell, the criterion is hardness-based and can be referenced in Subsection I of 20.6.4.900 NMAC.

(b) Where the letter "b" is indicated in a cell, the criterion can be referenced in Subsection C of 20.6.4.900 NMAC.

(c) Criteria are in µg/L unless otherwise indicated.

(d) Abbreviations are as follows: CAS - chemical abstracts service (see definition for "CAS number" in 20.6.4.7 NMAC); DWS - domestic water supply; Irr/Irr storage- irrigation [~~or~~]and irrigation storage; LW - livestock watering; WH - wildlife habitat; HH-OO - human health-organism only; C – criteria based on cancer-causing endpoint; P - persistent toxic pollutant.

(e) The criteria are based on analysis of an unfiltered sample unless otherwise indicated. The acute and chronic aquatic life criteria for aluminum are based on analysis of total recoverable aluminum in a sample that is filtered to minimize mineral phases as specified by the department.

(f) The criteria listed under human health-organism only (HH-OO) are intended to protect human health when aquatic organisms are consumed from waters containing pollutants. These criteria do not protect the aquatic life itself; rather, they protect the health of humans who ingest fish or other aquatic organisms.

(g) The dioxin criteria apply to the sum of the dioxin toxicity equivalents expressed as 2,3,7,8-TCDD dioxin.

(h) The criteria for polychlorinated biphenyls (PCBs) apply to the sum of all congeners, to the sum of all homologs or to the sum of all aroclors.

(i) The acute and chronic aquatic life criteria for dissolved aluminum only apply when the concurrent pH is less than 6.6 or greater than 9.0 S.U. If the concurrent pH is between 6.6 and 9.0 S.U. then the hardness-dependent total recoverable aluminum criteria in Paragraphs (1) and (2) of Subsection I of 20.6.4.900 NMAC apply.

~~**[K.** Acute aquatic life criteria for total ammonia are dependent on pH and the presence or absence of salmonids. The criteria in mg/L as N based on analysis of unfiltered samples are as follows:~~

~~pH~~	~~Where Salmonids Present~~	~~Where Salmonids Absent~~
~~6.5 and below~~	~~32.6~~	~~48.8~~
~~6.6~~	~~31.3~~	~~46.8~~
~~6.7~~	~~29.8~~	~~44.6~~

pH	Where Salmonids Present	Where Salmonids Absent
6.8	28.1	42.0
6.9	26.2	39.1
7.0	24.1	36.1
7.1	22.0	32.8
7.2	19.7	29.5
7.3	17.5	26.2
7.4	15.4	23.0
7.5	13.3	19.9
7.6	11.4	17.0
7.7	9.65	14.4
7.8	8.11	12.1
7.9	6.77	10.1
8.0	5.62	8.40
8.1	4.64	6.95
8.2	3.83	5.72
8.3	3.15	4.71
8.4	2.59	3.88
8.5	2.14	3.20
8.6	1.77	2.65
8.7	1.47	2.20
8.8	1.23	1.84
8.9	1.04	1.56
9.0 and above	0.885	1.32

L. Chronic aquatic life criteria for total ammonia are dependent on pH, temperature and whether fish in early life stages are present or absent. The criteria are based on analysis of unfiltered samples and are calculated according to the equations in Paragraphs (1) and (2) of this subsection. For temperatures from below 0 to 14°C, the criteria for 14°C apply; for temperatures above 30°C, the criteria for 30°C apply. For pH values below 6.5, the criteria for 6.5 apply; for pH values above 9.0, the criteria for 9.0 apply.

(1) Chronic aquatic life criteria for total ammonia when fish early life stages are present:

(a) The equation to calculate chronic criteria in mg/L as N is:

$$((0.0577/(1 + 10^{7.688-pH})) + (2.487/(1 + 10^{pH-7.688})) \times MIN (2.85, 1.45 \times 10^{0.028 \times (25-T)})$$

(b) Selected values of calculated chronic criteria in mg/L as N:

pH	Temperature (°C)									
	14 and below	15	16	18	20	22	24	26	28	30 and above
6.5 and below	6.67	6.46	6.06	5.33	4.68	4.12	3.62	3.18	2.80	2.46
6.6	6.57	6.36	5.97	5.25	4.61	4.05	3.56	3.13	2.75	2.42
6.7	6.44	6.25	5.86	5.15	4.52	3.98	3.50	3.07	2.70	2.37
6.8	6.29	6.10	5.72	5.03	4.42	3.89	3.42	3.00	2.64	2.32
6.9	6.12	5.93	5.56	4.89	4.30	3.78	3.32	2.92	2.57	2.25
7.0	5.91	5.73	5.37	4.72	4.15	3.65	3.21	2.82	2.48	2.18
7.1	5.67	5.49	5.15	4.53	3.98	3.50	3.08	2.70	2.38	2.09
7.2	5.39	5.22	4.90	4.31	3.78	3.33	2.92	2.57	2.26	1.99
7.3	5.08	4.92	4.61	4.06	3.57	3.13	2.76	2.42	2.13	1.87
7.4	4.73	4.59	4.30	3.78	3.32	2.92	2.57	2.26	1.98	1.74
7.5	4.36	4.23	3.97	3.49	3.06	2.69	2.37	2.08	1.83	1.61
7.6	3.98	3.85	3.61	3.18	2.79	2.45	2.16	1.90	1.67	1.47
7.7	3.58	3.47	3.25	2.86	2.51	2.21	1.94	1.71	1.50	1.32
7.8	3.18	3.09	2.89	2.54	2.23	1.96	1.73	1.52	1.33	1.17

pH	Temperature (°C)									
	14 and below	15	16	18	20	22	24	26	28	30 and above
7.9	2.80	2.71	2.54	2.24	1.96	1.73	1.52	1.33	1.17	1.03
8.0	2.43	2.36	2.21	1.94	1.71	1.50	1.32	1.16	1.02	0.897
8.1	2.10	2.03	1.91	1.68	1.47	1.29	1.14	1.00	0.879	0.773
8.2	1.79	1.74	1.63	1.43	1.26	1.11	0.973	0.855	0.752	0.661
8.3	1.52	1.48	1.39	1.22	1.07	0.941	0.827	0.727	0.639	0.562
8.4	1.29	1.25	1.17	1.03	0.906	0.796	0.700	0.615	0.541	0.475
8.5	1.09	1.06	0.990	0.870	0.765	0.672	0.591	0.520	0.457	0.401
8.6	0.920	0.892	0.836	0.735	0.646	0.568	0.499	0.439	0.386	0.339
8.7	0.778	0.754	0.707	0.622	0.547	0.480	0.422	0.371	0.326	0.287
8.8	0.661	0.641	0.601	0.528	0.464	0.408	0.359	0.315	0.277	0.244
8.9	0.565	0.548	0.513	0.451	0.397	0.349	0.306	0.269	0.237	0.208
9.0 and above	0.486	0.471	0.442	0.389	0.342	0.300	0.264	0.232	0.204	0.179

(2) Chronic aquatic life criteria for total ammonia when fish early life stages are absent.

(a) The equation to calculate chronic criteria in mg/L as N is:

$$((0.0577/(1 + 10^{7.688 - pH})) + (2.487/(1 + 10^{pH - 7.688})) \times 1.45 \times 10^{0.028 \times (25 - MAX(T,7))}$$

(b) Selected values of calculated chronic criteria in mg/L as N:

pH	Temperature (°C)								
	7 and below	8	9	10	11	12	13	14	15 and above
6.5 and below	10.8	10.1	9.51	8.92	8.36	7.84	7.35	6.89	6.46
6.6	10.7	9.99	9.37	8.79	8.24	7.72	7.24	6.79	6.36
6.7	10.5	9.81	9.20	8.62	8.08	7.58	7.11	6.66	6.25
6.8	10.2	9.58	8.98	8.42	7.90	7.40	6.94	6.51	6.10
6.9	9.93	9.31	8.73	8.19	7.68	7.20	6.75	6.33	5.93
7.0	9.60	9.00	8.43	7.91	7.41	6.95	6.52	6.11	5.73
7.1	9.20	8.63	8.09	7.58	7.11	6.67	6.25	5.86	5.49
7.2	8.75	8.20	7.69	7.21	6.76	6.34	5.94	5.57	5.22
7.3	8.24	7.73	7.25	6.79	6.37	5.97	5.60	5.25	4.92
7.4	7.69	7.21	6.76	6.33	5.94	5.57	5.22	4.89	4.59
7.5	7.09	6.64	6.23	5.84	5.48	5.13	4.81	4.51	4.23
7.6	6.46	6.05	5.67	5.32	4.99	4.68	4.38	4.11	3.85
7.7	5.81	5.45	5.11	4.79	4.49	4.21	3.95	3.70	3.47
7.8	5.17	4.84	4.54	4.26	3.99	3.74	3.51	3.29	3.09
7.9	4.54	4.26	3.99	3.74	3.51	3.29	3.09	2.89	2.71
8.0	3.95	3.70	3.47	3.26	3.05	2.86	2.68	2.52	2.36
8.1	3.41	3.19	2.99	2.81	2.63	2.47	2.31	2.17	2.03
8.2	2.91	2.73	2.56	2.40	2.25	2.11	1.98	1.85	1.74
8.3	2.47	2.32	2.18	2.04	1.91	1.79	1.68	1.58	1.48
8.4	2.09	1.96	1.84	1.73	1.62	1.52	1.42	1.33	1.25
8.5	1.77	1.66	1.55	1.46	1.37	1.28	1.20	1.13	1.06
8.6	1.49	1.40	1.31	1.23	1.15	1.08	1.01	0.951	0.892
8.7	1.26	1.18	1.11	1.04	0.976	0.915	0.858	0.805	0.754
8.8	1.07	1.01	0.944	0.855	0.829	0.778	0.729	0.684	0.641
8.9	0.917	0.860	0.806	0.756	0.709	0.664	0.623	0.584	0.548
9.0 and above	0.790	0.740	0.694	0.651	0.610	0.572	0.536	0.503	0.471

~~pH~~	~~Temperature (°C)~~								
	~~7 and below~~	~~8~~	~~9~~	~~10~~	~~11~~	~~12~~	~~13~~	~~14~~	~~15 and above~~
~~At 15ºC and above, the criterion for fish early life stages absent is the same as the criterion for fish early life stages present (refer to table in Paragraph (1) of this subsection).~~									

~~]~~

K. The criteria for total ammonia consider sensitive freshwater mussel species in the family Unionidae, freshwater non-pulmonate snails, and *Oncorhynchus* spp. (a genus of fish in the family Salmonidae), hence further protecting the aquatic community. The total ammonia criteria magnitude is measured as Total Ammonia Nitrogen (TAN) mg/L. TAN is the sum of NH_4^+ and NH_3. TAN mg/L magnitude is derived as a function of pH and temperature (EPA 2013).

L . The acute aquatic life criteria for TAN (mg/L) was derived by the EPA (2013) as the one-hour average concentration of TAN mg/L that shall not be exceeded more than once every three years on average. The EPA acute criterion magnitude was derived using the following equation:

Acute TAN Criterion Magnitude for 1-hour average=

$$\text{MIN}\left(\left(\frac{0.275}{1+10^{7.204-pH}}+\frac{39}{1+10^{pH-7.204}}\right),\left(0.7249\text{x}\left(\frac{0.0114}{1+10^{7.204-pH}}+\frac{1.6181}{1+10^{pH-7.204}}\right)\text{x}\left(23.12\times 10^{0.036(20-T)}\right)\right)\right)$$

T (temperature °C) and *pH* are defined as the paired values associated with the TAN sample.

(1) Temperature and pH-dependent values of the acute TAN criterion magnitude -when *Oncorhynchus* spp. absent.

	Temperature (°C)																				
pH	**0-10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**
6.5	51	48	44	41	37	34	32	29	27	25	23	21	19	18	16	15	14	13	12	11	9.9
6.6	49	46	42	39	36	33	30	28	26	24	22	20	18	17	16	14	13	12	11	10	9.5
6.7	46	44	40	37	34	31	29	27	24	22	21	19	18	16	15	14	13	12	11	9.8	9
6.8	44	41	38	35	32	30	27	25	23	21	20	18	17	15	14	13	12	11	10	9.2	8.5
6.9	41	38	35	32	30	28	25	23	21	20	18	17	15	14	13	12	11	10	9.4	8.6	7.9
7.0	38	35	33	30	28	25	23	21	20	18	**17**	15	14	13	12	11	10	9.4	8.6	7.9	7.3
7.1	34	32	30	27	25	23	21	20	18	17	15	14	13	12	11	10	9.3	8.5	7.9	7.2	6.7
7.2	31	29	27	25	23	21	19	18	16	15	14	13	12	11	9.8	9.1	8.3	7.7	7.1	6.5	6
7.3	27	26	24	22	20	18	17	16	14	13	12	11	10	9.5	8.7	8	7.4	6.8	6.3	5.8	5.3
7.4	24	22	21	19	18	16	15	14	13	12	11	9.8	9	8.3	7.7	7	6.5	6	5.5	5.1	4.7
7.5	21	19	18	17	15	14	13	12	11	10	9.2	8.5	7.8	7.2	6.6	6.1	5.6	5.2	4.8	4.4	4
7.6	18	17	15	14	13	12	11	10	9.3	8.6	7.9	7.3	6.7	6.2	5.7	5.2	4.8	4.4	4.1	3.8	3.5
7.7	15	14	13	12	11	10	9.3	8.6	7.9	7.3	6.7	6.2	5.7	5.2	4.8	4.4	4.1	3.8	3.5	3.2	2.9
7.8	13	12	11	10	9.3	8.5	7.9	7.2	6.7	6.1	5.6	5.2	4.8	4.4	4	3.7	3.4	3.2	2.9	2.7	2.5
7.9	11	9.9	9.1	8.4	7.7	7.1	6.6	3	5.6	5.1	4.7	4.3	4	3.7	3.4	3.1	2.9	2.6	2.4	2.2	2.1
8.0	8.8	8.2	7.6	7	6.4	5.9	5.4	5	4.6	4.2	3.9	3.6	3.3	3	2.8	2.6	2.4	2.2	2	1.9	1.7
8.1	7.2	6.8	6.3	5.8	5.3	4.9	4.5	4.1	3.8	3.5	3.2	3	2.7	2.5	2.3	2.1	2	1.8	1.7	1.5	1.4
8.2	6	5.6	5.2	4.8	4.4	4	3.7	3.4	3.1	2.9	2.7	2.4	2.3	2.1	1.9	1.8	1.6	1.5	1.4	1.3	1.2
8.3	4.9	4.6	4.3	3.9	3.6	3.3	3.1	2.8	2.6	2.4	2.2	2	1.9	1.7	1.6	1.4	1.3	1.2	1.1	1	0.96
8.4	4.1	3.8	3.5	3.2	3	2.7	2.5	2.3	2.1	2	1.8	1.7	1.5	1.4	1.3	1.2	1.1	1	0.93	0.86	0.79
8.5	3.3	3.1	2.9	2.7	2.4	2.3	2.1	1.9	1.8	1.6	1.5	1.4	1.3	1.2	1.1	0.98	0.9	0.83	0.77	0.71	0.65
8.6	2.8	2.6	2.4	2.2	2	1.9	1.7	1.6	1.5	1.3	1.2	1.1	1	0.96	0.88	0.81	0.75	0.69	0.63	0.58	0.54
8.7	2.3	2.2	2	1.8	1.7	1.6	1.4	1.3	1.2	1.1	1	0.94	0.87	0.8	0.74	0.68	0.62	0.57	0.53	0.49	0.45
8.8	1.9	1.8	1.7	1.5	1.4	1.3	1.2	1.1	1	0.93	0.86	0.79	0.73	0.67	0.62	0.57	0.52	0.48	0.44	0.41	0.37
8.9	1.6	1.5	1.4	1.3	1.2	1.1	1	0.93	0.85	0.79	0.72	0.67	0.61	0.56	0.52	0.48	0.44	0.4	0.37	0.34	0.32
9.0	1.4	1.3	1.2	1.1	1	0.93	0.86	0.79	0.73	0.67	0.62	0.57	0.52	0.48	0.44	0.41	0.37	0.34	0.32	0.29	0.27

(2) Temperature and pH-dependent values for the acute TAN criterion magnitude-when *Oncorhynchus* spp. are present.

	Temperature (°C)

pH	0-14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
6.5	33	33	32	29	27	25	23	21	19	18	16	15	14	13	12	11	9.9
6.6	31	31	30	28	26	24	22	20	18	17	16	14	13	12	11	10	9.5
6.7	30	30	29	27	24	22	21	19	18	16	15	14	13	12	11	9.8	9
6.8	28	28	27	25	23	21	20	18	17	15	14	13	12	11	10	9.2	8.5
6.9	26	26	25	23	21	20	18	17	15	14	13	12	11	10	9.4	8.6	7.9
7.0	24	24	23	21	20	18	17	15	14	13	12	11	10	9.4	8.6	8	7.3
7.1	22	22	21	20	18	17	15	14	13	12	11	10	9.3	8.5	7.9	7.2	6.7
7.2	20	20	19	18	16	15	14	13	12	11	9.8	9.1	8.3	7.7	7.1	6.5	6
7.3	18	18	17	16	14	13	12	11	10	9.5	8.7	8	7.4	6.8	6.3	5.8	5.3
7.4	15	15	15	14	13	12	11	9.8	9	8.3	7.7	7	6.5	6	5.5	5.1	4.7
7.5	13	13	13	12	11	10	9.2	8.5	7.8	7.2	6.6	6.1	5.6	5.2	4.8	4.4	4
7.6	11	11	11	10	9.3	8.6	7.9	7.3	6.7	6.2	5.7	5.2	4.8	4.4	4.1	3.8	3.5
7.7	9.6	9.6	9.3	8.6	7.9	7.3	6.7	6.2	5.7	5.2	4.8	4.4	4.1	3.8	3.5	3.2	3
7.8	8.1	8.1	7.9	7.2	6.7	6.1	5.6	5.2	4.8	4.4	4	3.7	3.4	3.2	2.9	2.7	2.5
7.9	6.8	6.8	6.6	6	5.6	5.1	4.7	4.3	4	3.7	3.4	3.1	2.9	2.6	2.4	2.2	2.1
8.0	5.6	5.6	5.4	5	4.6	4.2	3.9	3.6	3.3	3	2.8	2.6	2.4	2.2	2	1.9	1.7
8.1	4.6	4.6	4.5	4.1	3.8	3.5	3.2	3	2.7	2.5	2.3	2.1	2	1.8	1.7	1.5	1.4
8.2	3.8	3.8	3.7	3.5	3.1	2.9	2.7	2.4	2.3	2.1	1.9	1.8	1.6	1.5	1.4	1.3	1.2
8.3	3.1	3.1	3.1	2.8	2.6	2.4	2.2	2	1.9	1.7	1.6	1.4	1.3	1.2	1.1	1	1
8.4	2.6	2.6	2.5	2.3	2.1	2	1.8	1.7	1.5	1.4	1.3	1.2	1.1	1	0.9	0.9	0.8
8.5	2.1	2.1	2.1	1.9	1.8	1.6	1.5	1.4	1.3	1.2	1.1	1	0.9	0.8	0.8	0.7	0.7
8.6	1.8	1.8	1.7	1.6	1.5	1.3	1.2	1.1	1	1	0.9	0.8	0.8	0.7	0.6	0.6	0.5
8.7	1.5	1.5	1.4	1.3	1.2	1.1	1	0.9	0.9	0.8	0.7	0.7	0.6	0.6	0.5	0.5	0.5
8.8	1.2	1.2	1.2	1.1	1	0.9	0.9	0.8	0.7	0.7	0.6	0.6	0.5	0.5	0.4	0.4	0.4
8.9	1	1	1	0.9	0.9	0.8	0.7	0.7	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.3	0.3
9.0	0.88	0.9	0.9	0.8	0.7	0.7	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3

M. The chronic aquatic life criteria for TAN (mg/L) was derived by the EPA (2013) as a thirty-day rolling average concentration of TAN mg/L that shall not be exceeded more than once every three years on average. In addition, the highest four-day average within the 30-day averaging period should not be more than 2.5 times the CCC (e.g., 2.5 x 1.9 mg TAN/L at pH 7 and 20°C, or 4.8 mg TAN/L) more than once in three years on average. The EPA chronic criterion magnitude was derived using the following equation:

Chronic TAN Criterion Magnitude for 30-day average=

$$0.8876 \times \left(\frac{0.0278}{1+10^{7.688-pH}} + \frac{1.1994}{1+10^{pH-7.688}}\right) \times \left(2.126 \times 10^{0.028\times(20-MAX(T,7))}\right)$$

T (temperature °C) and *pH* are defined as the paired values associated with the TAN sample.

(1) Temperature and pH-Dependent Values of the Chronic TAN Criterion Magnitude.

	Temperature (°C)																								
pH	0-7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
6.5	4.9	4.6	4.3	4.1	3.8	3.6	3.3	3.1	2.9	2.8	2.6	2.4	2.3	2.1	2	1.9	1.8	1.6	1.5	1.5	1.4	1.3	1.2	1.1

6.6	4.8	4.5	4.3	4	3.8	3.5	3.3	3.1	2.9	2.7	2.5	2.4	2.2	2.1	2	1.8	1.7	1.6	1.5	1.4	1.3	1.3	1.2	1.1
6.7	4.8	4.5	4.2	3.9	3.7	3.5	3.2	3	2.8	2.7	2.5	2.3	2.2	2.1	1.9	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.2	1.1
6.8	4.6	4.4	4.1	3.8	3.6	3.4	3.2	3	2.8	2.6	2.4	2.3	2.1	2	1.9	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.1	1.1
6.9	4.5	4.2	4	3.7	3.5	3.3	3.1	2.9	2.7	2.5	2.4	2.2	2.1	2	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.2	1.1	1
7.0	4.4	4.1	3.8	3.6	3.4	3.2	3	2.8	2.6	2.4	2.3	2.2	2	**1.9**	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.1	1.1	1
7.1	4.2	3.9	3.7	3.5	3.2	3	2.8	2.7	2.5	2.3	2.2	2.1	1.9	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.2	1.1	1	1
7.2	4	3.7	3.5	3.3	3.1	2.9	2.7	2.5	2.4	2.2	2.1	2	1.8	1.7	1.6	1.5	1.4	1.3	1.3	1.2	1.1	1	1	0.9
7.3	3.8	3.5	3.3	3.1	2.9	2.7	2.6	2.4	2.2	2.1	2	1.8	1.7	1.6	1.5	1.4	1.3	1.3	1.2	1.1	1	1	0.9	0.9
7.4	3.5	3.3	3.1	2.9	2.7	2.5	2.4	2.2	2.1	2	1.8	1.7	1.6	1.5	1.4	1.3	1.3	1.2	1.1	1	1	0.9	0.9	0.8
7.5	3.2	3	2.8	2.7	2.5	2.3	2.2	2.1	1.9	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.2	1.1	1	1	0.9	0.8	0.8	0.7
7.6	2.9	2.8	2.6	2.4	2.3	2.1	2	1.9	1.8	1.6	1.5	1.4	1.4	1.3	1.2	1.1	1.1	1	0.9	0.9	0.8	0.8	0.7	0.7
7.7	2.6	2.4	2.3	2.2	2	1.9	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.1	1.1	1	0.9	0.9	0.8	0.8	0.7	0.7	0.6	0.6
7.8	2.3	2.2	2.1	1.9	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.2	1.1	1	1	0.9	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.5
7.9	2.1	1.9	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.2	1.1	1	1	0.9	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.5	0.5	0.5
8.0	1.8	1.7	1.6	1.5	1.4	1.3	1.2	1.1	1.1	1	0.9	0.9	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.4	0.4	0.4
8.1	1.5	1.5	1.4	1.3	1.2	1.1	1.1	1	0.9	0.9	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4
8.2	1.3	1.2	1.2	1.1	1	1	0.9	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3
8.3	1.1	1.1	1	0.9	0.9	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3
8.4	1	0.9	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2
8.5	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2
8.6	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
8.7	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1
8.8	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1
8.9	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
9.0	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1

[20.6.4.900 NMAC - Rp 20 NMAC 6.1.3100, 10/12/2010; A, 10/11/2002; A, 5/23/2005; A, 7/17/2005; A, 12/1/2010; A, 3/2/2017; A, XX/XX/XXXX]

20.6.4.901 PUBLICATION REFERENCES: These documents are intended as guidance and are available for public review during regular business hours at the offices of the surface water quality bureau. Copies of these documents have also been filed with the New Mexico state records center in order to provide greater access to this information.

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[20.6.4.901 NMAC - Rp 20 NMAC 6.1.4000, 10/12/2010; A, 5/23/2005; A, 12/1/2010; A, 3/2/2017; A, XX/XX/XXXX]

HISTORY of 20.6.4 NMAC:

Pre-NMAC History:

Material in the part was derived from that previously filed with the commission of public records - state records center and archives:

WQC 67-1, Water Quality Standards, filed 7-17-67, effective 8-18-67
WQC 67-1, Amendment Nos. 1-6, filed 3-21-68, effective 4-22-68
WQC 67-1, Amendment No. 7, filed 2-27-69, effective 3-30-69
WQC 67-1, Amendment No. 8, filed 7-14-69, effective 8-15-69
WQC 70-1, Water Quality Standards for Intrastate Waters and Tributaries to Interstate Streams, filed July 17, 1970;
WQC 67-1, Amendment Nos. 9 and 10, filed 2-12-71, effective 3-15-71
WQC 67-1, Amendment No. 11, filed 3-4-71, effective 4-5-71
WQC 73-1, New Mexico Water Quality Standards, filed 9-17-73, effective 10-23-73
WQC 73-1, Amendment Nos. 1 and 2, filed 10-3-75, effective 11-4-75
WQC 73-1, Amendment No. 3, filed 1-19-76, effective 2-14-76
WQC 77-2, Amended Water Quality Standards for Interstate and Intrastate Streams in New Mexico, filed 2-24-77, effective 3-11-77
WQC 77-2, Amendment No. 1, filed 3-23-78, effective 4-24-78
WQC 77-2, Amendment No. 2, filed 6-12-79, effective 7-13-79
WQCC 80-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, filed 8-28-80, effective 9-28-80
WQCC 81-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, filed 5-5-81, effective 6-4-81
WQCC 81-1, Amendment No. 1, filed 5-19-82, effective 6-18-82
WQCC 81-1, Amendment No. 2, filed 6-24-82, effective 7-26-82
WQCC 85-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, filed 1-16-85, effective 2-15-85
WQCC 85-1, Amendment No. 1, filed 8-28-87, effective 9-28-87
WQCC 88-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, filed 3-24-88, effective 4-25-88
WQCC 91-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, filed 5-29-91, effective 6-29-91

WQCC 91-1, Amendment No. 1, filed 10-11-91, effective 11-12-91

History of the Repealed Material:
WQC 67-1, Water Quality Standards, - Superseded, 10-23-73
WQC 73-1, New Mexico Water Quality Standards, - Superseded, 3-11-77
WQC 77-2, Amended Water Quality Standards for Interstate and Intrastate Streams in New Mexico, - Superseded, 9-28-80
WQCC 80-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, - Superseded, 6-4-81
WQCC 81-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, - Superseded, 2-15-85
WQCC 85-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, - Superseded, 4-25-88
WQCC 88-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, - Superseded, 6-29-91
WQCC 91-1, Water Quality Standards for Interstate and Intrastate Streams in New Mexico, - Superseded, 1-23-95
20 NMAC 6.1, Standards for Interstate and Intrastate Streams, - Repealed, 2-23-00
20 NMAC 6.1, Standards for Interstate and Intrastate Surface Waters, - Repealed, 10/12/2000

FEDERAL WATER POLLUTION CONTROL ACT

(33 U.S.C. 1251 et seq.)

AN ACT To provide for water pollution control activities in the Public Health Service of the Federal Security Agency and in the Federal Works Agency, and for other purposes.

Be it enacted by the Senate and House of Representatives of the United States of America in Congress assembled,

TITLE I—RESEARCH AND RELATED PROGRAMS

DECLARATION OF GOALS AND POLICY

SEC. 101. (a) The objective of this Act is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. In order to achieve this objective it is hereby declared that, consistent with the provisions of this Act—

(1) it is the national goal that the discharge of pollutants into the navigable waters be eliminated by 1985;

(2) it is the national goal that wherever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water be achieved by July 1, 1983;

(3) it is the national policy that the discharge of toxic pollutants in toxic amounts be prohibited;

(4) it is the national policy that Federal financial assistance be provided to construct publicly owned waste treatment works;

(5) it is the national policy that areawide treatment management planning processes be developed and implemented to assure adequate control of sources of pollutants in each State;

(6) it is the national policy that a major research and demonstration effort be made to develop technology necessary to eliminate the discharge of pollutants into the navigable waters, waters of the contiguous zone and the oceans; and

(7) it is the national policy that programs for the control of nonpoint sources of pollution be developed and implemented in an expeditious manner so as to enable the goals of this Act to be met through the control of both point and nonpoint sources of pollution.

(b) It is the policy of the Congress to recognize, preserve, and protect the primary responsibilities and rights of States to prevent, reduce, and eliminate pollution, to plan the development and use (including restoration, preservation, and enhancement) of land and water resources, and to consult with the Administrator in the exercise of his authority under this Act. It is the policy of Congress that the States manage the construction grant program under this Act and implement the permit programs under sections 402 and 404 of

pollutants if the applicant demonstrates at such hearing that (whether or not technology or other alternative control strategies are available) there is no reasonable relationship between the economic and social costs and the benefits to be obtained (including attainment of the objective of this Act) from achieving such limitation.

(B) REASONABLE PROGRESS.—The Administrator, with the concurrence of the State, may issue a permit which modifies the effluent limitations required by subsection (a) of this section for toxic pollutants for a single period not to exceed 5 years if the applicant demonstrates to the satisfaction of the Administrator that such modified requirements (i) will represent the maximum degree of control within the economic capability of the owner and operator of the source, and (ii) will result in reasonable further progress beyond the requirements of section 301(b)(2) toward the requirements of subsection (a) of this section.

(c) The establishment of effluent limitations under this section shall not operate to delay the application of any effluent limitation established under section 301 of this Act.

(33 U.S.C. 1312)

WATER QUALITY STANDARDS AND IMPLEMENTATION PLANS

SEC. 303. (a)(1) In order to carry out the purpose of this Act, any water quality standard applicable to interstate waters which was adopted by any State and submitted to, and approved by, or is awaiting approval by, the Administrator pursuant to this Act as in effect immediately prior to the date of enactment of the Federal Water Pollution Control Act Amendments of 1972, shall remain in effect unless the Administrator determined that such standard is not consistent with the applicable requirements of this Act as in effect immediately prior to the date of enactment of the Federal Water Pollution Control Act Amendments of 1972. If the Administrator makes such a determination he shall, within three months after the date of enactment of the Federal Water Pollution Control Act Amendments of 1972, notify the State and specify the changes needed to meet such requirements. If such changes are not adopted by the State within ninety days after the date of such notification, the Administrator shall promulgate such changes in accordance with subsection (b) of this section.

(2) Any State which, before the date of enactment of the Federal Water Pollution Control Act Amendments of 1972, has adopted, pursuant to its own law, water quality standards applicable to intrastate waters shall submit such standards to the Administrator within thirty days after the date of enactment of the Federal Water Pollution Control Act Amendments of 1972. Each such standard shall remain in effect, in the same manner and to the same extent as any other water quality standard established under this Act unless the Administrator determines that such standard is inconsistent with the applicable requirements of this Act as in effect immediately prior to the date of enactment of the Federal Water Pollution Control Act Amendments of 1972. If the Administrator makes such a determination he shall not later than the one hun-

dred and twentieth day after the date of submission of such standards, notify the State and specify the changes needed to meet such requirements. If such changes are not adopted by the State within ninety days after such notification, the Administrator shall promulgate such changes in accordance with subsection (b) of this section.

(3)(A) Any State which prior to the date of enactment of the Federal Water Pollution Control Act Amendments of 1972 has not adopted pursuant to its own laws water quality standards applicable to intrastate waters shall, not later than one hundred and eighty days after the date of enactment of the Federal Water Pollution Control Act Amendments of 1972, adopt and submit such standards to the Administrator.

(B) If the Administrator determines that any such standards are consistent with the applicable requirements of this Act as in effect immediately prior to the date of enactment of the Federal Water Pollution Control Act Amendments of 1972, he shall approve such standards.

(C) If the Administrator determines that any such standards are not consistent with the applicable requirements of this Act as in effect immediately prior to the date of enactment of the Federal Water Pollution Control Act Amendments of 1972, he shall, not later than the ninetieth day after the date of submission of such standards, notify the State and specify the changes to meet such requirements. If such changes are not adopted by the State within ninety days after the date of notification, the Administrator shall promulgate such standards pursuant to subsection (b) of this section.

(b)(1) The Administrator shall promptly prepare and publish proposed regulations setting forth water quality standards for a State in accordance with the applicable requirements of this Act as in effect immediately prior to the date of enactment of the Federal Water Pollution Control Act Amendments of 1972, if—

(A) the State fails to submit water quality standards within the times prescribed in subsection (a) of this section,

(B) a water quality standard submitted by such State under subsection (a) of this section is determined by the Administrator not to be consistent with the applicable requirements of subsection (a) of this section.

(2) The Administrator shall promulgate any water quality standard published in a proposed regulation not later than one hundred and ninety days after the date he publishes any such proposed standard, unless prior to such promulgation, such State has adopted a water quality standard which the Administrator determines to be in accordance with subsection (a) of this section.

(c)(1) The Governor of a State or the State water pollution control agency of such State shall from time to time (but at least once each three year period beginning with the date of enactment of the Federal Water Pollution Control Act Amendments of 1972) hold public hearings for the purpose of reviewing applicable water quality standards and, as appropriate, modifying and adopting standards. Results of such review shall be made available to the Administrator.

(2)(A) Whenever the State revises or adopts a new standard, such revised or new standard shall be submitted to the Adminis-

trator. Such revised or new water quality standard shall consist of the designated uses of the navigable waters involved and the water quality criteria for such waters based upon such uses. Such standards shall be such as to protect the public health or welfare, enhance the quality of water and serve the purposes of this Act. Such standards shall be established taking into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and agricultural, industrial, and other purposes, and also taking into consideration their use and value for navigation.

(B) Whenever a State reviews water quality standards pursuant to paragraph (1) of this subsection, or revises or adopts new standards pursuant to this paragraph, such State shall adopt criteria for all toxic pollutants listed pursuant to section 307(a)(1) of this Act for which criteria have been published under section 304(a), the discharge or presence of which in the affected waters could reasonably be expected to interfere with those designated uses adopted by the State, as necessary to support such designated uses. Such criteria shall be specific numerical criteria for such toxic pollutants. Where such numerical criteria are not available, whenever a State reviews water quality standards pursuant to paragraph (1), or revises or adopts new standards pursuant to this paragraph, such State shall adopt criteria based on biological monitoring or assessment methods consistent with information published pursuant to section 304(a)(8). Nothing in this section shall be construed to limit or delay the use of effluent limitations or other permit conditions based on or involving biological monitoring or assessment methods or previously adopted numerical criteria.

(3) If the Administrator, within sixty days after the date of submission of the revised or new standard, determines that such standard meets the requirements of this Act, such standard shall thereafter be the water quality standard for the applicable waters of that State. If the Administrator determines that any such revised or new standard is not consistent with the applicable requirements of this Act, he shall not later than the ninetieth day after the date of submission of such standard notify the State and specify the changes to meet such requirements. If such changes are not adopted by the State within ninety days after the date of notification, the Administrator shall promulgate such standard pursuant to paragraph (4) of this subsection.

(4) The Administrator shall promptly prepare and publish proposed regulations setting forth a revised or new water quality standard for the navigable waters involved—

(A) if a revised or new water quality standard submitted by such State under paragraph (3) of this subsection for such waters is determined by the Administrator not to be consistent with the applicable requirements of this Act, or

(B) in any case where the Administrator determines that a revised or new standard is necessary to meet the requirements of this Act.

The Administrator shall promulgate any revised or new standard under this paragraph not later than ninety days after he publishes such proposed standards, unless prior to such promulgation, such

State has adopted a revised or new water quality standard which the Administrator determines to be in accordance with this Act.

(d)(1)(A) Each State shall identify those waters within its boundaries for which the effluent limitations required by section 301(b)(1)(A) and section 301(b)(1)(B) are not stringent enough to implement any water quality standard applicable to such waters. The State shall establish a priority ranking for such waters, taking into account the severity of the pollution and the uses to be made of such waters.

(B) Each State shall identify those waters or parts thereof within its boundaries for which controls on thermal discharges under section 301 are not stringent enough to assure protection and propagation of a balanced indigenous population of shellfish, fish, and wildlife.

(C) Each State shall establish for the waters identified in paragraph (1)(A) of this subsection, and in accordance with the priority ranking, the total maximum daily load, for those pollutants which the Administrator identifies under section 304(a)(2) as suitable for such calculation. Such load shall be established at a level necessary to implement the applicable water quality standards with seasonal variations and a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality.

(D) Each State shall estimate for the waters identified in paragraph (1)(D) of this subsection the total maximum daily thermal load required to assure protection and propagation of a balanced, indigenous population of shellfish, fish and wildlife. Such estimates shall take into account the normal water temperatures, flow rates, seasonal variations, existing sources of heat input, and the dissipative capacity of the identified waters or parts thereof. Such estimates shall include a calculation of the maximum heat input that can be made into each such part and shall include a margin of safety which takes into account any lack of knowledge concerning the development of thermal water quality criteria for such protection and propagation in the identified waters or parts thereof.

(2) Each State shall submit to the Administrator from time to time, with the first such submission not later than one hundred and eighty days after the date of publication of the first identification of pollutants under section 304(a)(2)(D), for his approval the waters identified and the loads established under paragraphs (1)(A), (1)(B), (1)(C), and (1)(D) of this subsection. The Administrator shall either approve or disapprove such identification and load not later than thirty days after the date of submission. If the Administrator approves such identification and load, such State shall incorporate them into its current plan under subsection (e) of this section. If the Administrator disapproves such identification and load, he shall not later than thirty days after the date of such disapproval identify such waters in such State and establish such loads for such waters as he determines necessary to implement the water quality standards applicable to such waters and upon such identification and establishment the State shall incorporate them into its current plan under subsection (e) of this section.

(3) For the specific purpose of developing information, each State shall identify all waters within its boundaries which it has

not identified under paragraph (1)(A) and (1)(B) of this subsection and estimate for such waters the total maximum daily load with seasonal variations and margins of safety, for those pollutants which the Administrator identifies under section 304(a)(2) as suitable for such calculation and for thermal discharges, at a level that would assure protection and propagation of a balanced indigenous population of fish, shellfish and wildlife.

(4) LIMITATIONS ON REVISION OF CERTAIN EFFLUENT LIMITATIONS.—

(A) STANDARD NOT ATTAINED.—For waters identified under paragraph (1)(A) where the applicable water quality standard has not yet been attained, any effluent limitation based on a total maximum daily load or other waste load allocation established under this section may be revised only if (i) the cumulative effect of all such revised effluent limitations based on such total maximum daily load or waste load allocation will assure the attainment of such water quality standard, or (ii) the designated use which is not being attained is removed in accordance with regulations established under this section.

(B) STANDARD ATTAINED.—For waters identified under paragraph (1)(A) where the quality of such waters equals or exceeds levels necessary to protect the designated use for such waters or otherwise required by applicable water quality standard, any effluent limitation based on a total maximum daily load or other waste load allocation established under this section, or any water quality standard established under this section, or any other permitting standard may be revised only if such revision is subject to and consistent with the antidegradation policy established under this section.

(e)(1) Each State shall have a continuing planning process approved under paragraph (2) of this subsection which is consistent with this Act.

(2) Each State shall submit not later than 120 days after the date of the enactment of the Water Pollution Control Amendments of 1972 to the Administrator for his approval a proposed continuing planning process which is consistent with this Act. Not later than thirty days after the date of submission of such a process the Administrator shall either approve or disapprove such process. The Administrator shall from time to time review each State's approved planning process for the purpose of insuring that such planning process is at all times consistent with this Act. The Administrator shall not approve any State permit program under title IV of this Act for any State which does not have an approved continuing planning process under this section.

(3) The Administrator shall approve any continuing planning process submitted to him under this section which will result in plans for all navigable waters within such State, which include, but are not limited to, the following:

(A) effluent limitations and schedules of compliance at least as stringent as those required by section 301(b)(1), section 301(b)(2), section 306, and section 307, and at least as

stringent as any requirements contained in any applicable water quality standard in effect under authority of this section;

(B) the incorporation of all elements of any applicable areawide waste management plans under section 208, and applicable basin plans under section 209 of this Act;

(C) total maximum daily load for pollutants in accordance with subsection (d) of this section;

(D) procedures for revision;

(E) adequate authority for intergovernmental cooperation;

(F) adequate implementation, including schedules of compliance, for revised or new water quality standards, under subsection (c) of this section;

(G) controls over the disposition of all residual waste from any water treatment processing;

(H) an inventory and ranking, in order of priority, of needs for construction of waste treatment works required to meet the applicable requirements of sections 301 and 302.

(f) Nothing in this section shall be construed to affect any effluent limitation, or schedule of compliance required by any State to be implemented prior to the dates set forth in sections 301(b)(1) and 301(b)(2) nor to preclude any State from requiring compliance with any effluent limitation or schedule of compliance at dates earlier than such dates.

(g) Water quality standards relating to heat shall be consistent with the requirements of section 316 of this Act.

(h) For the purposes of this Act the term "water quality standards" includes thermal water quality standards.

(i) COASTAL RECREATION WATER QUALITY CRITERIA.—

(1) ADOPTION BY STATES.—

(A) INITIAL CRITERIA AND STANDARDS.—Not later than 42 months after the date of the enactment of this subsection, each State having coastal recreation waters shall adopt and submit to the Administrator water quality criteria and standards for the coastal recreation waters of the State for those pathogens and pathogen indicators for which the Administrator has published criteria under section 304(a).

(B) NEW OR REVISED CRITERIA AND STANDARDS.—Not later than 36 months after the date of publication by the Administrator of new or revised water quality criteria under section 304(a)(9), each State having coastal recreation waters shall adopt and submit to the Administrator new or revised water quality standards for the coastal recreation waters of the State for all pathogens and pathogen indicators to which the new or revised water quality criteria are applicable.

(2) FAILURE OF STATES TO ADOPT.—

(A) IN GENERAL.—If a State fails to adopt water quality criteria and standards in accordance with paragraph (1)(A) that are as protective of human health as the criteria for pathogens and pathogen indicators for coastal recreation waters published by the Administrator, the Administrator shall promptly propose regulations for the State setting forth revised or new water quality standards

for pathogens and pathogen indicators described in paragraph (1)(A) for coastal recreation waters of the State.

(B) EXCEPTION.—If the Administrator proposes regulations for a State described in subparagraph (A) under subsection (c)(4)(B), the Administrator shall publish any revised or new standard under this subsection not later than 42 months after the date of the enactment of this subsection.

(3) APPLICABILITY.—Except as expressly provided by this subsection, the requirements and procedures of subsection (c) apply to this subsection, including the requirement in subsection (c)(2)(A) that the criteria protect public health and welfare.

(33 U.S.C. 1313)

INFORMATION AND GUIDELINES

SEC. 304. (a)(1) The Administrator, after consultation with appropriate Federal and State agencies and other interested persons, shall develop and publish, within one year after the date of enactment of this title (and from time to time thereafter revise) criteria for water quality accurately reflecting the latest scientific knowledge (A) on the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, esthetics, and recreation which may be expected from the presence of pollutants in any body of water, including ground water; (B) on the concentration and dispersal of pollutants, or their byproducts, through biological, physical, and chemical processes; and (C) on the effects of pollutants on biological community diversity, productivity, and stability, including information on the factors affecting rates of eutrophication and rates of organic and inorganic sedimentation for varying types of receiving waters.

(2) The Administrator, after consultation with appropriate Federal and State agencies and other interested persons, shall develop and publish, within one year after the date of enactment of this title (and from time to time thereafter revise) information (A) on the factors necessary to restore and maintain the chemical, physical, and biological integrity of all navigable waters, ground waters, waters of the contiguous zone, and the oceans; (B) on the factors necessary for the protection and propagation of shellfish, fish, and wildlife for classes and categories of receiving waters and to allow recreational activities in and on the water; and (C) on the measurement and classification of water quality; and (D) for the purpose of section 303, on and the identification of pollutants suitable for maximum daily load measurement correlated with the achievement of water quality objectives.

(3) Such criteria and information and revisions thereof shall be issued to the States and shall be published in the Federal Register and otherwise made available to the public.

(4) The Administrator shall, within 90 days after the date of enactment of the Clean Water Act of 1977 and from time to time thereafter, publish and revise as appropriate information identifying conventional pollutants, including but not limited to, pollut-

for pathogens and pathogen indicators described in paragraph (1)(A) for coastal recreation waters of the State.

(B) EXCEPTION.—If the Administrator proposes regulations for a State described in subparagraph (A) under subsection (c)(4)(B), the Administrator shall publish any revised or new standard under this subsection not later than 42 months after the date of the enactment of this subsection.

(3) APPLICABILITY.—Except as expressly provided by this subsection, the requirements and procedures of subsection (c) apply to this subsection, including the requirement in subsection (c)(2)(A) that the criteria protect public health and welfare.

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INFORMATION AND GUIDELINES

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(2) The Administrator, after consultation with appropriate Federal and State agencies and other interested persons, shall develop and publish, within one year after the date of enactment of this title (and from time to time thereafter revise) information (A) on the factors necessary to restore and maintain the chemical, physical, and biological integrity of all navigable waters, ground waters, waters of the contiguous zone, and the oceans; (B) on the factors necessary for the protection and propagation of shellfish, fish, and wildlife for classes and categories of receiving waters and to allow recreational activities in and on the water; and (C) on the measurement and classification of water quality; and (D) for the purpose of section 303, on and the identification of pollutants suitable for maximum daily load measurement correlated with the achievement of water quality objectives.

(3) Such criteria and information and revisions thereof shall be issued to the States and shall be published in the Federal Register and otherwise made available to the public.

(4) The Administrator shall, within 90 days after the date of enactment of the Clean Water Act of 1977 and from time to time thereafter, publish and revise as appropriate information identifying conventional pollutants, including but not limited to, pollut-

ants classified as biological oxygen demanding, suspended solids, fecal coliform, and pH. The thermal component of any discharge shall not be identified as a conventional pollutant under this paragraph.

(5)(A) The Administrator, to the extent practicable before consideration of any request under section 301(g) of this Act and within six months after the date of enactment of the Clean Water Act of 1977, shall develop and publish information on the factors necessary for the protection of public water supplies, and the protection and propagation of a balanced population of shellfish, fish and wildlife, and to allow recreational activities, in and on the water.

(B) The Administrator, to the extent practicable before consideration of any application under section 301(h) of this Act and within six months after the date of enactment of Clean Water Act of 1977, shall develop and publish information on the factors necessary for the protection of public water supplies, and the protection and propagation of a balanced indigenous population of shellfish, fish and wildlife, and to allow recreational activities, in and on the water.

(6) The Administrator shall, within three months after enactment of the Clean Water Act of 1977 and annually thereafter, for purposes of section 301(h) of this Act publish and revise as appropriate information identifying each water quality standard in effect under this Act of State law, the specific pollutants associated with such water quality standard, and the particular waters to which such water quality standard applies.

(7) GUIDANCE TO STATES.—The Administrator, after consultation with appropriate State agencies and on the basis of criteria and information published under paragraphs (1) and (2) of this subsection, shall develop and publish, within 9 months after the date of the enactment of the Water Quality Act of 1987, guidance to the States on performing the identification required by section 304(l)(1) of this Act.

(8) INFORMATION ON WATER QUALITY CRITERIA.—The Administrator, after consultation with appropriate State agencies and within 2 years after the date of the enactment of the Water Quality Act of 1987, shall develop and publish information on methods for establishing and measuring water quality criteria for toxic pollutants on other bases than pollutant-by-pollutant criteria, including biological monitoring and assessment methods.

(9) REVISED CRITERIA FOR COASTAL RECREATION WATERS.—

(A) IN GENERAL.—Not later than 5 years after the date of the enactment of this paragraph, after consultation and in cooperation with appropriate Federal, State, tribal, and local officials (including local health officials), the Administrator shall publish new or revised water quality criteria for pathogens and pathogen indicators (including a revised list of testing methods, as appropriate), based on the results of the studies conducted under section 104(v), for the purpose of protecting human health in coastal recreation waters.

(B) REVIEWS.—Not later than the date that is 5 years after the date of publication of water quality criteria under

this paragraph, and at least once every 5 years thereafter, the Administrator shall review and, as necessary, revise the water quality criteria.

(b) For the purposes of adopting or revising effluent limitations under this Act the Administrator shall, after consultation with appropriate Federal and State agencies and other interested persons, publish within one year of enactment of this title, regulations, providing guidelines for effluent limitations, and, at least annually thereafter, revise, if appropriate, such regulations. Such regulations shall—

(1)(A) identify, in terms of amounts of constituents and chemical, physical, and biological characteristics of pullutants, the degree of effluent reduction attainable through the application of the best practicable control technology currently available for classes and categories to point sources (other than publicly owned treatment works); and

(B) specify factors to be taken into account in determining the control measures and practices to be applicable to point sources (other than publicly owned treatment works) within such categories of classes. Factors relating to the assessment of best practical control technology currently available to comply with subsection (b)(1) of section 301 of this Act shall include consideration of the total cost of application of technology in relation to the effluent reduction benefits to be achieved from such application, and shall also take into account the age of equipment and facilities involved, the process employed, the engineering aspects of the application of various types of control techniques, process changes, non-water quality environmental impact (including energy requirements), and such other factors as the Administrator deems appropriate;

(2)(A) identify, in terms of amounts of constituents and chemical, physical, and biological characteristics of pollutants, the degree of effluent reduction attainable through the application of the best control measures and practices achievable including treatment techniques, process and procedure innovations, operating methods, and other alternatives for classes and categories of point sources (other than publicly owned treatment works); and

(B) specify factors to be taken into account in determining the best measures and practices available to comply with subsection (b)(2) of section 301 of this Act to be applicable to any point source (other than publicly owned treatment works) within such categories of classes. Factors relating to the assessment of best available technology shall take into account the age of equipment and facilities involved, the process employed, the engineering aspects of the application of various types of control techniques, process changes, the cost of achieving such effluent reduction, non-water quality environmental impact (including energy requirements), and such other factors as the Administrator deems appropriate;

(3) identify control measures and practices available to eliminate the discharge of pollutants from categories and classes of point sources, taking into account the cost of achieving such elimination of the discharge of pollutants; and

NMSA 1978, § 74-6-3 Water Quality Act
As extracted from NMOneSource.com

74-6-3. Water quality control commission created. (Repealed effective July 1, 2026.)

A. There is created the "water quality control commission" consisting of:

(1) the secretary of environment or a member of the secretary's staff designated by the secretary;

(2) the secretary of health or a member of the secretary's staff designated by the secretary;

(3) the director of the department of game and fish or a member of the director's staff designated by the director;

(4) the state engineer or a member of the state engineer's staff designated by the state engineer;

(5) the chair of the oil conservation commission or a member of the chair's staff designated by the chair;

(6) the director of the state parks division of the energy, minerals and natural resources department or a member of the director's staff designated by the director;

(7) the director of the New Mexico department of agriculture or a member of the director's staff designated by the director;

(8) the chair of the soil and water conservation commission or a soil and water conservation district supervisor designated by the chair;

(9) the director of the bureau of geology and mineral resources at the New Mexico institute of mining and technology or a member of the director's staff designated by the director;

(10) a municipal or county government representative; and

(11) four representatives of the public to be appointed by the governor for terms of four years and who shall be compensated from the budgeted funds of the department of environment in accordance with the provisions of the Per Diem and Mileage Act [10-8-1 to 10-8-8 NMSA 1978]. At least one member appointed by the governor shall be a member of a New Mexico Indian tribe or pueblo.

B. A member of the commission shall not receive, or shall not have received during the previous two years, a significant portion of the member's income directly or indirectly from permit holders or applicants for a permit. A member of the commission shall, upon the acceptance of the member's appointment and prior to the performance of any of the member's duties, file a statement of disclosure with the secretary of state disclosing any amount of money or other valuable consideration, and its source, the value of which is in excess of ten percent of the member's gross personal income in each of the preceding two years, that the member received directly or indirectly from permit holders or applicants for permits required under the Water Quality Act. A member of the commission shall not participate in the consideration of an appeal if the subject of the appeal is an application filed or a permit held by an entity that either employs the commission member or from which the commission member received more than ten percent of the member's gross personal income in either of the preceding two years.

C. The commission shall elect a chair and other necessary officers and shall keep a record of its proceedings.

D. A majority of the commission constitutes a quorum for the transaction of business, but no action of the commission is valid unless concurred in by six or more members present at a meeting.

E. The commission is the state water pollution control agency for this state for all purposes of the federal act and the wellhead protection and sole source aquifer programs of the federal Safe Drinking Water Act and may take all action necessary and appropriate to secure to this state, its political subdivisions or interstate agencies the benefits of that act and those programs.

F. The commission is administratively attached, as defined in the Executive Reorganization Act [9-1-1 to 9-1-10 NMSA 1978], to the department of environment.

History: 1953 Comp., § 75-39-3, enacted by Laws 1967, ch. 190, § 3; 1970, ch. 64, § 2; 1971, ch. 277, § 50; 1973, ch. 326, § 2; 1977, ch. 253, § 74; 1987, ch. 234, § 81; 1993, ch. 291, § 3; 1997, ch. 82, § 1; 2001, ch. 246, § 14; 2001, ch. 267, § 1; 2003, ch. 165, § 2; 2007, ch. 183, § 1.

NMSA 1978, § 74-6-4 Water Quality Act
As extracted from NMOneSource.com

74-6-4. Duties and powers of commission. (Repealed effective July 1, 2026.)

The commission:

A. may accept and supervise the administration of loans and grants from the federal government and from other sources, public or private, which loans and grants shall not be expended for other than the purposes for which provided;

B. shall adopt a comprehensive water quality management program and develop a continuing planning process;

C. shall not adopt or promulgate a standard or regulation that exceeds a grant of rulemaking authority listed in the statutory section of the Water Quality Act authorizing the standard or regulation;

D. shall adopt water quality standards for surface and ground waters of the state based on credible scientific data and other evidence appropriate under the Water Quality Act. The standards shall include narrative standards and, as appropriate, the designated uses of the waters and the water quality criteria necessary to protect such uses. The standards shall at a minimum protect the public health or welfare, enhance the quality of water and serve the purposes of the Water Quality Act. In making standards, the commission shall give weight it deems appropriate to all facts and circumstances, including the use and value of the water for water supplies, propagation of fish and wildlife, recreational purposes and agricultural, industrial and other purposes;

E. shall adopt, promulgate and publish regulations to prevent or abate water pollution in the state or in any specific geographic area, aquifer or watershed of the state or in any part thereof, or for any class of waters, and to govern the disposal of septage and sludge and the use of sludge for various beneficial purposes. The regulations governing the disposal of septage and sludge may include the use of tracking and permitting systems or other reasonable means necessary to assure that septage and sludge are designated for disposal in, and arrive at, disposal facilities, other than facilities on the premises where the septage and sludge is generated, for which a permit or other authorization has been issued pursuant to the federal act or the Water Quality Act. Regulations may specify a standard of performance for new sources that reflects the greatest reduction in the concentration of water contaminants that the commission determines to be achievable through application of the best available demonstrated control technology, processes, operating methods or other alternatives, including where practicable a standard permitting no discharge of pollutants. In making regulations, the commission shall give weight it deems appropriate to all relevant facts and circumstances, including:

(1) the character and degree of injury to or interference with health, welfare, environment and property;

(2) the public interest, including the social and economic value of the sources of water contaminants;

(3) the technical practicability and economic reasonableness of reducing or eliminating water contaminants from the sources involved and previous experience with equipment and methods available to control the water contaminants involved;

(4) the successive uses, including domestic, commercial, industrial, pastoral, agricultural, wildlife and recreational uses;

(5) feasibility of a user or a subsequent user treating the water before a subsequent use;

(6) property rights and accustomed uses; and

(7) federal water quality requirements;

F. shall assign responsibility for administering its regulations to constituent agencies so as to assure adequate coverage and prevent duplication of effort. To this end, the commission may make such classification of waters and sources of water contaminants as will facilitate the assignment of administrative responsibilities to constituent agencies. The commission shall also hear and decide disputes between constituent agencies as to jurisdiction concerning any matters within the purpose of the Water Quality Act. In assigning responsibilities to constituent agencies, the commission shall give priority to the primary interests of the constituent agencies. The department of environment shall provide technical services, including certification of permits pursuant to the federal act, and shall maintain a repository of the scientific data required by the Water Quality Act;

G. may enter into or authorize constituent agencies to enter into agreements with the federal government or other state governments for purposes consistent with the Water Quality Act and receive and allocate to constituent agencies funds made available to the commission;

H. may grant an individual variance from any regulation of the commission whenever it is found that compliance with the regulation will impose an unreasonable burden upon any lawful business, occupation or activity. The commission may only grant a variance conditioned upon a person effecting a particular abatement of water pollution within a reasonable period of time. Any variance shall be granted for the period of time specified by the commission. The commission shall adopt regulations specifying the procedure under which variances may be sought, which regulations shall provide for the holding of a public hearing before any variance may be granted;

I. may adopt regulations to require the filing with it or a constituent agency of proposed plans and specifications for the construction and operation of new sewer systems, treatment works or sewerage systems or extensions, modifications of or additions to new or existing sewer systems, treatment works or sewerage systems. Filing with and approval by the federal housing administration of plans for an extension to an existing or construction of a new sewerage system intended to serve a subdivision solely residential in nature shall be deemed compliance with all provisions of this subsection;

J. may adopt regulations requiring notice to it or a constituent agency of intent to introduce or allow the introduction of water contaminants into waters of the state;

K. shall specify in regulations the measures to be taken to prevent water pollution and to monitor water quality. The commission may adopt regulations for particular industries. The commission shall adopt regulations for the dairy industry and the copper industry. The commission shall consider, in addition to the factors listed in Subsection E of this section, the best available scientific information. The regulations may include variations in requirements based on site-specific factors, such as depth and distance to ground water and geological and hydrological conditions. The constituent agency shall establish an advisory committee composed of persons with knowledge and expertise particular to the industry category and other interested stakeholders to advise the constituent agency on appropriate regulations to be proposed for adoption by the commission. The regulations shall be developed and adopted in accordance with a schedule approved by the commission. The schedule shall incorporate an opportunity for public input and stakeholder negotiations;

L. may adopt regulations establishing pretreatment standards that prohibit or control the introduction into publicly owned sewerage systems of water contaminants that are not susceptible to treatment by the treatment works or that would interfere with the operation of the treatment works;

M. shall not require a permit respecting the use of water in irrigated agriculture, except in the case of the employment of a specific practice in connection with such irrigation that documentation or actual case history has shown to be hazardous to public health or the environment or for the use of produced water;

N. shall not require a permit for applying less than two hundred fifty gallons per day of private residential gray water originating from a residence for the resident's household gardening, composting or landscape irrigation if:

(1) a constructed gray water distribution system provides for overflow into the sewer system or on-site wastewater treatment and disposal system;

(2) a gray water storage tank is covered to restrict access and to eliminate habitat for mosquitos or other vectors;

(3) a gray water system is sited outside of a floodway;

(4) gray water is vertically separated at least five feet above the ground water table;

(5) gray water pressure piping is clearly identified as a nonpotable water conduit;

(6) gray water is used on the site where it is generated and does not run off the property lines;

(7) gray water is applied in a manner that minimizes the potential for contact with people or domestic pets;

(8) ponding is prohibited, application of gray water is managed to minimize standing water on the surface and to ensure that the hydraulic capacity of the soil is not exceeded;

(9) gray water is not sprayed;

(10) gray water is not discharged to a watercourse; and

(11) gray water use within municipalities or counties complies with all applicable municipal or county ordinances enacted pursuant to Chapter 3, Article 53 NMSA 1978;

O. shall coordinate application procedures and funding cycles for loans and grants from the federal government and from other sources, public or private, with the local government division of the department of finance and administration pursuant to the New Mexico Community Assistance Act [11-6-1 NMSA 1978 et seq.];

P. shall adopt regulations to be administered by the department of environment for the discharge, handling, transport, storage, recycling or treatment for the disposition of treated produced water, including disposition in road construction maintenance, roadway ice or dust control or other construction, or in the application of treated produced water to land, for activities unrelated to the exploration, drilling, production, treatment or refinement of oil or gas; and

Q. may adopt regulations to be administered by the department of environment for surface water discharges.

History: 1953 Comp., § 75-39-4, enacted by Laws 1967, ch. 190, § 4; 1970, ch. 64, § 3; 1971, ch. 277, § 51; 1973, ch. 326, § 3; 1981, ch. 347, § 1; 1984, ch. 5, § 13; 1993, ch. 291, § 4; 2001, ch. 240, § 1; 2001, ch. 281, § 1; 2003, ch. 7, § 2; 2009, ch. 194, § 1; 2019, ch. 197, § 11.

NMSA 1978, § 74-6-6 Water Quality Act
As extracted from NMOneSource.com

74-6-6. Adoption of regulations and standards; notice and hearing.

A. No regulation or water quality standard or amendment or repeal thereof shall be adopted until after a public hearing.

B. Any person may petition in writing to have the commission adopt, amend or repeal a regulation or water quality standard. The commission shall determine whether to hold a hearing within ninety days of submission of the petition. The denial of such a petition shall not be subject to judicial review.

C. Hearings on regulations or water quality standards of statewide application shall be held in Santa Fe. Hearings on regulations or standards that are not of statewide application may be held within the area that is substantially affected by the regulation or standard. At least thirty days prior to the hearing date, notice of the hearing shall be published in the New Mexico register and a newspaper of general circulation in the area affected and mailed to all persons who have made a written request to the commission for advance notice of hearings and who have provided the commission with a mailing address. The notice shall state the subject, the time and the place of the hearing and the manner in which interested persons may present their views. The notice shall also state where interested persons may secure copies of any proposed regulation or water quality standard.

D. At the hearing, the commission shall allow all interested persons reasonable opportunity to submit data, views or arguments orally or in writing and to examine witnesses testifying at the hearing. The commission may designate a hearing officer to take evidence in the hearing. Any person heard or represented at the hearing shall be given written notice of the action of the commission.

E. No regulation or water quality standard or amendment or repeal thereof adopted by the commission shall become effective until thirty days after its filing in accordance with the provisions of the State Rules Act [Chapter 14, Article 4 NMSA 1978].

History: 1953 Comp., § 75-39-5, enacted by Laws 1967, ch. 190, § 5; 1982, ch. 73, § 26; 1993, ch. 291, § 6.

NMSA 1978, § 74-6-10 Water Quality Act
As extracted from NMOneSource.com

74-6-10. Penalties enforcement; compliance orders; penalties; assurance of discontinuance.

A. Whenever, on the basis of any information, a constituent agency determines that a person violated or is violating a requirement, regulation or water quality standard adopted pursuant to the Water Quality Act or a condition of a permit issued pursuant to that act, the constituent agency may:

(1) issue a compliance order requiring compliance immediately or within a specified time period or issue a compliance order assessing a civil penalty, or both; or

(2) commence a civil action in district court for appropriate relief, including injunctive relief.

B. A compliance order issued pursuant to Paragraph (1) of Subsection A of this section may include a suspension or termination of the permit allegedly violated.

C. A compliance order shall state with reasonable specificity the nature of the violation. Any penalty assessed in the compliance order shall not exceed:

(1) fifteen thousand dollars ($15,000) per day of noncompliance with the provisions in Section 74-6-5 NMSA 1978, including a regulation adopted or a permit issued pursuant to that section; or

(2) ten thousand dollars ($10,000) per day for each violation of a provision of the Water Quality Act other than the provisions in Section 74-6-5 NMSA 1978 or of a regulation or water quality standard adopted pursuant to the Water Quality Act.

D. In assessing a penalty authorized by this section, the constituent agency shall take into account the seriousness of the violation, any good faith efforts to comply with the applicable requirements and other relevant factors.

E. For purposes of this section, a single operational event that leads to simultaneous violations of more than one standard shall be treated as a single violation.

F. If a person fails to take corrective actions within the time specified in a compliance order, the constituent agency may:

(1) assess a civil penalty of not more than twenty-five thousand dollars ($25,000) for each day of continued noncompliance with the compliance order; and

(2) suspend or terminate the permit violated by the person.

G. Any compliance order issued by a constituent agency pursuant to this section shall become final unless, no later than thirty days after the compliance order is served, any person named in the

compliance order submits a written request to the commission for a public hearing. The commission shall conduct a public hearing within ninety days after receipt of a request.

H. The commission may appoint an independent hearing officer to preside over any public hearing held pursuant to Subsection F of this section. The hearing officer shall:

(1) make and preserve a complete record of the proceedings; and

(2) forward to the commission a report that includes recommendations, if recommendations are requested by the commission.

I. The commission shall consider the findings of the independent hearing officer and, based on the evidence presented at the hearing, the commission shall make a final decision regarding the compliance order.

J. In connection with any proceeding under this section, the commission may:

(1) adopt rules for discovery procedures; and

(2) issue subpoenas for the attendance and testimony of witnesses and for relevant papers, books and documents.

K. Penalties collected pursuant to this section shall be deposited in the general fund.

L. As an additional means of enforcing the Water Quality Act or any regulation or standard of the commission, the commission may accept an assurance of discontinuance of any act or practice deemed in violation of the Water Quality Act or any regulation or standard adopted pursuant to that act, from any person engaging in, or who has engaged in, such act or practice, signed and acknowledged by the chairman of the commission and the party affected. Any such assurance shall specify a time limit during which the discontinuance is to be accomplished.

History: 1953 Comp., § 75-39-9, enacted by Laws 1967, ch. 190, § 9; 1970, ch. 64, § 5; 1993, ch. 291, § 9.

NMSA 1978, § 11-18-3 State-Tribal Collaboration Act

As extracted from NMOneSource.com

11-18-3. Collaboration with Indian nations, tribes or pueblos.

A. By December 31, 2009, every state agency shall develop and implement a policy that:

(1) promotes effective communication and collaboration between the state agency and Indian nations, tribes or pueblos;

(2) promotes positive government-to-government relations between the state and Indian nations, tribes or pueblos;

(3) promotes cultural competency in providing effective services to American Indians or Alaska Natives; and

(4) establishes a method for notifying employees of the state agency of the provisions of the State-Tribal Collaboration Act and the policy that the state agency adopts pursuant to this section.

B. In the process of developing the policy set forth in Subsection A of this section, state agencies shall consult with representatives designated by the Indian nations, tribes or pueblos.

C. A state agency shall make a reasonable effort to collaborate with Indian nations, tribes or pueblos in the development and implementation of policies, agreements and programs of the state agency that directly affect American Indians or Alaska Natives.

D. The Indian affairs department shall maintain for public reference an updated list of the names and contact information for the chief executives of the Indian nations, tribes or pueblos and for the state agency tribal liaisons.

E. Every state agency shall designate a tribal liaison, who reports directly to the office of the head of the state agency, to:

(1) assist the head of the state agency with developing and ensuring the implementation of the policy as set forth in Subsection A of this section;

(2) serve as a contact person who shall maintain ongoing communication between the state agency and affected Indian nations, tribes or pueblos; and

(3) ensure that training is provided to the staff of the state agency as set forth in Subsection B of Section 4 [11-18-4 NMSA 1978] of the State-Tribal Collaboration Act. Nothing in this subsection shall preclude tribal liaisons from providing or facilitating additional training.

History: Laws 2009, ch. 15, § 3.

NMSA 1978, § 14-4-2 State Rules Act
As extracted from NMOneSource.com

14-4-2. Definitions.

As used in the State Rules Act:

A. "agency" means any agency, board, commission, department, institution or officer of the state government except the judicial and legislative branches of the state government;

B. "person" includes individuals, associations, partnerships, companies, business trusts, political subdivisions and corporations;

C. "proceeding" means a formal agency process or procedure that is commenced or conducted pursuant to the State Rules Act;

D. "proposed rule" means a rule that is provided to the public by an agency for review and public comment prior to its adoption, amendment or repeal, and for which there is specific legal authority authorizing the proposed rule;

E. "provide to the public" means for an agency to distribute rulemaking information by:

(1) posting it on the agency website, if any;

(2) posting it on the sunshine portal;

(3) making it available in the agency's district, field and regional offices, if any;

(4) sending it by electronic mail to persons who have made a written request for notice from the agency of announcements addressing the subject of the rulemaking proceeding and who have provided an electronic mail address to the agency;

(5) sending it by electronic mail to persons who have participated in the rulemaking and who have provided an electronic mail address to the agency;

(6) sending written notice that includes, at a minimum, an internet and street address where the information may be found to persons who provide a postal address; and

(7) providing it to the New Mexico legislative council for distribution to appropriate interim and standing legislative committees;

F. "rule" means any rule, regulation, or standard, including those that explicitly or implicitly implement or interpret a federal or state legal mandate or other applicable law and amendments thereto or repeals and renewals thereof, issued or promulgated by any agency and purporting to affect one or more agencies besides the agency issuing the rule or to affect persons not members or employees of the issuing agency, including affecting persons served by the agency. An order or decision or other document issued or promulgated in connection with the disposition of any case or agency decision upon a particular matter as applied to a specific set of facts shall not be deemed such a rule, nor shall it constitute specific adoption thereof by the agency. "Rule" does not include rules relating to the management, confinement, discipline or release of inmates of any penal or charitable institution, the New Mexico boys' school, the girls' welfare home or any hospital; rules made relating to the management of any particular educational institution, whether elementary or otherwise; or rules made relating to admissions, discipline, supervision, expulsion or graduation of students from any educational institution; and

G. "rulemaking" means the process for adoption of a new rule or the amendment, readoption or repeal of an existing rule.

History: 1953 Comp., § 71-7-2, enacted by Laws 1967, ch. 275, § 2; 1969, ch. 92, § 1; 2017, ch. 137, § 1.

NMSA 1978, § 14-4-5.2 State Rules Act
As extracted from NMOneSource.com

14-4-5.2. Notice of proposed rulemaking.

A. Not later than thirty days before a public rule hearing, the agency proposing the rule shall provide to the public and publish in the New Mexico register a notice of proposed rulemaking. The notice shall include:

(1) a summary of the full text of the proposed rule;

(2) a short explanation of the purpose of the proposed rule;

(3) a citation to the specific legal authority authorizing the proposed rule and the adoption of the rule;

(4) information on how a copy of the full text of the proposed rule may be obtained;

(5) information on how a person may comment on the proposed rule, where comments will be received and when comments are due;

(6) information on where and when a public rule hearing will be held and how a person may participate in the hearing; and

(7) a citation to technical information, if any, that served as a basis for the proposed rule, and information on how the full text of the technical information may be obtained.

B. An agency may charge a reasonable fee for providing any records in nonelectronic form when provided to a person pursuant to this section. An agency shall not charge a fee for providing any records in electronic form when provided to a person pursuant to this section.

C. An internet link providing free access to the full text of the proposed rule shall be included on the notice of proposed rulemaking.

D. If the agency changes the date of the public rule hearing or the deadline for submitting comments as stated in the notice, the agency shall provide notice to the public of the change.

E. The state records administrator or the administrator's designee shall timely publish the notice of proposed rulemaking in the next publication of the New Mexico register.

NMSA 1978, § 14-4A-4 Small Business Regulatory Relief Act
As extracted from NMOneSource.com

14-4A-4. Rules affecting small business.

A. Prior to the adoption of a proposed rule that may have an adverse effect on small business, an agency shall provide a copy of the proposed rule to the commission at the same time as persons who have requested advance notice of rulemaking.

B. Prior to the adoption of a proposed rule that the agency deems to have an adverse effect on small business, the agency shall consider regulatory methods that accomplish the objectives of the applicable law while minimizing the adverse effects on small business.

History: Laws 2005, ch. 244, § 4.

e-CFR data is current as of February 11, 2021

Title 40: Protection of Environment
PART 131—WATER QUALITY STANDARDS
Subpart C—Procedures for Review and Revision of Water Quality Standards

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§131.20 State review and revision of water quality standards.

(a) *State review.* The State shall from time to time, but at least once every 3 years, hold public hearings for the purpose of reviewing applicable water quality standards adopted pursuant to §§131.10 through 131.15 and Federally promulgated water quality standards and, as appropriate, modifying and adopting standards. The State shall also re-examine any waterbody segment with water quality standards that do not include the uses specified in section 101(a)(2) of the Act every 3 years to determine if any new information has become available. If such new information indicates that the uses specified in section 101(a)(2) of the Act are attainable, the State shall revise its standards accordingly. Procedures States establish for identifying and reviewing water bodies for review should be incorporated into their Continuing Planning Process. In addition, if a State does not adopt new or revised criteria for parameters for which EPA has published new or updated CWA section 304(a) criteria recommendations, then the State shall provide an explanation when it submits the results of its triennial review to the Regional Administrator consistent with CWA section 303(c)(1) and the requirements of paragraph (c) of this section.

(b) *Public participation.* The State shall hold one or more public hearings for the purpose of reviewing water quality standards as well as when revising water quality standards, in accordance with provisions of State law and EPA's public participation regulation (40 CFR part 25). The proposed water quality standards revision and supporting analyses shall be made available to the public prior to the hearing.

(c) *Submittal to EPA.* The State shall submit the results of the review, any supporting analysis for the use attainability analysis, the methodologies used for site-specific criteria development, any general policies applicable to water quality standards and any revisions of the standards to the Regional Administrator for review and approval, within 30 days of the final State action to adopt and certify the revised standard, or if no revisions are made as a result of the review, within 30 days of the completion of the review.

[48 FR 51405, Nov. 8, 1983, as amended at 80 FR 51049, Aug. 21, 2015]

Title 40: Protection of Environment

PART 131—WATER QUALITY STANDARDS

§131.10 Designation of uses.

(a) Each State must specify appropriate water uses to be achieved and protected. The classification of the waters of the State must take into consideration the use and value of water for public water supplies, protection and propagation of fish, shellfish and wildlife, recreation in and on the water, agricultural, industrial, and other purposes including navigation. If adopting new or revised designated uses other than the uses specified in section 101(a)(2) of the Act, or removing designated uses, States must submit documentation justifying how their consideration of the use and value of water for those uses listed in this paragraph appropriately supports the State's action. A use attainability analysis may be used to satisfy this requirement. In no case shall a State adopt waste transport or waste assimilation as a designated use for any waters of the United States.

(b) In designating uses of a water body and the appropriate criteria for those uses, the State shall take into consideration the water quality standards of downstream waters and shall ensure that its water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters.

(c) States may adopt sub-categories of a use and set the appropriate criteria to reflect varying needs of such sub-categories of uses, for instance, to differentiate between cold water and warm water fisheries.

(d) At a minimum, uses are deemed attainable if they can be achieved by the imposition of effluent limits required under sections 301(b) and 306 of the Act and cost-effective and reasonable best management practices for nonpoint source control.

(e) [Reserved]

(f) States may adopt seasonal uses as an alternative to reclassifying a water body or segment thereof to uses requiring less stringent water quality criteria. If seasonal uses are adopted, water quality criteria should be adjusted to reflect the seasonal uses, however, such criteria shall not preclude the attainment and maintenance of a more protective use in another season.

(g) States may designate a use, or remove a use that is *not* an existing use, if the State conducts a use attainability analysis as specified in paragraph (j) of this section that demonstrates attaining the use is not feasible because of one of the six factors in this paragraph. If a State adopts a new or revised water quality standard based on a required use attainability analysis, the State shall also adopt the highest attainable use, as defined in §131.3(m).

(1) Naturally occurring pollutant concentrations prevent the attainment of the use; or

(2) Natural, ephemeral, intermittent or low flow conditions or water levels prevent the attainment of the use, unless these conditions may be compensated for by the discharge of sufficient volume of effluent discharges without violating State water conservation requirements to enable uses to be met; or

(3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place; or

(4) Dams, diversions or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use; or

(5) Physical conditions related to the natural features of the water body, such as the lack of a proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses; or

(6) Controls more stringent than those required by sections 301(b) and 306 of the Act would result in substantial and widespread economic and social impact.

(h) States may not remove designated uses if:

(1) They are existing uses, as defined in §131.3, unless a use requiring more stringent criteria is added; or

(2) Such uses will be attained by implementing effluent limits required under sections 301(b) and 306 of the Act and by implementing cost-effective and reasonable best management practices for nonpoint source control.

(i) Where existing water quality standards specify designated uses less than those which are presently being attained, the State shall revise its standards to reflect the uses actually being attained.

(j) A State must conduct a use attainability analysis as described in §131.3(g), and paragraph (g) of this section, whenever:

(1) The State designates for the first time, or has previously designated for a water body, uses that do not include the uses specified in section 101(a)(2) of the Act; or

(2) The State wishes to remove a designated use that is specified in section 101(a)(2) of the Act, to remove a sub-category of such a use, or to designate a sub-category of such a use that requires criteria less stringent than previously applicable.

(k) A State is not required to conduct a use attainability analysis whenever:

(1) The State designates for the first time, or has previously designated for a water body, uses that include the uses specified in section 101(a)(2) of the Act; or

(2) The State designates a sub-category of a use specified in section 101(a)(2) of the Act that requires criteria at least as stringent as previously applicable; or

(3) The State wishes to remove or revise a designated use that is a non-101(a)(2) use. In this instance, as required by paragraph (a) of this section, the State must submit documentation justifying how its consideration of the use and value of water for those uses listed in paragraph (a) appropriately supports the State's action, which may be satisfied through a use attainability analysis.

[48 FR 51405, Nov. 8, 1983, as amended at 80 FR 51047, Aug. 21, 2015]

e-CFR data is current as of April 26, 2021

PART 131—WATER QUALITY STANDARDS

§131.6 Minimum requirements for water quality standards submission.

The following elements must be included in each State's water quality standards submitted to EPA for review:

(a) Use designations consistent with the provisions of sections 101(a)(2) and 303(c)(2) of the Act.

(b) Methods used and analyses conducted to support water quality standards revisions.

(c) Water quality criteria sufficient to protect the designated uses.

(d) An antidegradation policy consistent with §131.12.

(e) Certification by the State Attorney General or other appropriate legal authority within the State that the water quality standards were duly adopted pursuant to State law.

(f) General information which will aid the Agency in determining the adequacy of the scientific basis of the standards which do not include the uses specified in section 101(a)(2) of the Act as well as information on general policies applicable to State standards which may affect their application and implementation.

e-CFR data is current as of April 26, 2021

PART 131—WATER QUALITY STANDARDS

§131.2 Purpose.

A water quality standard defines the water quality goals of a water body, or portion thereof, by designating the use or uses to be made of the water and by setting criteria that protect the designated uses. States adopt water quality standards to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water Act (the Act). "Serve the purposes of the Act" (as defined in sections 101(a)(2) and 303(c) of the Act) means that water quality standards should, wherever attainable, provide water quality for the protection and propagation of fish, shellfish and wildlife and for recreation in and on the water and take into consideration their use and value of public water supplies, propagation of fish, shellfish, and wildlife, recreation in and on the water, and agricultural, industrial, and other purposes including navigation.

Such standards serve the dual purposes of establishing the water quality goals for a specific water body and serve as the regulatory basis for the establishment of water-quality-based treatment controls and strategies beyond the technology-based levels of treatment required by sections 301(b) and 306 of the Act.

[48 FR 51405, Nov. 8, 1983, as amended at 80 FR 51046, Aug. 21, 2015]

§131.11 Criteria.

(a) *Inclusion of pollutants:* (1) States must adopt those water quality criteria that protect the designated use. Such criteria must be based on sound scientific rationale and must contain sufficient parameters or constituents to protect the designated use. For waters with multiple use designations, the criteria shall support the most sensitive use.

(2) *Toxic pollutants.* States must review water quality data and information on discharges to identify specific water bodies where toxic pollutants may be adversely affecting water quality or the attainment of the designated water use or where the levels of toxic pollutants are at a level to warrant concern and must adopt criteria for such toxic pollutants applicable to the water body sufficient to protect the designated use. Where a State adopts narrative criteria for toxic pollutants to protect designated uses, the State must provide information identifying the method by which the State intends to regulate point source discharges of toxic pollutants on water quality limited segments based on such narrative criteria. Such information may be included as part of the standards or may be included in documents generated by the State in response to the Water Quality Planning and Management Regulations (40 CFR part 130).

(b) *Form of criteria:* In establishing criteria, States should:

(1) Establish numerical values based on:

(i) 304(a) Guidance; or

(ii) 304(a) Guidance modified to reflect site-specific conditions; or

(iii) Other scientifically defensible methods;

(2) Establish narrative criteria or criteria based upon biomonitoring methods where numerical criteria cannot be established or to supplement numerical criteria.

[48 FR 51405, Nov. 8, 1983, as amended at 51047, Aug. 21, 2015]

[48 FR 51405, Nov. 8, 1983, as amended at 80 FR 51046, Aug. 21, 2015]

§131.3 Definitions.

(a) *The Act* means the Clean Water Act (Pub. L. 92-500, as amended (33 U.S.C. 1251 *et seq.*)).

(b) *Criteria* are elements of State water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use.

(c) *Section 304(a) criteria* are developed by EPA under authority of section 304(a) of the Act based on the latest scientific information on the relationship that the effect of a constituent concentration has on particular aquatic species and/or human health. This information is issued periodically to the States as guidance for use in developing criteria.

(d) *Toxic pollutants* are those pollutants listed by the Administrator under section 307(a) of the Act.

(e) *Existing uses* are those uses actually attained in the water body on or after November 28, 1975, whether or not they are included in the water quality standards.

(f) *Designated uses* are those uses specified in water quality standards for each water body or segment whether or not they are being attained.

(g) *Use attainability analysis* is a structured scientific assessment of the factors affecting the attainment of the use which may include physical, chemical, biological, and economic factors as described in §131.10(g).

(h) *Water quality limited segment* means any segment where it is known that water quality does not meet applicable water quality standards, and/or is not expected to meet applicable water quality standards, even after the application of the technology-based effluent limitations required by sections 301(b) and 306 of the Act.

(i) *Water quality standards* are provisions of State or Federal law which consist of a designated use or uses for the waters of the United States and water quality criteria for such waters based upon such uses. Water quality standards are to protect the public health or welfare, enhance the quality of water and serve the purposes of the Act.

(j) *States* include: The 50 States, the District of Columbia, Guam, the Commonwealth of Puerto Rico, Virgin Islands, American Samoa, the Commonwealth of the Northern Mariana Islands, and Indian Tribes that EPA determines to be eligible for purposes of the water quality standards program.

(k) *Federal Indian Reservation, Indian Reservation,* or *Reservation* means all land within the limits of any Indian reservation under the jurisdiction of the United States Government, notwithstanding the issuance of any patent, and including rights-of-way running through the reservation."

(l) *Indian Tribe* or *Tribe* means any Indian Tribe, band, group, or community recognized by the Secretary of the Interior and exercising governmental authority over a Federal Indian reservation.

(m) *Highest attainable use* is the modified aquatic life, wildlife, or recreation use that is both closest to the uses specified in section 101(a)(2) of the Act and attainable, based on the evaluation of the factor(s) in §131.10(g) that preclude(s) attainment of the use and any other information or analyses that were used to evaluate attainability. There is no required highest attainable use where the State demonstrates the relevant use specified in section 101(a)(2) of the Act and sub-categories of such a use are not attainable.

(n) *Practicable*, in the context of §131.12(a)(2)(ii), means technologically possible, able to be put into practice, and economically viable.

(o) A *water quality standards variance* (WQS variance) is a time-limited designated use and criterion for a specific pollutant(s) or water quality parameter(s) that reflect the highest attainable condition during the term of the WQS variance.

(p) *Pollutant Minimization Program*, in the context of §131.14, is a structured set of activities to improve processes and pollutant controls that will prevent and reduce pollutant loadings.

(q) *Non-101(a)(2) use* is any use unrelated to the protection and propagation of fish, shellfish, wildlife or recreation in or on the water.

[48 FR 51405, Nov. 8, 1983, as amended at 56 FR 64893, Dec. 12, 1991; 59 FR 64344, Dec. 14, 1994; 80 FR 51046, Aug. 21, 2015]

§131.4 State authority.

(a) States (as defined in §131.3) are responsible for reviewing, establishing, and revising water quality standards. As recognized by section 510 of the Clean Water Act, States may develop water quality standards more stringent than required by this regulation. Consistent with section 101(g) and 518(a) of the Clean Water Act, water quality standards shall not be construed to supersede or abrogate rights to quantities of water.

(b) States (as defined in §131.3) may issue certifications pursuant to the requirements of Clean Water Act section 401. Revisions adopted by States shall be applicable for use in issuing State certifications consistent with the provisions of §131.21(c).

(c) Where EPA determines that a Tribe is eligible to the same extent as a State for purposes of water quality standards, the Tribe likewise is eligible to the same extent as a State for purposes of certifications conducted under Clean Water Act section 401.

[56 FR 64893, Dec. 12, 1991, as amended at 59 FR 64344, Dec. 14, 1994]

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Title 40: Protection of Environment

PART 131—WATER QUALITY STANDARDS

§131.12 Antidegradation policy and implementation methods.

(a) The State shall develop and adopt a statewide antidegradation policy. The antidegradation policy shall, at a minimum, be consistent with the following:

(1) Existing instream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected.

(2) Where the quality of the waters exceeds levels necessary to support the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water, that quality shall be maintained and protected unless the State finds, after full satisfaction of the intergovernmental coordination and public participation provisions of the State's continuing planning process, that allowing lower water quality is necessary to accommodate important economic or social development in the area in which the waters are located. In allowing such degradation or lower water quality, the State shall assure water quality adequate to protect existing uses fully. Further, the State shall assure that there shall be achieved the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable best management practices for nonpoint source control.

(i) The State may identify waters for the protections described in paragraph (a)(2) of this section on a parameter-by-parameter basis or on a water body-by-water body basis. Where the State identifies waters for antidegradation protection on a water body-by-water body basis, the State shall provide an opportunity for public involvement in any decisions about whether the protections described in paragraph (a)(2) of this section will be afforded to a water body, and

the factors considered when making those decisions. Further, the State shall not exclude a water body from the protections described in paragraph (a)(2) of this section solely because water quality does not exceed levels necessary to support all of the uses specified in section 101(a)(2) of the Act.

(ii) Before allowing any lowering of high water quality, pursuant to paragraph (a)(2) of this section, the State shall find, after an analysis of alternatives, that such a lowering is necessary to accommodate important economic or social development in the area in which the waters are located. The analysis of alternatives shall evaluate a range of practicable alternatives that would prevent or lessen the degradation associated with the proposed activity. When the analysis of alternatives identifies one or more practicable alternatives, the State shall only find that a lowering is necessary if one such alternative is selected for implementation.

(3) Where high quality waters constitute an outstanding National resource, such as waters of National and State parks and wildlife refuges and waters of exceptional recreational or ecological significance, that water quality shall be maintained and protected.

(4) In those cases where potential water quality impairment associated with a thermal discharge is involved, the antidegradation policy and implementing method shall be consistent with section 316 of the Act.

(b) The State shall develop methods for implementing the antidegradation policy that are, at a minimum, consistent with the State's policy and with paragraph (a) of this section. The State shall provide an opportunity for public involvement during the development and any subsequent revisions of the implementation methods, and shall make the methods available to the public.

[48 FR 51405, Nov. 8, 1983, as amended at 80 FR 51047, Aug. 21, 2015]

ELECTRONIC CODE OF FEDERAL REGULATIONS
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Title 40 → Chapter I → Subchapter A → Part 25
Title 40: Protection of Environment

PART 25—PUBLIC PARTICIPATION IN PROGRAMS UNDER THE RESOURCE CONSERVATION AND RECOVERY ACT, THE SAFE DRINKING WATER ACT, AND THE CLEAN WATER ACT

Contents

AUTHORITY: Sec. 101(e), Clean Water Act, as amended (33 U.S.C. 1251(e)); sec. 7004(b), Resource Conservation and Recovery Act (42 U.S.C. 6974(b)); sec. 1450(a)(1), Safe Drinking Water Act, as amended (42 U.S.C. 300j-9).

SOURCE: 44 FR 10292, Feb. 16, 1979, unless otherwise noted.

§25.1 Introduction.

This part sets forth minimum requirements and suggested program elements for public participation in activities under the Clean Water Act (Pub. L. 95-217), the Resource Conservation and Recovery Act (Pub. L. 94-580), and the Safe Drinking Water Act (Pub. L. 93-523). The applicability of the requirements of this part is as follows:

(a) Basic requirements and suggested program elements for public information, public notification, and public consultation are set forth in §25.4. These requirements are intended to foster public awareness and open processes of government decisionmaking. They are applicable to all covered activities and programs described in §25.2(a).

(b) Requirements and suggested program elements which govern the structure of particular public participation mechanisms (for example, advisory groups and responsiveness

summaries) are set forth in §§25.5, 25.6, 25.7, and 25.8. This part does not mandate the use of these public participation mechanisms. It does, however, set requirements which those responsible for implementing the mechanisms must follow if the mechanisms are required elsewhere in this chapter.

(c) Requirements which apply to Federal financial assistance programs (grants and cooperative agreements) under the three acts are set forth in §§25.10 and 25.12(a).

(d) Requirements for public involvement which apply to specific activities are set forth in §25.9 (Permit enforcement), §25.10 (Rulemaking), and §25.12 (Assuring compliance with requirements).

§25.2 Scope.

(a) The activities under the three Acts which are covered by this part are:

(1) EPA rulemaking, except non-policy rulemaking (for example publication of funding allotments under statutory formulas); and State rulemaking under the Clean Water Act and Resource Conservation and Recovery Act;

(2) EPA issuance and modification of permits, and enforcement of permits as delineated by §25.9;

(3) Development by EPA of major informational materials, such as citizen guides or handbooks, which are expected to be used over several years and which are intended to be widely distributed to the public;

(4) Development by EPA of strategy and policy guidance memoranda when a Deputy Assistant Administrator determines it to be appropriate;

(5) Development and implementation of plans, programs, standards, construction, and other activities supported with EPA financial assistance (grants and cooperative agreements) to State, interstate, regional and local agencies (herein after referred to as “State, interstate, and substate agencies”);

(6) The process by which EPA makes a determination regarding approval of State administration of the Construction Grants program in lieu of Federal administration; and the administration of the Construction Grants Program by the State after EPA approval;

(7) The process by which EPA makes a determination regarding approval of State administration of the following programs in lieu of Federal administration: The State Hazardous Waste Program; the NPDES Permit Program; the Dredge and Fill Permit Program; and the Underground Injection Control Program;

(8) Other activities which the Assistant Administrator for Water and Waste Management, the Assistant Administrator for Enforcement, or any EPA Regional Administrator deems

appropriate in view of the Agency's responsibility to involve the public in significant decisions.

(b) Activities which are not covered by this part, except as otherwise provided under (a)(8) or (c) of this section, are activities under parts 33 (Subagreements), 39 (Loan Guarantees for Construction of Treatment Works), 40 (Research and Development Grants), 45 (Training Grants and Manpower Forecasting) and 46 (Fellowships) of this chapter.

(c) Some programs covered by these regulations contain further provisions concerning public participation. These are found elsewhere in this chapter in provisions which apply to the program of interest. Regulations which govern the use and release of public information are set forth in part 2 of this chapter.

(d) Specific provisions of court orders which conflict with requirements of this part, such as court-established timetables, shall take precedence over the provisions in this part.

(e) Where the State undertakes functions in the construction grants program, the State shall be responsible for meeting these requirements for public participation, and any applicable public participation requirements found elsewhere in this chapter, to the same extent as EPA.

(f) Where the State undertakes functions in those programs specifically cited in §25.2(a)(7), the State shall be responsible for meeting the requirements for public participation included in the applicable regulations governing those State programs. The requirements for public participation in State Hazardous Waste Programs, Dredge and Fill Permit programs, Underground Injection Control programs and NPDES permit programs are found in part 123 of this chapter. These regulations embody the substantive requirements of this part.

(g) These regulations apply to the activities of all agencies receiving EPA financial assistance which is awarded after [the effective date of final regulations], and to all other covered activities of EPA, State, interstate, and substate agencies which occur after that date. These regulations will apply to ongoing grants or other covered activities upon any significant change in the activity (for example, upon a significant proposed increase in project scope of a construction grant). Parts 105 (Public Participation in Water Pollution Control) and 249 (Public Participation in Solid Waste Management) will no longer appear in the Code of Federal Regulations; however, they will remain applicable, in uncodified form, to grants awarded prior to the effective date of this part and to all other ongoing activities.

§25.3 Policy and objectives.

(a) EPA, State, interstate, and substate agencies carrying out activities described in §25.2(a) shall provide for, encourage, and assist the participation of the public. The term, “the public” in the broadest sense means the people as a whole, the general populace. There are a number of identifiable “segments of the public” which may have a particular interest in a given program or decision. Interested and affected segments of the public may be affected directly by a decision, either beneficially or adversely; they may be affected indirectly; or they may have some other concern about the decision. In addition to private citizens, the public

may include, among others, representatives of consumer, environmental, and minority associations; trade, industrial, agricultural, and labor organizations; public health, scientific, and professional societies; civic associations; public officials; and governmental and educational associations.

(b) Public participation is that part of the decision-making process through which responsible officials become aware of public attitudes by providing ample opportunity for interested and affected parties to communicate their views. Public participation includes providing access to the decision-making process, seeking input from and conducting dialogue with the public, assimilating public viewpoints and preferences, and demonstrating that those viewpoints and preferences have been considered by the decision-making official. Disagreement on significant issues is to be expected among government agencies and the diverse groups interested in and affected by public policy decisions. Public agencies should encourage full presentation of issues at an early stage so that they can be resolved and timely decisions can be made. In the course of this process, responsible officials should make special efforts to encourage and assist participation by citizens representing themselves and by others whose resources and access to decision-making may be relatively limited.

(c) The following are the objectives of EPA, State, interstate, and substate agencies in carrying out activities covered by this part:

(1) To assure that the public has the opportunity to understand official programs and proposed actions, and that the government fully considers the public's concerns;

(2) To assure that the government does not make any significant decision on any activity covered by this part without consulting interested and affected segments of the public;

(3) To assure that government action is as responsive as possible to public concerns;

(4) To encourage public involvement in implementing environmental laws;

(5) To keep the public informed about significant issues and proposed project or program changes as they arise;

(6) To foster a spirit of openness and mutual trust among EPA, States, substate agencies and the public; and

(7) To use all feasible means to create opportunities for public participation, and to stimulate and support participation.

§25.4 Information, notification, and consultation responsibilities.

(a) *General.* EPA, State, interstate, and substate agencies shall conduct a continuing program for public information and participation in the development and implementation of activities covered by this part. This program shall meet the following requirements:

(b) *Information and assistance requirements.* (1) Providing information to the public is a necessary prerequisite to meaningful, active public involvement. Agencies shall design informational activities to encourage and facilitate the public's participation in all significant decisions covered by §25.2(a), particularly where alternative courses of action are proposed.

(2) Each agency shall provide the public with continuing policy, program, and technical information and assistance beginning at the earliest practicable time. Informational materials shall highlight significant issues that will be the subject of decision-making. Whenever possible, consistent with applicable statutory requirements, the social, economic, and environmental consequences of proposed decisions shall be clearly stated in such material. Each agency shall identify segments of the public likely to be affected by agency decisions and should consider targeting informational materials toward them (in addition to the materials directed toward the general public). Lengthy documents and complex technical materials that relate to significant decisions should be summarized for public and media uses. Fact sheets, news releases, newsletters, and other similar publications may be used to provide notice that materials are available and to facilitate public understanding of more complex documents, but shall not be a substitute for public access to the full documents.

(3) Each agency shall provide one or more central collections of reports, studies, plans, and other documents relating to controversial issues or significant decisions in a convenient location or locations, for example, in public libraries. Examples of such documents are catalogs of documents available from the agency, grant applications, fact sheets on permits and permit applications, permits, effluent discharge information, and compliance schedule reports. Copying facilities at reasonable cost should be available at the depositories.

(4) Whenever possible, agencies shall provide copies of documents of interest to the public free of charge. Charges for copies should not exceed prevailing commercial copying costs. EPA requirements governing charges for information and documents provided to the public in response to requests made under the Freedom of Information Act are set forth in part 2 of this chapter. Consistent with the objectives of §25.3(b), agencies may reserve their supply of free copies for private citizens and others whose resources are limited.

(5) Each agency shall develop and maintain a list of persons and organizations who have expressed an interest in or may, by the nature of their purposes, activities or members, be affected by or have an interest in any covered activity. Generally, this list will be most useful where subdivided by area of interest or geographic area. Whenever possible, the list should include representatives of the several categories of interests listed under §25.3(a). Those on the list, or relevant portions if the list is subdivided, shall receive timely and periodic notification of the availability of materials under §25.4(b)(2).

(c) *Public notification.* Each agency shall notify interested and affected parties, including appropriate portions of the list required by paragraph (b)(5) of this section, and the media in advance of times at which major decisions not covered by notice requirements for public meetings or public hearings are being considered. Generally, notices should include the timetable in which a decision will be reached, the issues under consideration, any alternative

courses of action or tentative determinations which the agency has made, a brief listing of the applicable laws or regulations, the location where relevant documents may be reviewed or obtained, identification of any associated public participation opportunities such as workshops or meetings, the name of an individual to contact for additional information, and any other appropriate information. All advance notifications under this paragraph must be provided far enough in advance of agency action to permit time for public response; generally this should not be less than 30 days.

(d) *Public consultation.* For the purposes of this part, "public consultation" means an exchange of views between governmental agencies and interested or affected persons and organizations in order to meet the objectives set forth in §25.3. Requirements for three common forms of public consultation (public hearings, public meetings, and advisory groups) are set forth in §§25.5, 25.6, and 25.7. Other less formal consultation mechanisms may include but are not limited to review groups, ad hoc committees, task forces, workshops, include but are not limited to review groups, ad hoc committees, task forces, workshops, seminars and informal personal communications with individuals and groups. Public consultation must be preceded by timely distribution of information and must occur sufficiently in advance of decision-making to allow the agency to assimilate public views into agency action. EPA, State, interstate, and substate agencies shall provide for early and continuing public consultation in any significant action covered by this part. Merely conferring with the public after an agency decision does not meet this requirement. In addition to holding hearings and meetings as specifically required in this chapter, a hearing or meeting shall be held if EPA, the State, interstate, or substate agency determines that there is significant public interest or that a hearing or meeting would be useful.

(e) *Public information concerning legal proceedings.* EPA, State, interstate, and substate agencies shall provide full and open information on legal proceedings to the extent not inconsistent with court requirements, and where such disclosure would not prejudice the conduct of the litigation. EPA actions with regard to affording opportunities for public comment before the Department of Justice consents to a proposed judgment in an action to enjoin discharges of pollutants into the environment shall be consistent with the Statement of Policy issued by the Department of Justice (see title 28, CFR, chapter 1, §50.7).

§25.5 Public hearings.

(a) *Applicability.* Any non-adjudicatory public hearing, whether mandatory or discretionary, under the three Acts shall meet the following minimum requirements. These requirements are subordinate to any more stringent requirements found elsewhere in this chapter or otherwise imposed by EPA, State, interstate, or substate agencies. Procedures developed for adjudicatory hearings required by this chapter shall be consistent with the public participation objectives of this part, to the extent practicable.

(b) *Notice.* A notice of each hearing shall be well publicized, and shall also be mailed to the appropriate portions of the list of interested and affected parties required by §25.4(b)(5). Except as otherwise specifically provided elsewhere in this chapter, these actions must occur at least 45 days prior to the date of the hearing. However, where EPA determines that there

are no substantial documents which must be reviewed for effective hearing participation and that there are no complex or controversial matters to be addressed by the hearing, the notice requirement may be reduced to no less than 30 days. EPA may further reduce or waive the hearing notice requirement in emergency situations where EPA determines that there is an imminent danger to public health. To the extent not duplicative, the agency holding the hearing shall also provide informal notice to all interested persons or organizations that request it. The notice shall identify the matters to be discussed at the hearing and shall include or be accompanied by a discussion of the agency's tentative determination on major issues (if any), information on the availability of a bibliography of relevant materials (if deemed appropriate), and procedures for obtaining further information. Reports, documents and data relevant to the discussion at the public hearing shall be available to the public at least 30 days before the hearing. Earlier availability of materials relevant to the hearing will further assist public participation and is encouraged where possible.

(c) *Locations and time.* Hearings must be held at times and places which, to the maximum extent feasible, facilitate attendance by the public. Accessibility of public transportation, and use of evening and weekend hearings, should be considered. In the case of actions with Statewide interest, holding more than one hearing should be considered.

(d) *Scheduling presentations.* The agency holding the hearing shall schedule witnesses in advance, when necessary, to ensure maximum participation and allotment of adequate time for all speakers. However, the agency shall reserve some time for unscheduled testimony and may consider reserving blocks of time for major categories of witnesses.

(e) *Conduct of hearing.* The agency holding the hearing shall inform the audience of the issues involved in the decision to be made, the considerations the agency will take into account, the agency's tentative determinations (if any), and the information which is particularly solicited from the public. The agency should consider allowing a question and answer period. Procedures shall not unduly inhibit free expression of views (for example, by onerous written statement requirements or qualification of witnesses beyond minimum identification).

(f) *Record.* The agency holding the hearing shall prepare a transcript, recording or other complete record of public hearing proceedings and make it available at no more than cost to anyone who requests it. A copy of the record shall be available for public review.

§25.6 Public meetings.

Public meetings are any assemblies or gathering, (such as conferences, informational sessions, seminars, workshops, or other activities) which the responsible agency intends to be open to anyone wishing to attend. Public meetings are less formal than public hearings. They do not require formal presentations, scheduling of presentations and a record of proceedings. The requirements of §25.5 (b) and (c) are applicable to public meetings, except that the agency holding the meeting may reduce the notice to not less than 30 days if there is good reason that longer notice cannot be provided.

§25.7 Advisory groups.

(a) *Applicability.* The requirements of this section on advisory groups shall be met whenever provisions of this chapter require use of an advisory group by State, interstate, or substate agencies involved in activities supported by EPA financial assistance under any of the three Acts.

(b) *Role.* Primary responsibility for decision-making in environmental programs is vested by law in the elected and appointed officials who serve on public bodies and agencies at various levels of government. However, all segments of the public must have the opportunity to participate in environmental quality planning. Accordingly, where EPA identifies a need for continued attention of an informed core group of citizens in relation to activities conducted with EPA financial assistance, program regulations elsewhere in this chapter will require an advisory group to be appointed by the financially assisted agency. Such advisory groups will not be the sole mechanism for public participation, but will complement other mechanisms. They are intended to assist elected or appointed officials with final decision-making responsibility by making recommendations to such officials on important issues. In addition, advisory groups should foster a constructive interchange among the various interests present on the group and enhance the prospect of community acceptance of agency action.

that part of the record, shall note on the New Mexico sunshine portal that the part of the record is not displayed, and shall provide instructions for accessing or inspecting that part of the record.

B. Cost of duplication: The cost of duplicating documents shall be borne by the person seeking copies of such documents, but the commission administrator shall not charge a fee for providing the notice of proposed rulemaking in electronic form.

[20.1.6.105 NMAC - Rp, 20.1.6.105 NMAC, 03/16/2018]

20.1.6.106 - 20.1.1.199 [RESERVED]

20.1.6.200 PREHEARING PROCEDURES - PETITION FOR REGULATORY CHANGE:

A. Any person may file a petition with the commission to adopt, amend, or repeal any regulation within the jurisdiction of the commission.

B. The petition shall be in writing and shall include a statement of the reasons for the regulatory change. The petition shall cite the relevant statutes that authorize the commission to adopt the proposed rules and shall estimate the time that will be needed to conduct the hearing. A copy of the entire rule, including the proposed regulatory change, indicating any language proposed to be added or deleted, shall be attached to the petition. The entire rule and its proposed changes shall be submitted to the commission in redline fashion, and shall include line numbers. Any document that does not include all the items required to be in a petition shall be returned to the petitioner along with a copy of these rules and a check-off list of required items, and the petitioner will be asked to resubmit their petition in the form required by these rules.

C. The commission shall determine, at a public meeting occurring no later than 90 days after receipt of the petition, whether or not to hold a public hearing on the proposal. Any person may respond to the petition either in writing prior to the public meeting or in person at the public meeting.

D. If the commission determines to hold a public hearing on the petition, it may issue such orders specifying procedures for conduct of the hearing, in addition to those provided by this part, as may be necessary and appropriate to fully inform the commission of the matters at issue in the hearing or control the conduct of the hearing. Such orders may include requirements for giving additional public notice, holding pre-hearing conferences, filing direct testimony in writing prior to the hearing, or limiting testimony or cross-examination.

[20.1.6.200 NMAC - Rp, 20.1.6.200 NMAC, 03/16/2018]

20.1.6.201 NOTICE OF HEARINGS:

A. Unless otherwise allowed by governing law and specified by the commission, the commission shall provide to the public notice of the proposed rulemaking at least 60 days prior to the hearing.

B. Public notice for proposed regulatory changes of general application to the state shall include publication in at least one newspaper of general circulation in the state, publication in the New Mexico register, and such other means of providing notice as the commission may direct or are required by law. Notice for proposed regulatory changes that are confined in effect to a specific geographic area shall also be published in a newspaper of general circulation in the area affected.

C. The notice of proposed rulemaking shall state:

(1) the subject of the proposed rule, including a summary of the full text of the proposed rule and a short explanation of the purpose of the proposed rule;

(2) a citation to the specific legal authority authorizing the proposed rule and the adoption of the rule;

(3) a citation to technical information, if any, that served as a basis for the proposed rule, and information on how the full text of the technical information may be obtained;

(4) the statutes, regulations, and procedural rules governing the conduct of the hearing;

(5) the manner in which persons may present their views or evidence to the commission including information on participating in the public hearing;

(6) the location where persons may secure copies of the proposed regulatory change;

(7) an internet link providing free access to the full text of the proposed rule; and

(8) if applicable, that the commission may make a decision on the proposed regulatory change at the conclusion of the hearing.

[20.1.6.201 NMAC - Rp, 20.1.6.201 NMAC, 03/16/2018]

20.1.6.202 TECHNICAL TESTIMONY:

A. Any person, including the petitioner, who intends to present technical testimony at the hearing shall, no later than 20 days prior to the hearing, file a notice of intent to present technical testimony. The notice shall:

(1) identify the person for whom the witness(es) will testify;

(2) identify each technical witness the person intends to present, and state the qualifications of that witness, including a description of their educational and work background;

(3) if the hearing will be conducted at multiple locations, indicate the location or locations at which the witnesses will be present;

(4) include a copy of the direct testimony of each technical witness in narrative form, and state the estimated duration of the direct oral testimony of that witness;

(5) include the text of any recommended modifications to the proposed regulatory change; and

(6) list and attach all exhibits anticipated to be offered by that person at the hearing.

B. The hearing officer may enforce the provisions of this section through such action as the hearing officer deems appropriate, including, but not limited to, exclusion of the technical testimony of any witness for whom a notice of intent was not timely filed. If such testimony is admitted, the hearing officer may keep the record open after the hearing to allow responses to such testimony. The hearing officer may also require that written rebuttal testimony be submitted prior to hearing.

[20.1.6.202 NMAC - Rp, 20.1.6.202 NMAC, 03/16/2018]

20.1.6.203 ENTRY OF APPEARANCE: Any person may file an entry of appearance as a party. The entry of appearance shall be filed no later than 20 days before the date of the hearing on the petition. In the event of multiple entries of appearance by those affiliated with one interest group, the hearing officer may consolidate the entries, or divide the service list to avoid waste of resources.

[20.1.6.203 NMAC - Rp, 20.1.6.203 NMAC, 03/16/2018]

20.1.6.204 PARTICIPATION BY GENERAL PUBLIC:

A. Any member of the general public may testify at the hearing. No prior notification is required to present non-technical testimony at the hearing. Any such member may also offer non-technical exhibits in connection with their testimony, so long as the exhibit is not unduly repetitious of the testimony.

B. A member of the general public who wishes to submit a written statement for the record, in lieu of providing oral testimony at the hearing, shall file the written statement prior to the hearing or submit it at the hearing. Written comment must be mailed or delivered to the commission administrator.

C. If the commission changes the date of the hearing or the deadline for submitting comments as stated in the notice of proposed rulemaking, the commission shall provide to the public notice of the change.

[20.1.6.204 NMAC - Rp, 20.1.6.204 NMAC, 03/16/2018]

20.1.6.205 LOCATION OF HEARING: Unless otherwise provided by governing law, the commission shall hold hearings on proposed regulatory changes of statewide application in Santa Fe, and at other places the commission may prescribe. The commission may hold hearings on proposed regulatory changes that are not of statewide application within the area substantially affected by the proposal.

[20.1.6.205 NMAC - Rp, 20.1.6.205 NMAC, 03/16/2018]

20.1.6.206 PARTICIPATION BY CONFERENCE TELEPHONE OR OTHER SIMILAR DEVICE:

A. A member of the commission may participate in a meeting or hearing of the commission by means of a conference telephone or other similar communications equipment when it is otherwise difficult or impossible for the member to attend the meeting or hearing in person, provided that each member participating by conference telephone can be identified when speaking, all participants are able to hear each other at the same time and members of the public attending the meeting or hearing are able to hear any member of the commission who speaks at the meeting or hearing. A commission member's participation by such means shall constitute presence in person at the meeting or hearing. A commission member who needs to participate in this manner must notify the commission administrator sufficiently in advance so as to permit the commission administrator to arrange for the appropriate communications equipment.

B. A witness may participate in a hearing of the commission by means of a conference telephone or other similar communications equipment when an emergency or circumstances make it impossible for the witness to attend the hearing in person. A witness who needs to participate in this manner must receive permission from the

material to the variance. At such time as the department determines the report is administratively complete, the department shall post the report on its website, and mail or e-mail notice of its availability to those persons on a general and facility-specific list maintained by the department who have requested notice of discharge permit applications, and any person who participated in the variance process. If such conditions are not being met, or there is evidence indicating changed circumstances or newly-discovered facts or conditions that were unknown at the time the variance was initially granted, any person, including the department, may request a hearing before the commission to revoke, modify, or otherwise reconsider the variance within 90 days of the issuance of the notice of availability of the report.

F. An order of the commission is final and bars the petitioner from petitioning for the same variance without special permission from the commission. The commission may consider, among other things, the development of new information and techniques to be sufficient justification for a second petition. If the petitioner, or his authorized representative, fails to appear at the public hearing on the variance petition, the commission shall proceed with the hearing on the basis of the petition. A variance may not be extended or renewed unless a new petition is filed and processed in accordance with the procedures established by this section.

[7-19-68, 11-27-70, 9-3-72, 2-20-81, 11-15-96; 20.6.2.1210 NMAC - Rn, 20 NMAC 6.2.I.1210, 1-15-01; A, 12-21-18]

20.6.2.1211 - 20.6.2.1219: [RESERVED]

[12-1-95; 20.6.2.1211 - 20.6.2.1219 NMAC - Rn, 20 NMAC 6.2.I.1211-1219, 1-15-01]

20.6.2.1220 PENALTIES ENFORCEMENT, COMPLIANCE ORDERS, PENALTIES, ASSURANCE OF DISCONTINUANCE.: Failure to comply with the Water Quality Act, or any regulation or standard promulgated pursuant to the Water Quality Act is a prohibited act. If the secretary determines that a person has violated or is violating a requirement of the Water Quality Act or any regulation promulgated thereunder or is exceeding any water quality standard or ground water standard contained in commission regulations, or is not complying with a condition or provision of an approved or modified abatement plan, discharge plan, or permit issued pursuant to the Water Quality Act, the secretary may issue a compliance order, assess a penalty, commence a civil action in district court, or accept an assurance of discontinuance in accordance with NMSA 1978, Section 74-6-10 of the Water Quality Act.

[12-1-95; 20.6.2.1220 NMAC - Rn, 20 NMAC 6.2.I.1220, 1-15-01]

20.6.2.1221 - 20.6.2.1999: [RESERVED]

[12-1-95; 20.6.2.1221 - 20.6.2.1999 NMAC - Rn, 20 NMAC 6.2.I.1221-2099, 1-15-01]

20.6.2.2000 SURFACE WATER PROTECTION:

[12-1-95; 20.6.2.2000 NMAC - Rn, 20 NMAC 6.2.II, 1-15-01]

20.6.2.2001 PROCEDURES FOR CERTIFICATION OF FEDERAL NATIONAL POLLUTANT DISCHARGE ELIMINATION SYSTEM (NPDES) PERMITS:

A. This section applies to the state certification of draft national pollutant discharge elimination system (NPDES) permits under Section 401 of the federal Clean Water Act. The purpose of such certification is to reasonably ensure that the permitted activities will be conducted in a manner that will comply with applicable water quality standards, including the antidegradation policy, and the statewide water quality management plan.

B. After review of a draft permit, the department will either: (1) certify that the discharge will comply with the applicable provisions of Sections 208(e), 301, 302, 303, 306 and 307 of the federal Clean Water Act and with appropriate requirements of state law; (2) certify that the discharge will comply with the applicable provisions of Sections 208(e), 301, 302, 303, 306 and 307 of the Clean Water Act and with appropriate requirements of state law upon inclusion of specified conditions in the permit and include the justification for the conditions; or (3) deny certification and include reasons for the denial. If the department does not act on the certification within the time prescribed by the federal permitting agency for such action, the authority to do so shall be waived.

C. Pursuant to federal regulations at 40 CFR 124.10(c), the U.S. environmental protection agency provides notice of draft NPDES permits to the applicant (except for general permits); various local, state,

federal, tribal and pueblo government agencies; and other interested parties, and it allows at least 30 days of public comment. To the extent practicable, the department will provide public notice that the department is reviewing a draft NPDES permit for the purpose of preparing a state certification or denial pursuant to Section 401 of the federal Clean Water Act jointly with the notice provided by the U.S. environmental protection agency. The department will also post notice on its website.

D. When joint notice is impractical, the department shall provide notice that the department is reviewing a draft NPDES permit for purpose of preparing a state certification or denial pursuant to Section 401 of the federal Clean Water Act as follows:

(1) for general permits by:

(a) posting notice on the department's website;

(b) publishing notice in at least one newspaper of general circulation;

(c) mailing or e-mailing notice to those persons on the general mailing list maintained by the department who have requested such notice; and

(d) mailing or e-mailing notice to any affected local, state, federal, tribal, or pueblo government agency, as identified by the department; or

(2) for individual permits by:

(a) posting notice on the department's website;

(b) publishing notice in a newspaper of general circulation in the location of the discharge;

(c) mailing notice to the applicant;

(d) mailing or e-mailing notice to those persons on the general and facility-specific mailing list maintained by the department who have requested such notice; and

(e) mailing notice to any affected local, state, federal, tribal, or pueblo government agency, as identified by the department.

E. Public notices may describe more than one permit or permit action. The notice provided under Subsections C and D of 20.6.2.2001 NMAC shall include:

(1) for general permits:

(a) a statement that the department will accept written comments on the draft permit during the comment period including the address where comments may be submitted;

(b) a brief description of the activities that produce the discharge; and

(c) a description of the geographic area to be covered by the permit; or

(2) for individual permits:

(a) a statement that the department will accept written comments on the draft permit during the comment period including the address where comments may be submitted;

(b) the name and address of the permittee or permit applicant and, if different, of the facility or activity regulated by the permit;

(c) a brief description of the activities that produce the discharge; and

(d) a general description of the location of the discharge and the name of the receiving water.

F. Following the public notice provided under Subsections C or D of 20.6.2.2001 NMAC, there shall be a period of at least 30 days during which interested persons may submit written comments to the department. The 30-day comment period shall begin on the date of the public notice provided under Subsections C or D of 20.6.2.2001 NMAC. The department shall consider all pertinent comments.

G. Following the public comment period provided under Subsection F of 20.6.2.2001 NMAC, the department shall issue a final permit certification including any conditions that the department places on the certification, or issue a statement of denial including the reasons for the denial. The final certification will generally be issued within 45 days from the date a request to grant, deny or waive certification is received by the department, unless the department in consultation with the U.S. environmental protection agency regional administrator finds that unusual circumstances require a longer time. The department shall send a copy of the final permit certification or denial to the U.S. environmental protection agency, the applicant (except for general permits), and those members of the public who submitted comments to the department.

(1) The permit certification shall be in writing and shall include:

(a) the name of the applicant (except for general permits) and the NPDES permit number;

(b) a statement that the department has examined the application or other relevant information and bases its certification upon an evaluation of the information contained in such application or other information which is relevant to water quality considerations;

(c) a statement that there is a reasonable assurance that the activity will be conducted in a manner which will not violate applicable water quality standards;

(d) a statement of any conditions which the department deems necessary or desirable with respect to the discharge of the activity;

(e) identification of any condition more stringent than that in the draft permit required to assure compliance with the applicable provisions of Sections 208(e), 301, 302, 303, 306 and 307 of the Clean Water Act and with appropriate requirements of state law citing the Clean Water Act or state law upon which the condition is based;

(f) a statement of the extent to which each condition of the draft permit can be made less stringent without violating the requirements of state law, including water quality standards; and

(g) such other information as the department may determine to be appropriate.

(2) With justification, including any of the reasons listed in the New Mexico Water Quality Act, NMSA 1978, Section 74-6-5(E), the department may deny permit certification. Denial of permit certification shall be in writing and shall include:

(a) the name of the applicant (except for general permits) and the NPDES permit number;

(b) a statement that the department has examined the application or other relevant information and bases its denial upon an evaluation of the information contained in such application or other information which is relevant to water quality considerations;

(c) a statement of denial including the reasons for the denial; and

(d) such other information as the department may determine to be appropriate.

H. Any person who is adversely affected by the certification or denial of a specific permit may appeal such certification or denial by filing a petition for review with the secretary within 30 days after the department issues the final permit certification or statement of denial. Such petition shall be in writing and shall include a concise statement of the reasons for the appeal and the relief requested. The secretary may hold a hearing on the appeal. In any such appeal hearing, the procedures of 20.1.4 NMAC shall not apply. The department shall give notice of the appeal hearing at least 30 days prior to the hearing. The notice shall state the date, time, and location of the appeal hearing and shall include the pertinent information listed in Subparagraphs (b), (c), and (d) of Paragraph (2) of Subsection E of 20.6.2.2001 NMAC. The secretary shall appoint a hearing officer to preside over the appeal hearing. Any person may present oral or written statements, data, technical information, legal arguments, or other information on the permit certification or denial during the appeal hearing. Any person may present oral or written statements, data, technical information, legal arguments, or other information in rebuttal of that presented by another person. Reasonable time limits may be placed on oral statements, and the submission of written statements may be required. The hearing officer may question persons presenting oral testimony. Cross examination of persons presenting oral statements shall not otherwise be allowed. Within 30 days after the completion of the hearing, or such other time as the secretary may order given the complexities of the case, the hearing officer shall submit recommendations to the secretary. The secretary shall issue a final decision on the appeal within 30 days after receiving the recommendation, or such other time as the secretary may order given the complexities of the case.

I. Pursuant to the New Mexico Water Quality Act, NMSA 1978, Section 74-6-5(O), any person who is adversely affected by the secretary's final decision may file with the commission a petition for review of that decision based on the administrative record.

[20.6.2.2001 NMAC - N, 5-18-11]

20.6.2.2002 PROCEDURES FOR CERTIFICATION OF FEDERAL PERMITS FOR DISCHARGE OF DREDGED OR FILL MATERIAL:

A. This section applies to the state certification of draft permits or permit applications for the discharge of dredged or fill material under Section 401 of the federal Clean Water Act. The purpose of such certification is to reasonably ensure that the permitted activities will be conducted in a manner that will comply

with applicable water quality standards, including the antidegradation policy, and the statewide water quality management plan.

B. After review of a draft permit or permit application, the department will either: (1) certify that the discharge will comply with the applicable provisions of Sections 301, 302, 303, 306 and 307 of the federal Clean Water Act and with appropriate requirements of state law; (2) certify that the discharge will comply with the applicable provisions of Sections 301, 302, 303, 306 and 307 of the Clean Water Act and with appropriate requirements of state law upon inclusion of specified conditions in the permit and include the justification for the conditions; or (3) deny certification and include reasons for the denial. If the department does not act on the certification within the time prescribed by the federal permitting agency for such action, the authority to do so shall be waived.

C. Pursuant to federal regulations at 33 CFR 325.3 and 33 CFR 330.5, the U.S. army corps of engineers provides notice of draft dredged or fill permits and permit applications to the applicant (except for general or nationwide permits); various local, state, federal, tribal and pueblo government agencies; and other interested parties, and it allows at least 15 days of public comment. To the extent practicable, the department will provide public notice that the department is reviewing a draft permit or permit application for the purpose of preparing a state certification or denial pursuant to Section 401 of the federal Clean Water Act jointly with the notice provided by the U.S. army corps of engineers. The department will also post notice on its website.

D. When joint notice is impractical, the department shall provide notice that the department is reviewing a draft dredged or fill permit or permit application for purpose of preparing a state certification or denial pursuant to Section 401 of the federal Clean Water Act as follows:

(1) for general permits by:

(a) posting notice on the department's website;

(b) publishing notice in at least one newspaper of general circulation;

(c) mailing or e-mailing notice to those persons on the general mailing list maintained by the department who have requested such notice; and

(d) mailing or e-mailing notice to any affected local, state, federal, tribal, or pueblo government agency, as identified by the department; or

(2) for individual permit applications by:

(a) posting notice on the department's website;

(b) publishing notice in a newspaper of general circulation in the location of the discharge;

(c) mailing notice to the applicant;

(d) mailing or e-mailing notice to those persons on the general and facility-specific mailing list maintained by the department who have requested such notice; and

(e) mailing notice to any affected local, state, federal, tribal, or pueblo government agency, as identified by the department.

E. Public notices may describe more than one permit or permit action. The notice provided under Subsections C and D of 20.6.2.2002 NMAC shall include:

(1) for general permits:

(a) a statement that the department will accept written comments on the draft permit during the comment period including the address where comments may be submitted;

(b) a brief description of the activities that produce the discharge; and

(c) a description of the geographic area to be covered by the permit; or

(2) for individual permit applications:

(a) a statement that the department will accept written comments on the permit application during the comment period including the address where comments may be submitted;

(b) the name and address of the permittee or permit applicant and, if different, of the facility or activity regulated by the permit;

(c) a brief description of the activities that produce the discharge; and

(d) a general description of the location of the discharge and the name of the receiving water.

F. Following the public notice provided under Subsections C or D of 20.6.2.2002 NMAC, there shall be a period of at least 30 days during which interested persons may submit written comments to the

department. The 30-day comment period shall begin on the date of the public notice provided under Subsections C or D of 20.6.2.2002 NMAC. The department shall consider all pertinent comments.

G. The public notice provisions in Subsection C and D of Section 20.6.2.2002 NMAC and the public comment provisions in Subsection F of Section 20.6.2.2002 NMAC shall not apply to permits issued using emergency procedures under 33 CFR 325.2(e)(4). However, even in emergency situations, reasonable efforts shall be made to receive comments from interested state and local agencies and the affected public.

H. Following the public comment period provided under Subsection F of 20.6.2.2002 NMAC, the department shall issue a final permit certification including any conditions that the department places on the certification, or issue a statement of denial including the reasons for the denial. The final certification will generally be issued within 60 days from the date a request to grant, deny or waive certification is received by the department, unless the department in consultation with the U.S. army corps of engineers district engineer finds that unusual circumstances require a longer time. The department shall send a copy of the final permit certification or denial to the army corps of engineers, the applicant (except for general or nationwide permits), and those members of the public who submitted comments to the department.

(1) The permit certification or denial shall be in writing and shall include:

(a) the name of the applicant (except for general permits) and the permit number;

(b) a statement that the department has examined the application or other relevant information and bases its certification upon an evaluation of the information contained in such application or other information which is relevant to water quality considerations;

(c) a statement that there is a reasonable assurance that the activity will be conducted in a manner which will not violate applicable water quality standards;

(d) a statement of any conditions which the department deems necessary or desirable with respect to the discharge of the activity; and

(e) such other information as the department may determine to be appropriate.

(2) With justification, including any of the reasons listed in the New Mexico Water Quality Act, NMSA 1978, Section 74-6-5(E), the department may deny permit certification. Denial of permit certification shall be in writing and shall include:

(a) the name of the applicant (except for general permits) and the permit number;

(b) a statement that the department has examined the application or other relevant information and bases its denial upon an evaluation of the information contained in such application or other information which is relevant to water quality considerations;

(c) a statement of denial including the reasons for the denial; and

(d) such other information as the department may determine to be appropriate.

I. Any person who is adversely affected by the certification or denial of a specific permit may appeal such certification or denial by filing a petition for review with the secretary within 30 days after the department issues the final permit certification or statement of denial. Such petition shall be in writing and shall include a concise statement of the reasons for the appeal and the relief requested. The secretary may hold a hearing on the appeal. In any such appeal hearing, the procedures of 20.1.4 NMAC shall not apply. The department shall give notice of the appeal hearing at least 30 days prior to the hearing. The notice shall state the date, time, and location of the appeal hearing and shall include the pertinent information listed in Subparagraphs (b), (c), and (d) of Paragraph (2) of Subsection E of 20.6.2.2002 NMAC. The secretary shall appoint a hearing officer to preside over the appeal hearing. Any person may present oral or written statements, data, technical information, legal arguments, or other information on the permit certification or denial during the appeal hearing. Any person may present oral or written statements, data, technical information, legal arguments, or other information in rebuttal of that presented by another person. Reasonable time limits may be placed on oral statements, and the submission of written statements may be required. The hearing officer may question persons presenting oral testimony. Cross examination of persons presenting oral statements shall not otherwise be allowed. Within 30 days after the completion of the hearing, or such other time as the secretary may order given the complexities of the case, the hearing officer shall submit recommendations to the secretary. The secretary

shall issue a final decision on the appeal within 30 days after receiving the recommendation, or such other time as the secretary may order given the complexities of the case.

J. Pursuant to the New Mexico Water Quality Act, NMSA 1978, Section 74-6-5(O), any person who is adversely affected by the secretary's final decision may file with the commission a petition for review of that decision based on the administrative record.

[20.6.2.2002 NMAC - N, 5-18-11]

20.6.2.2003 PROCEDURES FOR CERTIFICATION OF OTHER FEDERAL PERMITS:

A. This section applies to the state certification of draft federal permits, permit applications or licenses under Section 401 of the federal Clean Water Act, except for NPDES permits or permits for the discharge of dredged or fill material. For example, this section applies to certification of permits or licenses issued by the federal energy regulatory commission (FERC) and to permits or licenses issued under the Rivers and Harbors Act of 1899. The purpose of such certification is to reasonably ensure that the permitted activities will be conducted in a manner that will comply with applicable water quality standards, including the antidegradation policy, and the statewide water quality management plan.

B. After review of a draft permit, permit application or license, the department will either: (1) certify that the activity will comply with the applicable provisions of Sections 301, 302, 303, 306 and 307 of the federal Clean Water Act and with appropriate requirements of state law; (2) certify that the activity will comply with the applicable provisions of Sections 301, 302, 303, 306 and 307 of the Clean Water Act and with appropriate requirements of state law upon inclusion of specified conditions in the permit and include the justification for the conditions; or (3) deny certification and include reasons for the denial. If the department does not act on the certification within the time prescribed by the federal permitting agency for such action, the authority to do so shall be waived.

C. To the extent practicable, the department will provide public notice that the department is reviewing a draft federal permit, permit application or license for the purpose of preparing a state certification or denial jointly with the notice provided by the federal permitting or licensing agency. The department will also post notice on its website.

D. When joint notice is impractical, the department shall provide notice that the department is reviewing a draft federal permit, permit application or license for purpose of preparing a state certification or denial pursuant to Section 401 of the federal Clean Water Act as follows:

(1) for general permits or licenses by:

(a) posting notice on the department's website;

(b) publishing notice in at least one newspaper of general circulation;

(c) mailing or e-mailing notice to those persons on the general mailing list maintained by the department who have requested such notice; and

(d) mailing or e-mailing notice to any affected local, state, federal, tribal, or pueblo government agency, as identified by the department; or

(2) for individual permits or licenses by:

(a) posting notice on the department's website;

(b) publishing notice in a newspaper of general circulation in the location of the permitted or licensed activity;

(c) mailing notice to the applicant;

(d) mailing or e-mailing notice to those persons on the general and facility-specific mailing list maintained by the department who have requested such notice; and

(e) mailing notice to any affected local, state, federal, tribal, or pueblo government agency, as identified by the department.

E. Public notices may describe more than one license, permit or permit action. The notice provided under Subsections C and D of 20.6.2.2003 NMAC shall include:

(1) for general permits or licenses:

(a) a statement that the department will accept written comments on the permit or license during the comment period including the address where comments may be submitted; and

(b) a brief description of the permitted or licensed activities; and

(c) a description of the geographic area to be covered by the permit; or

(2) for individual permits or licenses:

WATER QUALITY CONTROL COMMISSION

DELEGATION OF RESPONSIBILITIES TO ENVIRONMENTAL IMPROVEMENT DIVISION AND OIL CONSERVATION DIVISION

In an effort to prevent duplication of effort and to clarify the division of responsibilities pursuant to the provisions of the Water Quality Act, NMSA Sections 74-6-1 et seq. (1978), as administered and enforced by the Water Quality Control Commission, the Commission hereby approves the following list of delegated duties and responsibilities for two of the agencies that are constituent agencies to which authority can be delegated, the Environmental Improvement Division ("EID") and the Oil Conservation Division ("OCD"). The Commission is specifically authorized to take this action by NMSA Section 74-6-4E (1978) and by other general provisions of the Water Quality Act. The Commission notes that pursuant to NMSA Section 74-6-9C (1978), constituent agencies may "report to the Commission and to other constituent agencies water pollution conditions that are believed to require action where the circumstances are such that the responsibility appears to be outside the responsibility assigned to the agency making the report." The Commission encourages OCD and EID to continue close communication and cooperation where responsibility is unclear, to ensure that water pollution is prevented or abated quickly, efficiently and consistently. In situations involving discharges or facilities under the jurisdiction of both agencies, the agencies shall mutually agree which shall be the lead agency and shall determine the method by which the discharge plan shall be evaluated and approved. In preparing this delegation statement, the Commission is cognizant of the limitations imposed on its authority by the Water Quality Act, especially NMSA Section 74-6-12G (1978) which prohibits it from taking any action which would "interfere with the exclusive authority of the Oil Conservation Commission over all persons and things necessary to prevent water pollution as a result of oil or gas operations...."

This delegation shall supersede all previous delegations to EID and OCD; reference to the dates and minutes of Commission meetings in which previous delegations were made are in parentheses and the minutes are attached. The specific grants of authority are not intended to be comprehensive. When a question of authority and jurisdiction arises, which is not specifically delegated, the general provisions below shall control.

1. General Provisions

As a general rule, OCD will administer and enforce applicable Commission regulations pertaining to surface and ground water discharges at oil and natural gas production sites, oil refineries, natural gas processing plants, geothermal installations, carbon dioxide facilities, natural gas transmission lines, and discharges

associated with activities of the oil field service industry. The Commission recognizes that OCD also administers regulations under both the Oil and Gas Act and the Geothermal Resources Act, and that OCD shall have discretion as to which regulations to enforce in any given situation. OCD shall have jurisdiction over all activities associated with exploration for or development, production, transportation before refinement, refinement, storage or treatment of unrefined oil and natural gas, or oil or gas products on refinery premises.

EID will administer and enforce Commission regulations regarding discharges from transmission, transportation and storage facilities for oil or oil by-products after refinement (including but not limited to gasoline stations), except those within refinery premises. EID will administer and enforce all Commission regulations pertaining to all other discharges to surface and ground water which are not specifically delegated to other departments and agencies. (Source: 1/13/69 and 5/8/84 Commission minutes)

2. Specific Grants of Authority

A. EID shall certify Section 404 dredge and fill material permits under the Clean Water Act ("CWA"). (Source: 1/13/76 and 6/14/83 Commission minutes)

B. EID shall administer the Wastewater Construction Grants program pursuant to Section 205 of the CWA. (Source: 6/14/83 Commission minutes)

C. EID shall certify NPDES permits pursuant to Title IV of the Federal Water Pollution Control Act Amendments of 1972 and S402 of the CWA. (Source: 10/1/74 and 8/14/84 Commission minutes)

D. EID shall certify hydropower licenses issued by the Federal Energy Regulatory Commission. (Source: 8/14/84 Commission minutes)

E. EID shall administer and enforce Commission regulations pertaining to the disposal of human excrement and bath water at oil and natural gas production sites, oil refineries, natural gas processing plants, geothermal installations, carbon dioxide facilities and natural gas transmission lines when the treatment facilities for the sewage are a separate and isolated discharge unmixed with any produced water, oil field waste or oil field service waste. (Such an isolated discharge would include: a small sewage treatment plant, package plant, or septic tank and drainfield.) If, on the other hand, sewage is in a discharge combined or mixed with produced water, oil field waste or oil field service waste, OCD shall have jurisdiction. (Source: 5/8/84 Commission minutes)

F. OCD shall administer and enforce Commission regulations at brine manufacturing operations and concerning discharges to ground or surface water at brine manufacturing operations, including all brine production wells, holding ponds and tanks. OCD shall have jurisdiction over all manufactured brine once it is transported, used or disposed of off brine plant premises for use in or directly related to oil and gas operations regulated by OCD. OCD shall regulate brine injection through its Class II Underground Injection control (UIC) Program if the brine is used in the drilling for or production of oil and gas. EID shall regulate brine injection through its UIC Program if the brine is used for other purposes. (Source: 6/23/89 Commission minutes)

G. EID shall administer and enforce all programs implemented by the state under PL 92-500 (The Federal Water Pollution Control Act) and its Amendments, unless directed otherwise by the Commission. (Source: 7/8/75 Commission minutes)

H. OCD shall have general jurisdiction over the oil field service industry. Many activities that would ordinarily be regulated by EID are regulated by OCD when those activities occur in the oil field service industry. The following list, which is not intended to be inclusive, serves to help clarify this delegation:

OCD	EID
waste oil handled or processed by oil field service companies or treating plants	used motor oil handlers
all underground and above-ground tanks on refinery premises, unless the tanks contain unmixed sewage; all underground and above-ground tanks not on refinery premises which contain crude petroleum, produced water or oil field service chemicals	all underground and above-ground tanks not on refinery premises, unless the tanks contain crude petroleum, produced water or oil field service chemicals
tanker trucks hauling, spilling or disposing of well-service chemicals, kill water, produced water, crude oil, tank bottom sludge and other oil field wastes and oil field service materials	tanker trucks spilling or disposing of non-oil and gas production wastes, non-oil and gas service materials, or refined petroleum products
washings from trucks and other equipment used in the transport, production or refining of oil and gas crude products, production wastes or service materials	washings from trucks and other equipment not used for oil and gas production related purposes

Both EID and OCD are authorized to continue to take appropriate legal action in their respective areas of delegation (including initiating proceedings in court) on behalf of the Commission on a finding of good cause to believe any person is violating or is threatening to violate a Commission regulation or the Water Quality Act. The agencies shall send a copy of each Complaint, Settlement Agreement and Judgment to the Commission Secretary for distribution to Commission members. (Source: NMSA Section 74-1-8.2(B) (1978), 2/8/71 and 1/11/83 Commission minutes)

WATER QUALITY CONTROL COMMISSION

By: Richard Mitzelfelt, Chairman

Date Aug 21, 1989

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 6
1445 ROSS AVENUE, SUITE 1200
DALLAS, TEXAS 75202 – 2733

AUG 11 2017

Mr. Larry Dominguez
Chair
New Mexico Water Quality Control Commission
P.O. Box 5469
Santa Fe, NM 87502

RE: New Mexico 2013 Triennial Revisions to 20.6.4 NMAC

Dear Mr. Dominguez:

I am writing in regards to the U.S. Environmental Protection Agency's (EPA's) review of revisions to New Mexico's *Standards for Interstate and Intrastate Surface Waters* 20.6.4 New Mexico Administrative Code (NMAC), which were effective for state purposes on March 2, 2017, and submitted by the New Mexico Water Quality Control Commission for EPA review on March 14, 2017 ("the 2013 Triennial Revisions"). Subsequent to EPA's June 8, 2017 action on the state's revised water quality standards (WQS), EPA discovered that the Commission had not approved amendments that we believed had been adopted, and as a result is revising its action.

In making our initial determination, EPA believed the Commission had approved use and supporting criteria changes for segments 20.6.4.103, 116, 124, 204, 206, 207, 213, 219, and 308 NMAC. As a result, our June 8, 2017 approval included these proposed revisions. EPA has subsequently learned that these proposed modifications were not approved by the Commission and therefore never became New Mexico regulation. Under federal regulations at 40 C.F.R. §§ 131.5 and 131.21, EPA must review officially adopted revisions to state WQS. Because these proposed revisions were not officially adopted by the Commission, these provisions are not approved by EPA and not effective for CWA purposes. See 40 CFR §131.21(c), which states that state WQS are not effective for CWA purposes unless and until approved by EPA.

Section II of the enclosed amended Technical Support Document (TSD) details EPA's actions pursuant to CWA §303(c) and its implementing regulations at 40 CFR Part 131 concerning these segments and supersedes the TSD that accompanied our June 8, 2017 action. The TSD has also been amended to clarify the applicability of aluminum criteria. There are no other changes to our June 8, 2017 action. Please be advised that EPA is not taking any action on the new or revised WQS for those waters or portions of waters located in Indian Country, as defined in 18 U.S.C. § 1151.

NMED Exhibit 32

We appreciate the collaborative effort by the Surface Water Quality Bureau to complete the triennial review process. If you have any questions or concerns, please contact me at (214) 665-7101, or have your staff contact Russell Nelson at (214) 665-6646.

Sincerely,

William K. Honker, P.E.
Director
Water Division

cc: Pam Castaneda
Administrator for Boards & Commissions
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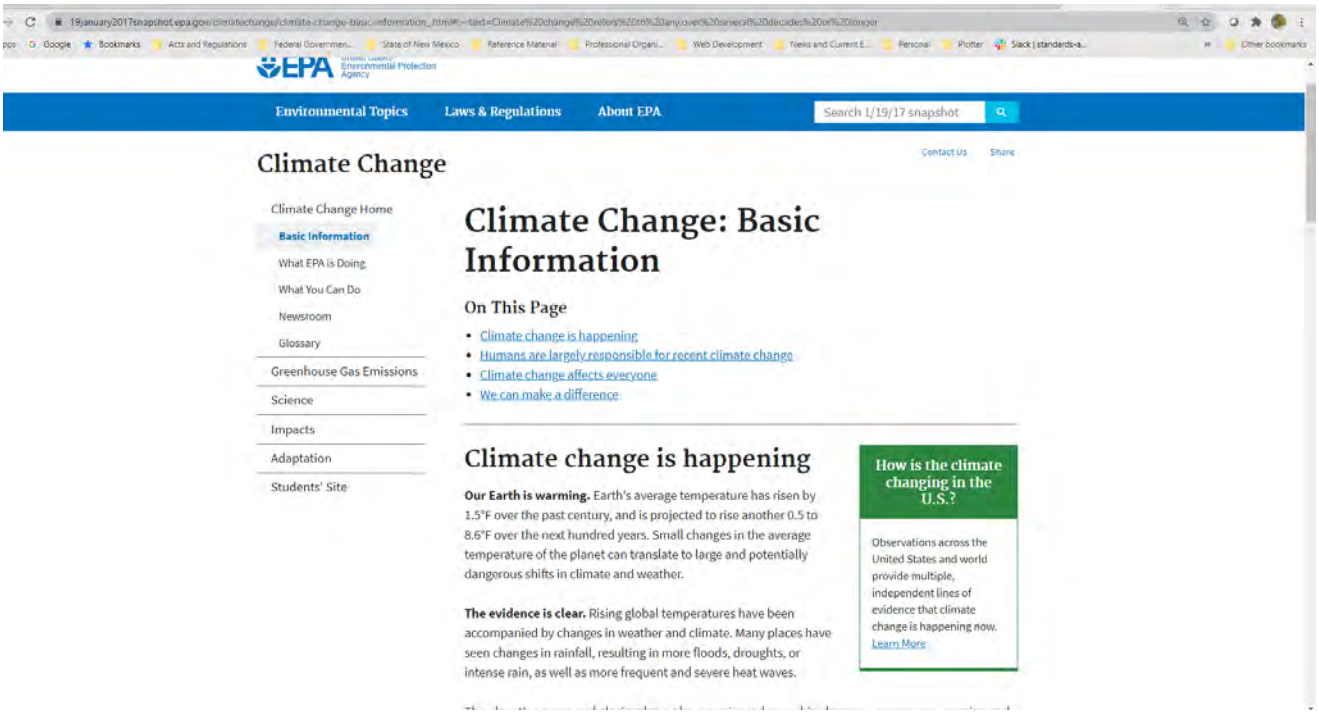

What are climate change and global warming?

Global warming refers to the recent and ongoing rise in global average temperature near Earth's surface. It is caused mostly by increasing concentrations of greenhouse gases in the atmosphere. Global warming is causing climate patterns to change. However, global warming itself represents only one aspect of climate change.

Climate change refers to any significant change in the measures of climate lasting for an extended period of time. In other words, climate change includes major changes in temperature, precipitation, or wind patterns, among other effects, that occur over several decades or longer.

Learn more about the signs of climate change in the United States.

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Effects of watershed topography, soils, land use, and climate on baseflow hydrology in humid regions: A review

Katie Price
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Abstract
Baseflow is the portion of streamflow that is sustained between precipitation events, fed to stream channels by delayed (usually subsurface) pathways. Understanding baseflow processes is critical to issues of water quality, supply, and habitat. This review synthesizes the body of global literature investigating relationships between baseflow and watershed characteristics of geomorphology, soil, and land use, as well as the potential effects of climate change, with an emphasis on humid, tropical and temperate (non-snowpack-dominated) regions. Such factors are key controls on baseflow through their influence on infiltration, rates of water removal from the catchment, and subsurface storage properties. The literature shows that there is much that remains to be resolved in gaining a solid understanding of the influence of watershed characteristics on baseflow. While it is clear that watershed geomorphology influences baseflow, there is no consensus on which geomorphic parameters are most closely linked to subsurface storage and baseflow. Many studies associate higher watershed forest cover with lower baseflows, attributed to high evapotranspiration rates of forests, while other studies indicate increased baseflow with higher watershed forest cover due to higher infiltration and recharge of subsurface storage. The demonstrated effects of agriculture and urbanization are also inconsistent, due to varied additions of imported water and extremely variable background conditions. This review underscores the need for more research that addresses multiple aspects of the watershed system in explaining baseflows, and for methodological consistency to allow for more fruitful comparisons across case studies. These needs are of immediate demand, given scientific and management emphasis on environmental flows required for maintenance of key ecosystem services.

Keywords
baseflow, catchment, climate change, ecosystem services, environmental flows, watershed

I Introduction

Baseflow is influenced by natural factors such as climate, geology, relief, soils, and vegetation. Human impacts on the landscape may modify some or all of these factors, in turn affecting baseflow timing and quantity. The need for a greater understanding of streamflow response

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to external change has been recognized for decades, but previous research has tended to emphasize flood response to increased human pressures on the landscape (e.g. Choi, 2008; Knox, 2001). In this respect, the literature is lacking with regard to studies investigating baseflow response to human impact. A scientific understanding of watershed processes and baseflow is critical to effective water policy and management. Population growth is associated with increasing demands on freshwater resources for industry, agriculture, and human consumption, and water shortages are not uncommon, even in humid regions (Hornbeck et al., 1993). A firmer grasp on the controls of baseflow is pivotal in issues of contaminant dilution (Barnes and Kalita, 2001; Jordan et al., 1997; Novotny and Olem, 1994), stream ecology (Boulton, 2003; Klein, 1979; Konrad and Booth, 2005), and adequate water supply to population centers (Hornbeck et al., 1993; Illinois EPA, 2002). Ensuring safe concentrations of contaminants associated with wastewater effluent requires accurate estimation of baseflow discharge (Smakhtin, 2001), and contaminants that enter stream systems via soil or groundwater storage are most highly concentrated during baseflow. These factors carry negative implications for stream biota and human consumption if baseflows are reduced (Barnes and Kalita, 2001; Dewson et al., 2007; Novotny and Olem, 1994). Reduced baseflow is also associated with reduced stream width, warmer temperatures, lower dissolved oxygen, and higher nutrient concentrations that may promote excessive growth of habitat-choking algae (Leigh, 2010; Price and Leigh, 2006a). These conditions are often fatally stressful for sensitive, endemic species, and low water levels in streams have been associated with decreases in richness of aquatic macroinvertebrate and fish species (Boulton, 2003; Mote et al., 2003).

The objective of this review was to synthesize research from various water resources disciplines, in order to provide a cohesive summary of the current state of research knowledge regarding the influences of watershed characteristics on stream baseflows and to address the potential impacts of climate change in this context. Water resource management requires a firmer understanding of baseflow processes, and a secondary objective of this review is to identify key research questions that remain unanswered. This review emphasizes literature covering geomorphic and anthropogenic effects on baseflow in humid, temperate and tropical regions of the world. Baseflow-controlling processes in polar and arid settings are sufficiently unique to merit specific treatment elsewhere. The introductory section covers a basic definition of baseflow, as well as discussion of primary controls on baseflow and various approaches to quantification. Next, a section on geomorphic controls on baseflow discharge covers the influences of basin geology, surface topography, subsurface topography, and soils. This section is followed by an overview of anthropogenic effects on baseflow, with emphases on forest removal, agriculture, and urbanization, because of the large body of research on those topics. Next, a summary of current research evaluating and predicting baseflow response to climate change is presented. The review concludes with a discussion of key research topics, the results of which would fill large gaps in our understanding of watershed hydrology and baseflow.

1 Baseflow overview

Within the literature, there is inconsistent terminology usage, with ‘baseflow’ and ‘low flow’ commonly used interchangeably to denote streamflow occurring between precipitation and/or snowmelt events, resulting from sustained subsurface inputs to the stream channel. These and other terms are also inconsistently differentiated within the literature to specify the lowest annual streamflow within a watershed or region. In this review, the term ‘baseflow’ will be used

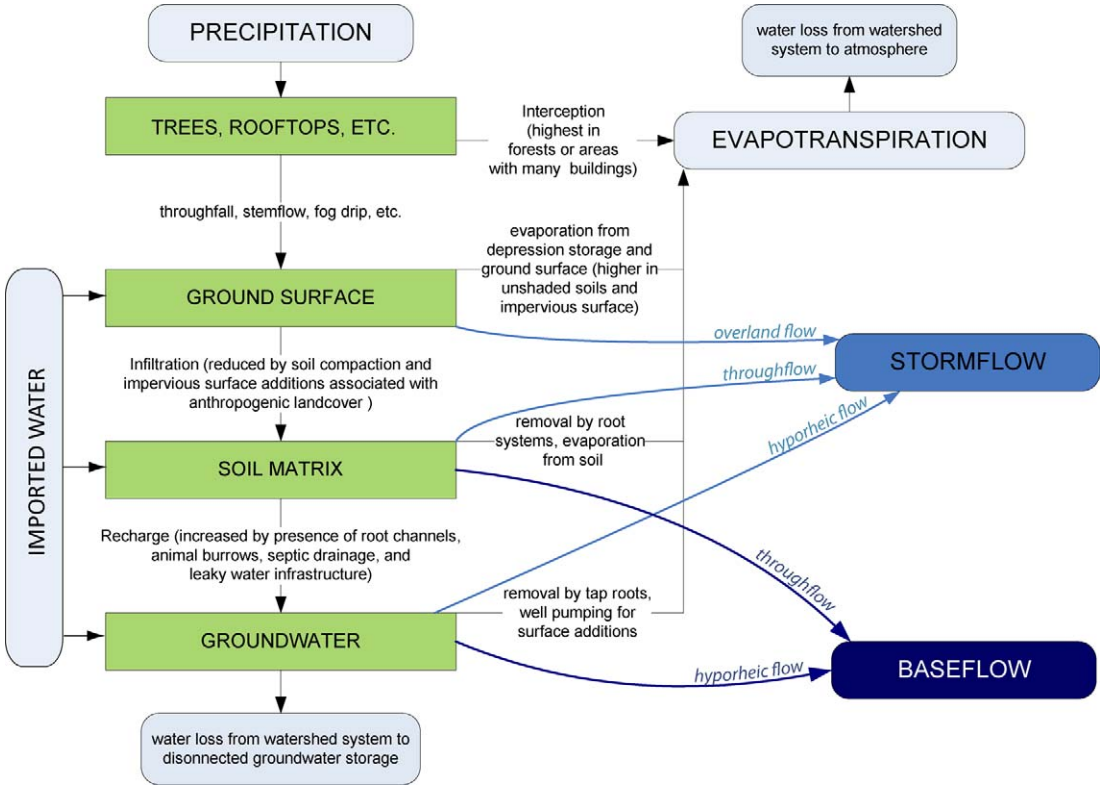

Figure 1. Conceptual model of watershed inputs, storage, and losses, and their roles in determining baseflow quantity. The primary input is precipitation, with imported water serving as an important input in some developed and agricultural watersheds. Factors of land use and climate change that increase infiltration and recharge are positively associated with baseflow, while those that increase evapotranspirative loss are negatively associated with baseflow. Prediction of baseflow response to environmental change requires consideration of both types of factors.

generally to represent streamflow fed from deep subsurface and delayed shallow subsurface storage between precipitation and/or snowmelt events (Ward and Robinson, 1990), and 'low flow' will specify dry season minimum flows (Smakhtin, 2001).

Several sources emphasize that 'baseflow' is not synonymous with groundwater flow, as it includes water transmitted from shallow unsaturated storage in addition to significant contributions as hyporheic flow from phreatic storage (Anderson and Burt, 1980; Brutsaert, 2005; Buttle, 1998; Ward and Robinson, 1990). In addition to bedrock water storage, baseflow is also derived from the drainage of near-surface valley soils and riparian zones, as water concentrates in these areas during and following precipitation events (Brutsaert, 2005; Smakhtin, 2001). Factors that promote infiltration and recharge of subsurface storage will increase baseflows, while factors associated with higher evapotranspiration (ET) will reduce baseflows (Figure 1). Baseflow is naturally influenced by a wide range of factors (Brutsaert, 2005):

- Basin physiographic characteristics;
- Distribution of storage in river channels and groundwater aquifers;
- Evapotranspiration (ET) from stream banks and throughout the catchment;

- Geomorphology of the landscape and stream network;
- Configuration and nature of the riparian aquifers and near-surface soils.

Many of these factors may be altered with human impact on the landscape, and it thus becomes critical to understand not only the relationships between basin physical properties and stream baseflow, but also the ways in which direct anthropogenic watershed impacts and climate change affect these physical properties.

2 Methods of identifying baseflow sources and residence/transit times

Many types of tracers are used for both source-apportionment ('fingerprinting') of baseflow sources and for estimation of transit times of water from the time it enters the watershed as precipitation to its exit at the stream outlet. Stable and radioactive environmental isotopes provide information on the hydrogeological characteristics of aquifers including the origin, time, and rate of recharge, and aquifer interconnections (Gonfiantini et al., 1998). Tracers as naturally occurring solutes, 'injected' solutes, and the isotopic composition of the water molecule itself have all been successfully used in baseflow studies. Solutes that originate in distinct areas of watersheds (based on geochemical or landcover differences) can be used for source-apportionment of baseflow (e.g. Gburek and Folmar, 1999; Lindgren et al., 2004). Geochemical solutes related to weathering reactions can be used to identify whether water is sourced from bedrock, where weathered ions are readily dissolved into the water, versus the regolith and soil mantle, from which these ions were long ago removed during the weathering process (Tetzlaff et al., 2007; Velbel, 1985). End-member mixing analysis (EMMA) is a method commonly used for source-apportionment of water, based on distinct ratios of multiple solutes originating in different parts of the catchment, generally based on mineralogical and geological differences (Christophersen et al., 1990; Genereaux et al., 1993). The ability to use natural geochemical signatures for source-apportionment varies with geologic setting, particularly the mineralogy and age of the landscape. In the absence of naturally occurring geochemical signatures, or to supplement such analyses, tracers may be injected into different portions of the watershed, in order to identify flowpaths and transit times. Dissolved gases, such as chloride and bromide, and plugs of highly saline water are commonly used as injected tracers (Solomon et al., 1998; Wang et al., 2009). Radioactive isotopes, such as radon, have also been used to identify baseflow sources and timing (e.g. Genereaux et al., 1993). A very active area of hydrologic research involves the use of stable isotopes in water molecules, which is thoroughly covered by Kendall and McDonnell (1998), and in more recent reviews by McGuire and McDonnell (2006) and Michel (2009). The varied concentrations of oxygen and hydrogen isotopes in precipitation versus stored water are used to fingerprint stream water for source-apportionment, and to distinguish stored water (or 'old water') from recent precipitation and surface runoff ('new water'). The potential exists for tracers to be used for the estimation of catchment water storage volumes, although to date this application is underexplored (Soulsby et al., 2009). A thorough review of the quantitative methods and issues surrounding estimation of water transit times is presented by McGuire and McDonnell (2006).

3 Quantifying baseflow and low flow

There is no standard method for quantification of baseflow, due to the large variety of research and management objectives and lengths of available streamflow records. There are four major categories of metrics used to summarize baseflow from an existing streamflow record: (1) event-based low flow statistics; (2) flow-duration curve

statistics; (3) metrics that express the proportion of baseflow to total flow; and (4) baseflow recession statistics. Additionally, many researchers have attempted to build predictive equations, based on watershed characteristics and meteorological conditions, to estimate baseflow in ungauged basins.

Event-based low flow statistics associated with varied return frequencies are used in many water quality and aquatic habitat management applications. These include calculations of 'environmental flows', or the flow regime required to sustain suitable habitat conditions for a given organism (O'Keeffe, 2009; Poff et al., 2010), as well as waste-load allocations, point source discharge permits, and withdrawal allowances related to water supply planning (Stedinger et al., 1993). One of the most commonly used metrics designed to express a minimum flow over a period is the 7Q10 statistic, which is the lowest streamflow for seven consecutive days that would be expected to occur once every 10 years (US EPA, 1997). This metric targets extreme low flow and is widely used for regulatory and modeling applications, especially with respect to point-source pollution and determination of Total Maximum Daily Load (TMDL) values for contaminants and nutrients (Ames, 2006). The 7Q10 statistic can only be calculated if there is sufficient length of flow record to calculate a reasonable 10-year recurrence probability. In addition, many researchers and managers seeking to establish environmental flows for aquatic biota may be concerned with flows of a more frequent return interval than decadal. Thus, other event-based statistics are used according to research needs and data availability, such as the 7Q2 (lowest seven-day flow with a two-year recurrence interval), average annual minimum daily flow, the minimum seven-day flow over a study period, etc. (Ouarda et al., 2008; Price et al., 2011; Stedinger et al., 1993).

These event-based low flow statistics, by definition, highlight extreme low flows. Many aspects of environmental flow management and water supply planning, however, benefit from information about sustained conditions as well. For these purposes, flow-duration statistics are used to identify exceedence probabilities of all flow observations in a given period of record (Stedinger et al., 1993). For emphasis on baseflow, flows that are exceeded a high proportion of the time are generally isolated. For example, managers might be interested in the 1, 5, or 25 percentile flow magnitude, which are exceeded 99, 95, and 75% of the time, respectively, during the entire period of analysis (Patel, 2007). These statistics are often referred to as Q_x, with Q representing discharge, and x representing the probability of exceedence (e.g. Q_{99}, Q_{95}, and Q_{75}).

A representation of sustained conditions, as opposed to extremes and events, is the baseflow index (BFI), which is the proportion of baseflow to total streamflow over a continuous period of record (Bloomfield et al., 2009). This metric is widely used in recent literature and has been indicated as an important variable for linking watershed characteristics to baseflow, addressing water quality concerns characterizing instream habitat availability, and drawing inferences about subsurface storage capacities (Lampadariou et al., 2008; Lee et al., 2006; Tesoriero et al., 2009). Determination of BFI requires separation of baseflow from stormflow, for which many methods have been used. Eckhardt (2008) provided a thorough review and analysis of seven baseflow separation methods. If data are available, concentrations of environmental isotopes such as oxygen-18 and deuterium can be used to separate event and pre-event water in streamflow (Buttle, 1994; Didszun and Uhlenbrook, 2008; Tetzlaff et al., 2007).

For most methods of baseflow separation, some analysis or index of a stream's recession characteristics is usually necessary, and recession analysis can offer fruitful insights in its own right (Wittenberg, 2003). A review of methods of baseflow recession analysis is presented by Tallaksen (1995). Since the publication of that review, additional computational resources for

recession analysis have become available, such as those described by Rutledge (1998), the spreadsheet method presented in Posavec et al. (2006), and the RECESS program created by the US Geological Survey (Rutledge, 2007). Gottschalk et al. (1997) described a method for combining recession analysis and low flow frequency analysis that has been successfully used for regionalization of low flow distribution functions. Ivanowski (2009) used the RECESS program to evaluate variability of recession characteristics of 20 watersheds in the Piedmont physiographic province of the southeastern USA, and found watershed relief to be a more important determinant of recession form than climatic factors. Wang and Cai (2010) demonstrated that recession characteristics can be used to evaluate the relative impacts of climate change and land-use change.

All four types of baseflow metrics are sometimes estimated using predictive statistical models for ungauged basins, based on regional empirical relationships between watershed characteristics and baseflow at gauged sites. There typically is a great deal of uncertainty associated with such approaches (Clausen, 1995), but they can be useful in the absence of observed data. An example of this approach is available from the US Geological Survey (Bingham, 1986), in which regionalized equations are presented for predicting low flows in Tennessee streams. These equations contain variables related to underlying geology and drainage area, and are associated with standard error ranging from 24 to 33%. Similar approaches have been used in other areas of North America and Europe (Bloomfield et al., 2009; Clausen, 1995; Gustard et al., 1989; Kent, 1999; Longobardi and Villani, 2008; Nathan et al., 1996; Neff et al., 2005; Thomas and Benson, 1970; Vogel and Kroll, 1992; Zhu and Day, 2005). These studies indicate that explanatory variables included in statistical models that best explain baseflow variability differ considerably among the various baseflow metrics. This implies that the specific watershed characteristics that influence extreme event low flows may be different than those that influence sustained baseflows and recession characteristics, and underscores the need for establishment of a consistent set of baseflow metrics to facilitate cross-study comparisons. Ouarda et al. (2008) presented a review of statistical approaches for predicting low flows based on watershed characteristics.

II Geomorphic controls on baseflow

I Geology

Catchment geology is a primary control on baseflow-generating processes (Bloomfield et al., 2009; Farvolden, 1963; Freeze, 1972; Neff et al., 2005; Smakhtin, 2001; Tague and Grant, 2004). In regions underlain by permeable, soluble, or highly fractured bedrock, groundwater storage volumes within the bedrock itself may be highly significant, and the connectivity to the surface water network may be extremely complex. In contrast, areas underlain by crystalline or massive bedrock with minor fracturing may not store significant quantities of water and thus contribute to relatively short water residence times (McGuire et al., 2005; Smith, 1981). In addition to bedrock type, geologic structure is also of great importance to baseflow hydrology in some regions (Delinom, 2009), and boundaries between geologic units have been shown to be important zones of groundwater-surface water interaction (Arnott et al., 2009; Konrad, 2006). Smith (1981) showed that low flows in shale and sandstones in Virginia were highly dependent on the degree of bedrock folding, with massively folded regions yielding higher low flows than non-folded zones. In some settings, bedrock fractures more readily transmit water to deep subsurface storage that is not connected to the surface stream network (hereafter 'disconnected storage'), than to more shallow storage that feeds baseflow (Seaton and Burbey, 2005). In some areas of extreme karst

development, a losing effect on baseflow has been observed, due to the often very high storage capacities in limestone and dolomite solution cavities (White, 1977). Baseflow losses have also been observed in areas of highly porous sandstone (Arnott et al., 2009). Catchment geology also indirectly affects basin hydrology in its influence on drainage network structure. Easily eroded bedrock lends itself more readily to channel formation and pedogenesis, both affecting storage capacities and rates of water transmission (Farvolden, 1963; Mwakalila et al., 2002). In some regions, weathered overburden (e.g. saprolite or other regolith), may serve as a more important baseflow-sustaining reservoir than the underlying solid bedrock (Smith, 1981; Witty et al., 2003). This can lead to complications with interpreting the influence of bedrock type on baseflows, because in many areas crystalline bedrock is associated with very low porosity and storage, but lends itself to the development of thick saprolite overburden that may store and transmit substantial quantities of water (Mwakalila et al., 2002). In addition to bedrock and saprolite, substantial quantities of baseflow may also originate from the near-surface valley bottom storage, such as bank soils, alluvial fills, and wetlands, where short-term storage levels are maintained to allow continuous lateral drainage into channels (Brutsaert, 2005; Smakhtin, 2001). This variably saturated throughflow zone, which may consist of a combination of regolith, alluvium, and/or soil, is often a more important source of baseflow than deeper groundwater (Ambroise et al., 1996; Mwakalila et al., 2002).

2 Surface topography

Meaningful assessment of basin topography is often missing from watershed analyses. Surface topography is a key control on baseflow (Vivoni et al., 2007), both directly and indirectly, and the influence of topography is most pronounced in relatively high relief settings (Tetzlaff et al., 2009). Exceptions exist in karst or highly porous settings, such as volcanic or glacial terrain, where water can move freely in the subsurface below surface drainage divides (Devito et al., 2005). Topographic gradients control the rate at which soil water moves downslope, thereby determining whether stormwater is flushed to the channel network or retained in the soil post-event (Figure 1). The effect of land-use and climate change on streamflow may be mitigated or amplified by basin surface and/or subsurface topography, and ideally these factors should be considered in assessment of stream response to human impact (Dubé et al., 1995; Iroumé et al., 2005). Little is known regarding which specific topographic variables are most useful for predicting baseflow and/or explaining baseflow variability response to land-use change, but many metrics have been demonstrated as beneficial components of hydrologic models.

Metrics of surface topography in hydrologic modeling are often reduced to single indices, with Beven and Kirkby's (1979) topographic index (TI) the most common. TI is computed as $\ln(\alpha/\tan\beta)$, where α = specific contributing area to a given site, and β = the local slope angle at that site. TI increases as contributing area increases and slope angle decreases. Increasing drainage area should increase groundwater contributions, and decreasing slope angle should reduce the rate of groundwater transmission, assuming that surface topography approximates the hydraulic gradient for shallow groundwater systems (Buttle et al., 2001). Troch et al. (1993) reported that the TOPMODEL approach using TI and soil transmissivity yielded accurate depths to shallow water tables. However, many studies that test predicted versus observed water table depths, streamflows, or other related factors using this approach have reported limited success (Burt and Butcher, 1985; Buttle et al., 2001; Jordan, 1994; Moore and Thompson, 1996; Rodhe et al., 1996). Furthermore, the index is so highly generalized that mean basin TI values may not vary greatly within a study region (McGuire et al., 2005; Price et al.,

2011), limiting its use in cross-site comparisons. The lack of total success of such an approach does not by any means negate the importance of surface topography in the storage and transmission of baseflow, although some of these authors arrive at that conclusion. The lack of success is at least partially due to the insufficiency of the index in characterizing elements of basin topography that directly relate to watershed storage and transmission rates. Though obviously simplistic, TI is readily computed from digital terrain data and incorporated into spatial models, and is thus widely used in popular applications, such as TOPMODEL (Beven and Kirkby, 1979).

Several studies have demonstrated that parameters expressing catchment geometry (e.g. hypsometric integral, metrics expressing degree of stream network development, and indices of flowpath length and gradient) are beneficial in prediction and analysis of baseflow and related factors (Farvolden, 1963; McGuire et al, 2005; Woods et al., 1997). Among many influences addressed, Farvolden (1963) found potential discharge (a flow component related to baseflow) to be most strongly correlated to basin geometry in a mountainous region of Nevada. Woods et al. (1997) devised a subsurface flow index based on surface topography, which the authors report to efficiently describe the time-varying spatial pattern in subsurface runoff generation, ideal for use in steep forested catchments in humid climates. Corroborating the idea that catchment-scale flow path distribution is largely a function of catchment geometry (Kirchner et al., 2001; Lindgren et al., 2004), McGuire et al. (2005) found strong correlations between catchment terrain indices representing flow path distance and gradient to the stream network in the Oregon Cascades. Santhi et al. (2008) found topographic relief to be a predictor of BFI on a regional scale. However, dimensionless topographic parameters were shown to have no relationship with BFI in southeastern Australia (Lacey and Grayson, 1998). Drainage density, or the length of stream network per unit watershed area, has been shown to have a negative relationship to baseflow in many settings (Farvolden, 1963; Gregory and Walling, 1968; Marani et al., 2001; Price et al., 2011; Tague et al., 2008; Warner et al., 2003). Higher drainage density is synonymous with greater contact area between subsurface storage and stream channels. This greater contact area may facilitate removal of water and reduce baseflows during drier times of year. Additionally, drainage density may be related to subsurface storage characteristics, with higher drainage density possibly negatively correlated with storage capacity.

In addition to its influence on subsurface flowpath distribution and transit times, surface topography also relates to the distribution of shallow storage. Surface topographic characteristics may express the amount of alluvial bottomland and floodplain storage (Brown et al., 2005), and the presence and extent of colluvium available for subsurface water storage. Alluvial aquifers are understood to be a key source of streamflow in many settings (Larkin and Sharp, 1992). In theory, the presence and extent of alluvial valleys is closely linked with baseflow quantity, though few studies have directly addressed this relationship (Brown et al., 2005; Soulsby et al., 2006). Schilling (2009) showed that groundwater recharge quantity was highly dependent on topographic position, with the greatest quantities of recharge observed in alluvial zones. Using geochemical and isotopic tracers, Tetzlaff and Soulsby (2008) demonstrated that the upper 54% of a large river catchment in Scotland supplied 71% of the river's baseflow, and that the groundwater of the lower slopes of montane headwaters (where colluvium deposits occur) provide a major source of baseflow to the river system. Colluvium has also been shown to be an important shallow reservoir in the Cascades (Galster and Leprade, 1991; Schulz et al., 2008), and was an important variable for explaining variability of baseflow

magnitudes in the southern Blue Ridge Mountains of the USA (Price et al., 2011). This review has emphasized GIS-based evaluations of the influence of surface topographic characteristics on baseflows. It is important to note that ongoing research indicates that variation in digital elevation model (DEM) resolution can have a pronounced effect on rainfall-runoff analyses, and more research needs to be conducted to link DEM-based topographic characteristics with baseflow at multiple resolutions (Dixon and Earls, 2009; Lee et al., 2009).

3 Subsurface topography and soil characteristics

Subsurface topography, in addition to surface relief, exerts strong influence on water storage and throughflow pathways, and thus influences baseflow. Throughflow processes require a confining layer through which water cannot easily infiltrate, thereby initiating lateral subsurface flow (Hutchinson and Moore, 2000). It is these confining layers that prevent continued infiltration of water, thereby allowing shallow storage contributions to baseflow. In hydrologic modeling, topographic indices to estimate soil moisture properties and rates of throughflow are generally limited to metrics of surface topography, despite the known influence of confining layers on flowpaths and soil moisture characteristics (e.g. Chaplot and Walter, 2003; Chaplot et al., 2004; Gburek and Folmar, 1999; Hutchinson and Moore, 2000; McDonnell et al., 1996). During or immediately following storm or snowmelt events, when water table elevations are relatively high, the soil moisture surface is more likely to parallel the surface topography than that of the confining layer (Hutchinson and Moore, 2000). However, the influence of subsurface topography is of particular importance during relatively low moisture conditions, when the topography of the confining layer may be the predominant control on moisture retention, and, thus, an important factor for baseflow. However, no known studies have specifically addressed the influence of subsurface topographic characteristics on stream baseflows.

Subsurface strata that induce throughflow are widely varied, but are most often associated with pedogenically unaltered parent material. Bedrock with negligible fracturing and porosity (Hatcher, 1988), impermeable saprolite (Chaplot et al., 2004), heavily compacted till (Hutchinson and Moore, 2000; Reuter and Bell, 2003), and hydraulically restrictive loess layers (O'Geen et al., 2003) have all been demonstrated to influence soil and hillslope hydrology. Additionally, pedogenic features such as claypans (Wilkison and Blevins, 1999) and well-developed argillic horizons (Perillo et al., 1999) have been shown to limit vertical infiltration, although the effect is rarely widespread enough to significantly impact meso- or macroscale hydrology. Pedogenic features generally fail to function as true confining layers, primarily due to macropore and preferential flow path development across the hydraulically restrictive horizon (Bryan and Jones, 1997). Tree root growth, animal burrowing, and other bioturbation processes affect soil horizons to a much greater extent than seen with parent material confining layers such as bedrock, saprolite, or compacted till. Wilkison and Blevins (1999) used chemical tracers to demonstrate vertical preferential flow paths through a claypan to outweigh lateral throughflow above the claypan. Similarly, Perillo et al. (1999) identified vertical preferential flow pathways created by decayed roots through a well-developed argillic horizon that partially induced lateral flow. Thus, it seems that extreme circumstances are required for pedogenic features to serve as broadly influential confining layers. These circumstances seem particularly unlikely to be met in vegetated environments, where biological activity is abundant and disruptive to hydraulically resistant horizons. Thus, it is generally assumed that lithologic contacts underlying soil, such as the

soil/bedrock or saprolite/bedrock interface (Hatcher, 1988; McDonnell et al., 1996), are more important in governing subsurface flow and contributions to baseflow than pedogenic features in the soil itself.

4 Combined influences of topography and soils

Soil properties influence the distribution of water storage, but correlations between soil properties and topography typically hinder isolation of the influence of soil characteristics on water storage and baseflow. Primarily, variation in soil texture plays a significant role in the rate of moisture loss due to surface or subsurface topographic gradients (Dodd and Lauenroth, 1997; Yeakley et al., 1998). Spatial variability of soil moisture is most pronounced during unsaturated conditions between storm events (Hutchinson and Moore, 2000; Kim et al., 2005; Sidle et al., 2000; van Ommen et al., 1989), and such variability is partially attributable to soil texture. However, determining the strength of this influence is complicated by the correlations between topography and soil texture. Systematic downslope variation in soil texture commonly occurs, as the result of decreasing slope and corresponding slowed rates of water movement from ridge to toeslope positions (Schaetzl and Anderson, 2005). Thus, correlations between soil texture and hillslope position are likely to exist, with finer particle size, thicker soils, and low slope gradients combining their influences to encourage soil moisture retention. Conversely, steep upper slopes are likely characterized by coarser, less developed, and thinner soils, thereby more rapidly transmitting water. Furthermore, soil hydrology is strongly affected by spatial variability of soil moisture, which may be predominantly controlled by surface and/or subsurface topography (Woods et al., 1997). From this perspective, isolating the influence of soil characteristics from topography is problematic.

III Effects of human land use on baseflow

Widespread vegetation change and soil disturbance accompany most forms of land-use change, and such impacts are often sufficient to alter the timing and quantity of baseflow (Figure 1). Additionally, human impact may involve direct water removal (abstractions) or inputs to streams or catchments. Table 1 summarizes baseflow response to several common forms of human impact. Extreme impact (e.g. urbanization) may be associated with a total rearrangement of surface and subsurface pathways, in addition to changes in soil properties, vegetation, etc. This section on anthropogenic controls on baseflow addresses patterns observed with forest removal, urbanization, and agriculture.

1 Forest removal

Globally inclusive literature investigating the role of basin forest cover on flow in small headwater catchments (i.e. < 2 km^2) indicates an increase in mean annual flow in response to removal of basin vegetation (examples of reviews: Bosch and Hewlett, 1982; Brown et al., 2005; Hibbert, 1967; Johnson, 1998; Jones and Post, 2004; Sahin and Hall, 1996; Swank et al., 1988), with many studies specifically indicating increases in baseflow (Harr et al., 1982; Hicks et al., 1991; Keppeler and Ziemer, 1990; Smith, 1991). This relationship is due to greater interception and evapotranspiration rates associated with forest cover (Bosch and Hewlett, 1982; Calder, 1990; McCulloch and Robinson, 1993). The negative relationship between watershed forest cover and baseflow volume for headwater streams results from experimentation methods where the surface infiltration characteristics are not drastically altered, thus isolating evapotranspiration changes as the key influence on recharge and baseflow (Figure 1) (Brown et al., 2005; Bruijnzeel, 2004). In some cases, these results have been interpreted as a

Table 1. Summary of studies assessing the response of baseflow and recharge to various human impacts

Impact	Baseflow response	Attributed effect	Reference(s)
Groundwater abstraction	Decrease	Lowers water tables	Owen (1991)
Wetland drainage	Decrease	accelerated removal of water from valley bottoms	Riggs (1976)
Valley bottom vegetation change	Increase or decrease	ET change, dependent on specific impact	Keppeler and Ziemer (1990); Swank et al. (1988)
Catchment afforestation	Decrease	Increased ET	Gustard and Wesselink (1993); Trimble et al. (1987)
Catchment forest harvest	Increase	Decreased ET	Harr et al. (1982); Hicks et al. (1991); Swank et al. (1988)
Catchment forest conversion	Increase or decrease	Decreased ET, decreased infiltration	Costa et al. (2003); Wilk et al. (2001)
River abstraction	Decrease	Direct removal of water from channel	Kottegoda and Natale (1994)
Effluent discharge to rivers	Increase	Direct input of water to channel	Pirt and Simpson (1983)
Irrigation return flow	Increase	Direct input of water to channel	Blodgett et al. (1992); Dow (2007)
Importation of water	Increase	Surface and subsurface water inputs	Davies et al. (1993)
Flow regulation	Increase or decrease	Channel impoundment with regulated release	Gustard et al. (1989)

potentially dangerous suggestion that watershed management approaches could include deforestation to increase water yield for public use (Brooks et al., 1991; Chang, 2003). However, because forest cover is associated with high infiltration and recharge of basin subsurface storage (Figure 1), more permanent canopy decreases associated with pasture, agriculture, or suburban land uses my decrease baseflows due to soil compaction, reduction of soil organic matter, and increase in impervious surface (Gregory et al., 2006; Ohnuki et al., 2008; Price et al., 2010; Woltemade, 2010; Zimmermann et al., 2006). Studies investigating permanent land-use change have shown decreased baseflow from conversion of forest to non-forest land use (e.g. Bruijnzeel, 2004; Line and White, 2007), or baseflow increases associated with afforestation (Ma et al., 2009). Studies relating baseflow of 30+ streams in the Piedmont and Blue Ridge provinces of the southern Appalachian Highlands indicate a significant positive relationship between basin forest cover and baseflow discharge (Price and Jackson, 2007; Price et al., 2011).

2 Urbanization

Urbanization involves a wide range of impacts, and specific stream response depends on many factors (Doyle et al., 2000). Anthropogenic impacts on watershed hydrology accompanying urbanization involve widespread and drastic reorganization of surface and subsurface pathways, and frequently are complicated by importation of water from other watersheds or previously disconnected storage (Figure 1). Following urbanization, water is more quickly flushed through catchments due to reduced hydraulic resistance of land surfaces and

Table 2. Recharge response to various aspects of urbanization (modified from Meyer, 2002)

Increased recharge	Decreased recharge
Surface distribution of imported water (irrigation and other outdoor water use)	Impervious surface coverage and soil compaction
Infrastructure leakage of imported water	Rapid transmission of event water through storm sewers and modified channels
Stormwater detention	Leakage of shallow groundwater into storm sewers
Leakage of event water into shallow groundwater via storm sewers	Shallow groundwater withdrawal
	Removal of wastewater outside of catchment

channels, as a result of impervious surface coverage, compacted soils, channelization, and subsurface storm drainage networks. Intuitively, it follows that accelerating water removal from stream systems would be linked with corresponding decreases in recharge and baseflow in urban systems. This assumption dominated hydrologic understanding of urban impacts for decades, largely due to the influence of Leopold's (1968) widely cited urban hydrology guidebook (Brandes et al., 2005). In this benchmark publication, management implications center on baseflow reduction associated with urbanization, based more on theory than observed trends. While the assumption that increased impervious surface decreases infiltration, recharge, and ultimately baseflow is theoretically solid, Leopold's conceptual model has proven to be overly simplistic and is not well supported by published data (Ferguson and Suckling, 1990). While event flows do consistently increase and result in faster recession to baseflow with increased impervious surface (Brandes et al., 2005; Burns et al., 2005; Ferguson and Suckling, 1990; Konrad, 2003), the corollary of baseflow decline does not behave quite as neatly, as a result of additional urban effects on subsurface recharge. The complete picture of hydrologic response to urbanization is extremely complex, with some factors acting to reduce recharge and others to increase recharge (Table 2).

Assumptions that urbanization decreases baseflow are generally based on reduced recharge due to increased impervious surface, which is indeed a dominant factor in urban hydrology. Impervious surface coverage in urban basins drastically exceeds that of basins with other land-use types. Road networks, parking lots, rooftops, etc., all contribute to increased impervious percentages, with individual cities demonstrating different degrees of greenspace to offset the impacts of impervious surface (Carter and Jackson, 2007). Impervious coverage undoubtedly has an enormous effect on urban hydrology, with stream corridor impervious cover having a particularly detrimental effect on baseflow quantity and quality (Landers et al., 2007). However, it is unrealistic to view urban systems in a surface-based framework as is commonly applied to systems experiencing lower-intensity impacts. In more moderately impacted settings, surface hydrology remains dominated by natural processes (e.g. evapotranspiration, soil hydrology) following landscape change. In most urban settings, however, water is completely redistributed to accommodate human activities and prevent flood damage. Water is routed across the surface and through the subsurface via ditching, storm drains, water mains, wastewater sewers, and other means, altering the rates and paths of water transmission through urban basins. Such reworking of the hydrologic system precludes explanation of baseflow response to urban land use solely in terms of the effects of vegetation removal and increased impervious surface (Lerner, 2002;

Meyer, 2005), although such simplification is still commonplace.

A major additional complication occurs in urban systems: virtually all major cities import water (Lerner, 2002). The importation of water may include pumping from deep groundwater that is otherwise disconnected from the surface water system, piping of water from other watersheds, and/or withdrawal of water from downstream reservoirs. This water is redistributed throughout cities via pipe networks that often lose substantial quantities of water (Lerner, 2002; Roy et al., 2009). Lerner (1986) reports water main leakage rates of 20–25% to be common, with rates reaching as high as 50%. Wastewater sewer systems may also leak substantial amounts of water, which often originates outside the drainage basin. Such leakage, along with surface inputs of imported water (e.g. septic drainage, lawn/garden watering, and other forms of outdoor domestic water usage) may enter subsurface storage and can significantly offset or overshadow storage losses due to other urbanization effects. Sustained baseflow with urbanization has also been attributed to ET reduction associated with vegetation removal (e.g. Appleyard et al., 1999; Rose and Peters, 2001). However, the role of ET in urban systems remains largely unresolved. For example, Oke (1979) showed that ET rates remain steady despite decreased vegetation cover in Vancouver, BC, due to heat advection from non-vegetated surfaces. While such processes may be significant in suburban areas or cities with abundant vegetation, they cannot be assumed to dominate in all urban areas.

All of the factors addressed above may be expressed to varying degrees in different cities or regions, resulting in inconsistent hydrologic response to urbanization throughout the world (Table 3). It seems that there is no predictable response of annual low flow, proportion of baseflow to total streamflow, or groundwater recharge to urbanization, as demonstrated by the case studies outlined below. Of the studies reviewed that directly address annual low flow response to urbanization, none demonstrated a pronounced decrease in discharge (e.g. Harris and Rantz, 1964; Konrad and Booth, 2002; Rose and Peters, 2001). Harris and Rantz (1964) attribute increased annual low flow to distribution and leakage of imported water, an insight issued decades before most hydrologists accepted such a source to be significant. Rose and Peters (2001) attribute the lack of annual low flow response in Atlanta, Georgia, to an offsetting of the effects of impervious surface by reduced ET associated with vegetation removal. Finally, Konrad and Booth (2002) interpret inconsistent annual low flow response in the Puget Sound basin to varying degrees of development, implying that in some cases a development threshold necessary to induce response had not yet been reached.

The response of baseflow proportion shows a weak tendency toward decline among the case studies reviewed. Streams in Pennsylvania, New York, Georgia, and Oregon all demonstrated baseflow reduction associated with urbanization (Chang, 2007; Leopold, 1968; Rose and Peters, 2001; Simmons and Reynolds, 1982). In all cases, the authors attribute observed declines to recharge loss associated with impervious surface coverage, and Simmons and Reynolds (1982) additionally cite the removal of wastewater from stream basins. In contrast, streams in Harlow, Great Britain, and southern New York demonstrated baseflow increases with urbanization, presumably due to distribution and leakage of imported water (Burns et al., 2005; Hollis, 1977). The wide variety of factors controlling baseflow discharge and system response to urbanization likely explains the disagreement among these studies. A lack of consistent results or no response was observed in the majority of the reviewed studies addressing baseflow (Beran and Gustard, 1977; Brandes et al., 2005; Ferguson and Suckling, 1990; Konrad and Booth, 2005). Explanations for the lack of clear trends include effects from pronounced seasonality in

Table 3. Summary of studies investigating baseflow and recharge response to urbanization

Location	Response to urbanization	Attributed mechanism(s)	Reference
Atlanta, Georgia	Decrease	Reduced infiltration	Rose and Peters (2001)
Coatesville, Pennsylvania	Decrease	Reduced infiltration	Leopold (1968)
Long Island, New York	Decrease	Reduced infiltration + export of sewerage water	Simmons and Reynolds (1982)
Portland, Oregon	Decrease	Reduced infiltration	Chang (2007)
Long Island, New York	Decrease	Export of sewerage water	Koszalska (1975)
Western Washington	Inconsistent	Insufficient impact in some of the study basins	Konrad and Booth (2002)
Western Washington	Inconsistent	Seasonality effects	Konrad and Booth (2005)
Delaware River Basin	Inconsistent	Varied influences among basins	Brandes et al. (2005)
Long Island, New York	Inconsistent	Seasonality effects	Ku et al. (1992)
Santa Clara County, California	Increase	Distribution and leakage of imported water	Harris and Rantz (1964)
Southern New York state	Increase	Septic effluent	Burns et al. (2005)
Harlow, Great Britain	Increase		Hollis (1977)
Caracas, Venezuela	Increase	Infrastructure leakage	Seiler and Alvarado-Rivas (1999)
Northeastern Illinois	Increase	Distribution and leakage of imported water	Meyer (2005)
Perth, Australia	Increase	Reduced ET + distribution and leakage of imported water	Appleyard et al. (1999)
Wolverhampton, U.K.	Increase	Distribution and leakage of imported water	Hooker et al. (1999)
Atlanta, Georgia	No response	Reduced infiltration offset by Reduced summer ET	Rose and Peters (2001)
Great Britain	No response		Beran and Gustard (1977)
Atlanta, Georgia	No response	Reduced infliltration offset by distribution and leakage of imported water	Ferguson and Suckling (1990)
Southern New York state	No response	Insufficient impact (suburban)	Burns et al. (2005)

the Pacific Northwest (Konrad and Booth, 2005), marked variability of background conditions and specific impacts in the Mid-Atlantic region (Brandes et al., 2005), and the offsetting of rapid transmission of stormwater by distribution and leakage of imported water (Ferguson and Suckling, 1990).

Additional case studies were reviewed that address recharge to subsurface storage, as this is inextricably linked with baseflow. Results from these studies generally indicate a more consistent response to urbanization than seen with annual low flow or baseflow proportion. Four of the studies reviewed, conducted in Caracas (Venezuela), Perth (Australia), Wolverhampton (UK), and northeastern Illinois demonstrate increased recharge with urbanization (Appleyard et al., 1999; Hooker et al., 1999; Meyer, 2005; Seiler and Alvarado-Rivas, 1999). In all of these cases, recharge increases are attributed to distribution of imported water and/or infrastructure leakage, with Appleyard et al. (1999) additionally citing reduced ET as a factor. Decreases in recharge were observed in Long Island, New York (Koszalska, 1975), Atlanta, Georgia (Rose and Peters, 2001), and the Kleine Nete basin in Belgium (Dams et al., 2008), attributed to export of wastewater in New York and reduced infiltration in the latter two studies. Two studies in southern New York failed to demonstrate a clear direction of response to urbanization (Burns et al., 2005; Ku et al., 1992). It is noteworthy that a larger percentage of recharge studies demonstrated increase than was seen in the baseflow studies. The fact that increases in recharge were slightly more common than increases in baseflow may indicate that urban manipulation detectibly complicates the pathways between subsurface recharge and channel flow. However, the only study that explicitly addressed both baseflow and recharge demonstrated the same direction of response in both components (Rose and Peters, 2001), which suggests that the discrepancies seen among recharge and baseflow studies may simply be further evidence of lack of consistent response to urbanization in different settings.

Interpretation of baseflow response to urbanization is further complicated by several considerations. Comparison of urban response across cities and regions is problematic, based on differences in natural hydrologic background variability, unique infrastructure systems, and varied management approaches. Research design and choice of parameters assessed is not universally consistent, clouding cross-study comparison. Investigators often seek clear trends in response to urbanization, and in the process may overlook complex patterns associated with geographic variability in physical setting, a point reinforced by more comprehensive analyses (e.g. Ferguson and Suckling, 1990; Konrad and Booth, 2005; Rose and Peters, 2001). Relatively intense, long-term urbanization has been the focus of most urban hydrology research, and far less is known about the impacts of lower-density or carefully mediated urban development. Land-use activities associated with moderate impact or episodic disturbance may not result in detectible stream response, given other background sources of hydrologic variability (Konrad and Booth, 2002). The conceptual model outlined by Leopold (1968) does not include consideration of these and other factors, and it unfortunately appears that baseflow response to urbanization cannot be predicted by a highly simplified set of parameters.

3 Agriculture

As seen with urbanization, baseflow response to agricultural land use may be positive or negative, depending on management practices. First, there is the obvious confounding factor of irrigation (Dow, 2007; He et al., 2009). If crops are irrigated from surface water resources linked to the stream network, increased ET may reduce baseflows (Figure 1). However, increases in baseflow may occur if irrigation water is drawn

from disconnected storage resources or from outside the drainage basin. Furthermore, varied management practices are associated with a wide range of soil impacts (e.g. conventional tillage practices versus no-till and conservation tillage), differing temporal patterns to intensive cropping (e.g. perennial versus seasonal cultivation), and whether or not crop residue or other soil cover are used during the fallow season (Kent, 1999). Drainage tiling, which speeds removal of moisture from the near-surface soil layers, may also have strong impacts on baseflow in agricultural areas (Schilling and Helmers, 2008).

Accordingly, studies investigating baseflow response to agricultural land use have demonstrated mixed results. Schilling and Libra (2003) showed that many Iowa rivers have seen increases in annual baseflow magnitude and proportion, and additional work has shown that these increases were significantly related to increasing row crop intensity (Schilling, 2005). Increases in baseflow over the past 60 years within the upper Mississippi River basin have been attributed to reductions in ET associated with conversion from perennial to seasonal cultivation (Lins and Slack, 2005; Zhang and Schilling, 2006), and changes in tillage practices (Kent, 1999; Potter, 1991). Using rainfall simulation experiments, Rasiah and Kay (1995) showed that minimized tillage practices were associated with lower overland flow and increased infiltration compared with conventional tillage of corn crops in Canada. Charlier et al. (2008) showed that greater overland flow in agricultural areas of Guadeloupe reduced recharge and decreased baseflows. Decreased agricultural land use in Georgia and Wisconsin has been linked with increased baseflows attributed to higher infiltration rates (Juckem et al., 2008; Knox, 2001), while large-scale conversion of forest to agricultural land in Thailand demonstrated no significant changes in baseflow (Wilk et al., 2001). Despite the inconsistency in results from these studies, two main inferences can be drawn from the literature addressing baseflow response to agricultural influence: (1) watersheds that have been under agricultural land use for extended periods show baseflow increases in response to improved cropping and tillage practices; (2) comparison of baseflows under agricultural land use versus other land uses is precluded by the variety of management practices, variable uses and sources of irrigation, and other background sources of variability.

IV Effects of climate change on baseflow

For most of the planet, temperatures are projected to rise as a result of continually increasing atmospheric greenhouse gas concentrations (IPCC, 2007). It is unlikely that temperature increases will occur in isolation, and there is limited predictability of atmospheric feedbacks that will accompany warming due to increased greenhouse gas concentrations. At local scales, higher summer temperatures and, by extension, evaporation rates, could lead to increased convective precipitation, offsetting baseflow reductions. At regional scales, changes in global circulation patterns and higher evaporation over large water bodies will likely translate to changes in precipitation regimes in many regions of the world, but the major global circulation models (GCMs) do not agree on what these changes will be. The likely climate changes that will affect the majority of the globe will involve some combination of temperature increase and either precipitation decrease or increase, and any specific baseflow response to climate change will depend on the magnitude and direction of changes in both precipitation and temperature (Choi et al., 2009; Smakhtin, 2001; Tague et al., 2008). Another important complication to understanding the effects of climate change on baseflow is that empirical studies evaluating baseflow response to changing climate typically are confounded by concurrent land-use change during the period of record

(Choi, 2008; Juckem et al., 2008). As a result, hydrologic simulations with projections of climate change are required to evaluate baseflow response to climate change in true isolation of land-use change, and both the hydrologic and climate models are associated with substantial uncertainty. Furthermore, climate change and hydrologic response likely will exhibit considerable regional variability, such that it is impossible to make any single prediction about how, for example, continued greenhouse gas-related warming generally will affect baseflows (Lins and Slack, 2005).

Despite these obstacles, many researchers have designed studies offering insights into the issue of climate change impacts on baseflows. One recurrent prediction is that continued warming and subsequent changes in global circulation are likely to lead to more extreme hydrologic regimes in many regions, with wetter wet seasons and drier dry seasons (Nyenje and Batelaan, 2009). This, in turn, will lead to reductions in seasonal low flows, and a more pronounced impact on low flows than high flows (Choi, 2008; Smakhtin, 2001; Yang et al., 2009; Zhang et al., 2008). Multiple empirical and simulation studies suggest that this increased flow seasonality, along with warmer temperatures in summer, will lead to severe reductions in late summer baseflows (Cooper et al., 1995; Kim and Kaluarachchi, 2009; Reihan et al., 2007; Wegehenkel and Kersebaum, 2009; Xie et al., 2010; Yusoff et al., 2002). It should, however, be noted that regional analyses have shown streamflow increases across the USA from 1944 to 1999, attributed to greater warm season precipitation (Lins and Slack, 2005). It has been suggested that colder regions will experience more extreme baseflow response as a result of climate warming (Ma et al., 2009). Several empirical studies in colder regions that have recorded warming have shown that earlier snowmelt has led to reduced late-summer low flows (Barnett et al., 2008; Huntington et al., 2009; Luce and Holden, 2009; Pike et al., 2008; Poff, 1996; Schneider, 2008). In very high-latitude or high-altitude regions that are presently underlain by permafrost, baseflows may increase with warming, as a result of permafrost thaw and increased infiltration and recharge (Brabets and Walvoord, 2009).

Perhaps the greatest obstacle to predicting water quality and quantity response to climate change is the confounding factor of concurrent land-use change (Choi, 2008; Ma et al., 2009; Poff, 1996). A recent study by Wang and Cai (2010) evaluated climate versus human influences on baseflow recession in the Nebraska Sand Hills and found land-use change to be a more significant influence on recession than climate change throughout the second half of the 20th century. Juckem et al. (2008) offered the useful interpretation of their empirical analysis of baseflow changes in the Kickapoo River watershed, Wisconsin, that climate change predominantly affects baseflow timing (due to earlier snowmelt, etc.), while land-use change superimposes changes in magnitude upon these climatic effects. Additionally, climate change may be associated with changes in precipitation intensity, the hydrologic effect of which could be exacerbated by land-use change in the form of soil compaction and greater impervious surface coverage. Easterling et al. (2000) showed that most precipitation increases in global climate change are the result of increases in extreme, highly intense rainfall events. Even in the absence of concurrent land-use change, more frequent high-intensity events may lead to greater overland flow and reduced recharge, and these effects will be exacerbated if combined with anthropogenic decreases in watershed infiltration capacity.

Several studies attempting to evaluate hydrologic response to land-use change in the context of long-term climate fluctuations have shown that land-use change leads to much more drastic hydrologic response than is evident throughout prehistoric Holocene warming and cooling cycles (Knox, 2001; Leigh, 2008; Smakhtin,

2001). The results of these studies support Tomer and Schilling's (2009) observation that the impacts of anthropogenic climate change are subtle compared with persistent cycles of drought and precipitation surplus, as well as Smakhtin's (2001) recommendation that predictions of baseflow response to climate change be accompanied by as much paleoenvironmental context as possible. However, it is not clear that land-use change impacts exceed climate-change impacts in all settings, especially where land-use intensity is not extreme. It is possible that 21st-century climate change will exceed the ranges observed to date during the Holocene, in which case climate change could exert equal or greater baseflow response relative to land-use change. This is particularly the case where climate fluctuations lead to major changes in the hydrologic regime, e.g. from snow- to rain-dominated systems (Barnett et al., 2008; Schneider, 2008). There is also evidence that baseflow response will vary with hydrogeologic and geomorphic setting (Tague et al., 2008; Wang et al., 2009). Watersheds with high drainage efficiency (as a result of highly permeable bedrock or high drainage density) may show exacerbated reductions in baseflow associated with higher atmospheric temperature and ET (Tague et al., 2008; van Wateren-de Hoog, 1998). Conversely, watersheds in settings that favor higher storage and baseflow proportion, and/or those underlain by large, productive aquifers will likely demonstrate mediated response (Schneider, 2008; Wang et al., 2009).

V Summary and conclusions

Understanding how land-use and climate change will affect baseflow quantity, in the context of watershed geomorphology, will aid watershed managers and stream ecologists in the protection of adequate water supply for human needs and habitat availability for stream biota. In addition to introducing challenges in meeting agricultural, municipal, and industrial water needs, reduced baseflows contribute to impairments known to affect fish, invertebrates, and algal assemblages (James et al., 2009; Kennan and Ayers, 2002; Roy et al., 2009; Wenger et al., 2009). Even in regions characterized by relatively low-intensity land-use change, there have been detectible reductions in baseflow quantity and quality, as well as impairments to aquatic species assemblages (Price and Leigh, 2006b; Roy et al., 2003; Sutherland et al., 2002; Walters et al., 2003).

This review of the literature has shown that watershed topography and geomorphology influence baseflow by affecting the storage properties and rates of water transmission within a catchment. The influence of factors of slope, relief, and drainage density are particularly noteworthy. However, it remains unclear whether these factors are themselves strong drivers of baseflow (Price et al., 2011), or whether they instead correlate to other aquifer properties that more directly control baseflow. More research is needed to understand the role of subsurface topography on baseflow, and very little is known about water storage in varied geomorphic units (e.g. colluvial deposits and alluvial bottomlands) and their linkages to baseflow.

Research investigating anthropogenic controls on baseflow has tended to disproportionately emphasize forestry experimentation and urbanization, and within these studies the natural background controls on baseflow are often downplayed or ignored. Several recent studies emphasize the importance of considering changes in soil hydrology when assessing streamflow response to land-use change (Bruijnzeel, 2004; Price et al., 2010; Woltemade, 2010). Very little is known about baseflow response to land-use change in larger, more complex systems, or in settings affected by development of moderate intensity, information which is essential for effective water resources protection and management. It is increasingly clear that the results of forestry experimentation studies demonstrating baseflow increase with forest removal should not be

extrapolated to more complex systems with long-term land-use change and extensive soil disturbance.

It is difficult to draw overarching conclusions regarding the influence of watershed characteristics on baseflow from the existing body of literature, given the enormous diversity of natural background conditions, watershed parameters, and baseflow metrics among case studies. This highlights a clear need for more studies investigating the relative influences of watershed geomorphology and land use within a given natural template, and for efforts to be made toward developing consistent methodologies for watershed characterization and baseflow quantification. Few predictions can be made from the current knowledge base of how greenhouse gas-induced warming will affect baseflows, because our current modeling capabilities cannot resolve significant uncertainty in state variable projections (e.g. climate and land cover), as well as the unknown dynamics concerning the interaction of climate and land-cover change. It can be inferred from empirical and simulation-based studies that earlier spring snowmelt in high-latitude and high-altitude regions will threaten summer and fall low flows (Barnett et al., 2008).

From this review, seven key needs for future research have emerged that could broadly benefit the water resources community, and without which our understanding of watershed function will remain limited:

(1) Experimental studies specifically designed to evaluate the influence of subsurface topography on baseflow.
(2) Improvement of methods to determine distribution of shallow subsurface storage at scales relevant to policy and management.
(3) Comprehensive empirical comparisons that link soil hydrology and baseflows under land-use gradients that incorporate more detail than the broad categories of forest, agriculture, and urban land use.
(4) Modeling and empirical studies that address multiple aspects of watershed hydrology in a single study, such as a comparative watershed study in which ET, soil moisture, subsurface storage recharge, and streamflow are all evaluated. There is a clear need for enhanced understanding of watershed function, and addressing the complete system should be a high priority.
(5) Modeling and empirical studies that explore baseflow response to varied land-use change, planned growth, and mitigation strategies.
(6) Under a given experimental design, do research conclusions differ with the specific baseflow metric analyzed? Are there optimal baseflow separation methods, recession statistics, and low flow statistics?
(7) Ensemble modeling studies that explore multiple working hypotheses of atmospheric feedbacks that will accompany warming, and various interactions between land-use and climate change, in order to ensure mitigation plans are in place for any scenario that is likely to occur.

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Contaminants of Emerging Concern including Pharmaceuticals and Personal Care Products

Contaminants of emerging concern (CECs), including pharmaceuticals and personal care products (PPCPs), are increasingly being detected at low levels in surface water, and there is concern that these compounds may have an impact on aquatic life. It is important for EPA to be able to evaluate the potential impact of CECs and PPCPs on aquatic life and have an approach for determining protective levels for aquatic organisms.

These chemicals have features that require additional consideration when applying existing ambient water quality criteria for the protection of aquatic life, using EPA's 1985 *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Life and Their Uses*.

- Guidelines for Deriving Numerical National Water Quality Criteria

There are many CECs and PPCPs that act as so-called endocrine disruptors (EDCs). EDCs are compounds that alter the normal functions of hormones resulting in a variety of health effects. EDCs can alter hormone levels leading to reproductive effects in aquatic organisms, and evaluating these effects may require testing methodologies not typically available along with endpoints not previously evaluated using current guidelines.

The emerging contaminants may also demonstrate low acute toxicity but cause significant reproductive effects at very low levels of exposure. In addition, the effects of exposure to aquatic organisms during the early stages of life may not be observed until adulthood. Therefore, traditional toxicity test endpoints may not be sufficiently comprehensive for criteria derivation for these chemicals and the chemicals may also have specific modes of action that may affect only certain types of aquatic animals (e.g., vertebrates such as fish).

Therefore, EPA developed a *White Paper Aquatic Life Criteria for Contaminants of Emerging Concern: Part I Challenges and Recommendations* detailing the technical issues and recommendations to serve as a basis for modifying the 1985 guidelines. These modifications should enable the Agency to better address CECs and develop ambient water quality criteria when appropriate for protection of aquatic life that makes the best use of available science.

EPA's Office of Water asked the Science Advisory Board (SAB) for advice on the scientific merit of a white paper that identifies and addresses technical issues in deriving aquatic life criteria for emerging contaminants such as pharmaceuticals and personal care products exhibiting endocrine disrupting activity or other toxic mechanisms.

- Science Advisory Board Review of the White Paper

You may need a PDF reader to view some of the files on this page. See EPA's About PDF page to learn more.

- White Paper Aquatic Life Criteria for Contaminants of Emerging Concern Part I General Challenges and Recommendations (PDF) (86 pp, 534 K, June 3, 2008)
- Water Quality Criteria Determination Memo (PDF) (6 pp, 53 K, June 2008)
- SAB Advisory on Aquatic Life WQC for Contaminants of Emerging Concern (PDF) (63 pp, 368 K, December 2008)
- EPA Response to SAB Comments on White Paper Titled Aquatic Life Criteria for Contaminants of Emerging Concern (PDF) (2 pp, 1 MB, May 2009)

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Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin

Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin

Prepared by:

U.S. Environmental Protection Agency
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Washington, DC

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NOTICES

This document has been drafted and approved for publication by the Health and Ecological Criteria Division, Office of Science and Technology, United States Environmental Protection Agency, and is approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

FOREWORD

Section 304(a) of the Clean Water Act (CWA) requires the Administrator of the U.S. Environmental Protection Agency (EPA) to publish water quality criteria that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that might be expected from the presence of pollutants in any body of water, including ground water.

The EPA is publishing these recommended values under CWA 304(a) for states to consider as the basis for swimming advisories for notification purposes in recreational waters to protect the public. The EPA envisions that if states decide to use the values as swimming advisory values they might do so in a manner similar to their current recreational water advisory programs. Alternatively, states may consider using these same values when adopting new or revised water quality standards (WQS). If adopted by states as WQS and approved by the EPA under CWA 303(c), the WQS could be used for all CWA purposes. States may also wish to consider using these values as both swimming advisory values and WQS.

This document has undergone an EPA intra-agency peer-review process. The Health and Ecological Criteria Division, Office of Science and Technology, Office of Water, U.S. Environmental Protection Agency has completed the final review and the document is approved for publication. The values were derived using the existing peer-reviewed and published science on the adverse human health effects of the toxins including previous EPA analysis, such as the EPA's *Health Effects Support Document for the Cyanobacterial Toxin Microcystins* and *Health Effects Support Document for the Cyanobacterial Toxin Cylindrospermopsin* (HESDs), and the EPA's *Drinking Water Health Advisory for the Cyanobacterial Microcystin Toxins* and *Drinking Water Health Advisory for the Cyanobacterial Toxin Cylindrospermopsin* (Drinking Water Health Advisories) (U.S. EPA 2015a, 2015b, 2015c, 2015d). The EPA used established criteria methodologies (U.S. EPA 2000) and recreation-specific exposure parameters from the EPA's *Exposure Factors Handbook* (EFH) (U.S. EPA 2011) to derive these values. Detailed information that can be found in the EPA's HESDs and Drinking Water Health Advisories is summarized in this document.

The term "water quality criteria" is used in two sections of the CWA section 304(a)(1) and section 303(c)(2). The term has a different legal meaning in each section. In section 304, the term represents a non-regulatory, scientific assessment of effects on human health or aquatic life. The criteria recommendations presented in this document are such a scientific assessment. If the state or authorized tribe adopts water quality criteria associated with specific designated uses as WQS under section 303, and approved by the EPA, they become applicable CWA WQS in ambient waters within that state or tribe. Water quality criteria adopted in state or tribal WQS could have the same numerical values as criteria developed by the EPA under section 304. Alternatively, states and authorized tribes may derive numeric criteria based on other scientifically defensible methods, but the criteria must be protective of designated uses. States and tribes can adopt criteria into their standards. When approved by the EPA, the criteria become Clean Water Act-applicable WQS. Guidelines to assist in modifying the criteria recommendations presented in this document are contained in the Water Quality Standards Handbook (U.S. EPA 2012).

This document provides recommendations only. It does not establish or affect legal rights or obligations. It does not establish a binding norm and cannot be finally determinative of the issues addressed. Agency decisions in any particular situation will be made by applying the CWA and EPA regulations on the basis of specific facts presented and scientific information then available.

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The EPA gratefully acknowledges the valuable contributions of the EPA internal technical reviewers who reviewed this document. Staff from the following EPA program and regional offices completed a formal review of these *Human Health Recreational Ambient Water Quality Criteria (AWQC) or Swimming Advisories for Microcystins and Cylindrospermopsin*.

U.S. EPA Office of Children's Health Protection

U.S. EPA Office of General Counsel

U.S. EPA Office of Policy

U.S. EPA Office of Research and Development

U.S. EPA Office of Water

Office of Ground Water and Drinking Water

Office of Science and Technology

Office of Wastewater Management

Office of Wetlands, Oceans, and Watersheds

U.S. EPA Regional Offices

- Region 1
- Region 4
- Region 5
- Region 7
- Region 8

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ACRONYMS AND ABBREVIATIONS

ACC	ambient cyanotoxin concentration
AWQC	ambient water quality criteria
AWWARF	American Water Works Association Research Foundation
BAX	BCL2 Associated X, Apoptosis Regulator
Bcl-2	BCL2 Apoptosis Regulator
BGAS	blue-green algae supplements
BID	BH3 interacting domain death agonist
BW	body weight
C.	*Cylindrospermopsis*
CalEPA	California Environmental Protection Agency
CAS	Chemical Abstracts Service
CAWS	Chicago Area Waterway System
CCD	cyanobacterial cell density
CDC	U.S. Centers for Disease Control and Prevention
CDEEP	Connecticut Energy and Environmental Protection
CDPH	Connecticut Department of Public Health
CFU	colony forming unit
CI	confidence interval
cm	centimeter
CTA	cell toxin amount
CWA	Clean Water Act
CyAN	Cyanobacteria Assessment Network
CYP450	Cytochrome P450
DFB	DeFlorio-Barker et al. (2017)
DIN	dissolved inorganic nitrogen
DIP	dissolved inorganic phosphorus
DON	dissolved organic nitrogen
dw	dry weight
E.	*Escherichia*
EFH	*Exposure Factors Handbook*
ELISA	Enzyme Linked Immunosorbent Assay
EPA	U.S. Environmental Protection Agency
FAQs	frequently asked questions
Fe	iron
fg	femtogram
g	grams
GI	gastrointestinal
GI2	more severe gastrointestinal symptom index
GM	geometric mean
GSD	geometric standard deviation

HAB	harmful algal bloom
HABISS	Harmful Algal Bloom-related Illness Surveillance System
HESD	Health Effects Support Document
HPLC	high performance liquid chromatography
IDEQ	Idaho Department of Environmental Quality
IQR	interquartile range
IR	ingestion rate
kg	kilograms
km	kilometer
K_{oc}	soil organic carbon-water partition coefficient
K_{ow}	octanol-water partition coefficient
L	liter
LA	leucine, alanine
LC/MS/MS	liquid chromatography with tandem mass spectrometry
LF	leucine, phenylalanine
LC_{50}	lethal concentration causing the death of 50 percent of a group of test animals
LD_{50}	lethal dose causing the death of 50 percent of a group of test animals
LOAEL	lowest-observed-adverse-effect-level
LOD	level of detection
LPS	lipopolysaccharide
LR	leucine, arginine
LW	leucine, tryptophan
LY	leucine, tyrosine
M.	*Microcystis*
m^3	cubic meter
mcy	three-letter nomenclature for genes that produce microcystins
mg	milligram
mL	milliliter
MW	molecular weight
MS	mass spectroscopy
n	sample size
N	nitrogen
N/A	not available
NASA	National Aeronautics and Space Administration
ng	nanogram
NHMRC	National Health and Medical Research Council
NLA	National Lakes Assessment
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect-level
NORS	National Outbreak Reporting System
NYSDOH	New York State Department of Health

OATP	organic anion transporting polypeptide
OHHABS	One Health Harmful Algal Bloom System
OPP	EPA Office of Pesticide Programs
OR	odds ratio
ORSANCO	Ohio River Valley Water Sanitation Commission
P	phosphorus
PCR	polymerase chain reaction
pg	picogram
pH	potential of hydrogen
ppb	parts per billion
PWS	public drinking water system
qPCR	quantitative polymerase chain reaction
rDNA	ribosomal deoxyribonucleic acid
RfD	reference dose
ROS	reactive oxygen species
RR	relative risk or when microcystin-RR it means arginine, arginine
RSC	relative source contribution
SDWA	Safe Drinking Water Act
SWIMODEL	Swimmers Exposure Assessment Model
t	event duration
TBD	to be determined
TDI	tolerable daily intake
TN:TP	total nitrogen ratio to total phosphorus
TOXLINE	Toxicology Literature Online
U.S.	United States of America
UF	uncertainty factor
URL	Uniform Resource Locator
µg	microgram
μm^3	cubic micrometer
USGS	U.S. Geological Survey
WHO	World Health Organization
WHOI	Woods Hole Oceanographic Institute
WoS	Web of Science
WQS	water quality standards
WSDE	Washington State Department of Ecology
YR	tyrosine, arginine

1.0 EXECUTIVE SUMMARY

Cyanobacteria, also commonly referred to as blue-green algae, are photosynthetic bacteria that are ubiquitous in nature and are found in surface waters. Environmental conditions that promote excessive growth of cyanobacteria in surface waters can lead to situations in which cyanobacterial cell density is high, known as blooms. Nitrogen and phosphorus levels, the ratio of nitrogen to phosphorus, water temperature, organic matter availability, light attenuation, pH, and water column stratification are environmental factors that play an important role in the development of cyanobacterial blooms and their production of cyanotoxins. Some cyanobacteria, but not all, have the ability to produce toxins. The toxin-producing cyanobacteria contain genes that confer the ability to produce toxins and are referred to as toxigenic cells. The abundance of toxigenic cyanobacteria can vary within the overall cyanobacteria population, between waterbody to waterbody, and over time within a single waterbody.

Microcystins can be produced by a variety of toxigenic cyanobacteria genera, including *Microcystis*, *Anabaena*, *Dolichospermum, Nostoc*, *Oscillatoria*, *Fischerella*, *Planktothrix*, and *Gloeotrichia*. Some of these species can be distributed through the water column, concentrate in the upper layers, or form surface scums depending on environmental conditions. More than 100 microcystin congeners exist, which vary based on amino acid composition. The majority of toxicological data on the effects of microcystins are available for microcystin featuring leucine and arginine (microcystin-LR), which is also a frequently monitored congener. Microcystins are water soluble and tend to remain contained within the toxigenic cyanobacterial cell until the cell breaks and they are released into the water. Microcystins typically have a half-life of four to 14 days in surface waters or may persist longer, depending on factors such as photodegradation, bacteria, and the presence of organic matter. Microcystins can persist even after a toxigenic cyanobacterial bloom is no longer visible.

Cylindrospermopsin can be produced by a variety of toxigenic cyanobacteria species, including *Cylindrospermopsis raciborskii*, *Aphanizomenon*, *Anabaena*, *Lyngbya wollei*, and *Raphidiopsis*. Some of these species tend not to form visible surface scums, and the highest concentrations of total cyanobacterial cells typically occur below the water surface. Two congeners of cylindrospermopsin, as well as two structural analogs, have been identified. Cylindrospermopsin can be retained within the cell or released into the water. The biodegradation of cylindrospermopsin in natural water bodies is a complex process that can be influenced by many environmental factors, including toxin concentration, water temperature, sunlight, and the presence of cell pigments and bacteria. Half-lives of 11 to 15 days and up to eight weeks have been reported for cylindrospermopsin in surface waters.

This document for microcystins and cylindrospermopsin focuses on the human health risks associated with incidental ingestion while recreating in freshwaters containing these harmful cyanotoxins. The recommended cyanotoxin values apply to freshwaters with the recreational designated use. The toxins that are produced by cyanobacteria growing in freshwaters can enter estuarine and marine waters as waters containing the toxins flow downstream. The EPA recognizes that there may be circumstances where harmful cyanobacterial blooms (also known as harmful algal blooms or HABs) can impact downstream marine and estuarine waters. This document provides information on occurrence and incidental ingestion in estuarine and marine waters for states to consider but does not provide recommendations for those waters. Exposure to cyanobacteria and their toxins can also occur through non-recreational pathways such as consumption of cyanotoxin-contaminated drinking water and food (including fish), and during bathing or showering. This document does not address or provide recommendations for non-recreational exposures.

The EPA is publishing these recommended values for microcystins[1] and cylindrospermopsin under the Clean Water Act (CWA) section 304(a) for states to consider as the basis for swimming advisories for notification purposes to protect public health in recreational waters. The EPA envisions that if states decide to use the values as swimming advisory values, they would do so in a manner similar to their current recreational water advisory programs. Alternatively, states may consider using these same values when adopting new or revised water quality standards (WQS). If adopted as WQS and approved by the EPA under the CWA section 303(c), the WQS could be used for all CWA purposes. States may also wish to consider using these values as both swimming advisory values and WQS.

The recommended values in this document leverage the information that the EPA collected and evaluated in its *Health Effects Support Document for the Cyanobacterial Toxin Microcystins* and *Health Effects Support Document for the Cyanobacterial Toxin Cylindrospermopsin* (HESDs), and the EPA's *Drinking Water Health Advisory for the Cyanobacterial Microcystin Toxins* and *Drinking Water Health Advisory for the Cyanobacterial Toxin Cylindrospermopsin* (Drinking Water Health Advisories) (U.S. EPA 2015a, 2015b, 2015c, 2015d).

The EPA evaluated the health effects of microcystins and derived a reference dose (RfD) in its 2015 *Health Effects Support Document for the Cyanobacterial Toxin Microcystins* (U.S. EPA 2015d). Exposure to elevated levels of microcystins can potentially lead to liver damage. The critical study for the derivation of the microcystins RfD was conducted by Heinze (1999) based on rat exposure to microcystin-LR in drinking water. The critical effect from this study was slight to moderate liver lesions with necrosis and increased liver weight and enzymes associated with tissue damage. The EPA established the RfD based on microcystin-LR and used it as a surrogate for other microcystin congeners. Monitoring and toxicity studies suggest that the microcystin-LR is the most frequently occurring congener and is more toxic than other congeners of microcystin evaluated (Loftin et al. 2016b; U.S. EPA 2015d; Ito et al. 2002; Rinehart et al. 1994; Vesterkvist and Meriluoto 2003; WHO 1999). The EPA used the RfD to derive its previously published Drinking Water Health Advisories for microcystins (U.S. EPA 2015a) and the recommended values in this document. The dose and critical effects that the EPA used from Heinze (1999) to establish the RfD are supported by a Guzman and Solter (1999) study, also conducted in rats.

The EPA evaluated the health effects of cylindrospermopsin and derived an RfD in its 2015 *Health Effects Support Document for the Cyanobacterial Toxin Cylindrospermopsin* (U.S. EPA 2015c). The kidneys and liver appear to be the primary target organs for cylindrospermopsin toxicity. The critical study that the EPA used to derive the cylindrospermopsin RfD was conducted by Humpage and Falconer (2002, 2003) based on drinking water exposure to mice. Adverse effects on the kidneys were manifested by decreases in urinary protein concentration and increased relative kidney weight. Upon considering all effects observed by Humpage and Falconer (2002, 2003), increased relative kidney weight was considered the most appropriate basis for quantitation (U.S. EPA 2015c). The EPA used the RfD to derive its previously published Drinking Water Health Advisories for cylindrospermopsin (U.S. EPA 2015b).

Based on available noncancer health effects information, the EPA is recommending values protective of primary contact recreation as follows:

[1] Microcystins comprise a class of over 100 congeners and unless specified otherwise, "microcystins" refers to total microcystins.

- For microcystins, the recommended recreational value is 8 micrograms (μg)/liter (L).
- For cylindrospermopsin, the recommended recreational value is 15 μg/L.

These values are based on the exposure experienced by recreating children due to their higher exposures compared with other age groups. Given that toxigenic cyanobacterial blooms typically are seasonal events, recreational exposures are likely to be episodic, and may be short term in nature. The EPA recommends that if used as a swimming advisory to protect swimmers at a beach, these values not be exceeded on any single day. If used as a water quality criterion for assessment and listing purposes, the EPA recommends a maximum of three excursions across a recreational season and observation of that pattern across multiple years to reflect seasonal dynamics and occurrence patterns of HABs.

At this time, available data are insufficient to develop quantitative recreational values for total cyanobacterial cell density related to inflammatory health endpoints. The reported epidemiological relationships between cell density exposure and specific health outcomes (e.g., dermal symptoms, eye/ear irritation, fever, gastrointestinal (GI) illness, and respiratory symptoms) are not consistent. The uncertainties related to the epidemiological study differences, such as study size, species and strains of cyanobacteria present, and the total cyanobacterial cell densities associated with significant health effects, do not provide sufficient information to determine a consistent association between total cyanobacterial densities associated with adverse inflammatory health effects. The EPA recognizes that some states have included total cyanobacterial cell density values as an important part of their HAB management strategy. Available information on health endpoints, cell density, and guidelines developed by other authorities on total cyanobacteria cells is described in the Effects Characterization section of the document (section 7.5) and in Appendix D.

Because the EPA's recommendations in this document are cyanotoxin concentrations, it can be helpful for risk-management purposes to understand how this relates to toxigenic cyanobacteria in the waterbody, as the abundance of toxigenic cells in a water body affects the amount of cyanotoxin produced. The number of toxigenic cyanobacteria relative to the number of total cyanobacteria can vary in time and space. Quantifying the abundance of toxigenic cyanobacteria is a better predictor of potential toxin production compared to quantifying total cyanobacteria. The EPA presents a toxigenic cell number based on the number of toxigenic cells that could produce microcystins equivalent to the recommended magnitude. The Effects Characterization section also describes gene-based detection methods (i.e., quantitative polymerase chain reaction (qPCR)) that can target and quantify the toxigenic subpopulation of cyanobacteria that are present in a waterbody.

2.0 INTRODUCTION AND BACKGROUND

Section 304(a) of the CWA requires the Administrator of the EPA to publish water quality criteria that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that might be expected from the presence of pollutants in any body of water.

Currently there are no U.S. federal water quality criteria or regulations for cyanobacteria or cyanotoxins in drinking water under the Safe Drinking Water Act (SDWA) or in ambient waters under the CWA. No cyanotoxins are included on EPA's priority pollutant list.[2] In 2015, the EPA published non-regulatory Drinking Water Health Advisories (U.S. EPA 2015a, 2015b) to provide information for public health officials or other interested groups on two cyanotoxins (microcystins and cylindrospermopsin) that can affect drinking water quality but are not regulated under SDWA.

The EPA is publishing these recommended values for microcystins and cylindrospermopsin under the CWA section 304(a) for states to consider as the basis for swimming advisories for notification purposes to protect public health in recreational waters. The EPA envisions that if states decide to use the values as swimming advisory values, they would do so in a manner similar to their current recreational water advisory programs. Alternatively, states may consider using these same values when adopting new or revised WQS. If adopted as WQS and approved by the EPA under the CWA section 303(c), the WQS could be used for all CWA purposes. States may also wish to consider using these values as both swimming advisory values and WQS.

The EPA-recommended values in this document leverage the information that the EPA collected and evaluated in its *Health Effects Support Document for the Cyanobacterial Toxin Microcystins* and *Health Effects Support Document for the Cyanobacterial Toxin Cylindrospermopsin* (HESDs), and its *Drinking Water Health Advisory for the Cyanobacterial Microcystin Toxins* and *Drinking Water Health Advisory for the Cyanobacterial Toxin Cylindrospermopsin* (Drinking Water Health Advisories) (U.S. EPA 2015a, 2015b, 2015c, 2015d).

This document for microcystins and cylindrospermopsin focuses on the human health risks associated with incidental ingestion while recreating in freshwaters containing these harmful cyanotoxins. The recommended cyanotoxin values apply to freshwaters with the recreational designated use. The toxins that are produced by cyanobacteria growing in freshwaters can enter estuarine and marine waters as waters containing the toxins flow downstream. The EPA recognizes that there may be circumstances where harmful cyanobacterial blooms (also known as HABs) can impact downstream marine and estuarine waters. This document provides information on occurrence and incidental ingestion in estuarine and marine waters for states to consider but does not provide recommendations for those waters. Exposure to cyanobacteria and their toxins can also occur through non-recreational pathways such as consumption of cyanotoxin-contaminated drinking water and food (including fish), and during bathing or showering. This document does not address or provide recommendations for non-recreational exposures.

International and State Guidelines

The World Health Organization (WHO 2003a) published a series of guideline values for recreational exposure to cyanobacteria associated with incremental severity and probability of health effects at increasing densities of total cyanobacteria and corresponding concentrations of chlorophyll *a* (if

[2] https://www.epa.gov/sites/production/files/2015-09/documents/priority-pollutant-list-epa.pdf

cyanobacteria dominate) (Table 2-1). The WHO also considered the potential for liver damage by microcystins in deriving the recommended total cyanobacterial cell densities. Potential concentrations of microcystins that could be associated with each guidance level are discussed in the WHO document. However, it should be noted that actual microcystin concentrations at each WHO action level could vary depending on the composition of toxigenic strains in the cyanobacterial community present and the dominant species of microcystin producer present in a bloom. For example, at a total cyanobacterial cell density of 100,000 cells/milliliter (mL), an estimated microcystin concentration of 20 µg/L could occur assuming all cells present are toxin-producing *Microcystis* species and the average cellular toxin content was 0.2 picogram (pg) microcystin per cell (WHO 2003a). Microcystin concentrations could range from 50 to 100 µg/L, or higher, if another toxin-producing species, such as *Planktothrix*, is present at the same cell density.

Table 2-1. WHO (2003a) Recreational Guidance/Action Levels for Cyanobacteria, Chlorophyll *a*, and Estimated Corresponding Microcystin Level

Relative Probability of Acute Health Effects	Cyanobacteria (cells/mL)	Chlorophyll *a* (µg/L)	Estimated Corresponding Microcystin Levels (µg/L)
Low	≤ 20,000	≤ 10	< 10[a]
Moderate	> 20,000–100,000	> 10–50	2–4 to 20[a,b]
High	> 100,000	> 50	> 20

[a] WHO estimated that 2 to 4 µg microcystins/L may be expected, with 10 µg/L possible, at a cell density of 20,000 cells/mL if microcystin-producing cyanobacteria are dominant.

[b] At 100,000 cyanobacterial cells/mL, a concentration of 20 µg microcystins/L is likely if the bloom consists of *Microcystis* and has an average toxin content of 0.2 pg/cell.

For these guidelines, the WHO recommended values that included the potential health effects from exposure to total cyanobacteria because it was "unclear whether all important cyanotoxins had been identified and that the health outcomes observed after recreational exposure could be related to cyanobacterial substances other than the well-known cyanotoxins" (WHO 2003a). The different guideline levels were an effort to distinguish between irritative or inflammatory-response symptoms associated with total cyanobacterial cells and the more severe hazard of exposure to elevated concentrations of cyanotoxins, particularly microcystins. The cell-associated inflammatory responses are represented by the low probability of adverse health effects category of ≤ 20,000 cells/mL, corresponding to ≤ 10 µg/L chlorophyll *a* if cyanobacteria dominate. According to the WHO, as the density of cyanobacteria increase above that level, the probability of inflammatory responses increases, and the potential for more severe adverse health effects associated with exposure to the cyanotoxins also increases. The WHO high-risk category includes both > 100,000 cells/mL, corresponding to 50 µg/L of chlorophyll *a*, if cyanobacteria dominate, and > 20 µg/L microcystin levels. Health effects at this level are expected to be primarily due to the toxic effects of microcystins. Very high densities of cells occurring in scums—for example, > 10 million cells/mL or > 5,000 chlorophyll *a*—can be associated with very high concentrations of toxin, for example 2,000 µg/L of microcystins in the top 4 cm of a water body (WHO 2003a). Scums that accumulate along the shoreline due to wind can be associated with a thousand-fold higher density of cells (WHO 2003a).

The WHO guideline value development was informed by results from a review conducted by Chorus and Bartram (1999). A primary study identified in this review was a prospective epidemiology study by Pilotto et al. (1997), which evaluated health effects after recreational exposure to total cyanobacteria and reported associations between total cyanobacterial cell densities and health. Pilotto et al. (1997) found a

significant association among recreators exposed to greater than 5,000 cells/mL. The WHO chose a guideline level of 20,000 cells/mL to represent the upper bound of the "low probability of adverse health effects" category (WHO 2003a). While the association among recreators exposed to greater than 5,000 cells/mL for more than one hour and one or more symptoms reported in Pilotto et al. (1997) was statistically significant, the WHO states that they represented less than 30 percent of the individuals exposed (Chorus and Bartram 1999). Therefore, the level of health effect and the small number of people affected at 5,000 cells/mL were not considered by the WHO to be a basis to justify action (WHO 2003a).

The WHO pointed out that the potential concentration of microcystins could vary depending on the composition of toxigenic strains within the overall cyanobacterial community present and the dominant species of microcystin producer present in a bloom. The WHO states that, at the same cyanobacterial cell density, cyanotoxin levels could approximately double if *Planktothrix agardhii* were the dominant member of the community.

Many countries have adopted the multiple parameters that the WHO discusses for recreational waters including cell density, biovolume, and cyanotoxin concentration (see Table 2-2). Some international authorities have multiple action levels. For brevity, Table 2-2 presents the guideline reflecting the lowest concentration of microcystins or density of cyanobacterial cells or narrative guidelines that recommended or triggered a health protective action for countries that have adopted action levels. For a more complete list of guideline or action levels and recommended actions for international jurisdictions, see Appendix A. The EPA did not identify any recreational guideline levels for cylindrospermopsin established by other international regulatory authorities.

Table 2-2. International Recreational Water Guideline or Action Levels for Cyanobacteria and Microcystins

Jurisdiction	Lowest Recreational Water Guideline/Action Level[a]	Reference
Australia[b]	microcystins (total): $\geq$ 10 µg/L or *Microcystis aeruginosa* (total): $\geq$ 500 to < 5,000 cells/mL or cyanobacteria (total): $\geq$ 0.4 to < 4 mm^3/L (where a known toxin producer is dominant in the total biovolume) or total biovolume of all cyanobacterial material $\geq$ 10 mm^3/L (where known toxins are not present)	Australian Government National Health and Medical Research Council (2008)
Canada	microcystins (total): ≥ 20 µg/L (expressed as microcystin-LR) or cyanobacteria (total): ≥ 100,000 cells/mL	Health Canada (2012)
Cuba	cyanobacteria: > 1 of the species known as potentially toxic or phytoplankton cells: > 20,000 – to < 100,000 cells/mL, > 50 percent of cells cyanobacteria	German Federal Environment Agency (2012)[c]
Czech Republic	cells: > 20,000 cells/mL	German Federal Environment Agency (2012)[c]
Denmark	chlorophyll *a*: > 50 µg/L, dominated by cyanobacteria or visible surface scum	German Federal Environment Agency (2012)[c]
European Union	appropriate monitoring must be implemented if there is a risk of proliferation of algae. Member state authorities responsible must take management measures and provide information immediately if a proliferation of cyanobacteria (or blue algae) occurs.	European Parliament and the Council of the European Union (2006)
Finland	algae (includes cyanobacteria): detected	German Federal Environment Agency (2012)[c]

Jurisdiction	Lowest Recreational Water Guideline/Action Level[a]	Reference
France[b]	microcystins: > 25 µg/L or cyanobacteria: > 20,000 to < 100,000 cells/mL (± 20 percent)	German Federal Environment Agency (2012)[c]
Germany	Secchi Disk reading > 1 m and (microcystins): ≥ 10 µg/L or chlorophyll *a* (with dominance by cyanobacteria): ≥ 40 µg/L or biovolume: ≥ 1 mm^3/L)	German Federal Environment Agency (2012)[c]
Hungary	microcystins: ≥ 4 to < 10 µg/L or cell count: ≥ 20,000 to < 50,000 cells/mL or chlorophyll *a* (with dominance by cyanobacteria): ≥ 10 to < 25 µg/L	German Federal Environment Agency (2012)[c]
Italy[b]	microcystin-LR: > 20 µg/L equivalents or cyanobacterial cell count for cyanotoxin-producing species other than those that produce microcystins (e.g., cylindrospermopsin) > 100,000 cells/mL (± 20 percent) or transparency ≤ 1 m and total phosphorus > 20 µg/L and total cyanobacterial cell count > 2,000 to ≤ 20,000 cells/mL (± 20 percent) or transparency ≥ 1 m and total phosphorus > 20 µg/L and total cyanobacterial cell count ≤ 2,000 cells/mL	Funari et al. (2017)
Netherlands	chlorophyll *a*: ≥ 12.5 to ≤ 75 µg/L or biovolume (cyanobacterial cell count): ≥ 2.5 to ≤ 15 mm^3/L	German Federal Environment Agency (2012)[c]
New Zealand[b]	microcystins (total): ≥ 12 µg/L or cyanobacteria (benthic): 20−50 percent coverage of potentially toxigenic cyanobacteria attached to substrate or cyanobacteria (total): > 0.5 to < 1.8 mm^3/L (biovolume equivalent of potentially toxic cyanobacteria) or cyanobacteria (total): > 0.5 to < 10 mm^3/L (biovolume equivalent of the combined total of all cyanobacteria)	Wood et al. (2008)
Poland	visible blooms	German Federal Environment Agency (2012)[c]
Scotland[b]	chlorophyll *a*: ≥ 10 µg/L with dominance of cyanobacteria or cyanobacteria: ≥ 20,000 cells/mL	Scottish Government Health and Social Care Directorates Blue-Green Algae Working Group (2012)
Spain	cyanobacteria proliferation potential (low)	German Federal Environment Agency (2012)[c]
Turkey	microcystin-LR: > 25 µg/L equivalents or cells: ≥ 20,000 to 100,000 cells/mL	German Federal Environment Agency (2012)[c]
World Health Organization (WHO)	cyanobacteria: 20,000 cells/mL or chlorophyll *a*: 10 µg/L (approximately 2-4 µg microcystins/L, assuming cyanobacteria dominance)	Chorus and Bartram (1999); WHO (2003a)

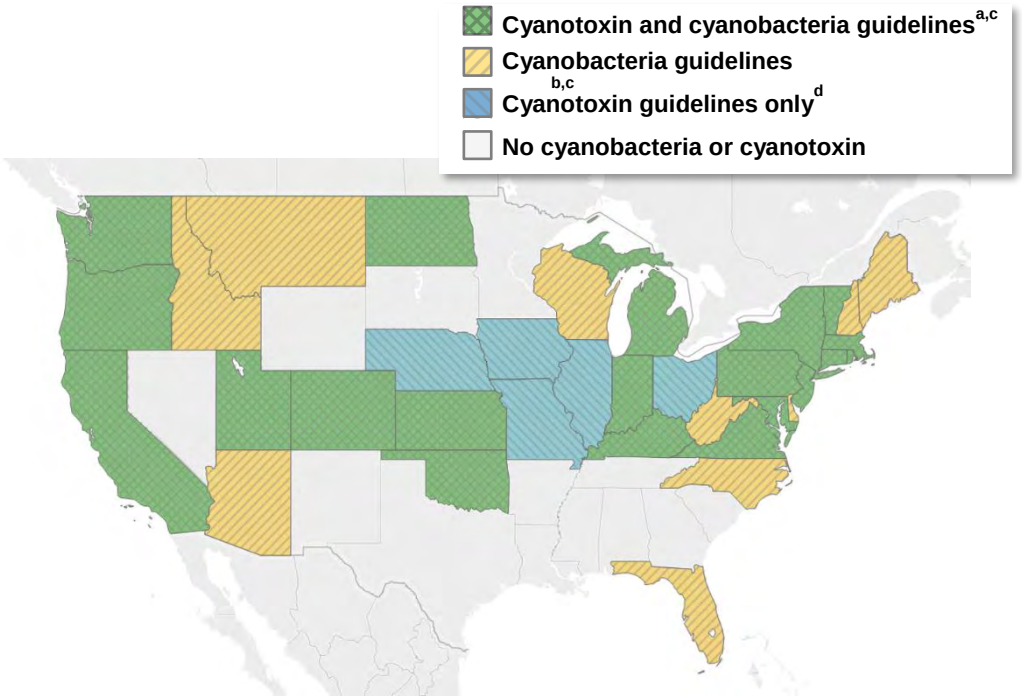
[a] More details are provided in Appendix A.

[b] The lowest guideline values for each quantitative parameter (i.e., cyanotoxin concentration, cyanobacterial cell density, biovolume) are not associated with the same action level. For example, for Australia, the lowest cyanobacterial cell density and biovolume criteria trigger the green level surveillance mode, and the lowest cyanotoxin concentration triggered the red level action mode.

[c] Following the VIIIth International Conference on Toxic Cyanobacteria, the German Federal Environmental Agency compiled and published in 2012 regulatory approaches to the assessment and management of cyanotoxin risks based on contributions by member countries.

As of March 2018, approximately 35 U.S. states have implemented cyanobacterial HAB guidelines for recreational waterways. As graphically shown in Figure 2-1, five states have quantitative or qualitative cyanotoxin guidelines only, and 20 states have quantitative guidelines for cyanotoxins, as well as either quantitative or qualitative guidelines for total cyanobacterial cell density. Qualitative guidelines for cell density use visual inspection rather than quantitative detection methods. In addition, 10 states had quantitative guidelines for cyanobacterial cell density only or had qualitative guidelines for cyanobacteria only. Seven states have guideline levels that address toxin-producing cyanobacteria as a proportion of the total cyanobacterial population or include a toxin-specific cyanobacteria cell density (California, Idaho, Maryland, New York, New Hampshire, Oregon, and Virginia). The Karuk Tribe, located in California, developed cell-based values for posting cyanotoxin public health warnings for the tribe's recreational waters (Kann 2014). Its values were based on the site-specific relationships between the cell densities of *Microcystis* and the level of microcystins observed in Karuk waters. For example, in the Klamath River, at 20,000 cells *Microcystis*/mL, the probability of exceeding 4 µg/L microcystins was 55 percent, while at 5,000 cells/mL there were no exceedances. Because the probability of exceeding the microcystins benchmark rapidly increased at cell densities above 5,000 *Microcystis*/mL, the Karuk Tribe uses that value to inform decision-making for health warnings (Kann 2014).

Figure 2-1. State Guidelines for Cyanotoxins and Cyanobacteria in Recreational Water by Type and Scope of Guidelines



[a] Includes states with quantitative cyanotoxin guidelines as well as either quantitative or qualitative cyanobacteria guidelines.

[b] Includes states that either have quantitative cyanobacteria guidelines only or qualitative guidelines only.

[c] The EPA found that Texas and North Carolina published guidelines in the past, but the guidelines were no longer on their websites.

[d] Missouri has presence/absence testing for cyanotoxins and quantitative thresholds.

For brevity, Table 2-3 lists the lowest recreational water guideline or narrative guidelines or action levels for microcystins, cylindrospermopsin, or total cyanobacteria that trigger or recommend a health protective action for U.S. states. For a more complete list of state guideline or action levels see

Appendix B. Parameters and values used as the basis for guidelines varied across states, as did the methodologies for developing the values.

Table 2-3. State Guideline or Action Levels for Microcystins, Cylindrospermopsin, and Cyanobacterial Cells in Recreational Water

State	Lowest Recreational Water Guideline or Action Level[a]	Reference
Arizona	Blue-green algae (mean value based on a minimum of two sample events within one peak season): 20,000 cells/mL and Chlorophyll *a* result (mean value based on a minimum of two sample events within one peak season) in target range	Arizona Department of Environmental Quality (2008)
California	Microcystins: 0.8 µg/L	Butler et al. (2012); Cyanobacteria Harmful Algal Bloom Network (2016a, 2016b)
	Cylindrospermopsin: 1 µg/L	
	Toxin-producing cyanobacteria: 4,000 cells/mL	
	Site-specific indicators of cyanobacteria (e.g., blooms, scums, mats)	
Colorado	Microcystin-LR: ≥ 10 µg/L and < 20 µg/L	Colorado Department of Public Health and Environment (2016)
	Cylindrospermopsin: ≥ 7 µg/L	
	Potentially toxic algae are visible	
Connecticut[b]	Combination of visual inspection, cell counts: Visual rank category 2: Blue-green algae cells > 20,000 cells/mL and < 100,000 cells/mL	Connecticut Department of Public Health (CDPH) and Connecticut Energy and Environmental Protection (CDEEP) (CDPH and CDEEP 2017; CDEEP 2017)
Delaware	Thick green, white, or red scum on surface of pond	Delaware Department of Natural Resources and Environmental Control: Division of Water (2016)
Florida	Cyanobacteria bloom	Florida Department of Environmental Protection (2019)
Idaho	*Microcystis* or *Planktothrix*: > 40,000 cells/mL	IDEQ (2015)
	Sum of all potentially toxigenic taxa: ≥ 100,000 cells/mL	
Illinois	Microcystin-LR: > 10 µg/L	Illinois Environmental Protection Agency (2018); Illinois Environmental Protection Agency (2013)
Indiana	Blue-green algae: 100,000 cells/mL	Indiana Department of Environmental Management (2018)
	Microcystin-LR: 4 µg/L	
	Cylindrospermopsin: 8 µg/L	
Iowa	Microcystin: ≥ 20 µg/L	Iowa Environmental Council (2018)
Kansas	Cyanobacteria: ≥ 80,000 and < 250,000 cells/mL	Kansas Department of Health and Environment (2015a); Kansas Department of Health and Environment (2015b)
	Microcystin: ≥ 4 and < 20 µg/L	
Kentucky	Blue-green algae: > 100,000 cells/mL	Kentucky Department for Environmental Protection (2014)

State	Lowest Recreational Water Guideline or Action Level[a]	Reference
	Microcystins: > 20 µg/L	Commonwealth of Kentucky Department for Environmental Protection Division of Water (2015)
Maine	Secchi disk reading < 2 meters caused by algae	Maine Department of Environmental Protection (2013)
Maryland	*Microcystis aeruginosa* or other potential microcystin-producing blue-green algae > 40,000 cells/mL, and samples contain microcystins: > 10 ppb	Wazniak personal communication (2016); Maryland Department of Natural Resources (2014)
Massachusetts	Blue-green algae: > 50,000 cells/mL	Massachusetts Bureau of Environmental Health (2015); Massachusetts Department of Public Health (2008)
	Microcystins: ≥ 14 µg/L	
Michigan	Microcystin: ≥ 20 µg/L	Michigan Department of Environmental Quality (2018); Kohlhepp G (2015)
	Chlorophyll *a*: > 30 µg/L and visible surface accumulations/scum are present, or cells are visible throughout the water column	
Missouri	Microcystins: presence (test strip range 0 to 10 ng/mL)	Missouri Department of Natural Resources (2017)
	Cylindrospermopsin: presence (test strip range 0 to 10 ng/mL)	
Montana	Reservoirs that seem stagnated and harbor large quantities of algae	State of Montana Newsroom (2015)
Nebraska	Microcystin: ≥ 20 µg/L	Nebraska Department of Environmental Quality and Nebraska Department of Health and Human Services: Division of Public Health (2018)
New Hampshire	Cyanobacteria: > 50 percent of total cell counts from toxigenic cyanobacteria	New Hampshire Department of Environmental Services (2014)
New Jersey	Microcystins (as total including -LR and other detectable congeners): 3 µg/L	New Jersey Department of Environmental Protection (2017)
	Cylindrospermopsin: 8 µg/L	
	Cyanobacterial cell count: ≥ 20,000 cells/mL	
New York	Bloom: credible report or digital imagery of a bloom determined as likely to be potentially toxic cyanobacteria by DEC or DOH staff	New York State Department of Environmental Conservation (2017)
	Blue-green chlorophyll *a*: > 25 µg/L	
	Microcystin-LR: > 4 µg/L	
North Carolina	Visible discoloration or surface scum	North Carolina Health and Human Services: Division of Public Health (2014)
North Dakota	Blue-green algae bloom is present over a significant portion of the lake AND microcystin-LR: ≥ 10 µg/L	North Dakota Department of Health: Division of Water Quality (2016)
Ohio	Microcystins: 6 µg/L	Ohio EPA (2016)
	Cylindrospermopsin: 5 µg/L	
Oklahoma	Cyanobacteria: 100,000 cell/mL	Oklahoma Legislature (2012)

State	Lowest Recreational Water Guideline or Action Level[a]	Reference
	Microcystin: > 20 µg/L	
Oregon	Cylindrospermopsin: ≥ 8 µg/L	Oregon Health Authority (2018)
	Microcystin: ≥ 4 µg/L	
	Microcystis: > 40,000 cells/mL	
	Planktothrix: > 40,000 cells/mL	
	Toxigenic species: > 100,000 cells/mL	
	Visible scum with documentation and testing	
Pennsylvania	Microcystin: > 6 µg/L	Pennsylvania Department of Environmental Protection (2014)
	Cylindrospermopsin: > 5 µg/L	
	HAB verified by visual observation	
Rhode Island	Cyanobacteria: > 70,000 cells/mL	Rhode Island Department of Environmental Management and Rhode Island Department of Health (2013)
	Microcystin-LR: ≥ 14 µg/L	
	Visible cyanobacteria scum or mat	
Utah	Cyanobacteria: 20,000–10,000,000 cells/mL	Utah Department of Environmental Quality and Department of Health (2017)
	Microcystin: 4–2,000 µg/L	
Vermont	Cylindrospermopsin: ≥ 10 µg/L	Vermont Department of Health (2015)
	Microcystin-LR (equivalents): ≥ 6 µg/L	
	Visible known blue-green algae bloom/scum or an unknown, potentially blue-green algae (i.e., not pollen), bloom/scum	
Virginia	Blue-green algal "scum" or "mats" on water surface	Virginia Department of Health (2012)
	Microcystin: > 6 µg/L	
	Microcystis: 5,000 to < 20,000 cells/mL	
Washington	Bloom is forming or a bloom scum is visible (toxic algae may be present); cyanotoxin levels do not exceed thresholds	Hardy and Washington State Department of Health (2008); Hardy and Washington State Department of Health (2011)
	Microcystins: 6 µg/L	
	Cylindrospermopsin: 4.5 µg/L	
West Virginia	Blue-green algal blooms observed and monitored	West Virginia Department of Health and Human Resources (2015)
Wisconsin	Cyanobacteria: > 100,000 cells/mL	Wisconsin Department of Natural Resources (2012); Wisconsin Department of Health Services (2016)
	Visible scum layer	Werner and Masnado (2014) ; Wisconsin Department of Health Services (2016)

[a] More details are provided in Appendix B.
[b] Connecticut states "based on US EPA's draft recreational criterion, CT DPH suggests a cyanotoxin threshold of 4 µg/L microcystin."
[c] The EPA found that Texas published guidelines in the past, but the guidelines were no longer on its website.

3.0 NATURE OF THE STRESSORS

This section describes cyanobacteria and cyanobacterial blooms that have the potential to produce microcystins and cylindrospermopsin. It also describes the chemical and physical properties, sources and occurrence information in different media, environmental fate, and toxicokinetics for the cyanotoxins. The information in this section is based on information the EPA presented in its HESDs and Drinking Water Health Advisories (U.S. EPA 2015a, 2015b, 2015c, 2015d). The EPA conducted supplemental literature searches in September 2015 to capture new references related to the following topics:

- Levels of human exposure to cylindrospermopsin or microcystins through recreational water activities.
- Health effects for humans or animals exposed to cylindrospermopsin or microcystins.
- State and international safety levels or criteria for microcystins or cylindrospermopsin.
- Recreational exposure ingestion rates for children's age groups.
- Incidents of pet or livestock adverse health effects, including mortality, due to exposure to cyanotoxins.

For detailed information on these supplemental literature searches and the five research questions that correspond to the bullets above, see Appendix C.

Cyanobacteria are a group of microorganisms that naturally occur in freshwater and marine environments and can be found at higher densities in eutrophic or nutrient-enriched water bodies. Many cyanobacteria are capable of producing toxins, referred to as cyanotoxins, which can adversely affect human health. Under the right conditions of water temperature, light, pH, nutrient availability, and other factors, cyanobacteria can reproduce rapidly, forming what are commonly referred to as cyanobacterial HABs. Other microorganisms can form HABs, but for the purpose of this document the usage of "HABs" refers to cyanobacterial HABs unless otherwise specified.

3.1 Cyanobacteria and Cyanobacterial Blooms

Cyanobacteria are photosynthetic prokaryotes (Seckbach and Oren 2007) and are ubiquitous in the environment. Cyanobacteria smaller than 2.0 μm are known as picocyanobacteria (Jakubowska and Szeląg-Wasielewska 2015). The chloroplast, found in photosynthetic eukaryotes like algae and plants, evolved from an endosymbiotic relationship with cyanobacteria (Kutschera and Niklas 2005). Ecologists historically grouped cyanobacteria, often referred to as "blue-green algae," with eukaryotic algae because they contain chlorophyll *a* and can perform oxygenic photosynthesis. However, cyanobacteria are prokaryotes (i.e., no discrete membrane-bound nucleus or membrane-bound subcellular organelles) and are genetically related to other bacteria in the eubacteria domain. Taxonomically, they are classified in the phylum Cyanobacteria or Cyanophyceae (Carmichael 2008; O'Neil et al. 2012).

Cyanobacteria, including picocyanobacteria, can produce bioactive compounds including toxins, which can be harmful. These biomolecules include hepatotoxic, neurotoxic, and cytotoxic compounds and compounds that can result in allergic reactions (Burkholder and Glibert 2006; Carmichael 1994; Jaiswal et al. 2008; Jakubowska and Szeląg-Wasielewska 2015; Śliwińska-Wilczewska et al. 2018; Volk and

Mundt 2007). Studies have shown that exposure to cyanobacterial cells can cause health effects that are independent of the cyanotoxins; this information is detailed in Appendix D.

Under certain conditions, cyanobacteria possessing the toxin synthesis genes, also referred to as toxigenic cyanobacteria, begin producing cyanotoxins. Numerous biotic and abiotic factors can influence not only the dominance of cyanobacteria within the overall phytoplankton community, but also the proportion of toxigenic cyanobacteria relative to non-toxin-producing cyanobacteria (Davis et al. 2009; Hyenstrand et al. 1998; McCarthy et al. 2009; Neilan et al. 2013; Gobler et al. 2016). Multiple species of cyanobacteria are capable of producing the same toxin, such as the microcystins, which can pose a risk to human and animal health (Crawford et al. 2017). Although scientists have observed a generalized relationship between cyanobacteria density or chlorophyll *a* and cyanotoxin concentration, these relationships are affected by the dominance of the toxin-producing cyanobacteria within the overall cyanobacterial community (Zhang et al. 2014; Loftin et al. 2016b).

Members of the genera *Microcystis*, *Dolichospermum* (*Anabaena*), *Nostoc*, *Fischerella*, *Planktothrix* (formerly *Oscillatoria*), and *Gloeotrichia* can produce microcystins (Carey et al. 2012b; Codd et al. 2005; Duy et al. 2000; Stewart et al. 2006c). *Microcystis aeruginosa* occurs mostly at the surface with higher light intensities and in shallow lakes. Kosten et al. (2012) surveyed 143 shallow lakes along a latitudinal gradient (between 5–55°S and 38–68°N) from subarctic Europe to southern South America. *Microcystis* have been documented to occur in blooms on all continents except Antarctica and often dominate phytoplankton assemblages in the summer (O'Neil et al. 2012). *Microcystis* have been documented throughout the United States (Carmichael 2001; Jacoby et al. 2000). Species of cyanobacteria, like *Microcystis*, that occur at or near the surface due to buoyancy and wind, can accumulate on shores and bays where they can form scums (Australian Government National Health and Medical Research Council 2008; WHO 2003b).

Cylindrospermopsin can be produced by a number of cyanobacterial species including *Raphidiopsis raciborskii* (formerly Cylindrospermopsis raciborskii),[3] *Aphanizomenon flos-aquae*, *Aphanizomenon gracile*, *Aphanizomenon ovalisporum*, *Umezakia natans*, *Anabaena bergii*, *Anabaena lapponica*, *Anabaena planctonica*, *Lyngbya wollei*, *Raphidiopsis curvata*, and *Raphidiopsis mediterranea* (B-Béres et al. 2015; Kokocinski et al. 2013; McGregor et al. 2011; Moreira et al. 2013). These species do not tend to form visible surface scums and the highest concentrations of cyanobacterial cells occurs below the water surface (Falconer 2005).

Cylindrospermopsin-producing cyanobacteria occur in tropical or subtropical regions, as well as warmer temperate regions. For example, *Cylindrospermopsis raciborskii* occurs in freshwater ponds, rivers, reservoirs, and eutrophic lakes and has been found in Australia, Asia, Europe, Africa, and South, Central, and North America (Fuentes et al. 2010). According to a survey conducted in Florida in 1999 from June to November, the most frequently observed toxigenic cyanobacteria were *Microcystis* (43.1 percent), *Cylindrospermopsis* (39.5 percent), and *Anabaena* (28.7 percent) (Burns 2008).

Research indicates that cyanotoxins can confer competitive advantage for survival and replication and are associated with physiological functions of cyanobacterial cell signaling, nutrient uptake, iron scavenging, maintenance of homeostasis, and protection against oxidative stress (Holland and Kinnear 2013). Cylindrospermopsin production provides a competitive advantage to cyanobacteria when phosphorus becomes scarce. Bar-Yosef et al. (2010) observed that when phosphorus is scarce, the

[3] Cyanobacteria taxonomy is continuously being revised. The genus *Cylindrospermopsis* has been renamed to *Raphidiopsis*. This document mostly maintains the genus name of *Cylindrospermopsis.*

cyanobacterium *Aphanizomenon ovalisporum* releases cylindrospermopsin, which causes other microorganisms to release alkaline phosphatase, a compound that will increase available free phosphorus. Subsequently, *Aphanizomenon* can gain access to phosphorus made available by other microorganisms while simultaneously conserving the energy and resources required to express and excrete alkaline phosphatase (Bar-Yosef et al. 2010). The precise ecological function of microcystins has not been determined conclusively (Zurawell et al. 2005). Studies comparing wild-types and mutants of a microcystin-producing species, examining the genes involved in microcystin biosynthesis, and evaluating *Microcystis* colony size have suggested that microcystins play important physiological roles in cyanobacteria, including colony formation (Kaplan et al. 2012; Zurawell et al. 2005). Gobler et al. (2007) observed decreased zooplankton grazing when toxigenic *Microcystis* were actively producing microcystin. Although cyanotoxins can negatively affect humans and other animals, research suggests that the primary functions of cyanotoxins are in cyanobacterial physiology and microbial ecology.

Cyanobacteria can regulate their buoyancy; thus, they can actively seek water depths with optimal growth conditions and will enlarge their gas vesicles to adapt to turbulent conditions. When weather conditions shift from turbulent to strongly stratified, excessively buoyant cells may accumulate at the surface because the regulation of buoyancy takes a few days (Australian Government National Health and Medical Research Council 2008, WHO 2003b). When the rate of cyanobacterial cell growth exceeds the loss rate for a population, positively buoyant, floating cyanobacterial cells can also accumulate at the surface (Falconer 1998). This accumulation can form a visibly colored scum on the water surface, which can contain more than 10,000 cells/mL (Falconer 1998). Scums can pose an elevated health risk to recreational users. The floating scum can be concentrated by prevailing winds in certain surface water areas, especially at the shore as is the case for *Microcystis*. Scums have frequently been reported to accumulate cells and cyanotoxin concentration by a factor of 1,000 or more, with million-fold accumulations resulting in pea soup consistency (Australian Government National Health and Medical Research Council 2008; WHO 2003b).

The microbial community can be complex and variable. It can consist of multiple different species and strains of cyanobacteria and other microbes. Microbial interactions can occur within blooms, such as competition and adaptation between toxic and nontoxic cyanobacterial strains, as well as impacts from viruses and zooplankton grazers like *Daphnia* (large generalist grazers), copepods, and cladocerans (Ger et al. 2014). Each of these microbial-related factors can cause fluctuations in bloom development and composition.

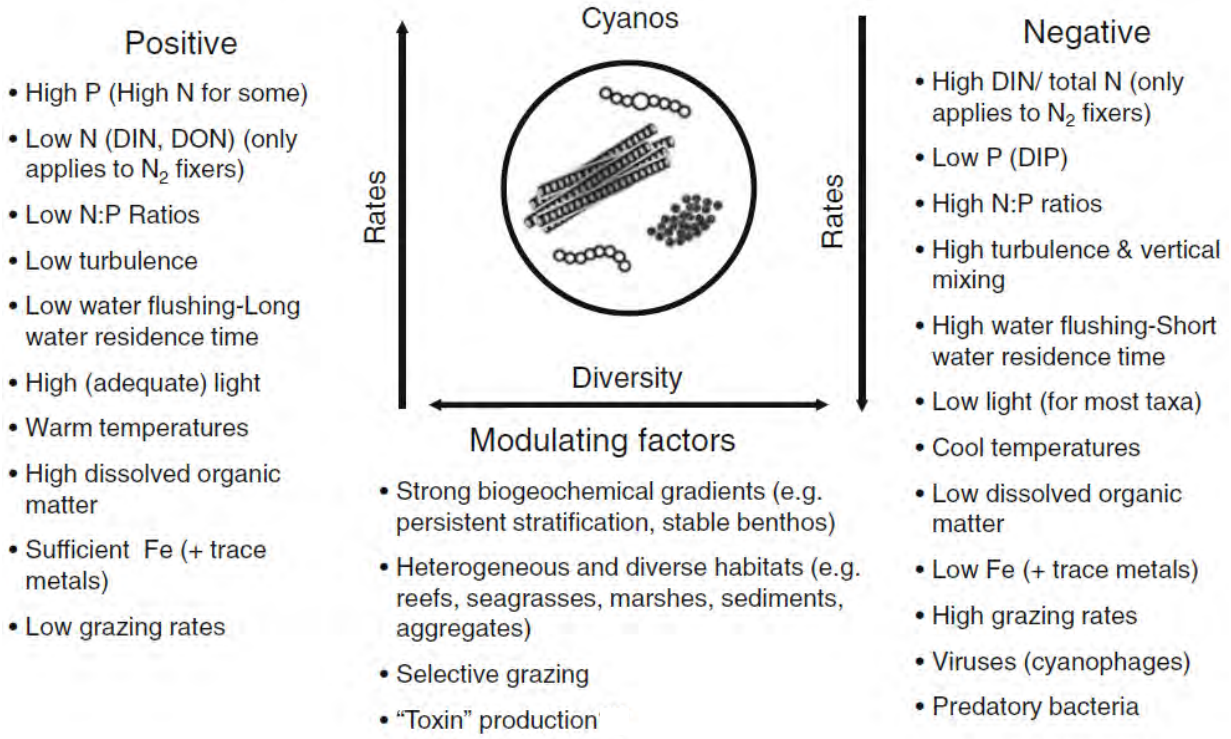
3.1.1 Environmental Factors Influencing Occurrence of Cyanobacteria and Cyanotoxins

A variety of physical, chemical, and environmental factors can influence both cyanobacteria proliferation and toxin production, including nutrient (e.g., nitrogen and phosphorus) concentrations, water temperature, light levels, and pH. Other factors include water turbulence, mixing, and flushing, oxidative stressors, and interactions with other biota (e.g., viruses, bacteria, and animal grazers), as well as their combined effects (Paerl and Otten 2013a, 2013b). See Figure 3-1.

Total cyanobacterial density in a bloom and cyanotoxin concentration are not always closely related. Cyanotoxin concentrations depend on the dominance and diversity of species and strains within the bloom along with environmental and ecosystem influences on bloom dynamics (Chorus et al. 2000; Hitzfeld et al. 2000; WHO 1999). Cyanotoxin production can vary among strains and clones of a single species (Carmichael 1994; Utkilen and Gjølme 1992) and within and between blooms (Codd and Bell 1985). Growth phase also can influence cyanotoxin production (Jaiswal et al. 2008). Biomass and toxin production do not necessarily coincide (section 7.5.2.3). Francy et al. (2016) modeled the relationship of

environmental variables compared to cyanotoxin levels. They demonstrated that some environmental factors such as measures of the algal community (e.g., phycocyanin, cyanobacterial biovolume, and cyanobacterial gene concentrations) and pH are strongly correlated with microcystin concentrations.

Figure 3-1. Environmental Factors Influencing Total Cyanobacterial Blooms, Reproduced from Paerl and Otten (2013b)



Cyanotoxins can be found inside the cell (i.e., intracellular) or external to the cell in the water (i.e., extracellular). The proportion of intracellular versus extracellular cyanotoxin can vary. Extracellular microcystins (either dissolved in water or bound to other materials) typically are less than 30 percent of the total microcystin concentration in source water (Graham et al. 2010). Most of the microcystins are intracellular and released into the water when the toxigenic cyanobacterial cells rupture or die. Cylindrospermopsin can be retained within the cyanobacterial cell or released. The ratio of intracellular to extracellular cyanotoxin can change depending on the growth phase with as much as 50 percent of cylindrospermopsin produced by *Cylindrospermopsis raciborskii* released extracellularly (Griffiths and Saker 2003).

A complex interplay of environmental factors dictates the spatial and temporal changes in the concentration of cyanobacterial cells and their toxins with respect to the dominant species. Factors such as the amount and timing of nutrient supply (i.e., nutrient concentration and nutrient loading), the relative proportions of nutrients (i.e., nitrogen to phosphorus ratio), dissolved organic matter availability, temperature, and light attenuation, as well as other physico-chemical processes, can play a role in shaping cyanobacterial bloom composition and cyanotoxin production (Paerl and Huisman 2008; Paerl and Otten 2013b).

Some cyanobacteria possess toxin genes that enable them to produce toxins, while other cyanobacteria do not contain toxin genes and therefore cannot produce toxins. For example, cyanobacteria that can produce microcystins contain a collection of genes, called "*mcy*" genes, that when expressed produce

microcystins. Multiple species of cyanobacteria can contain this set of genes. Together these species comprise microcystin-producing toxigenic cyanobacteria. Ten genes are in the microcystin gene cluster, *mcyA* through *mcyJ* (Tillett et al. 2000). Different researchers have studied the occurrence and prevalence of these genes within cyanobacteria populations.

Environmental factors can provide competitive advantages to *Microcystis* relative to other phytoplankton (Jacoby et al. 2000; Marmen et al. 2016). Evidence suggests that these environmental factors also affect the relative abundance of microcystin-producing strains and non-microcystin-producing strains (Marmen et al. 2016). *Microcystis* thrive in warmer temperatures, with optimal growth and photosynthesis occurring above 25°C (O'Neil et al. 2012). A Japanese study between May and November 2006 found that the toxin-producing species, *Microcystis aeruginosa*, dominated in months with relatively higher water temperatures, while the non-toxin-producing species, *M. wesenbergii*, dominated in months with lower water temperatures (Imai et al. 2009). The genetic composition of the bloom can also influence the degree of toxicity associated with an algal bloom. Lee et al. (2015) found that *Microcystis* typically comprised less than one percent of the total cyanobacterial abundance in Vancouver Lake, Washington, but the majority of the *Microcystis* cells contained the toxin-producing gene. Despite comprising a small percentage of the total cyanobacterial community in this lake, *Microcystis* were the sole microcystin-producing cyanobacteria and were responsible for microcystin concentrations that exceeded the WHO guidelines several times throughout the sampling period. In addition, increases in phosphate concentrations were associated with increases in both toxigenic and non-toxigenic *Microcystis* and with toxin production. The authors note that quantifying *Microcystis mcyE* gene (one of the genes responsible for toxin production) copy number, rather than relying solely on visual cell counts, might be a better estimate of overall cyanotoxin concentration (Lee et al. 2015).

Zhang et al. (2015) observed that low flow conditions favored total cyanobacteria and higher flow conditions favored green algae. Loftin et al. (2016a) suggest that low stream flow, shallow depth, and high water-column light penetration in Piedmont streams favored periphyton occurrence (mixture of algae, cyanobacteria, heterotrophic bacteria, and detritus).

Phytoplankton competition and food web interactions occur as blooms develop, persist, and decline, thereby impacting cyanotoxin concentrations in surface waters. In addition, potential warming of surface waters and changes in precipitation could result in changes in ecosystem dynamics that lead to more frequent formation of cyanobacteria blooms and their associated toxins (Paerl et al. 2011; Paerl and Huisman 2008; Paerl and Otten 2013b).

3.1.1.1 Nutrients

Nutrients, particularly nutrient over-enrichment, are key environmental drivers that influence the proportion of cyanobacteria in the phytoplankton community, the cyanobacterial biovolume, cyanotoxin production, and the impact that cyanobacteria may have on ecosystem function and water quality (Yang et al. 2016a; Beaulieu et al. 2013; Paerl et al. 2011). Cyanobacteria have been shown to dominate the phytoplankton communities in eutrophic lakes (Downing et al. 2001; Monchamp et al. 2014). Phosphorus loading has been linked to the proliferation of cyanobacteria and the shift toward cyanobacterial dominance of the phytoplankton community (O'Neil et al. 2012). However, it is important to consider both phosphorus and nitrogen when considering the occurrence of toxigenic cyanobacterial blooms. Cyanobacterial toxin concentrations are also associated with nutrient levels (Wang et al. 2002); however, different cyanobacteria species use organic and inorganic nutrient forms differently. Dolman et al. (2012) found that total cyanobacterial biomass was higher in lakes with above-

average nitrogen and phosphorus concentrations and that concentrations of all cyanotoxin groups were higher in lakes with higher total nitrogen and total phosphorus concentrations.

Paerl (2008) demonstrated that nitrogen and phosphorus additions, both independently and together, can stimulate primary productivity and *Cylindrospermopsis raciborskii* biomass. Elevated nitrogen and phosphorus loading can enhance the growth and cyanotoxin levels of *Microcystis* blooms and microcystin synthetase gene expression (Gobler et al. 2007; O'Neil et al. 2012; Marmen et al. 2016). Gobler et al. (2007) found that *Microcystis* dominance and toxin production was stimulated by elevated nitrogen and suppressed by nitrogen limiting conditions. Toxin production may cause the inhibition of grazing by mesozooplankton and further accumulation of cyanobacterial cells. Willis et al. (2015) found the highest growth rates for environmental isolates of *Cylindrospermopsis raciborskii* were observed with the addition of nitrogen.

The relative abundance of nitrogen and phosphorus can be an important consideration in regards to toxigenic cyanobacterial blooms. Loadings of nitrogen, or phosphorus, or both, to water bodies from agricultural, industrial, and urban sources influences the development of total cyanobacterial blooms and are associated with cyanotoxin production (Paerl et al. 2011). Smith (1983) was the first to describe a strong relationship between the relative amounts of nitrogen and phosphorus in surface waters and toxigenic cyanobacterial blooms. Smith proposed that cyanobacteria should be superior competitors under conditions of nitrogen limitation because of their unique capacity for nitrogen fixation, although many cyanobacteria like *Microcystis* that produce toxins do not fix nitrogen. Many toxigenic cyanobacterial blooms are comprised of non-nitrogen-fixing genera and in the presence of elevated phosphorus, nitrogen can be a limiting factor for biomass proliferation and microcystin production (Gobler et al. 2007). Schindler et al. (2008) demonstrated that lower nitrogen inputs relative to phosphorus loadings can lead to dominance of nitrogen-fixing cyanobacteria in mesocosm- and ecosystem-scale experiments in prairie and boreal lakes. Otten et al. (2012) reported higher average microcystin concentrations and a higher prevalence of toxigenic *Microcystin* biomass at sites that had narrower TN:TP ratios (< 20) in Lake Taihu, China. Fortin et al. (2015) demonstrated that the dominance of *Microcystis* depended on the ratio of nitrogen to phosphorus, with a (mass) ratio 11:1 resulting in the highest abundance of *Microcystis*, whereas the concentrations of each nutrient were significant factors affecting the amount of biomass that could be generated.

Cyanotoxin concentration can be related to cyanobacterial cell abundance, which is facilitated by nutrient availability (Welker 2008), so nutrient concentration can be correlated to cyanotoxin concentration. Yuan et al. (2014; 2015) developed nutrient thresholds related to microcystin concentrations, cyanobacterial biovolume, and chlorophyll *a*. Nutrient availability, environmental conditions, and ecosystem interactions can affect the production and amount of toxins that cells produce and release (Bar-Yosef et al. 2010; Dolman et al. 2012; Graham et al. 2004; Paerl et al. 2001). For example, both nitrogen and phosphorus have been shown to promote the production of microcystins during bloom events (Davis et al. 2009; Gobler et al. 2016; Ha et al. 2009). Horst et al. (2014) found a significant positive relationship between cellular microcystin amounts and nitrate concentration with nitrogen limitation related to lower cell quotas of microcystin. Ha et al. (2009) found that microcystin concentrations were highly associated with *mcyA* gene copies and that high concentrations of nitrates and ammonium increased microcystin production by promoting the growth of toxigenic *Microcystis*. Elevated phosphorus has been shown to favor toxigenic strains over non-toxin strains coupled with higher intracellular toxin concentrations (Boopathi and Ki 2014; Burford et al. 2016).

Soluble phosphates and nitrates may also result in the increased production of microcystins (ILS 2000; O'Neil et al. 2012; Paerl and Scott 2010; Wang et al. 2010). Davis et al. (2009) found that growth rates

of toxigenic *Microcystis* were higher than nontoxic strains as temperature increased in the presence of elevated soluble phosphorus and that toxigenic cells contained more copies of the gene *mcyD* under these conditions. The authors conclude that lakes experiencing this combination of factors could experience more toxic blooms (Davis et al. 2009). In the Sacramento-San Joaquin delta in California nitrogen and phosphorus are available in non-limiting amounts and facilitate persistence of total cyanobacterial blooms (Berg and Sutula 2015). A study by Lehman et al. (2015) characterizes nitrogen sources of a *Microcystis* bloom in the San Francisco Estuary using stable isotopes. They reported that ammonium from the Sacramento River was the likely sole source of the nitrogen for most of the bloom, overriding nitrate contributions from the San Joaquin River.

Jacoby et al. (2000) characterized multiple physical and chemical environmental factors associated with blooms in the summer of 1994 and 1995 at Steilacoom Lake, Washington. The dominance of *Microcystis aeruginosa* in the lake was associated with low nitrogen-to-phosphorus ratios and low nitrate-nitrogen with sufficient ammonium-nitrogen. Microcystin concentrations were positively correlated with increasing soluble reactive phosphorus concentrations with the highest microcystin concentrations associated with a low ratio of soluble nitrogen to soluble reactive phosphorus (less than five). The authors reported that microcystin production per gram cyanobacterial biomass was not consistent, thus no relationship was found between *Microcystis aeruginosa* abundance and microcystin concentration. A significant positive relationship between total phosphorus concentrations and total cyanobacteria densities was observed in both years of the study (Jacoby et al. 2000).

During bloom events, nutrients on a local scale are incorporated into the production of biomass and decrease in the water column within the bloom, even in eutrophic water bodies. Kuniyoshi et al. (2013) showed that phosphate deficiency resulting from exponential biomass production can result in approximately seven-fold increase in microcystin synthesis. Bar-Yosef et al. (2010) reported that cylindrospermopsin-producing *Aphanizomenon* excrete cylindrospermopsin when phosphorus-limiting conditions occur within the bloom, to induce other cells to produce and excrete alkaline phosphatase, thus increasing availability of extracellular inorganic phosphate. Cylindrospermopsin is energetically cheaper for the cell to produce relative to alkaline phosphatase (Raven 2010) and coupled with a high-affinity phosphorus uptake protein also found in these cells, allows *Aphanizomenon* to increase rapidly, outcompeting other cyanobacteria and dominate a bloom (Bar-Yousef et al. 2010). Preußel et al. (2014) observed that cylindrospermopsin is actively released from *Aphanizomenon ovalisporum* cells subjected to phosphorus limitation, a condition that occurs during the exponential biomass production in a bloom event.

Eutrophic systems already subject to bloom events are prone to further expansion of these blooms due to additional nitrogen inputs, especially if these nutrients are available from internal sources. As the trophic state increases, aquatic systems absorb higher concentrations of nitrogen (Paerl and Huisman 2008; Paerl and Otten 2013b). Recent surveys of cyanobacterial and algal productivity in response to nutrient pollution across geographically diverse eutrophic lakes, reservoirs, estuarine and coastal waters, and in different experimental enclosures of varying sizes demonstrate that greater stimulation is routinely observed in response to both nitrogen and phosphorus additions. Further, this evidence suggests that nutrient co-limitation is widespread (Elser et al. 2007; Lewis et al. 2011; Paerl et al. 2011). These results suggest that reductions in nutrient concentration would reduce eutrophication and cyanobacterial bloom expansion. For example, analysis of observational data collected at high spatial scales support the idea that controlling total phosphorus and total nitrogen could reduce the frequency of high microcystin contamination events by reducing the biomass of total cyanobacteria in the system (Orihel et al. 2012; Scott et al. 2013; Yuan et al. 2014). In addition, reduction of phosphorus in the absence of concurrent

reductions in nitrogen loading may not effectively control the growth, toxicity, or both of cyanobacteria such as *Microcystis* (Gobler et al. 2016). Study authors concluded that reduction of specific nutrient species, such as soluble forms of nitrogen and phosphorus, could reduce the dominance of toxigenic cyanobacteria in the lake microbial community, which could, in turn, decrease the incidences of elevated toxin levels (Davis et al., 2010; Gobler et al. 2016).

3.1.1.2 Temperature

Cyanobacterial blooms commonly occur from spring to early fall in various regions of the United States (Wynne and Stumpf 2015). Conditions such as elevated water temperatures and increased vertical stratification in lakes and reservoirs can support proliferation of total cyanobacteria (Paerl and Huisman 2008). The increasing body of laboratory and field data (Carey et al. 2012a; De Senerpont Domis et al. 2007; Huisman et al. 2005; Jeppesen et al. 2009; Kosten et al. 2012; Reynolds 2006; Wagner and Adrian 2009; Weyhenmeyer 2001) suggest that an increase in temperature may influence cyanobacterial dominance in phytoplankton communities. Some cyanobacteria have higher optimal growth temperatures compared with other phytoplankton and can proliferate at higher water temperatures by outcompeting these other phytoplankton groups (Elliott 2010; Paerl et al. 2011). Warmer water temperatures favor surface bloom-forming cyanobacterial genera because they are heat-adapted, and their maximal growth rates occur at relatively high temperatures, with optimum growth temperatures ranging from 30 to 35°C and optimum microcystin production ranging from 20 to 25°C (Giannuzzi 2018; Reynolds 2006; Robarts and Zohary 1987; WHO 2003b). As the growth rates of the eukaryotic taxa decline in response to warming water temperature, cyanobacterial growth rates reach their optima. Davis et al. (2009) found in four U.S. lakes that concurrent increases in temperature and phosphorus concentrations yielded the highest growth rates of toxic *Microcystis* cells, which led them to conclude that eutrophication and warm temperatures may promote the growth of toxic, rather than nontoxic, populations of *Microcystis* leading to blooms with higher microcystin content.

Cyanobacteria are typically known to proliferate in warm water environments such as tropical and temperate lakes and rivers, but they can also proliferate in cooler water environments under mesophilic and psychrophilic conditions (Seckback and Oren 2007). Cyanobacteria are also found in Antarctic habitats where they play a significant role in microbial ecosystem dynamics by providing fixed carbon via photosynthesis (Singh and Elster 2007). Cyanobacteria can grow in these extreme environments because they can adapt to survive freeze/thaw cycles and they can metabolize at near 0°C (Singh and Elster 2007).

The increase in water column stability associated with higher temperatures, less flow, and shallower water can also favor total cyanobacteria growth (Carey et al. 2012a; Wagner and Adrian 2009). In a study of 143 shallow lakes sampled along a latitudinal transect ranging from subarctic Europe to southern South America, Kosten et al. (2012) reported the percentage of cyanobacteria relative to total phytoplankton biovolume increased steeply with temperature in the lakes. The series of conditions most likely to result in cyanobacterial dominance begin with elevated winter–spring rainfall and runoff, followed by protracted periods of summer drought where temperatures, vertical stratification, and water residence times all increase simultaneously (Paerl and Otten 2013b).

Indirectly, warming can increase nutrient concentrations by enhancing mineralization (Gudasz et al. 2010; Kosten et al. 2009; Kosten et al. 2010) by temperature- or anoxia-mediated sediment phosphorus release (Jensen and Andersen 1992; Søndergaard et al. 2003). Thus, increases in temperature can indirectly increase cyanobacterial biomass through its effect on nutrient concentrations. Others have suggested that warmer conditions may raise total phytoplankton biomass through an alteration of top-

down regulation by selective grazing that favors larger size phytoplankton species and cyanobacterial blooms (Jeppesen et al. 2009; Jeppesen et al. 2010; Teixeira-de Mello et al. 2009). The relationship between temperature and cyanobacterial dominance can be explained not only through a temperature-related effect on the competitive advantage of cyanobacteria, but also by factors such as the percent area covered and the volume of the lake taken up by submerged macrophytes (Carey et al. 2012a; Kosten et al. 2012).

Cylindrospermopsis raciborskii was first identified in the tropics but has also been increasingly found in temperate regions since it was first found in North America in 1955 (Hong et al. 2006). *Cylindrospermopsis raciborskii* blooms are most likely to occur between the temperatures of 25 to 32°C but can sustain biomass at temperatures as low as 11°C (Antunes et al. 2015). In Florida, *C. raciborskii* was found to be the dominant cyanobacteria species in one lake all year round (Burns 2008). In 2006, *C. raciborskii* was detected in lakes in southern Louisiana (Fuentes et al. 2010). Conditions promoting its growth were shallow, warm surface water (over 30°C) and low light intensities. The highest densities of *C. raciborskii* were observed from June through August with densities ranging from 37,000 cells/mL to more than 160,000 cells/mL. In a study of two lakes directly connected to Lake Michigan, Hong et al. (2006) found low levels of *C. raciborskii* only in the late summer, and these were associated with elevated bottom water temperatures and phosphorus concentrations.

3.1.1.3 Sunlight

Sunlight availability and turbidity can have a strong influence on the cyanobacteria species that predominate, as well as the depth at which they occur (Carey et al. 2012a; Falconer 2005). The authors (Carey et al. and Falconer) found a greater proportion of the total phytoplankton biovolume attributable to cyanobacteria in lakes with high rates of light absorption. They could not establish cause and effect from their field data, but other controlled experiments and field data have demonstrated that light availability can affect the competitive balance among a large group of shade-tolerant species of cyanobacteria, primarily *Oscillatoriales* and other phytoplankton species (Scheffer et al. 1997; Smith 1986).

3.1.1.4 pH Levels

Total cyanobacterial blooms intensify and persist at pH levels between six and nine (Caraco and Miller 1998; WHO 2003a). Kosten et al. (2012) noted that pH affected cyanobacteria abundance in lakes along a latitudinal transect from Europe to southern South America. The percentage of cyanobacteria in the 143 shallow lakes sampled highly correlated with pH, increasing as the pH increased. Shapiro (1984) hypothesized that cyanobacteria have a competitive advantage over other phytoplankton species because they are efficient users of carbon dioxide in water. When dissolved carbon dioxide is high (low pH), conditions favor growth and replication of the green algal colonies over the blue-green cyanobacteria (Caraco and Miller 1998; Shapiro 1984). At alkaline pH levels, inorganic carbon is present as carbonate anion rather than as carbon dioxide, carbonic acid, or bicarbonate anion. This situation favors the growth of cyanobacteria because they can carry out photosynthesis when the levels of dissolved carbon dioxide are very low (high pH). The blue-green algae have a much higher photosynthetic demand for the dissolved carbon dioxide allowing them to out compete the green algae for the limited supply (Caraco and Miller 1998; Shapiro 1984). Thus, a higher water column pH can correlate with a higher proportion of cyanobacteria in an algal bloom.

The Caraco and Miller (1998) study suggests that pH and dissolved carbon dioxide, although chemically linked, are also independent factors in bloom dynamics because, even when dissolved carbon dioxide in

water is mechanically enriched, an alkaline pH still favors growth of the cyanobacteria over the green algae if nutrient inputs are constant.

3.2 Cyanotoxins

Much of the information and the studies summarized in this section for microcystins and cylindrospermopsin are described in detail in the EPA's HESDs and Drinking Water Health Advisories for microcystins and cylindrospermopsin (U.S. EPA 2015a, 2015b, 2015c, 2015d). The EPA's HESDs established the scientific basis for the EPA Drinking Water Health Advisories and also informed the EPA in developing these ambient water quality criteria (AWQC) or swimming advisories. This section summarizes the information that is provided in more detail in the EPA's HESDs. Additional information can be found in the EPA's HESDs for microcystins and cylindrospermopsin (U.S. EPA 2015c, 2015d).

3.2.1 Chemical and Physical Properties

Structurally, microcystins are monocyclic heptapeptides that contain seven amino acids joined end to end and then head to tail to form cyclic compounds that are comparatively large; molecular weights range from approximately 800 to 1,100 g/mole for the different congeners (e.g., microcystin-LR is 995.17 g/mole). The cyclic peptides include more than 100 congeners of microcystins (Niedermeyer 2014). Figure 3-2 provides the structure of microcystin where X and Y represent variable amino acids. Although substitutions mostly occur in positions X and Y, other modifications have been reported for all the amino acids (Puddick et al. 2015).

The microcystin congeners are named based on their two variable amino acids (Carmichael et al. 1988). For example, microcystin-LR, the most common congener (Carmichael 1992). The letters used to identify the variable amino acids are the standard single letter abbreviations for the amino acids found in proteins. The variable amino acids are usually the L-amino acids as found in proteins. In Figure 3-2, which shows the structure of microcystin-LR, leucine is in the X position and arginine is in the Y position. Table 3-1 lists the most common microcystin congeners, including the amino acids in the X and Y positions.

There are other variants of microcystins besides those that arise because of the two interchangeable amino acids on the microcystin ring. For example, demethylated congeners have been observed in Europe; Wejnerowski et al. (2018) identified demethylated forms of microcystin-RR and microcystin-LR in a toxigenic cyanobacterial bloom in Poland. Observations of demethylated microcystins suggest that more than 200 microcystin congeners are possible.

Figure 3-2. Structure of Microcystin (Kondo et al. 1992)

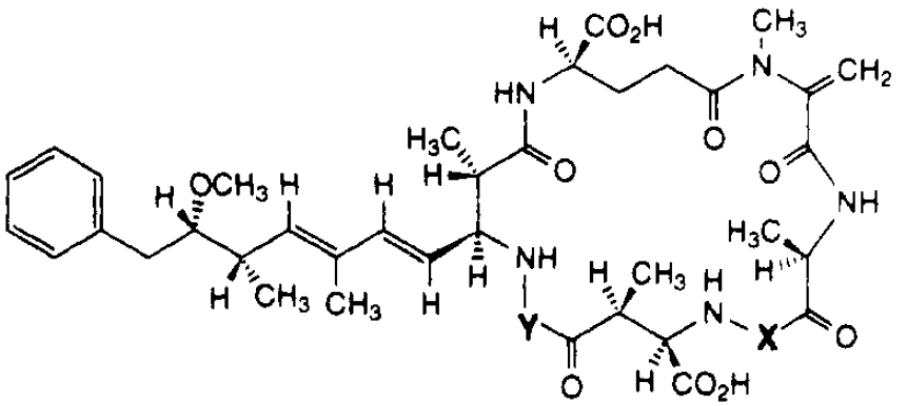


Table 3-1. Abbreviations for Selected Microcystins (Yuan et al. 1999)

Microcystin Congeners	Amino Acid in X	Amino Acid in Y
Microcystin-LR	Leucine	Arginine
Microcystin-RR	Arginine	Arginine
Microcystin-YR	Tyrosine	Arginine
Microcystin-LA	Leucine	Alanine
Microcystin-LY	Leucine	Tyrosine
Microcystin-LF	Leucine	Phenylalanine
Microcystin-LW	Leucine	Tryptophan

The preponderance of toxicological data on the effects of microcystins result from tests using the microcystin-LR congener. Toxicity data suggest that microcystin-LR is as potent as or more potent than other studied microcystins and that the most toxic microcystins are those with the more hydrophobic L-amino acids (e.g., -LA, -LR, and -YR); the least toxic are those with hydrophilic amino acids, such as microcystin-RR (U.S. EPA 2015d; Ito et al. 2002; Rinehart et al. 1994; Vesterkvist and Meriluoto 2003; WHO 1999). Data on the -RR, -YR, and -LA congeners, however, are limited, and toxicity values cannot be derived for them. Therefore, values developed from data specific to microcystin-LR can represent other present microcystin congeners.

Table 3-2 provides chemical and physical properties of microcystin-LR. Microcystins are water soluble. In aquatic environments, the cyclic peptides tend to remain contained within the cyanobacterial cell and are released in substantial amounts only when the cell walls are broken down (cell lysis).

Cylindrospermopsin is a tricyclic alkaloid with the molecular formula of $C_{15}H_{21}N_5O_7S$ (Ohtani et al. 1992) and a molecular weight of 415.43 g/mole. It is a dipolar ion with localized positive and negative charges (Ohtani et al. 1992). The chemical structure of cylindrospermopsin is presented in Figure 3-3(a). Two naturally occurring congeners of cylindrospermopsin have been identified, 7-epi-cylindrospermopsin (the epimer of cylindrospermopsin) and 7-deoxycylindrospermopsin; see Figure 3-3(b) and (c) (de la Cruz et al. 2013; Norris et al. 1999). Recently, Wimmer et al. (2014) identified two new analogs, 7-deoxy-desulfo-cylindrospermopsin and 7-deoxy-desulfo-12-acetylcylindrospermopsin,

from the Thai strain of *Cylindrospermopsis raciborskii*. However, it is not clear if these are cylindrospermopsin congeners, precursors, or degradation products.

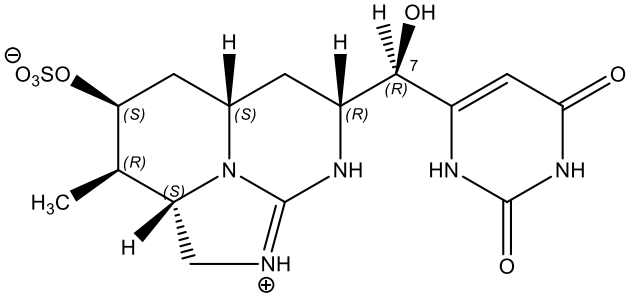
Table 3-2. Chemical and Physical Properites of Microcystin-LR

Property	Microcystin-LR
Chemical Abstracts Registry (CAS) Number	101043-37-2
Chemical Formula	$C_{49}H_{74}N_{10}O_{12}$
Molecular Weight	995.17 g/mole
Color/Physical State	Solid
Boiling Point	Not available (N/A)
Melting Point	N/A
Density	1.29 g/cm^3
Vapor Pressure at 25°C	N/A
Henry's Law Constant	N/A
Log Octanol-Water Partition Coefficient (K_{ow})	2.16; -1.41 to 1.67 as pH decreases
Soil Organic Carbon-Water Partition Coefficient (K_{oc})	N/A
Solubility in Water	Highly*
Other Solvents	Ethanol and methanol

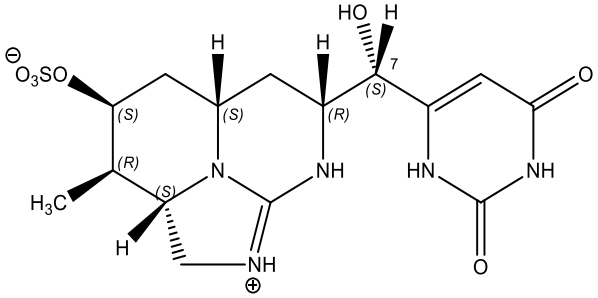
Sources: Chemical Book (2012); TOXLINE (2012); Ward and Codd (1999) and McCord et al. (2018) for log K_{ow}.
* Microcystin congeners vary in their relative solubility in water.

Figure 3-3. Structure of Cylindrospermopsin and Structurally Related Cylindrospermopsins (de la Cruz et al. 2013)

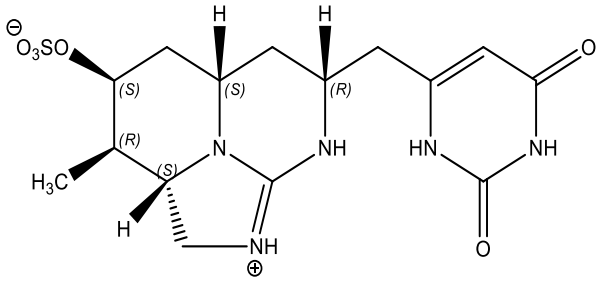
(a) Cylindrospermopsin



(b) 7-epi-cylindrospermopsin (the epimer of cylindrospermopsin)



(c) 7-deoxycylindrospermopsin



The physical and chemical properties of cylindrospermopsin are presented in Table 3-3. Cylindrospermopsin is highly soluble in water (Chiswell et al. 1999; Moore et al. 1998). It is isolated for commercial use mostly from *Cylindrospermopsis raciborskii*. Some relevant physico-chemical properties of cylindrospermopsin could not be identified, and no physico-chemical properties were found for the structurally related cylindrospermopsins.

Table 3-3. Chemical and Physical Properties of Cylindrospermopsin

Property	Cylindrospermopsin
CAS Registry Number	143545-90-8
Chemical Formula	$C_{15}H_{21}N_5O_7S$
Molecular Weight	415.43 g/mole
Color/Physical State	White powder
Boiling Point	N/A
Melting Point	N/A
Density	2.03 g/cm^3
Vapor Pressure at 25°C	N/A
Henry's Law Constant	N/A
K_{ow}	N/A
K_{oc}	N/A
Solubility in Water	Highly
Other Solvents	Dimethyl sulfoxide and methanol

Sources: Chemical Book (2012); TOXLINE (2012).

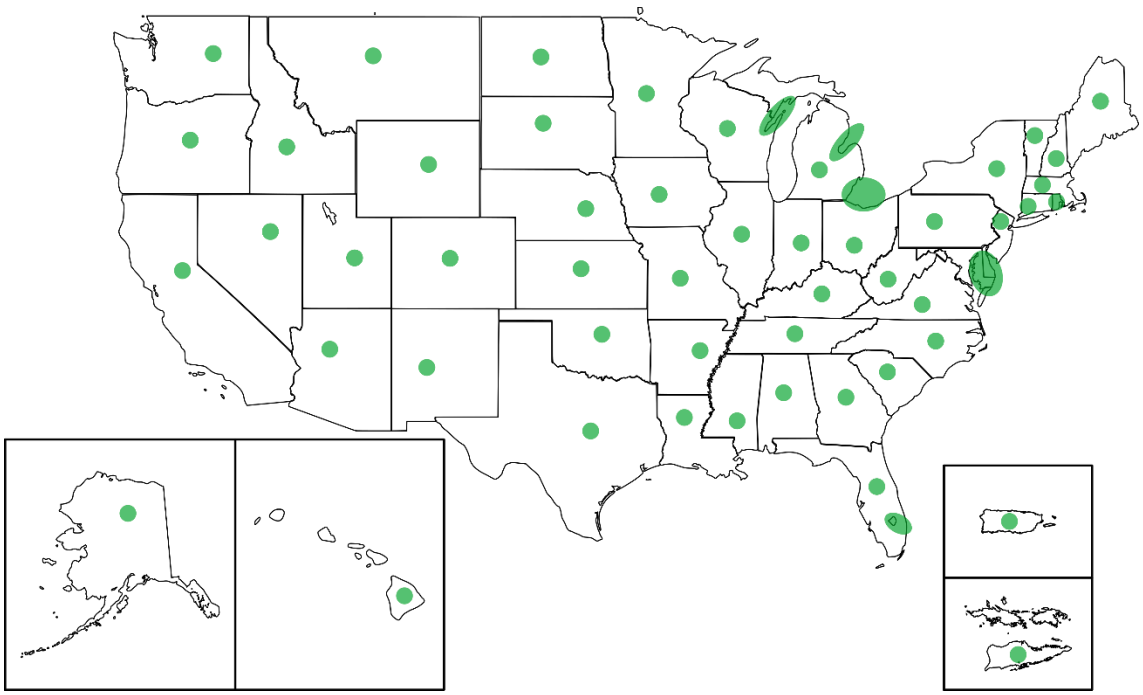
3.2.2 Sources and Occurrence in Surface Waters

Because they are a natural part of algal communities, cyanobacteria are commonly observed in freshwater systems. The occurrence of HABs has been documented in surface waters of all 50 states as well as U.S. territories between 2006 and 2015 as shown in Figure 3-4 (Richlen 2016; WHOI 2016). Figure 3-4 also identifies areas where more widespread HAB problems have occurred (e.g., parts of the Great Lakes).

Figure 3-5 shows the number of 2017 freshwater HAB recreational notices states publicly reported, organized by the EPA region between June 2 and August 1, 2017. To develop this regional summary map, the EPA researched and compiled publicly available reports posted on states' websites between these dates. During that time, states reported at least 281 notices for freshwater HABs with reported microcystin concentrations ranging from not detected (i.e., below the limit of detection) to 382 µg/L. These notices included cautions, warnings, public health advisories, and public health warnings due to

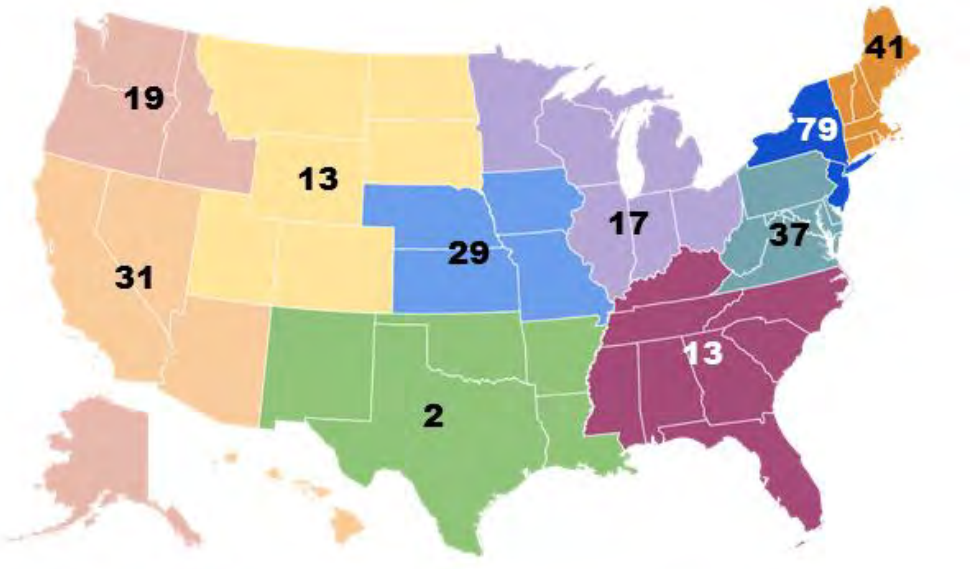
the presence of total cyanobacteria, cyanotoxins, or both. These notices can last for multiple days. The review was not exhaustive and might not reflect all the monitoring, beach, or general health advisories (e.g., some advisories at local or county-level may not be posted on the state website). Thus, the number of actual HAB notices during this time might be higher. In addition, many states have only recently begun to monitor HABs, so monitoring may be limited.

Figure 3-4. Generalized Distribution of Cyanobacterial HABs in the United States and Territories



[a] Graphic adapted from a Woods Hole Oceanographic Institute (WHOI) map of HABs that occurred between 2006 and 2015. It reflects input from HAB experts with broad experience in HAB events and reports to the U.S. National Office for Harmful Algal Blooms (Richlen 2016; WHOI 2016). Each state that has experienced one or more cyanobacterial HAB is indicated with a single green dot. Larger green ovals mark areas where more widespread cyanobacterial HAB problems occurred.

Figure 3-5. State-reported HAB Notices by EPA Region, June 2 to August 1, 2017



3.2.2.1 Microcystins

Microcystins are the most common cyanotoxins found worldwide and have been reported in surface waters in most of the states in the United States (Funari and Testai 2008; Loftin et al. 2016b; U.S. EPA 2009). Dry-weight concentrations of microcystins in surface freshwater toxigenic cyanobacterial blooms or surface freshwater samples reported worldwide between 1985 and 1996 ranged from 1 to 7,300 µg/g. Water concentrations of extracellular plus intracellular microcystins ranged from 0.04 to 25,000 µg/L. The remainder of this section provides examples of microcystin concentrations reported in ambient waters in the United States.

The EPA (U.S. EPA 2009) reported on the 2007 National Lakes Assessment (NLA), a national probability-based survey of the nation's lakes, ponds, and reservoirs. The NLA provided estimates of the condition of natural and man-made freshwater lakes, ponds, and reservoirs greater than 4 hectares (10 acres) and at least one meter deep. The 2007 NLA study surveyed 1,028 inland lakes and included measured microcystin concentrations, total cyanobacterial cell counts, and chlorophyll *a* concentrations. Microcystins were quantified using enzyme-linked immunosorbent assays (ELISA) with a detection limit of 0.1 µg/L (Loftin et al. 2016b). At each lake site, crews collected samples at a single station located at the deepest point in the lake and at ten stations around the lake perimeter. Due to the design of the survey, samples were taken at random and not necessarily where a bloom was occurring.

The 2007 NLA found that total cyanobacteria were detected in 98 percent of samples and were the dominant member of the phytoplankton community in 76 percent of samples (Loftin et al. 2016b; U.S. EPA 2009). Subsequent analysis indicated that potential microcystin-producing species occurred in 95 percent of samples (Loftin et al. 2016b). Microcystins were the most commonly detected class of cyanotoxins found in 32 percent of lakes in the contiguous United States (Loftin et al. 2016b; U.S. EPA 2009) and 39 percent of streams in the southeastern United States (Loftin et al. 2016a). Microcystins present in lakes ranged from the limit of detection (0.1 µg/L) to 225 µg/L with a mean concentration of 3.0 µg/L (detections only). Approximately 1.1 percent of lake samples exceeded 10 µg/L microcystins, and approximately 27 percent and 44 percent of lakes exceeded the WHO low-risk threshold for cyanobacterial abundance and chlorophyll *a*, respectively (Loftin et al. 2016b).

Lakes in states with microcystins levels > 10 µg/L reported in the 2007 NLA are shown in Table 3-4. The 2007 NLA data show two states (North Dakota and Nebraska) had nine percent of samples above 10 µg/L. Other states including Iowa, Texas, South Dakota, and Utah also had samples that exceeded 10 µg/L, but the frequency of detection was lower. Several of the 2007 NLA samples in North Dakota, Nebraska, and Ohio exceeded 20 µg/L (192, 225, and 78 µg/L, respectively).

In 2012, the EPA expanded on the 2007 NLA to include smaller water bodies in this statistically designed survey. Results represent the population of natural lakes, ponds, and reservoirs across the lower 48 states (not including the Great Lakes or the Great Salt Lake). To be included, in the survey lakes had to be larger than 2.47 acres (1 hectare), at least 3.3 feet (1 meter) deep, with a minimum quarter acre (0.1 hectare) of open water (U.S. EPA 2016). Data were collected from 1,038 lakes selected from a stratified random sample based on ecoregion, state, and surface area in the larger inference population (the set of 111,818 lakes). The NLA used thresholds established by the WHO to determine risk of exposure to cyanotoxins. Microcystins were detected in 39 percent of lakes monitored, but less than one percent exceeded the WHO estimates for microcystins at moderate or high risk of exposure. Less than one percent of lakes are in the most and moderately disturbed condition (i.e., have a high or moderate risk of exposure), and 99 percent are either least disturbed or show no detection of microcystins. Between 2007 and 2012, the percentage of lakes categorized as most disturbed for

microcystins did not change (U.S. EPA 2016), even though there was a significant increase in the detection of microcystins (+9.5 percent).

Table 3-4. States Surveyed as Part of the 2007 NLA with Water Body Microcystin Concentrations above 10 µg/L (U.S. EPA 2009)

State	Number of Sites Sampled	Percentage of Samples with Detection of Microcystins > 10 µg/L	Maximum Detection of Microcystins
North Dakota	38	9.1 percent	192 µg/L
Nebraska	42	9.1 percent	225 µg/L
South Dakota	40	4.9 percent	33 µg/L
Ohio	21	4.5 percent	78 µg/L*
Iowa	20	4.5 percent	38 µg/L*
Utah	26	3.6 percent	15 µg/L*
Texas	51	1.8 percent	28 µg/L*

*Single sample.

The NLA used total cyanobacterial cell counts as an indicator of water quality impacts of microcystins; 15 percent of lakes were classified in the most disturbed condition, 23 percent were classified as moderately disturbed, and 61 percent were classified as least disturbed. Between 2007 and 2012, there was a statistically significant increase (+8.3 percent) in the number of lakes in the most disturbed category for cyanobacterial cell counts. Lakes that were considered most disturbed exceeded the WHO recreational levels of concern (20 µg of microcystins/L).

A survey conducted during the spring and summer of both 1999 and 2000 in more than 50 lakes in New Hampshire found measurable microcystin concentrations in all samples (Haney and Ikawa 2000). Microcystins were analyzed by ELISA and were found in all the lakes sampled with a mean concentration of 0.1 µg/L.

A survey conducted in Florida in 1999 found potential microcystin-producing genera in water samples, including, *Microcystis* (43.1 percent), *Anabaena* (28.7 percent), *Planktothrix* (13.8 percent), *Aphanizomenon* (7.2 percent), and *Coelosphaerium* (3.6 percent) (Burns 2008). Although *Planktothrix* and *Aphanizomenon* were found less frequently than were the other genera, at times they accounted for a significant portion of the cyanobacterial community present. Microcystins were the most commonly found toxins in Florida waters, occurring in all samples analyzed containing cyanotoxins (Burns 2008).

In 2002, the Monitoring and Event Response to Harmful Algal Blooms in the Lower Great Lakes project evaluated the occurrence and distribution of cyanotoxins in the lower Great Lakes region (Boyer 2007). Analysis for total microcystins was performed using protein phosphatase inhibition assay. Microcystins were detected in at least 65 percent of the samples, mostly in Lake Erie, Lake Ontario, and Lake Champlain.

A 2004 study of the Great Lakes found high levels of cyanotoxins during the month of August (Makarewicz et al. 2006). Microcystin-LR was analyzed by protein phosphatase inhibition assay (limit of detection of 0.003 µg/L) and was detected at levels of 0.008 µg/L in the nearshore and 0.076 µg/L in the bays and rivers. This study reported higher levels of microcystin-LR (1.6 to 10.7 µg/L) in smaller lakes in the Lake Ontario watershed.

In 2005, Washington State Department of Ecology developed the Ecology Freshwater Algae Program to focus on the monitoring and management of cyanobacteria in Washington lakes, ponds, and streams (WSDE 2012). Microcystin levels ranged from the detection limit (0.05 µg/L) to 4,620 µg/L in 2008, to 18,700 µg/L in 2009, to 853 µg/L in 2010, and to 26,400 µg/L in 2011 (Hamel 2009, 2011, 2012).

In 2006, the U.S. Geological Survey (USGS) conducted a study of 23 lakes in the midwestern United States in which total cyanobacterial blooms were sampled to determine the co-occurrence of cyanotoxins in cyanobacterial blooms (Graham et al. 2010). This study reported that microcystins were detected in 91 percent of the lakes sampled with 17 percent of microcystin-positive samples exceeding 20 µg/L. The researchers also found that cylindrospermopsin co-occurred with microcystins in nine percent of samples (Graham et al. 2010). Mixtures of all the microcystin congeners measured (-LA, -LF, -LR, -LW, -LY, -RR, and -YR) were common. Microcystin-LR and -RR were the dominant congeners detected with mean concentrations of 104 and 910 µg/L, respectively.

The Ohio EPA (2012) has been monitoring inland lakes since 2007 for cyanotoxins. Of the Ohio lakes sampled during the 2007 NLA, 36 percent had detectable levels of microcystins. In 2010, the Ohio EPA sampled Grand Lake St. Marys for cylindrospermopsin, microcystins, and other cyanotoxins. Microcystin levels ranged from below the detection limit (< 0.15 µg/L) to more than 2,000 µg/L. Follow-up samples taken in 2011 for microcystins indicated concentrations exceeded 50 µg/L in August. During the same month, sampling in Lake Erie found microcystin levels exceeding 100 µg/L.

The USGS monitored Lake Houston in Texas from 2006 to 2008 and found microcystins in 16 percent of samples and at concentrations less than or equal to 0.2 µg/L (Beussink and Graham 2011). The USGS also did a study in the Upper Klamath Lake in Oregon in 2007 and detected total microcystin concentrations between 1 µg/L and 17 µg/L (VanderKooi et al. 2010). In 2011, the USGS conducted a study on the upstream reservoirs of the Kansas River to characterize the transport of cyanobacteria and associated compounds (Graham et al. 2012). Concentrations of total microcystins were low in the majority of the tributaries with the exception of Milford Lake, which had higher total microcystin concentrations, some exceeding the Kansas recreational guideline level of 20 µg/L. Upstream from Milford Lake, a cyanobacterial bloom was observed with a total microcystin concentration of 150,000 µg/L. When sampled a week later, total microcystin concentrations were less than 1 µg/L. The study authors indicated that this might be due to dispersion of microcystins through the water column or to other areas, or by degradation of microcystins via abiotic and biological processes. Samples taken during the same time from outflow waters contained total microcystin concentrations of 6.2 µg/L.

In 2008, the National Oceanic and Atmospheric Administration (NOAA) began monitoring for total cyanobacterial blooms in Lake Erie using high temporal resolution satellite imagery. Using the Great Lakes Coastal Forecast System, forecasts of bloom transport are created to estimate the trajectory of the bloom, which are distributed as bulletins to local managers, health departments, researchers, and other stakeholders. To evaluate bloom toxicity, the Great Lakes Environmental Research Laboratory collected samples at six to eight stations each week for 24 weeks, measuring cyanotoxin concentrations as well as chlorophyll biomass and an additional 18 parameters (e.g., nutrients) to improve future forecasts of these blooms. Microcystins were separated into particulate (cell-bound) and dissolved (extracellular) phases

(Graham and Jones 2007; Zastepa et al. 2014). In 2014, particulate microcystin concentrations ranged from below detection to 36.7 µg/L. Samples taken in 2015 and 2016 showed particulate microcystin concentration ranges from below detection to 9.19 µg/L and from below detection to 21.26 µg/L, respectively. Particulate microcystin concentrations peaked in August 2014 at all sites. Dissolved microcystin concentrations were also collected at each site in 2014 from September until the end of the sampling period in November, as well as during the field sampling seasons in 2015 and 2016. During the final months of sampling in 2014 (October to November), dissolved microcystin concentrations were detected with peak concentrations of 0.8 µg/L (mean: 0.28 +/- 0.2 µg/L) whereas particulate microcystin concentrations were below detection limits on many dates, indicating that a majority of the microcystins (mean: 72 percent +/- 37 percent) were in the dissolved form, as the bloom declined in intensity. Measured dissolved microcystin concentrations in the following two years ranged from levels below detection to peaks of 0.69 µg/L in September 2015 and 1.76 µg/L in July 2016 (NOAA 2014).

A 2014 survey of southeastern U.S. streams detected microcystins in 39 percent of the samples (29 of 75 sites) (Loftin et al. 2016a). The stream sample concentrations ranged from the minimum reporting limit of 0.1 µg/L to 3.2 µg/L. In some cases, the source of the cyanobacteria in flowing water bodies was traced to an upstream water body such as a lake or reservoir.

From August to October 2015, a bloom identified as *Microcystis aeruginosa* occurred on the Ohio River (ORSANCO 2017). Patches of the bloom covered 636 miles of the river and peaked in late September. The Ohio River Valley Water Sanitation Commission (ORSANCO) collected over 150 river samples, which were analyzed for microcystins. Of the samples collected by ORSANCO, 15 (10 percent) were greater than 6 µg/L. The highest microcystin concentration was 1900 µg/L from a sample collected at river mile 468.8 (Cincinnati, Ohio). No toxins were detected in finished drinking water (tested by utilities and state agencies). Ohio, West Virginia, Kentucky, and Indiana issued recreation notices for the Ohio River as the bloom extended into their areas. Illinois issued a precautionary statement concerning recreation in the river due to concern that the bloom would reach their border. These recreation advisories were lifted after the bloom ended (ORSANCO 2017).

From July 14 to September 14, 2016, an extensive cyanobacterial bloom covering 100 square miles occurred in Utah Lake, Jordan River, and nearby canals. Microcystin-LR concentrations ranged from below the detection limit to 0.23 µg/L, and the highest total microcystin concentration reported was 176 µg/L (Utah Department of Environmental Quality 2016). Both maximum values were from samples collected at the surface near an accumulation of cyanobacteria. Cyanobacteria composition observed during the 2016 bloom varied in both time and space, but was primarily dominated by *Aphanizomenon* or *Dolichospermum*. Other taxa including *Geitlerinema*, *Pseudanabaena*, and *Phormidium* were also observed in significant densities in a few samples (Utah Department of Environmental Quality 2016).

Lake Okeechobee, located north of the Everglades, is the largest freshwater lake in Florida. It is subject to agricultural runoff from adjacent cattle farms and sugar cane fields, which contribute to the formation of massive cyanobacterial blooms (Parker 2016). Water may be pumped out of the lake to the coast through the St. Lucie River and the Caloosahatchee River to prevent the lake level from rising too high after periods of heavy rain (Parker 2016). In July 2016, a 239-square mile cyanobacterial bloom in Lake Okeechobee was discharged and flowed through canals, rivers, and estuaries to the ocean. As a result of the microcystin levels in the river and at the coast, and the visible cyanobacterial scum in the lake and river, a state of emergency was declared in the counties of Martin, St. Lucie, Palm Beach, and Lee. From May 4 to August 4, 2016, the Florida Department of Environmental Protection took approximately 200 water samples from the St. Lucie River and estuary, Caloosahatchee River and estuary, Lake Okeechobee, Indian River Lagoon, and other nearshore marine locations (Florida Department of

Environmental Protection 2016). The microcystin concentrations in freshwater were reported in Lake Okeechobee (from not detected to 382.3 µg/L). Elevated levels were also reported in the St. Lucie River and the St. Lucie Canal (from not detected to 80.3 µg/L). Among the cyanobacteria species identified were *Microcystis aeruginosa*, *Scrippsiella trochoidea*, *Planktolyngbya limnetica*, *Dolichospermum circinalis*, and *Plectonema wollei* (Florida Department of Environmental Protection 2016).

3.2.2.2 Cylindrospermopsin

In general, fewer surface water occurrence data were available for cylindrospermopsin compared with microcystins. During blooms, testing for microcystins is much more common than is testing for cylindrospermopsin.

In a 1999 study, *Cylindrospermopsis* was detected in 40 percent of 167 water samples taken from 87 water bodies in Florida (Burns 2008). The actual cylindrospermopsin concentrations were not reported, but all samples containing the organism *Cylindrospermopsis* were positive for the toxin cylindrospermopsin.

In 2005, the U.S. Army Corps of Engineers detected cylindrospermopsin at a maximum concentration of 1.6 µg/L in lake water samples from Oklahoma (Lynch and Clyde 2009).

The USGS detected cylindrospermopsin in nine percent of blooms sampled during a 2006 USGS survey of 23 lakes in the midwestern United States (Graham et al. 2010). The low concentrations of cylindrospermopsin detected (0.12 to 0.14 µg/L) in the study occurred in bloom communities dominated by the genera *Aphanizomenon* or *Anabaena* and *Microcystis*.

The USGS analyzed the stored samples collected during the 2007 EPA NLA (U.S. EPA 2009) and detected cylindrospermopsin in four percent of samples, with a mean concentration 0.56 µg/L and a range from the limit of detection, 0.01 µg/L, to a maximum of 4.4 µg/L (Loftin et al. 2016b). Potential cylindrospermopsin-producing species (*Anabaena*, *Aphanizomenon*, *Cylindrospermopsis*, *Lyngbya*, and *Raphidiopsis*) occurred in 67 percent of samples (Loftin et al. 2016b). Cylindrospermopsins occurred most frequently in the midwestern and south-central United States and parts of Florida.

In Grand Lake in St. Marys, Ohio, cylindrospermopsin concentrations as high as 9 µg/L were reported in 2010 (Ohio EPA 2012).

3.2.3 Estuarine and Marine Waters

In Japan, the Isahaya Reservoir discharges water into Isahaya Bay. The reservoir experiences algal blooms seasonally, with species including nontoxic cyanobacteria as well a microcystin-producing *Microcystis aeruginosa* (Umehara et al. 2012). Water from the reservoir is discharged to the bay after rainfall events, even during periods of *Microcystis aeruginosa* blooms. Between November 2008 and November 2009, Umehara et al. (2012) estimated that 64.5 kilograms (kg) of microcystins were discharged to the bay, of which only 0.7 kg deposited on the floor. The authors speculated that because the majority of microcystins remain in the water, it is likely that they are washed out to other coastal areas with strong tides (Umehara et al. 2012).

In 2007, Miller et al. (2010) confirmed the presence of *Microcystis* and microcystins in Lake Pinto's downstream tributaries within 1 kilometer (km) of Monterey Bay in California after a large cyanobacterial bloom in the lake, and detected microcystins in nearshore marine waters following the

rainy season. The same researchers observed sea otters dying from consuming microcystin-contaminated clams, mussels, and oysters near ocean outflows of freshwater systems (Miller et al. 2010). A follow-up study was designed by Gibble and Kudela (2014) to identify the potential pathways leading to microcystin contamination in coastal ecosystems in and around Monterey Bay. They surveyed 21 sites at the land-sea interface in 2010–2011 followed by a survey of four watersheds in 2011–2013. In the first year of a three-year study, microcystins were detected in 15 of 21 freshwater, estuarine, and marine locations. In the two subsequent years, monitoring focused on four major watersheds that feed into Monterey Bay. The authors observed high microcystin concentrations in both autumn and spring seasons and concluded that microcystins are likely present throughout the year and transfer to the coastal environment, with the potential to be a persistent issue in the Monterey Bay area. The authors also correlated anthropogenic nutrient loadings with microcystins. Concentrations ranged from undetectable up to 20 ng/g resin, which translates to approximately 20 parts per billion (ppb) microcystins in the water column.

Otten et al. (2015) used microbial source tracking techniques to trace the source of a toxic *Microcystis* bloom in the Klamath River in Oregon to a single upstream reservoir. The use of assays targeting gene sequences for phycocyanin and microcystin synthase allowed the quantification of total and toxigenic *Microcystis*. Their results showed that large quantities of cyanobacterial cells could withstand passage through hydroelectric installations and transport over 300 km. Microcystin concentrations ranged from 165 µg/L in a reservoir upstream to 3.6 µg/L within the lower estuary less than 1 km from the Pacific Ocean (Otten et al. 2015).

The large cyanobacterial bloom in Lake Okeechobee, Florida, in 2016 (described above) flowed downstream and impacted estuarine and marine waters, resulting in beach closures along the Atlantic (Chaney 2016; Florida Department of Environmental Protection 2016). From May 4 to August 4, 2016, the Florida Department of Environmental Protection sampled freshwater, estuarine waters, and nearshore marine waters. The highest concentration reported (414.3 µg/L) was collected in Martin County at Bathtub Reef, a beach along the Atlantic Ocean. Sampling efforts in estuarine water, for example at a marina in the St. Lucie River, reported a concentration of 78 µg/L. The majority of marine waters sampled had low levels of microcystins (not detected or approximately 1 µg/L).

3.2.4 Other Sources of Microcystins and Cylindrospermopsin

Cyanotoxins have the potential to occur in drinking water, ground water, fish, shellfish, dietary supplements, air, soil, and sediments. These potential sources of cyanotoxins are discussed briefly in section 7.6. Exposure to these toxins in finished drinking water is also characterized in the Drinking Water Health Advisories (U.S. EPA 2015a, 2015b).

3.3 Environmental Fate

Different physical and chemical processes are involved in the persistence, breakdown, and movement of microcystins and cylindrospermopsin in aquatic systems as described below.

3.3.1 Mobility

Microcystins may adsorb onto naturally suspended solids and dried crusts of cyanobacteria. They can precipitate out of the water column and reside in sediments for months (Falconer 1998; Han et al. 2012). A study conducted by the USGS and the University of Central Florida determined that microcystin-LR and cylindrospermopsin did not sorb in sandy aquifers and were transported along with ground water

(O'Reilly et al. 2011). The authors suggested that the removal of microcystin-LR was due to biodegradation.

Cyanotoxins that are produced by cyanobacteria growing in freshwaters can enter estuarine and marine waters as waters containing the toxins flow downstream. Studies have demonstrated that toxigenic cyanobacteria can travel long distances in freshwater and can reach estuarine and marine waters from coastal lakes, reservoirs, and rivers (Preece et al. 2017).

In sediments, cylindrospermopsin exhibits some adsorption to organic carbon, with little adsorption observed on sandy and silt sediments (Klitzke et al. 2011). The low adsorption of cylindrospermopsin reduces its residence time in sediments, thus reducing the opportunity for microbial degradation.

3.3.2 Persistence

3.3.2.1 Microcystins

Microcystins are relatively stable and resistant to chemical hydrolysis or oxidation at or near neutral pH. Elevated or low pH or temperatures above 30°C may cause slow hydrolysis. Microcystins have been observed to persist for 21 days to two to three months in solution and up to six months in dry scum (Funari and Testai 2008; Rapala et al. 2006). Environmental conditions such as temperature, pH, presence of light, salinity, and presence of certain aquatic bacteria can influence the rate of microcystin degradation (Schmidt et al. 2014). Microcystins can persist even after a cyanobacterial bloom is no longer visible (Lahti et al. 1997b; Zastepa et al. 2014). In a study by Zastepa et al. (2014), dissolved microcystin-LA was present at a concentration of 20 μg/L or greater for 9.5 weeks even though the *Microcystis* bloom was not visible after five weeks.

In the presence of full sunlight, microcystins undergo photochemical breakdown, but this varies by microcystin congener (Chorus et al. 2000; WHO 1999). Zastepa et al. (2014) suggest that microcystin-LA degrades at a slower rate than microcystin-LR, -RR, and -YR congeners. The presence of water-soluble cyanobacterial cell pigments, in particular phycobiliproteins, enhances this breakdown. Breakdown can occur in as few as two weeks to longer than six weeks, depending on the concentration of pigment and the intensity of the light (Tsuji et al. 1994, 1995).

Several other factors, including pH, wavelength of light (Schmidt et al. 2014), and whether microcystins are dissolved or present in particulate matter (Lahti et al. 1997b) can affect the rate of transformation or photodegradation. According to Tsuji et al. (1994, 1995), microcystin-LR was photodegraded with a half-life of about five days in the presence of 5 mg/L of extractable cyanobacterial pigment. Humic substances can act as photosensitizers and can increase the rate of microcystins breakdown in sunlight. Others have found that high concentrations of humic acids can slow the rate of microcystins transformation by sunlight (Schmidt et al. 2014). In deeper or turbid water, the breakdown rate is slower. Welker and Steinberg (2000) estimated the maximum rate of microcystin-LR degradation in the presence of humic substance photosensitizers. Extrapolating results from their small experimental tubes to a water column of 1 meter, Schmidt et al. (2014) estimated the half-life of microcystin-LR to be 90 to 120 days per meter of water depth in surface waters. The researchers demonstrated that the wavelength of light can also affect degradation rates; complete microcystins degradation was observed within one hour when exposed to 254-nm light and within five days using 365-nm light. According to Lahti et al. (1997b), microcystin-LR follows first-order decay kinetics, with a decimal reduction time of 30 days for dissolved microcystins compared with 15 days for microcystins found in particulate matter. Zastepa et

al. (2014) also found that dissolved microcystin-LA persists longer than microcystin-LA in particulates, with in situ half-lives of 15.8 days and 6.5 days, respectively.

Microcystins are susceptible to biodegradation by aquatic bacteria found naturally in surface waters (Jones et al. 1994). Bacteria isolates of *Arthrobacter, Brevibacterium, Rhodococcus, Paucibacter*, and various strains of the genus *Sphingomonas* (*Pseudomonas*) have been reported to be capable of degrading microcystin-LR (de la Cruz et al. 2011; Han et al. 2012). These degradative bacteria have also been found in sewage effluent (Lam et al. 1995), lake water (Cousins et al. 1996; Jones et al. 1994; Lahti et al. 1997b), and lake sediment (Lahti et al. 1997a; Rapala et al. 1994; U.S. EPA 2015a). Lam et al. (1995) reported that the biotransformation of microcystin-LR followed a first-order decay with a half-life of 0.2 to 3.6 days. In a study conducted by Jones et al. (1994) with microcystin-LR in different natural surface waters, microcystin-LR persisted for three days to three weeks; however, more than 95 percent loss occurred within three to four days. A study by Christoffersen et al. (2002) measured half-lives in the laboratory and in the field of approximately one day, driven largely by bacterial aerobic metabolism. These researchers found that approximately 90 percent of the initial amount of microcystins disappeared from the water phase within five days, irrespective of the starting concentration. Other researchers (Edwards et al. 2008) have reported half-lives of four to 14 days, with longer half-lives associated with a flowing stream and shorter half-lives associated with lakes. Microcystin-LR degradation by *Sphingopyxis* species was observed with an optimal degradation rate at pH values between 6.5 and 8.5 (Schmidt et al. 2014). Several studies have demonstrated bacterial degradation of microcystin-LR, but other congeners, such as microcystin-LF or -LA, were not significantly degraded (Zastepa et al. 2014). During periods of high toxigenic cyanobacterial densities, the composition of other bacteria in the community may shift in response. In a study of the San Juan reservoir in Spain, Lezcano et al. (2017) found that several classes, orders, and families of known biodegrading bacteria, such as the *Spirobacillales* order, increased by more than a factor of 1.5 during the peak of a cyanobacterial bloom. The increase in relative abundance suggests that these biodegraders may play a role in microcystins degradation in the environment. Although microcystin-degrading bacteria might be present, initial degradation rates could be slow because the bacteria need time to begin using the toxins as carbon or energy sources (Hyenstrand et al. 2003). Microcystins can accumulate in the water column if these biodegrading bacteria are not present at the time of a toxic bloom (Schmidt et al. 2014). Cousins et al. (1996) demonstrated that microcystin experimentally added into reservoir water has a half-life of three to four days, whereas microcystin spiked into the same matrix but sterilized (so biodegrading bacteria are dead) had no significant change in the 12 days of the experiment. The authors concluded that biodegradation was the primary mechanism of microcystin reductions in the raw reservoir water.

Where rivers discharge to the ocean, freshwater cyanobacteria, cyanotoxins, or both can enter the marine environment (Andersen et al. 1993; Miller et al. 2010). Miller et al. (2010) confirmed the transfer of freshwater microcystins to the marine environment; the researchers found that after introducing *Microcystis* cyanobacteria to a saline environment, cyanobacteria can survive for 48 hours before lysing and releasing microcystins. Microcystin concentrations in these experiments decreased in the range of 44 to 71 percent after one hour in the saline environment, but continued to be detected in the seawater for at least 21 days, based on a detection limit of 0.02 µg/L (Miller et al. 2010).

3.3.2.2 Cylindrospermopsin

Cylindrospermopsin is relatively stable in the dark and at temperatures from 4°C to 50°C for up to five weeks (ILS 2000). Cylindrospermopsin is also resistant to changes in pH and remains stable for up to eight weeks at pH 4, 7, and 10. In the absence of cyanobacterial cell pigments, cylindrospermopsin tends

to be relatively stable in sunlight, with a half-life of 11 to 15 days in surface waters (Funari and Testai 2008).

Like microcystins, degradation of cylindrospermopsin increases in the presence of cell pigments such as chlorophyll *a* and phycocyanin, a blue photosynthetic pigment found in cyanobacteria. When exposed to both sunlight and cell pigments, cylindrospermopsin breaks down rapidly—more than 90 percent within two to three days (Chiswell et al. 1999).

Bacteria have been shown to decompose cylindrospermopsin in laboratory studies; the biodegradation is influenced by the cyanotoxin concentration, temperature, and pH. Mohamed and Alamri (2012) reported that *Bacillus* bacteria degraded cylindrospermopsin and that degradation occurred in six days at the highest toxin concentration (300 µg/L) and in seven or eight days at lower concentrations (10 and 100 µg/L, respectively). The biodegradation rate was also reported to depend on temperature and pH, with the highest rates occurring in warm waters (25 and 30°C) and neutral to slightly alkaline conditions (pH 7 and 8). Klitzke and Fastner (2012) confirmed the observations of Mohamed and Alamri (2012), noting that a decrease in temperature from 20 to 10°C slowed down degradation by a factor of 10. They also found that degradation slowed significantly under anaerobic conditions, with half-lives of 2.4 days under aerobic conditions and 23.6 days under anaerobic conditions.

3.4 Toxicokinetics

Limited data are available regarding the toxicokinetics of microcystins in environmental exposure conditions (U.S. EPA 2015d). Available intestinal data indicate that the organic anion transporting polypeptide (OATP) family transporters facilitate the absorption of microcystins from the intestinal tract into liver, brain, and other tissues, as well as their export out of organs and tissues (Cheng et al. 2005; Fischer et al. 2005; Svoboda et al. 2011). However, bile acids and other physiologically relevant substrates compete with microcystins for transporter uptake by the liver (Thompson and Pace 1992); reduction or elimination of liver toxicity has been observed during in vivo or in vitro exposures when microcystin uptake by OATP transporters is limited or inhibited (Hermansky et al. 1990a, 1990b; Runnegar et al. 1995; Runnegar and Falconer 1982; Runnegar et al. 1981). Both in vivo and in vitro studies have shown biliary excretion of microcystins (Falconer et al. 1986; Pace et al. 1991; Robinson et al. 1991), possibly via conjugation with cysteine and glutathione (Kondo et al. 1996). Additional details of microcystin toxicokinetics can be found in the EPA's Drinking Water Health Advisory and HESD for microcystins (U.S. EPA 2015a, 2015d).

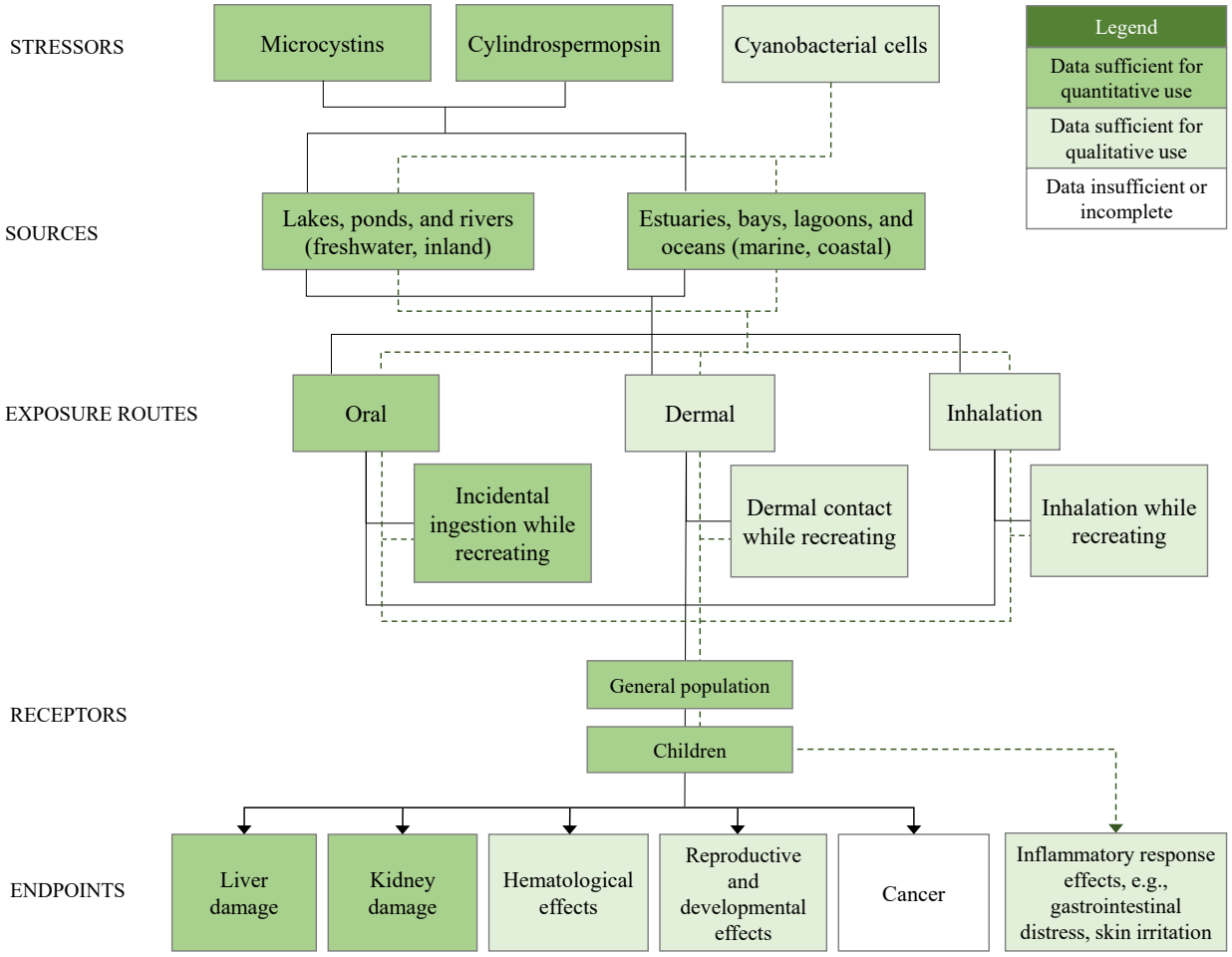
Limited toxicokinetic data for cylindrospermopsin are available and are derived from mice intraperitoneal studies and in vivo studies that do not necessarily reflect environmental exposure conditions (U.S. EPA 2015c; Pichardo et al. 2017). Cylindrospermopsin is absorbed from the GI tract (Humpage and Falconer 2003; Shaw et al. 2001; Shaw et al. 2000) and is distributed primarily to the liver but also to the kidneys and spleen (Norris et al. 2001). The metabolism and toxicity of cylindrospermopsin is mediated by hepatic cytochrome P450 (CYP450) enzymes, and the periacinar region of the liver appears to be the main target of toxicity where cylindrospermopsin and its metabolites bind to proteins (Norris et al. 2001; Runnegar et al. 1995; Shaw et al. 2001; Shaw et al. 2000). Elimination of cylindrospermopsin was continuous over a monitoring period of 24 hours, with a large mean total recovery primarily from urine, and to a smaller extent, feces, after 24 hours (Norris et al. 2001). Additional details of cylindrospermopsin toxicokinetics can be found in the EPA's Drinking Water Health Advisory and HESD for cylindrospermopsin (U.S. EPA 2015b, 2015c).

4.0 PROBLEM FORMULATION

4.1 Conceptual Model

This conceptual model provides useful information that characterizes and communicates the potential health risks related to exposure to microcystins and cylindrospermopsin in recreational waters. The model depicts the sources of the cyanotoxins in these waters, the recreational routes of exposure for sensitive biological receptors of concern, and the potential assessment endpoints (e.g., effects such as kidney and liver toxicity) (Figure 4-1).

Figure 4-1. Conceptual Model of Exposure Pathways to the Cyanotoxins, Microcystins and Cylindrospermopsin, and Cyanobacteria in Surface Waters While Recreating



4.1.1 Conceptual Model Diagram for Recreational Exposure

The conceptual model is intended to explore potential links of exposure to a contaminant or stressor with the adverse effects and toxicological endpoints important for management goals, including the development of recreational AWQC. Boxes that are shaded darker green indicate pathways that the EPA considered quantitatively in estimating the advisory level, whereas boxes shaded lighter green indicate data were sufficient for qualitative use and the white boxes did not have sufficient data for the EPA to

evaluate quantitatively or qualitatively. The solid lines are for the cyanotoxins and the dotted lines are for the cyanobacterial cells.

4.1.2 Factors Considered in the Conceptual Model for Microcystins and Cylindrospermopsin

Stressors

The stressors are microcystins and cylindrospermopsin concentrations in water. These toxins can be produced by cyanobacteria occurring in freshwater. The EPA concluded that although statistically significant associations with adverse health effects occur across a wide range of cyanobacterial cell densities, criteria cannot be derived based on cyanobacterial cell density at this time. Effects related to cyanobacterial cells are discussed in section 7.5.1 and Appendix D.

Sources

Cyanobacteria occur naturally in surface waters, such as lakes, ponds, rivers, estuaries, bays, lagoons, and oceans in or surrounding the United States. Some genera of the cyanobacteria, including *Microcystis*, *Cylindrospermopsis*, *Anabaena*, *Planktothrix*, and *Nostoc*, can produce the cyanotoxins microcystins and cylindrospermopsin. This assessment focuses on cyanotoxins produced by these cyanobacteria in freshwater. These toxins have the potential to affect downstream waters, including coastal areas where surface water containing the toxins discharges into estuarine and marine waters.

Routes of Exposure

Exposure to cyanotoxins from recreational water sources can occur via oral exposure (incidental ingestion while recreating); dermal exposure (contact of exposed parts of the body with water containing cyanotoxins during recreational activities such as swimming, wading, or water skiing); and inhalation exposure to contaminated aerosols (while recreating). The route of exposure considered quantitatively in this assessment is the oral exposure to microcystins and cylindrospermopsin via incidental ingestion while swimming. Inhalation can occur from exposures from personal watercraft and boat spray. Dermal exposure can occur through recreational water contact; however, significant dermal absorption of microcystins and cylindrospermopsin is not expected due to the large size and charged nature of these molecules and the lack of dermal receptor sites capable of uptake (Butler et al. 2012; U.S. EPA 2004; U.S. EPA 2007). Sufficient data to quantify toxicity via the inhalation and dermal exposure routes were not available. The dermal and inhalation routes of exposure are discussed further in the Effects Characterization section (7.4).

Receptors

Anyone who recreates in a water body where cyanotoxins are present could be exposed to cyanotoxins through ingestion, dermal contact, and inhalation of aerosols while recreating in contaminated surface waters. Recreating children can be at greater risk from exposure to microcystins or cylindrospermopsin because they have smaller body mass compared to adults, they spend more time in contact with the water compared to adults, and they incidentally ingest more water than adults while recreating. Therefore, the EPA has determined that childhood is the most vulnerable lifestage due to potential increased exposure while recreating when compared with adults. The EPA evaluates and discusses differences between lifestages in the Effects Characterization section (7.3).

While there are examples in the literature and reports of animal poisonings and death due to exposure of cyanotoxins, values protective of animals such as dogs and livestock are not generated in this document. However, section 7.8 discusses some animal-specific issues, including a summary of guidelines that several states have developed for animals.

Endpoints

Available microcystin toxicity data indicate that the primary target organ for microcystins is the liver as described in the EPA's HESD for microcystins (U.S. EPA 2015d).

Available cylindrospermopsin toxicity data are described in the EPA's HESD for cylindrospermopsin (U.S. EPA 2015c). For cylindrospermopsin, the EPA selected kidney effects as the endpoint on which to quantify the measure of effect. However, in both the critical study and the supporting studies there is evidence that cylindrospermopsin can also alter the shape of red blood cells.

Clinical, epidemiological, and outbreak study results (see Appendix D) suggest a link between an increase in adverse inflammatory symptoms among recreators and elevated cyanobacterial cell densities. However, there is considerable uncertainty and variability associated with the epidemiological results, which did not identify consistent effects at similar cyanobacterial densities. Specifically, significant associations occur across a wide range of cell densities; associations vary with different self-reported health endpoints or combined symptom categories. Potential inflammatory health effects related to exposure to total cyanobacterial cells are described in the Effects Characterization section (7.5.1) and in Appendix D, both of which include a discussion of the uncertainties related to associations with cyanobacterial cells.

4.2 Analysis Plan

The EPA's 2000 *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (2000 Human Health Methodology) outlines the Agency's process for deriving AWQC and guides the development of these recreational criteria and swimming advisory recommendations (U.S. EPA 2000).

The 2000 Human Health Methodology includes identifying the population subgroup that should be protected and evaluating cancer and non-cancer endpoints, measures of effect, measures of exposure, and relative source contribution (RSC). In this analysis plan, the EPA describes: (1) the RfD previously derived for microcystins and cylindrospermopsin (measure of effect); (2) the calculation for the recreational criteria; (3) incidental ingestion exposure in terms of volume ingested, duration of exposure, and body weight (measure of exposure) described in the EPA's *Exposure Factors Handbook* (EFH) and data reported in the peer-reviewed scientific literature; and (4) discusses the RSC. These criteria focus on human exposure as a result of primary contact recreation activities, such as swimming, during which immersion and incidental ingestion of ambient water are likely.

The EPA's HESD for microcystins and HESD for cylindrospermopsin (U.S. EPA 2015c, 2015d) provide the health effects basis for the development of the Drinking Water Health Advisories for microcystins and cylindrospermopsin (U.S. EPA 2015a, 2015b), including the basis for estimating the point of departure. To develop its HESDs for microcystins and cylindrospermopsin, the EPA assembled available information on toxicokinetics, acute, short-term, subchronic, and chronic toxicity along with developmental and reproductive toxicity, neurotoxicity, immunotoxicity, genotoxicity, and cancer in

humans and animals. For detailed descriptions of the literature search strategies, see the EPA's HESDs for microcystins and cylindrospermopsin (U.S. EPA 2015c, 2015d).

The EPA's HESDs were subject to rigorous internal and external peer review before being finalized in 2015. The information evaluated for these documents also supports the development of the recreational criteria and swimming advisories for microcystins and cylindrospermopsin, which evaluate exposure via recreational water ingestion. The EPA conducted supplemental literature searches to capture new references, including effects related to recreational exposure to cells. For detailed information on the search terms, see Appendix C.

4.2.1 Approach for Recreational AWQC and Swimming Advisory Derivation

The recreational AWQC and swimming advisory recommendations for microcystins and cylindrospermopsin are calculated as described in the 2000 Human Health Methodology and presented in the equation below:

$$\text{Recreational AWQC } (\mu g/L) = \text{RfD} \times \frac{\text{BW}}{\text{IR}}$$

Where:

RfD = reference dose (μg/kg body weight/day)

BW = mean body weight (kg)

IR = ingestion rate (L/day) (discussed in section 4.2.3.1)

4.2.1.1 Magnitude, Duration, and Frequency

Recreational criteria, like other 304(a) criteria, consist of a magnitude, duration, and frequency. Magnitude is the numeric expression of the maximum amount of the contaminant that may be present in a water body that supports the designated use. Duration is the period over which the magnitude is calculated. Frequency of excursion describes the number of times the contaminant may be present above the magnitude over the specified period (duration). A criterion is derived such that the combination of magnitude, duration, and frequency protect the designated use (e.g., primary contact recreation).

4.2.2 Measures of Effect

The EPA's HESDs for microcystins and cylindrospermopsin (U.S. EPA 2015c, 2015d), provide the health effects basis for development of an oral toxicity value or the RfD, including the selection of the critical study and critical endpoints and application of uncertainty factors (UFs). In derivation of the recreational criteria and swimming advisory recommendations, the EPA uses these toxicity values as the measure of effect for oral exposure through incidental ingestion while recreating. The RfDs described in the EPA's HESDs are based on short-term and subchronic studies and therefore are an estimate (with uncertainties spanning perhaps an order of magnitude) of the daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a short-term exposure period.

4.2.3 Measures of Exposure

The EPA selected incidental ingestion during primary contact activities (such as swimming) in derivation of the recreational criteria and swimming advisories because data suggest that incidental ingestion can be considered the highest potential exposure pathway for cyanotoxins while recreating. Dorevitch et al. (2011) studied the volume of water ingested during a range of recreational activities in the Chicago Area Waterway System (CAWS) and at a public outdoor swimming pool. Study participants took part in one of the following activities on the CAWS: canoeing, fishing, kayaking, motor boating, or rowing. In the swimming pool, participants took part in canoeing, fishing, kayaking, swimming, or wading/splashing. The results indicate that the odds of ingesting a teaspoon or more of water are significantly higher among swimmers than among those who just immersed their head in a swimming pool or those who participated in the other, more limited contact activities on surface waters. Therefore, the EPA determined that using a swimmer scenario for exposure as the basis for the criteria is protective of these other aquatic activities.

Inhalation exposure occurs during swimming; however, data are not sufficient to quantify health effects resulting from inhalation exposure to cyanotoxins at this time. See section 7.4.1 for a characterization of potential effects from inhalation exposure.

Dermal exposure happens during swimming; however, significant dermal absorption of the toxins microcystins and cylindrospermopsin is not expected due to the large size and charged nature of these molecules (Butler et al. 2012; U.S. EPA 2004; U.S. EPA 2007). Because available data are not sufficient, the EPA is not quantifying effects resulting from dermal exposure to cyanotoxins. See section 7.4.2 for a characterization of dermal exposure to these cyanotoxins.

Dermal exposure to cyanobacterial cells can also result in adverse health effects, such skin rashes, eye irritation, and ear irritation. Because adequate effects data are not available, the EPA is not quantifying effects resulting from exposure to cells at this time; effects are described qualitatively. Available epidemiological study results do not provide consistent associations between cell densities and the inflammatory health endpoints. See section 7.5.1 for a characterization of potential effects from recreational exposure to cyanobacterial cells.

All recreational exposure studies that included both children and adults found that age tended to influence incidental ingestion exposure while recreating. More specifically, children tend to ingest more water and spend more time in the water compared with adults (Dufour et al. 2017; Dufour et al. 2006; Schets et al. 2011; U.S. EPA 2011). Data supporting the selected exposure factors are described in the sections that follow.

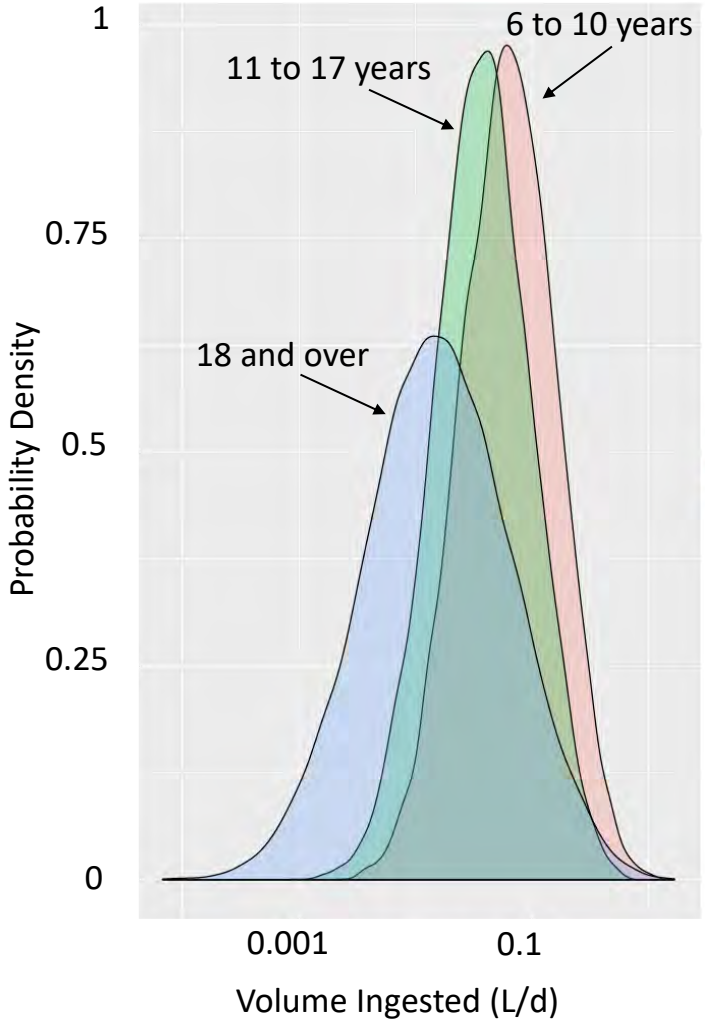
The measure of exposure is the 90th percentile of the daily incidental ingestion rate (volume of surface water incidentally ingested per day) and body weight (kg). Both body weight and incidental ingestion while recreating are parameters that vary with age. The EPA's 2000 Human Health Methodology (2000) outlines EPA's process for deriving AWQC and guides the development of these recreational criteria and swimming advisory recommendations.

4.2.3.1 Incidental Ingestion

To calculate the recreational incidental ingestion rate in units of volume per day, the EPA combined a distribution of incidental ingestion volumes (volume per event normalized to volume per hour) and a distribution of exposure durations (hours per day). The EPA uses the 90th percentile of the combined

distribution of ingestion rate and exposure duration to represent incidental ingestion per day, consistent with the EPA's Human Health Methodology (U.S. EPA 2000). Probability density plots of the combined distributions are shown in Figure 4-2. The ingestion data demonstrate that the mean ingestion rate for children six to 10 years is higher than for older children and adults. These data are discussed in the following sections.

Figure 4-2. Combined Distributions for Age Groups

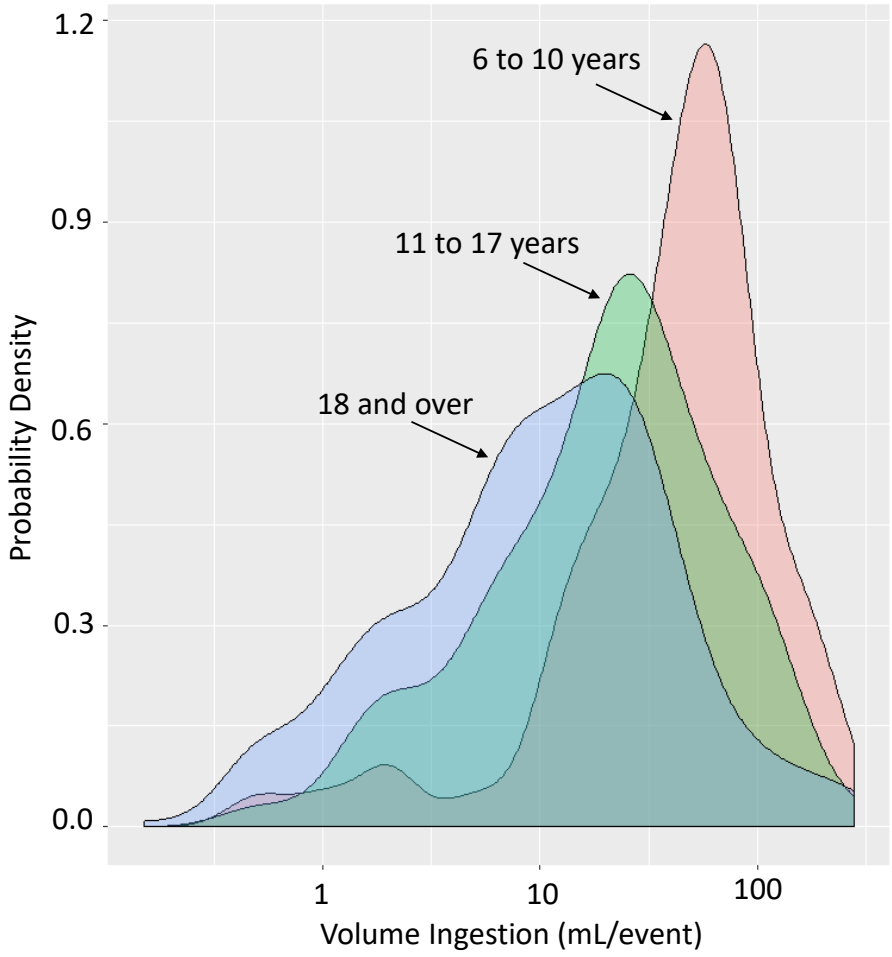


Ingestion Volume Studies

The EPA evaluated seven studies on ingestion and selected the dataset collected and analyzed by Dufour et al. (2017) for development of these AWQC or swimming advisory recommendations. This study used the same methodology as an earlier study (Dufour et al. 2006) but included 10 times more participants. Both studies used cyanuric acid as an indicator of amount of pool water ingested while swimming in an outdoor pool. Pool water samples were collected before the start of swimming activities, and participants' urine was collected for 24 hours after the swimming event ended; pool water and urine samples were analyzed for cyanuric acid. The dataset collected by Dufour et al. (2017) included age information for each particpant ages six to 81 years, whereas the 2006 study classified individuals as over or under 18 years old. Both studies did not include children younger than six years old. The 2017

study recorded time spent in the water for each participant. The 2017 study results highlighted that younger children tested ingested more than older children or adults. The EPA selected the Dufour et al. (2017) dataset to calculate incidental ingestion volume because of the larger number of particpants, the inclusion of additional age groups, and recording of the duration exposure of each participant. The raw data collected and analyzed by Dufour et al. (2017) was provided by the study authors (U.S. EPA 2018a). The EPA adjusted (i.e., normalized) the volume ingested by each participant to one hour based on the length of time that participant reported being in the water. The summary statistics the EPA calculated using this dataset are shown in Appendix E (Table E-1). Figure 4-3 shows the raw data density plots for the Appendix E Dufour data separately grouped as age groups six to 10, 11 to 17, and 18 years and over. The density plots show the volume of incidental ingestion (mL) per recreational event on a log scale. To develop the distribution, each participant's volume ingested was adjusted to one hour based on the length of time that participant reported being in the water. Incidental ingestion was recorded for 66 individuals in the six- to 10-year category.

Figure 4-3. Incidential Ingestion for Age Groups Based on Appendix E Dufour Data



Appendix F describes seven studies that reported incidental ingestion while recreating, but only three others reported ingestion estimates for children (Dufour et al. 2006; Schets et al. 2011; Suppes et al. 2014). These other studies reported children's ingestion volumes similar to Dufour et al. (2017). Although these other studies corroborate the Dufour et al. (2017) findings, they were not selected for deriving the ingestion rate. Dufour et al. (2006) had fewer age groups (i.e., six to 17 and 18+ years), smaller sample size, and did not record time spent in water for each participant, making it a less robust

study than Dufour et al. (2017). Schets et al. (2011) collected data in the Netherlands, which may not be representative of the United States due to different behavioral trends in the resident population, including effects of temperature on recreating patterns. In addition, Schets et al. (2011) ingestion volumes are based on self-reported estimates; parents estimated volumes for children five and younger. Self- and parent-reported estimates are more uncertain than the methods used by Dufour et al. (2017). Suppes et al. (2014) used video and urine analysis to estimate ingestion volume. In Suppes et al. (2014) quantitative data were available for 35 participants, which is much lower than the sample size for Dufour et al. (2017). In addition, Suppes et al. (2014) only reported two age groups, children (five to 17 years) and adults (18+ years), which does not allow for the finer discernment of exposure patterns that is possible with the Appendix E and U.S. EPA (2018a) data.

Appendix F also describes the methodology used by the EPA's Office of Pesticide Programs (OPP) to calculate exposures to pool chemicals during swimming to support registration decisions. The Swimmers Exposure Assessment Model (SWIMODEL) (U.S. EPA 2003) uses incidental ingestion values for children that are twice the values used for noncompetitive adult swimmers. The model assumes an incidental ingestion rate of 0.050 L/hour for children ages seven to 10 years and 11 to 14 years while swimming noncompetitively. Incidental ingestion rates among adults while swimming competitively and noncompetitively are 0.0125 L/hour and 0.025 L/hour, respectively.

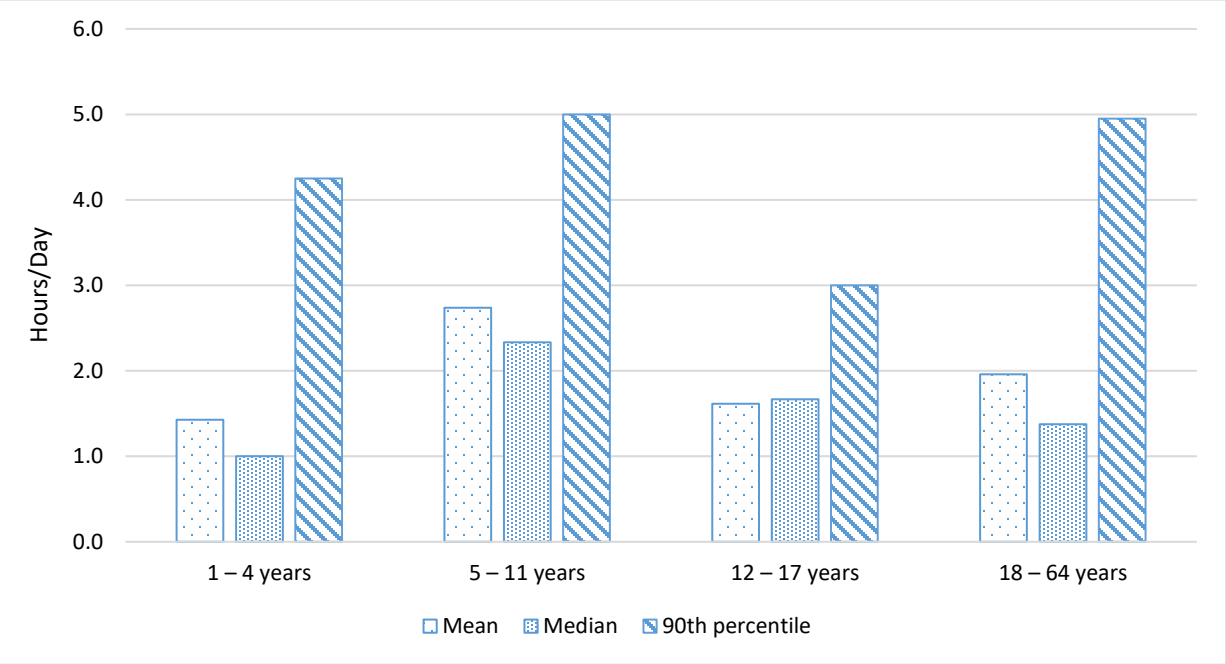
Duration of Recreational Exposure

Duration of recreational exposure quantifies the length of time people might be exposed to cyanotoxins during their primary contact recreational use. Duration is needed to convert recreational ingestion rates in units of volume per hour to an amount incidentally ingested per day, which is the exposure parameter needed to derive the recommended cyanotoxin values.

The EPA selected recreational exposure data from the EFH (U.S. EPA 2011) for the development of these criteria/swimming advisories. The EPA's EFH (2011) lists time spent per 24 hours in an outdoor spa or pool for different age groups. The data are based on analysis of the National Human Activity Pattern Survey (U.S. EPA 1996). Figure 4-4 compares point estimates for the recreational duration data for different age groups and shows that recreators ages five to 11 years (n = 15) tend to spend more time in the water than other child age groups and adults. A duration was not provided for children younger than age one year.

The EPA investigated available exposure parameters for children younger than six years old, but they have large uncertainties given the lack of measured incidental ingestion data for this age group (see section 7.3.2). See section 7.2 (Recreational Exposure Duration) for further discussion of the available data for recreational exposure duration. The EPA used the distribution of exposure durations for children ages five to 11 years (n = 15; units are hour/day) as described below to calculate incidental ingestion per day.

Figure 4-4. Direct Contact Recreational Exposure Duration by Age Group, Based on Table 16-20 in U.S. EPA (2011)[a]



[a] This figure shows a comparison of point estimates. The EPA used the whole distribution for ages five to 11 years in deriving the AWQC and swimming advisory magnitudes.

Determination of Incidental Ingestion per Day

The incidental ingestion volume per day the EPA used to calculate the AWQC or swimming advisories is the product of the distribution of children's incidental ingestion rate for children ages six to 10 years (Appendix E; U.S. EPA 2018a) and the distribution of exposure durations for children ages five to 11 years (U.S. EPA 2011). The lifestage grouping for the duration data include children one year older and one year younger than the lifestage group for the incidental ingestion data.

The individual ingestion rate data points (adjusted to L/hour) were used to calculate a mean and standard deviation of the log-normal transformed dataset. This distribution was combined with the distribution of hours of recreation per day (ages five to 11 years) from the 2011 EFH (Table 16-20 *Time Spent (minutes/day) in Selected Outdoor Locations, Doers Only, At Home in the Outdoor Pool or Spa*). The mathematical relationship between the two variables and the daily incidental ingestion rate is shown in this equation:

$$\text{Ingestion Volume (L/hour)} \times \text{Recreation Duration (hour/day)} = \text{Daily Incidental Ingestion Rate (L/day)}$$

The EPA used probabilistic (Monte Carlo) simulation to develop the combined distribution of these variables as follows:

- Estimated statistical distributions for hourly ingestion rate and recreation duration for different age groups.
- Sampled randomly one value from each of these distributions.

- Multiplied the two sampled values.
- Repeated a large number of times (i.e., 100,000 times) to populate the distribution for daily ingestion rate (L/day) or the combined distribution.
- Reported results as summary statistics of the combined distribution.

The distribution shape that best fit the datasets was log-normal for both ingestion volume and exposure duration. Table 4-1 presents summary statistics for different age groups based on the combined distribution analysis. As per the EPA's 2000 Human Health Methodology (U.S. EPA 2000), the 90th percentile of exposure, represented by this combined distribution (0.21 L/day) was used as a point estimate for deriving the AWQC or swimming advisories. Details and the R code for this analysis are shown in Appendix E. Appendix E also includes the mean, median, and standard deviation for the distributions for ages six to 10, 11 to 17, and 18 years and older.

Table 4-1. Results of the Combined Distribution Analysis

Age Group	Summary Statistics for Ingestion Rate (L/day)		
	Median	Mean	90th Percentile
6 to 10 years	0.063	0.094	0.21
11 to 17 years	0.038	0.058	0.13
18+ years	0.015	0.04	0.10

4.2.3.2 Body Weight

Table 8-1 in the EPA's EFH (U.S. EPA 2011) reported body weight statistics based on the National Health and Nutrition Examination Survey, including for a range of age groups. The EPA selected children aged six to 10 years because it reflected the age group with higher ingestion volumes (Appendix E; U.S. EPA 2018a; U.S. EPA 2011) and exposure duration (U.S. EPA 2011).[4] As per the EPA's 2000 Human Health Methodology (U.S. EPA 2000), mean body weight (31.8 kg) was used for deriving the AWQC or swimming advisories. Section 7.3.2 provides a discussion of younger children's exposure factors.

4.2.4 Relative Source Contribution (RSC)

The RSC component of the AWQC calculation allows a percentage of the exposure to a contaminant to include other potential exposure sources. The RSC describes the portion of the RfD available for AWQC-related sources (U.S. EPA 2000); the remainder of the RfD is allocated to other sources of the contaminant. The EPA focused on recreational exposures to microcystins and cylindrospermopsin in ambient freshwaters. To derive recommendations protective of the recreational designated use, the EPA assumes all cyanotoxin exposure is from incidental ingestion of water while recreating; therefore, no RSC term is applied.

[4] The age group six to 10 years includes 10-year-old children. The EPA's *Exposure Factors Handbook* labels this age group as six to < 11 years.

5.0 EFFECTS ASSESSMENT

The health effects studies summarized below for microcystins and cylindrospermopsin are described in detail in the EPA's HESDs and Drinking Water Health Advisories for these two cyanotoxins (U.S. EPA 2015a, 2015b, 2015c, 2015d).

5.1 Hazard Identification

5.1.1 Noncancer Health Effects

5.1.1.1 Animal Toxicity Studies

Microcystins

The preponderance of animal toxicity data on the noncancer effects of microcystins is restricted to the microcystin-LR congener. Available data on the RR, YR, and LA congeners do not provide dose-response information sufficient for quantification. The EPA is using data on effects of microcystin-LR to represent other microcystin congeners (U.S. EPA 2015d). Observed effects in animals exposed orally or via intraperitoneal infusion to microcystin-LR include liver, reproductive, developmental, kidney, and GI effects (Chernoff et al. 2002; Falconer et al. 1998; Fawell et al. 1999; Fitzgeorge et al. 1994; Guzman and Solter 1999, 2002; Heinze 1999; Ito et al. 1997a, 1997b; Yoshida et al. 1997). Most oral and injection studies in laboratory animals have demonstrated that the liver is a primary target organ for microcystin toxicity. Liver effects, as well as kidney effects, have been reported in acute, short-term, and subchronic oral studies in laboratory animals exposed to microcystin-LR, in addition to reproductive effects following short-term and subchronic oral exposures. Studies evaluating the chronic toxicity of microcystins have not shown clinical signs of toxicity and are limited by study design and by the lack of quantitative data. For individual study details see the EPA's HESD for microcystins (U.S. EPA 2015d).

Available animal data on the acute oral toxicity of microcystin-LR provide evidence of hepatotoxicity. Liver effects described in the above studies are summarized in Table 5-1. A single oral dose of 500 μg microcystin-LR/kg resulted in diffuse hemorrhage in the liver of mice and rats; more pronounced liver damage occurred at higher doses (Ito et al. 1997a; Fawell et al. 1999; Yoshida et al. 1997). Studies that utilized parenteral administration of microcystin-LR show a steep dose-response with rapid onset of liver damage.

The findings in acute and subchronic studies support the liver as a target organ for microcystin-LR toxicity. The EPA identified a 28-day short-term study by Heinze (1999) as the critical study for derivation of an RfD. Male hybrid rats (10/group) were administered microcystin-LR in drinking water at doses of 0, 50, or 150 μg/kg body weight (Heinze 1999). Liver effects included increased liver weight, and slight to moderate liver necrosis lesions with or without hemorrhages at the low dose and with dose-related increases in necrotic severity. The necrosis was accompanied by changes in serum enzymes indicative of liver damage. All rats in each dose group had liver necrosis. Data were not collected prior to the end of the study so it is not known when during the 28-day study period these effects were manifested.

Table 5-1. Liver Effects in Animals Exposed to Microcystins in Selected Acute and Short-term Studies as Discusssed in the EPA's *Health Effects Support Document for the Cyanobacterial Toxin Microcystins* (U.S. EPA 2015d)

Species	Exposure Route	Dosing Regimen	Micro-cystin Congener	Description of Liver Effects	Study
Female BALB/c mice (n = 7)	Gavage	Single dose of 0, 8,000, 10,000, or 12,500 µg/kg Examination at 24 hours after treatment	LR	Centrilobular hemorrhage, hepatocyte degeneration	Yoshida et al. (1997)
Male ICR mice aged (n = 29 age 32 weeks) and young (n = 12 age 5 weeks)	Gavage	Single dose of 500 µg/kg Animals sacrificed at 2, 5, and 19 hours after treatment	LR	Bleeding and disappearance of hepatocytes in the whole liver or in centrilobular region, friable tissue, necrosis, or eosinophilic changes in the centrilobular region	Ito et al. (1997a)
CR1:CD-1(ICR)BR(VAF plus) mice and CR1:CD(SD)BR(VAF plus) rats (5 males and 5 females per group)	Gavage	Single dose of 500, 1,500, or 5,000 µg/kg (no control) Animals sacrificed at day 14 post treatment	LR	Darkly discolored and distended livers; moderate or marked centrilobular hemorrhage of liver; diffuse hemorrhage in the liver	Fawell et al. (1999)
Male ICR mice (n = 5 per group)	Gavage	Repeated doses of 0, 4.6, 23, 46, 93, or 186 µg/kg/day for 7 days Animals sacrificed at day 7	RR	Dose-dependent increase in apoptosis	Huang et al. (2011)
Male hybrid rats (F1 generation of female WELS/Fohm × male BDIX) (10 per group)	Drinking water	Repeated doses of 0, 50, or 150 µg/kg/day for 28 days	LR	Hepatocyte degeneration, hemorrhage, and necrosis; increase in periodic acid-Schiff-positive substances (indicating cell damage), Kupffer cell activation	Heinze (1999)
Male Sprague-Dawley rats (3 per group)	Intraperitoneal infusion	Repeated doses of 0, 16, 32, or 48 µg/kg/day for 28 days	LR	Fibrous tissue, cell death, necrosis, lipid vacuoles, Kupffer cell activation (+2 and +3 severity rating)	Guzman and Solter (1999)

The liver effects in the Heinze (1999) study were supported by additional data from a study by Guzman and Solter (1999). Rats exposed via intraperitoneal infusion displayed histological evidence of liver

damage (i.e., inflammation, fibrous tissue, necrosis, and apoptosis). The study authors identified a no-observed-adverse-effect-level (NOAEL) of 16 µg/kg/day and a lowest-observed-adverse-effect-level (LOAEL) of 32 µg/kg/day. Microcystin-LR was delivered directly to the livers of the animals in the study by implanted osmotic pumps and this may account for the liver effects observed at lower doses compared to Heinze (1999). Guzman and Solter (1999) only included three rats per group exposed to doses of 0, 16, 32, or 48 µg/kg/day of microcystin for 28 days, which is a limitation of the study design. Although adverse liver effects were observed, the limited numbers of animals per dose group (n = 3) and the exposure route, which bypassed intestinal barriers to absorption, resulted in greater uncertainty than Heinze (1999). Thus, Guzman and Solter (1999) was not used to derive the RfD.

Some studies observed other kinds of effects following short-term or subchronic oral or intraperitoneal exposures. These studies, including limitations, are discussed in the EPA's HESD for microcystins (U.S. EPA 2015d). Potential effects included reproductive toxicity in males (Chen et al. 2011), maternal mortality (Fawell et al. 1999; Chernoff et al. 2002), and fetal body weight changes (i.e., at 2,000 µg/kg, administered orally during gestational days six to 15, at which significant maternal mortality was observed) (Fawell et al. 1999). Chernoff et al. (2002) did not report adverse effects on fetal or pup weights in two separate intraperitoneal studies.

Cylindrospermopsin

The available acute, short-term, and subchronic studies for cylindrospermopsin (Bazin et al. 2012; Humpage and Falconer 2002; 2003; Reisner et al. 2004; Terao et al. 1994; Shaw et al. 2001) support the liver and kidneys as the primary targets for cylindrospermopsin toxicity (summarized in Table 5-2), with effects on red blood cells also evident. These effects were observed in mice given single or repeated doses of purified cylindrospermopsin via oral administration or intraperitoneal injection (Bazin et al. 2012; Humpage and Falconer 2002, 2003; Reisner et al. 2004; Terao et al. 1994). The EPA did not find health effects information for other cylindrospermopsin congeners or analogs.

No oral reproductive or developmental studies are available for cylindrospermopsin. Developmental toxicity studies following intraperitoneal administration of cylindrospermopsin provide some evidence for maternal toxicity and decreased postnatal pup survival and body weight (Chernoff et al. 2011; Rogers et al. 2007). For individual study details, see the EPA's HESD for cylindrospermopsin (U.S. EPA 2015c).

The RfD for cylindrospermopsin was derived from the 11-week critical study by Humpage and Falconer (2002, 2003). This study was an 11-week study in mice, and the critical effect identified was kidney toxicity. The short-term studies available for cylindrospermopsin (Shaw et al. 2001; Reisner et al. 2004), were also evaluated and are considered supportive of the critical study; however, the EPA concluded that they were not suitable for quantification based on limitations including the use of extract, lack of adequate numbers of animals, monitored endpoints, the limited number of doses tested and endpoints monitored.

Humpage and Falconer (2002, 2003) identified a NOAEL of 30 µg/kg/day and a LOAEL of 60 µg/kg/day for increases in relative kidney weight in mice treated with purified cylindrospermopsin by gavage for 11 weeks. There were indications of reduced renal function effects, decreased urinary protein, and red blood cell effects (including increased bilirubin, spleen weight and polychromasia, indicative of hemolysis) at doses above the LOAEL. Although effects on kidney weight and urine protein levels were observed in male mice, the biological relevance of the latter effect and whether it would also occur in female mice needs further investigation. Mice are known to excrete a group of

highly polymorphic, low-molecular-weight urinary proteins that play important roles in social recognition and mate assessment. The relevance of the urinary protein findings in mice to humans is unknown. Humpage and Falconer (2002, 2003) found signs indicative of hemolysis (e.g., increased bilirubin, spleen weight and polychromasia), however these changes were not statistically significant.

Results from Reisner et al. (2004) corroborate Humpage and Falconer (2002, 2003) with comparable effects observed in mice during a three-week study. The kidney and red blood cell effects observed by Reisner et al. (2004) occurred at a LOAEL of 66 μg/kg/day in drinking water. The study authors demonstrated significant increases in hematocrit, acanthocytes (abnormal red blood cells), and liver and testes weights in exposed animals and a duration-related nonsignificant increase in kidney weight. The red blood cell effects were seen as early as the end of the first week of dosing and were present in each of the three weekly blood samples collected. Sukenik et al. (2006) observed similar effects on red blood cells (increases in hematocrit from week 16 to 32 accompanied by increased numbers of acanthocytes up to week 42) in male and female mice exposed to gradually increasing concentrations of cylindrospermopsin (i.e., from 100 to 550 μg/L) in drinking water for 42 weeks. Mice were given cylindrospermopsin in the form of spent medium on which cultures of *Aphanizomenon ovalisporum* had been grown; other medium components were not characterized. The authors proposed a LOAEL of 20 μg/kg/day (equivalent to 200 μg/L) for male and female mice based on changes in hematocrit at 16 weeks (Sukenik et al. 2006). This study was not selected as a critical study because this study used a single dose; however, the kidney and red blood cell effects at that dose after three weeks were comparable to the effects seen in the Humpage and Falconer (2002, 2003) study at a slightly lower 60 mg/kg/day dose after 11 weeks.

The short-term study by Shaw et al. (2001) was also considered in the development of the RfD for cylindrospermopsin. Shaw et al. (2001) reported liver effects (fatty infiltration) in mice given 50 μg/kg purified cylindrospermopsin by gavage for 14 days; this dose is lower than the NOAEL identified in the key study by Humpage and Falconer (2002, 2003). However, the EPA concluded that the Shaw et al. (2001) study was not suitable for quantification based on the limited number of doses tested.

A 90-day oral toxicity study by Chernoff et al. (2018) demonstrated signs of hepatic and renal injury in mice at all dose levels (0, 75, 150, and 300 μg/kg/day). Liver toxicity effects were noted by elevated absolute and relative liver weights, increases in serum alanine aminotransferase activity, reduced serum blood urea nitrogen and cholesterol levels, and increased incidence of hepatocellular hypertrophy and cord disruption. Renal toxicity effects were demonstrated in elevated absolute and relative kidney weights and renal cellular hypertrophy, tubule dilation, and cortical tubule lesions. Males showed more susceptibility to toxic effects; liver and kidney/body weight ratios, reduced cholesterol levels, cellular signs of inflammation, and degree and extent of renal histopathological damage were all observed to be more prominent in males. A NOAEL was not determined for any dose level based on significant liver and kidney effects exhibited in the 75 μg/kg group. The LOAEL of 75 μg/kg observed by Chernoff et al. (2018) is higher than the Humpage and Falconer (2002, 2003) NOAEL of 30 μg/kg.

Table 5-2. Kidney and Liver Effects in Animals Exposed to Cylindrospermopsin (Purified) in Acute and Key Short-term Studies in the *Health Effects Support Document for the Cyanobacterial Toxin Cylindrospermopsin* (U.S. EPA 2015c)

Species	Exposure Route	Dosing Regimen	Description of Kidney and Liver Effects	Study
Male Swiss albino mice (10 mice per group, except the highest dose group, which included 6 mice)	Gavage	Repeated doses of 0, 30, 60, 120, or 240 µg/kg/day for 11 weeks	Kidney: dose-related increases in relative kidney weight, proximal renal tubular damage, decreased urinary protein Liver: necrosis, inflammatory foci, and bile duct changes	Humpage and Falconer (2002, 2003)
CD-1 (Swiss-Webster) mice (18 to 20 per group)	Gavage	Repeated doses of 0, 75, 150, or 300 µg/kg/day for 90 days	Kidney: elevated absolute and relative kidney weights, renal cellular hypertrophy, tubule dilation, cortical tubule lesions Liver: elevated absolute and relative liver weights, increases in serum alanine aminotransferase activity, reduced serum blood urea nitrogen and cholesterol levels, increased incidence of hepatocellular hypertrophy and cord disruption	Chernoff et al. (2018)
Male Swiss mice (3 per group)	Gavage	Single dose of 1,000, 2,000, or 4,000 µg/kg Examination at 24 hours after treatment	Liver: dark red liver, apoptosis in the liver and the kidneys	Bazin et al. (2012)
Male ICR mice (n = 24, single group)	Intraperitoneal injection	Single dose of 200 µg/kg Three animals sacrificed at 8 time points, 16–100 hours after treatment	Kidney: proliferation of the endoplasmic reticulum and fat droplet accumulation in cells along the brush borders of the tubules plus limited single cell necrosis Liver: necrosis in the centrilobular region	Terao et al. (1994)
Male ICR mice (4 per group)	Drinking water	Repeated doses of 0 or 0.6 mg/L (estimated at 66 µg/kg/day) for 3 weeks	Kidney: duration-related nonsignificant increase in kidney weight Liver: increases in relative weight	Reisner et al. (2004)
Quackenbush mice (4 per group)	Intraperitoneal injection	Single dose of 200 µg/kg	Liver: fatty infiltration and cell necrosis	Shaw et al. (2001)

Species	Exposure Route	Dosing Regimen	Description of Kidney and Liver Effects	Study
Quackenbush mice (4 per group)	Gavage or intraperitoneal injection	0 to 300 μg/kg/day (oral) or 0 to 25 μg/kg/day (intraperitoneal injection) for 14 days	Liver: fatty infiltration (oral), foamy hepatocellular cytoplasm (intraperitoneal injection)	Shaw et al. (2001)

The Humpage and Falconer (2002, 2003) study was determined to be the most appropriate for the quantitative assessment because the LOAEL at 11 weeks would be protective for the effects seen at three weeks in the shorter duration study. For these reasons, this RfD was deemed suitable for development of the short-term drinking water health advisory and for use in recreational exposure scenarios. The EPA's HESD and Health Advisory documents for cylindrospermopsin describe the selection of the critical study and effect in detail and provide the rationale for applicability of the longer-term duration study (U.S. EPA 2015c).

5.1.1.2 Human Studies

Microcystins

The EPA identified the available epidemiological, outbreak, and case study reports on adverse health effects from oral exposures to microcystins. Limited human studies examining microcystin effects on humans exposed via drinking water are available, and no dose response data from oral exposure to microcystins in ambient water were identified. The scant human data on the oral toxicity of microcystin-LR are limited by the potential co-exposure to other pathogens, cyanotoxins, and microorganisms; by the lack of quantitative information; and by the failure to control for confounding factors. Available human studies evidence is supportive of the liver as a target organ for toxicity (Carmichael 2001; Falconer et al. 1983; Giannuzzi et al. 2011; Hilborn et al. 2013; Jochimsen et al. 1998; Li et al. 2011b). The EPA identified four epidemiological studies, three case reports, and two outbreak summaries that evaluated human health effects associated with recreational exposures to cyanobacteria and microcystins. This human health effects information is summarized in the paragraphs that follow.

Backer et al. (2008) characterized microcystin concentrations in blood and reported symptoms in people recreating in a lake with a *Microcystis aeruginosa* bloom to those of people recreating in a nearby bloom-free lake. Low levels of total microcystins (detection limit = 0.08 ng/m^3) were detected in air samples collected above a lake bloom. Phytoplankton counts ranged from 175,000 to 688,000 cells per mL with > 95 percent of those cells being cyanobacteria. Cell densities of potentially toxigenic cyanobacteria ranged from approximately 54,000 to 144,000 cells/mL. Although a visible bloom was present and contained cyanobacterial species capable of producing microcystin, microcystin concentrations in water during the study were low and ranged from 2 to 5 μg/L. Recreational users of the lake at the time of the bloom had no detectable microcystins in their blood and did not report an increase in GI, dermal, respiratory, or neurological symptoms after spending time on the lake. Adenoviruses (level of detection (LOD) = 1,250 gene copy equivalents) and enteroviruses (LOD = 200 plaque forming units/10 L) were not detected in any water sample. This study was limited in the number of participants (n = 104) and included a limited number of exposure days in the analysis (three days). The study demonstrated that people recreating on or in a water body can be exposed to aerosolized microcystins. However, given the limited number of participants and exposure days, and the low levels of microcystins present in the water and as aerosols, there were no reported increases in self-reported symptoms following recreational exposures. Other symptoms consistent with microcystin intoxication (e.g., liver toxicity) were not included in the study.

Backer et al. (2010) applied the same experimental approach at three lakes in California. Two of the lakes experienced blooms producing much higher microcystin concentrations compared with the lakes studied in the Backer et al. (2008) study, and the third lake did not contain a toxin-producing bloom. Eighty-one people, aged 12 and older, participated in the study and engaged in waterskiing, using personal watercraft, swimming, or wading. Total microcystins present in the lake containing toxic blooms ranged from < 10 µg/L to > 500 µg/L. Measured microcystin concentrations from personal air samples ranged from the limit of detection (0.1 ng/m^3) to 2.89 ng/m^3; the mean air concentration was 0.4 ng/m^3. Similarly, nasal swabs ranged from below the limit of detection to 5 ng, and all blood samples were below the limit of detection. Recreators had a significantly higher amount of microcystins present in nasal swabs after exposure. No statistically significant differences were noted in the frequency of reported GI, dermal, or respiratory symptoms between participants immediately after they engaged in direct- or indirect-contact recreational activities in the lake with a cyanobacterial bloom and those in a lake without a cyanobacterial bloom. Other symptoms consistent with microcystin intoxication (e.g., liver toxicity) were not included in the study. Adenoviruses or enteroviruses were not detected at the study locations. The authors concluded that it is possible for microcystins to become aerosolized, which in turn represents a potential route of exposure to recreators. They recommended additional research studying larger populations and sensitive subgroups.

Lévesque et al. (2014) conducted a prospective study of residents living in proximity to three lakes in Canada affected by cyanobacteria and microcystins to investigate the relationship between recreational exposure, specifying full contact and limited contact with lake water, and the incidence of GI, dermal, respiratory, and other symptoms (e.g., ear pain, muscle pain). Full contact included swimming, waterskiing, windsurfing, use of watercraft involving launching, accidental falls, and similar activities, and limited contact included fishing, use of watercraft not involving launching, and other activities. The authors reported a dose-effect relationship (p-trend = 0.001) between total cyanobacterial cell counts and severe GI illness with a significant increase in reported symptoms starting at 20,000 cells/mL and above. The study reported a relative risk value of 3.28 (95 percent confidence interval (CI): 1.69–6.37) for the more severe GI symptom index (i.e., GI2, defined as diarrhea or vomiting or (nausea and fever) or (abdominal cramps and fever)) for exposures by full or limited contact to concentrations higher than 100,000 cells/mL (Lévesque et al. 2014). Adjusted relative risks of GI illness were significantly high for limited contact, but no relationship was found between GI symptoms and full contact. The authors explained that study participants avoided full contact with lake waters when high densities of cyanobacteria were visible, but continued to have limited contact. No significant fecal contamination measured by *Escherichia coli* (*E. coli*) was observed with geometric means in the lakes ranging from 8 to 145 colony forming units (CFU)/100 mL.[5] No associations were observed between any symptoms and recreational exposures to microcystins. Overall, the microcystin concentrations were low during the study, and the reported lower bound of the upper tertile was 0.2456 µg/L. The maximum microcystin concentrations for which recreational-related GI symptoms were reported was 7.65 g/L; however, microcystins occurred at much higher concentrations (e.g., maximum reported microcystin concentrations of 108 µg/L and 773 µg/L at two of the study locations), but there was no significant trend of increasing illness symptoms with elevated toxin concentrations. The study did not characterize the primary endpoint of concern for exposure to microcystins (i.e., liver toxicity) and did not conduct the necessary medical testing to determine liver function impairment.

Lévesque et al. (2016) provided additional analysis of the prospective study reported previously. Because GI illness was significantly associated with increasing cyanobacterial cell densities and GI

[5] Current Canadian recreational water guidelines for *E. coli*: geometric mean ≤ 200 *E. coli*/100 mL and single-sample maximum ≤ 400 *E. coli*/100 mL (Health Canada 2012).

symptoms can be related to cellular constituents, also termed *endotoxins* in the literature, the authors characterized the relationship between endotoxin exposure and illness in the study participants. Endotoxins include cell wall-associated lipopolysaccharides present in cyanobacteria and Gram negative bacteria. Frozen filters collected during the study were analyzed for endotoxins. The authors found a weak correlation between endotoxin levels and cyanobacteria cell density and reported a significant trend of increasing GI illness with increasing endotoxin concentrations. They also suggest that endotoxin concentrations could be a surrogate for another stressor. They cite other researchers that have suggested the endotoxins could be contributed by other members of the microbial community or the reported symptoms could be related to another stressor (Berg et al. 2008; Blahova et al. 2013; Rapala et al. 2002; Stewart 2006d).

In a recent case report by Vidal et al. (2017), a 20-month-old child and three adults reported GI symptoms several hours after engaging in bathing and other recreational activities at beaches in Montevideo, Uruguay, during January 2015. At that time, a cyanobacterial bloom of mainly *Microcystis* occurred in the River de la Plata. While the GI symptoms in the adults (i.e., diarrhea) rapidly resolved, the child's symptoms (i.e., diarrhea and vomiting) persisted. The child developed fatigue and jaundice, and five days after the exposure, she was admitted to hospital. Tests showed significant increases in bilirubin and serum liver enzymes, and a diagnosis of acute liver failure was given. The child was recommended for, and received, a liver transplant. The city government's beach monitoring program from April 2014 to March 2015 reported mean and maximum microcystin concentrations of 2.9 µg/L of 56 µg/L, respectively. These levels were reported in water samples from the beaches the family used with cyanobacteria presence but without cyanobacterial foam. Mean and maximum microcystin concentrations of 2,900 µg/L and 8,200 µg/L, respectively, were reported in water samples with cyanobacterial foam. The monitoring program also reported geometric means of fecal coliform values below the limit of 1,000 CFU/100 mL. After the child received a liver transplant, histological analysis of the explanted liver revealed liver damage characterized by hemorrhagic necrosis, intracytoplasmic cholestasis, large and multinucleated hepatocytes, proliferation, and nodular regeneration. The pathological findings and detection of microcystin-LR in the liver (2.4 ng microcystin-LR/g and 75.4 ng (D-Leu1) microcystin-LR/g liver) led to a diagnosis of acute liver failure related to exposure to microcystin-LR and cyanobacteria.

In another case report, acute intoxication with microcystin-producing cyanobacterial blooms in recreational water was reported in Argentina in 2007 (Giannuzzi et al. 2011). A male Jet Skier was exposed to a *Microcystis* bloom containing 33,680 and 35,740 cells/mL. A level of 48.6 µg/L of microcystin-LR concentrations was detected in water samples associated with the bloom. The subject was immersed for two hours as a result of an accident that required him to swim to the shoreline towing the Jet Ski. Four hours later the subject reported experiencing nausea and abdominal pain. Three days later the subject sought medical assistance because of respiratory distress requiring his hospitalization. One week after the exposure, the patient developed a hepatotoxicosis with a significant increase of serum alanine aminotransferase, aspartate aminotransferase, and γ-glutamyltransferase. With treatment, the patient recovered within 20 days.

An outbreak among army recruits undergoing canoe exercises who had consumed reservoir water containing a bloom of *Microcystis aeruginosa* reported symptoms of headache, sore throat, vomiting and nausea, stomach pain, dry cough, diarrhea, blistering around the mouth, and pneumonia (Turner et al. 1990). Microcystins, including microcystin-LR, were present in bloom samples. However, high levels of *E. coli* were also found in reservoir water after two weeks. The authors suggested that exposure

to microcystins may have had a role in some of the clinical symptoms; however, this case report information is insufficient to establish cause and effect.

Dziuban et al. (2006) and Hilborn et al. (2014) reported 10 outbreaks associated with recreational exposure to cyanobacteria in which microcystins were detected. Hilborn et al. (2014) reported that eight of these investigations evaluated the presence of cyanotoxins; eight detected microcystins; and two detected cylindrospermopsin. In four of the outbreaks, microcystin concentrations ranged from 0.2 µg/L to > 2,000 µg/L. Four outbreaks had microcystin concentrations > 20 µg/L. Cylindrospermopsin and anatoxin-a also were detected in three of the outbreaks. In one outbreak, 20.8 µg/L microcystins was measured, and other cyanotoxins were either not detected or measured. The nine persons reporting illness for this outbreak had symptoms that included abdominal cramps (3 people), diarrhea (3), nausea (3) vomiting (2), fever (2), headache (2), rash (8), eye irritation (1), ear ache (1), neurologic symptoms (2), tingling (2), confusion (1), and respiratory symptoms (1) (Hilborn et al. 2014). Dziuban et al. (2006) reported on two 2004 cyanobacteria-associated outbreaks in which 22 cases of illness were associated with elevated levels of microcystins in Nebraska lakes. The predominant illnesses in both outbreaks included dermatitis and gastroenteritis, and individuals who sought medical care showed a combination of rashes, diarrhea, cramps, nausea, vomiting, and fevers. Walker et al. (2008) also reported about a Nebraska outbreak. Levels of total microcystins at the east swimming beach of Pawnee Lake exceeded 15 ppb on July 12, 2004, and a health alert was issued. However, heavy public use of Pawnee Lake occurred that weekend and more than 50 calls were received from the public, complaining about symptoms such as skin rashes, lesions, blisters, vomiting, headaches, and diarrhea after swimming or water skiing in Pawnee Lake (Walker et al. 2008). The outbreak reports data are not sufficient to establish cause and effects for microcystins because of weaknesses in the nature of the data reported and the many potential confounding variables. The researchers concluded that the disease outbreak data suggest that the time to onset of effects might be rapid, that children might be at higher risk for illness, and that these types of outbreaks occur during the warmer months. Hilborn et al. (2014) noted that HAB-associated illness from recreational exposure might be underreported due to multiple possible exposure routes and the non-specific nature of potential health effects.

Graham et al. (2009) counted 36 states with anecdotal reports of acute cyanotoxin poisonings of animals, humans, or both as reported in journal articles and newspaper articles (Chorus and Bartram 1999; Hilborn et al. 2014; Huisman et al. 2005; Yoo et al. 1995).

Information on the human health effects of microcystins based on epidemiological studies related to drinking water exposures to microcystins are discussed in detail in the EPA's HESD for microcystins (U.S. EPA 2015d). These studies are summarized in the paragraphs that follow.

An epidemiology study done in Australia compared the hepatic enzyme levels from patients served by a public water supply contaminated with a *Microcystis aeruginosa* bloom with enzyme levels from patients living in areas served by water supplies uncontaminated by cyanobacteria (Falconer et al. 1983). Although the authors observed significant variability in enzyme levels between the two groups, the findings were attributed by the authors to the imprecise method of study participant selection and confounding factors such as alcoholism and chronic kidney disease among some of the participants.

A cross-sectional study conducted in China assessed the relationship between the consumption of drinking water and aquatic food (carp and duck) contaminated with microcystins and liver damage in children (Li et al. 2011b). The authors found that mean serum levels of microcystins ranged from below detection to 1.3 µg microcystin-LR equivalents/L. According to the authors, hepatitis B infection was a greater risk for liver damage among these children than the microcystins exposure.

An outbreak of acute liver failure occurred in a dialysis clinic in 1996 in Caruaru, Brazil, where dialysis water was contaminated with microcystins, and possibly cylindrospermopsin. Of the 130 patients who received their routine hemodialysis treatment (intravenously) at that time, 116 reported symptoms of headache, eye pain, blurred vision, nausea, and vomiting. Subsequently, 100 of the affected patients developed acute liver failure and, of these, 76 died (Carmichael et al. 2001; Jochimsen et al. 1998). Analyses of blood, sera, and liver samples from the patients revealed microcystins.

In another contamination event at a dialysis center in Rio de Janeiro, Brazil, in 2001, 44 dialysis patients were potentially exposed to microcystin concentrations of 0.32 μg/L, detected in the activated carbon filter used in an intermediate step for treating drinking water to prepare dialysate (Soares et al. 2005). Concentrations of 0.4 μg/L microcystin-LR were detected in the drinking water. Serum samples were collected from 13 dialysis patients 31 to 38 days after the detections in water samples, and patients were monitored for eight weeks. Concentrations of microcystin-LR in the serum ranged from 0.46 to 0.96 ng/mL. Although the biochemical outcomes varied among the patients, markers of hepatic cellular injury and of chlolestasis (elevations of AST, ALT bilirubin, ALP, and GGT) in serum during weeks one to eight after treatment frequently exceeded normal values (Hilborn et al. 2013). Because microcystin-LR was not detected in the dialysate during weekly monitoring after the first detection, the authors suggested that the patients were not continuously exposed to the toxin and that the toxin detected in the serum after eight weeks may have been present in the form of bound toxin in the liver (Soares et al. 2005). Results were consistent with a mild to moderate mixed liver injury (Hilborn et al. 2013). Although the patients in the study had pre-existing diseases, the direct intravenous exposure to dialysate prepared from surface drinking water supplies put them at risk for cyanotoxin exposure and resultant adverse effects (Hilborn et al. 2013).

Cylindrospermopsin

No epidemiological studies were identified for recreational exposure to cylindrospermopsin.

Hilborn et al. (2014) reported two outbreaks associated with recreational exposure to HABs in which cylindrospermopsin was detected between 2009 and 2010. However, cyanobacteria, microcystins, and other cyanotoxins were also present. As mentioned earlier, the results reported from the outbreaks should not be interpreted as cause and effect.

Human data on oral toxicity of cylindrospermopsin are limited, but results indicate that kidney and liver exhibit adverse effects due to cylindrospermopsin exposures. Information on the human health effects of cylindrospermopsin based on epidemiological studies related to drinking water are discussed in detail in the EPA's HESD for cylindrospermopsin (U.S. EPA 2015c). This information is summarized in the paragraphs that follow.

Reports of a hepatoenteritis-like outbreak (mostly in children) in Palm Island, Australia, in 1979 were attributed to consumption of drinking water with a bloom of *Cylindrospermopsis raciborskii*, a cyanobacteria that can produce cylindrospermopsin. No data are available on exposure levels or potential co-exposures to other cyanobacterial toxins and microorganisms. The majority of the cases, mostly children, required hospitalization. The clinical picture included fever, headache, vomiting, bloody diarrhea, hepatomegaly, and kidney damage with loss of water, electrolytes, and protein (Byth 1980; Griffiths and Saker 2003).

Dermal exposure to cylindrospermopsin was evaluated using skin-patch testing in humans (Pilotto et al. 2004; Stewart et al. 2006a). Exposed individuals showed mild irritation, but no statistically significant

dose-response relationship or reaction rates were found between skin reactions and increasing cell concentrations for either whole or lysed cells (Pilotto et al. 2004). No detectable skin reactions were observed in individuals exposed to lyophilized *Cylindrospermopsis raciborskii* (Stewart et al. 2006a).

5.1.1.3 Mode of Action for Noncancer Health Effects

Microcystins

Mechanistic studies have shown the importance of membrane transporters for systemic uptake and tissue distribution of microcystins by all exposure routes (Feurstein et al. 2010; Fischer et al. 2005). The importance of the membrane transporters to systemic uptake and tissue access is demonstrated by studies where there was either no liver damage or reduced damage when the hepatic organic anion transporting polypeptide (OATP) receptors were inhibited (Hermansky et al. 1990a, 1990b; Thompson and Pace 1992). OATPs are a transporter family that controls uptake of microcystins by the liver (Fischer et al. 2005).

The uptake of microcystins causes protein phosphatase inhibition and a loss of coordination between cytoskeletal protein phosphorylation by kinases and dephosphorylation by phosphatases. This event initiates altered cell function followed by cellular apoptosis and necrosis (Barford et al. 1998). Both cellular kinases and phosphatases keep the balance between phosphorylation and dephosphorylation of key cellular proteins controlling organization of the cytoskeleton, metabolic processes, gene regulation, cell cycle control, transport and secretory processes, and cell adhesion. Each of the microcystin congeners evaluated (LR, LA, and LL) interacts with catalytic subunits of protein phosphatases PP1 and PP2A, inhibiting their functions (Craig et al. 1996).

As a consequence of the microcystin-induced changes in cytoskeleton proteins, an increase in cellular reactive oxygen species (ROS) leads to cellular apoptosis. In both in vitro and in vivo studies, cellular pro-apoptotic Bax and Bid proteins increased whereas anti-apoptotic Bcl-2 decreased (Fu et al. 2005; Huang et al. 2011; Li et al. 2011a; Takumi et al. 2010; Weng et al. 2007; Xing et al. 2008). Mitochondrial membrane potential and permeability transition pore changes (Ding and Nam Ong 2003; Zhou et al. 2012) lead to membrane loss of cytochrome c, a biomarker for apoptotic events. Wei et al. (2008) identified a time-dependent increase in ROS production and lipid peroxidation in mice after exposure to microcystin-LR. Following intraperitoneal injection of 55 μg/kg of body weight microcystin-LR, the levels of hepatic ROS increased within 0.5 hours of treatment and continued to accumulate for up to 12 hours in a time-dependent manner.

Cylindrospermopsin

Despite the number of studies that have been published, the mechanisms for liver and kidney toxicity by cylindrospermopsin are not completely characterized.

In vitro and in vivo studies showed that cylindrospermopsin can inhibit hepatic protein synthesis (Froscio et al. 2003; Froscio et al. 2008; Terao et al. 1994), which could impact mouse urinary protein production leading to decreased urinary excretion of these proteins. Available evidence indicates that protein synthesis inhibition is not decreased by broad-spectrum CYP450 inhibitors, but they do reduce cytotoxicity (Bazin et al. 2010; Froscio et al. 2003). Hepatotoxicity appears to be CYP450-dependent, which indicates a possible involvement of oxidized or fragmented metabolites and mechanisms other than protein synthesis inhibition (Froscio et al. 2003; Humpage et al. 2005; Norris et al. 2002; Norris et al. 2001).

In the Reisner et al. (2004) and Sukenik et al. (2006) reports, microscopic examination of blood samples showed the presence of red blood cells with spiked surfaces rather than their normal biconcave-disc shape. The authors attributed the acanthocyte formation to an increase in the cholesterol to phospholipid ratio of the red blood cell membrane. Phospholipids constitute the matrix material of cell membranes. The authors hypothesized that this change was the consequence of decreased activity of plasma lecithin cholesterol acyl transferase, an enzyme associated with high-density lipoproteins and the esterification of plasma cholesterol. Effects on the cholesterol content of the red blood cell membrane can occur with inhibition of the enzyme increasing membrane fluidity and mean corpuscular volume. Removal of the abnormal blood cells by the spleen increases both spleen weight and serum bilirubin as well as stimulates hematopoiesis. Additional research is needed to examine the lecithin cholesterol acyl transferase enzyme inhibition hypothesis to confirm whether it accounts for the effects on the red blood cell as a result of cylindrospermopsin exposure.

Kidney necrosis and a decreased renal failure index at the high cylindrospermopsin doses in Humpage and Falconer (2002, 2003) are also indicative of an effect on the kidney. Numerous signs of renal damage including proteinuria, glycosuria, and hematuria were also observed in humans after a hepatoenteritis-like outbreak in Palm Island, Australia, in 1979 (Byth 1980). The outbreak was attributed to consumption of drinking water from source waters with a bloom of *Cylindrospermopsis raciborskii*. These effects have been shown to be related to impaired kidney function (Byth 1980); however, no mode of action information for kidney effects was observed in the available animal or human studies of cylindrospermopsin. Because all the studies were conducted in mice, a species that excretes low-molecular-weight proteins in urine, a study is needed of cylindrospermopsin in a laboratory species that does not excrete protein in the urine to determine whether there are comparable effects on kidney weight, protein excretion, and renal cellular damage.

5.1.2 Cancer

5.1.2.1 Weight of Evidence Classification

While there is evidence of an association between liver and colorectal cancers in humans and microcystins exposure and some evidence that microcystin-LR is a tumor promoter in mechanistic studies, there is "*inadequate information to assess carcinogenic potential"* of microcystins in humans (U.S. EPA 2005b). The human studies are limited by lack of exposure information and the uncertainty regarding whether these studies adequately controlled for confounding factors such as hepatitis B infection. No chronic cancer bioassays for microcystins in animals are available. The EPA (U.S. EPA 2005a) states that the descriptor of "*inadequate information to assess carcinogenic potential*" is appropriate when available data are judged inadequate for applying one of the other descriptors or for situations where there is little or no pertinent information or conflicting information. The guidelines also state that (p. 2-52) "Descriptors can be selected for an agent that has not been tested in a cancer bioassay if sufficient other information, e.g., toxicokinetic and mode of action information, is available to make a strong, convincing, and logical case through scientific inference." In the case of microcystins, the data suggest that microcystin-LR may be a tumor promoter but not an initiator. Without strong epidemiological data and a chronic bioassay of purified microcystin-LR, the data do not support classifying microcystin-LR as a carcinogen.

No chronic cancer bioassays of cylindrospermopsin were located in the literature. Limited data from an in vivo study showed no indication that the cyanobacterial extract containing cylindrospermopsin in the presence of a tumor promotor indicated preneoplastic changes consistent with its having tumorigenic

activity in mice (Falconer and Humpage 2001). Following the EPA guidelines (U.S. EPA 2005a), there is inadequate information to assess carcinogenic potential of cylindrospermopsin.

5.2 Dose-response Assessment

The RfD value for microcystins used to derive this recreational AWQC or swimming advisory is described in the EPA's HESD for microcystins (U.S. EPA 2015d). The EPA identified a 28-day study in male hybrid rats by Heinze (1999) as the critical study (described in section 5.1.1). A LOAEL of 50 µg/kg/day was identified based on increased liver weight, slight to moderate liver necrosis (necrotic severity was dose-related) with hemorrhages, and increased enzyme levels, which was used to derive an RfD of 0.05 µg/kg/day. The EPA selected the study by Heinze (1999) based on the appropriateness of the study duration, the use of multiple doses, dose-related toxicological responses, and histopathological evaluations of toxicity. After 28 days of exposure, rat organ weights (liver, kidneys, adrenals, thymus, and spleen) were measured, and hematology, serum biochemistry, and histopathology of liver and kidneys were evaluated. The critical effect in the Heinze (1999) study was supported by additional acute and subchronic data as described in the EPA's HESD for microcystins and summarized in section 5.1.1.1. The EPA's selection of uncertainty factors and derivation of the RfD are documented in its HESD for microcystins (U.S. EPA 2015d).

The RfD value for cylindrospermopsin used to derive the AWQC and swimming advisory is described in the EPA's HESD for cylindrospermopsin (U.S. EPA 2015c). The EPA identified an 11-week study in mice by Humpage and Falconer (2002, 2003) as the critical study for development of the RfD. The NOAEL was 30 µg/kg/day dose for increases in relative kidney weight seen at the LOAEL of 60 µg/kg/day. Increased relative kidney weights was the critical effect on which to base the point of departure. The EPA's selection of UFs and derivation of the RfD are documented in its HESD for cylindrospermopsin (U.S. EPA 2015c).

6.0 RECOMMENDED RECREATIONAL CRITERIA AND SWIMMING ADVISORY DERIVATION

This section summarizes the inputs and shows the calculation for the recommended recreational criteria and swimming advisories for microcystins and cylindrospermopsin.

6.1 Microcystins Magnitude

The magnitude of the recommended recreational criteria and swimming advisory for microcystin toxins is calculated as follows:

$$\text{Recreational value (µg/L)} = \text{RfD} \times \frac{\text{BW}}{\text{IR}}$$

Where:

RfD (µg/kg/day)	=	0.05 µg/kg/day (U.S. EPA 2015d)
BW (kg)	=	31.8 kg (mean body weight of children six to 10 years; U.S. EPA 2011)
IR (L/day)	=	0.21 L/day (90th percentile daily recreational water incidental ingestion rate for children age six to 10 years; Appendix E; U.S. EPA 2018a; U.S. EPA 2011; see section 4.2.3.1)

$$\text{Microcystins recommended recreational value} = 0.05\ \mu g/kg/day \times \frac{31.8\ kg}{0.21\ L/day} = 8\ \mu g/L$$

6.2 Cylindrospermopsin Magnitude

The magnitude of the recommended recreational criteria and swimming advisory values for cylindrospermopsin is calculated as follows:

$$\text{Recreational value (µg/L)} = \text{RfD} \times \frac{\text{BW}}{\text{IR}}$$

Where:

RfD (µg/kg/day)	=	0.1 µg/kg/day (U.S. EPA 2015c)
BW (kg)	=	31.8 kg (mean body weight of children six to 10 years; U.S. EPA 2011)
IR (L/day)	=	0.21 L/day (90th percentile daily recreational water incidental ingestion rate for children age six to 10 years; Appendix E; U.S. EPA 2018a; U.S. EPA 2011; see section 4.2.3.1)

$$\text{Cylindrospermopsin recommended recreational value} = 0.1\ \mu g/kg/day \times \frac{31.8\ kg}{0.21\ L/day} = 15\ \mu g/L$$

6.3 Frequency and Duration for Recreational Criteria

The frequency and duration components of a criterion describe how often and for how long a water body's conditions can exceed the magnitude and be protective of the designated use (U.S. EPA 2005c). HABs can occur naturally, but can be an uncommon event due to a convergence of climatic and other environmental factors that result in a single short-term bloom lasting days or a couple of weeks. In some cases, multiple HABs can occur in a single year. Alternatively, longer-term HABs can occur regularly in some waters lasting for a few weeks, months, or possibly all year. HABs can occur while conditions conducive to cyanobacterial proliferation exist and limit the use of the water body for primary recreation. Water bodies where a toxic HAB has occurred in the past may experience repeat occurrences of elevated toxins when bloom-promoting conditions reoccur. In some circumstances, anthropogenic inputs are identified and controlled, and the conditions that cause the bloom can be mitigated.

The EPA recognizes that a single sample above the cyanotoxin criteria magnitude does not necessarily indicate that the designated recreational use is not attained. However, when cyanotoxin concentrations exceed the criteria magnitude either in multiple short-term blooms within a year or from a single bloom that persists for an extended period within a year, and when these patterns occur in more than one year, the designated recreational use may not be attained. The frequency and duration components discussed in this section support the identification of a trend or pattern of cyanotoxin excursions that state decision makers can use to inform the evaluation of a water body. The EPA recommends that decisions on whether the designated recreational use is attained should be flexible enough to address both types of exposure patterns when patterns reoccur in more than one year (short-term blooms that occur frequently in a recreational season, or blooms that persist for an extended period during a recreational season). States may want to evaluate the pattern of bloom occurrence and toxin concentrations within and across years to determine if there is a trend toward degradation of the water quality.

The EPA's recommended criteria duration rely on the underlying toxicity data used to derive the criteria. For both toxins, animal toxicological studies consistently demonstrate adverse health effects at various dosages and relevant timeframes. See Tables 5-1 and 5-2. For microcystins, the key study (Heinze 1999) shows adverse liver effects from repeated microcystin exposures (50 and 150 µg/kg body weight) during a study duration of 28 days. Another supporting study showed similar effects (Guzman and Solter 1999). For cylindrospermopsin, the key study (Humpage and Falconer 2002, 2003) had a duration of 11 weeks. The shorter-term studies available for cylindrospermopsin (Shaw et al., 2001; Reisner et al., 2004) were not suitable for quantification due to study limitations; however, effects observed in these studies are the same or similar to the Humpage and Falconer study (2002, 2003) and occur at similar doses. The LOAEL derived from Humpage and Falconer (2002, 2003) was determined to be protective for the adverse effects observed in the shorter duration studies. For both key studies, adverse health effects were noted at the end of the study period and it is not known if those effects occurred earlier.

The criteria are based on the same science used to develop the EPA's Drinking Water Health Advisories for microcystins and cylindrospermopsin, which are 10-day advisories (U.S. EPA 2015a, 2015b). The 10-day drinking water health advisory values represent concentrations of cyanotoxins in finished drinking water below which adverse noncarcinogenic effects are not expected to result from ingestion of drinking water over a 10-day period. Following the detection and confirmation of microcystins or cylindrospermopsin in finished drinking water above the health advisory values, the EPA recommends that drinking water utilities initiate actions to reduce exposure to consumers including determining when to notify drinking water consumers who may be more susceptible to adverse outcomes (U.S. EPA 2015c). If the advisory level continues to be exceeded after 10 days, additional public health measures can be taken, including a do-not-drink and do-not-boil water advisory. Recreational water managers

have fewer options to reduce exposure to toxins in recreational waters than do drinking water treatment operators, as recreational water does not go through a treatment process.

The EPA recommends states use 10-day assessment periods over the course of a recreation season to evaluate ambient water body condition and recreational use attainment. The 10-day period links the water body assessment to the adverse health effects from ingestion of the toxins over short-term exposures, consistent with the EPA's Drinking Water Health Advisory (described in greater detail in section 5.1). Also, Cordell (2012) discussed decade-long trends in outdoor recreation activities showing a significant proportion (43 percent) of Americans visited a beach in 2005–2009, up almost 21 percent over the previous decade. Over the same timeframe, participation in swimming in lakes and streams (42 percent of the population) increased by 14 percent (Cordell 2012). Beach visitation surveys have shown that nearly half (47 percent) of the local population are regular beach users with five or more visits in a recreation season (Caldwell et al. 2013). The recommended assessment period is reasonable considering beach visitation rates for recreators living in proximity to a beach or vacationing at a beach for a week or two with daily beach visits expected. Exposure to recreational waters containing microcystins or cylindrospermopsin at or below the recommended magnitude concentrations over the short-term 10-day duration would not be expected to result in the adverse health effects discussed in section 5.

The EPA recommends that if toxin concentrations are higher than the criterion magnitude in a sample collected during a 10-day assessment period, that period should be considered an excursion from the recreational criteria. Elevated toxin concentrations can occur over hours, days, or a couple of weeks and are counted as excursions in a recreational season. A short-term HAB that does not reoccur can result in a small number of excursions of the criteria but is not expected to result in impairment of the recreational use. Such algal blooms may result from conditions that occur naturally (e.g., as a result of unusually hot conditions), but not frequently. Following an excursion (an exceedance during the 10-day assessment period), the EPA recommends increasing the monitoring frequency to better understand the temporal and spatial nature of cyanotoxin occurrence in the affected waterbody.

In some waterbodies, longer-term HABs can persist for many weeks to months with conditions conducive to cyanobacterial proliferation. This can result in many excursions of the recommended toxin values during a recreation season. The EPA recommends that when more than three excursions (an exceedance during the 10-day assessment period) occur within a recreational season and that pattern reoccurs in more than one year, it is an indication the water quality is or is becoming degraded such that the water body no longer supports the recreational use. Recreational freshwaters at lower latitudes can have longer recreational seasons compared with those waters found at higher latitudes. For those waters in more temperate areas with a recreational season of approximately 100 days (i.e., from Memorial Day to Labor Day), three excursions could translate into a maximum of 30 percent of the recreational season not supporting the designated recreational use. Surface waters in areas with longer recreational seasons can also experience conditions that can support HAB proliferation and cyanotoxin occurrence for a longer period of the year. A maximum of three excursions across a recreational season reflects seasonal dynamics and occurrence patterns of HABs within years and the potential for adverse health effects over a short-term duration of exposure (i.e., approximately 30 days).

The EPA recognizes that multiple environmental factors can cause variability in bloom formation and toxin production, and that some years may produce HABs that occur for long periods, or HABs of shorter duration that occur repeatedly throughout a single recreational season, but such events may not occur every year. Therefore, the EPA concludes that it is appropriate to consider a pattern of multiple excursions within a recreational season as well as in multiple years (i.e., more than one year) when

determining whether the use is attained. It is important to note that the years with multiple excursions do not have to be consecutive to indicate a water quality problem. The upper-bound frequency (e.g., one year out of three years) is a risk-management decision that states need to determine when developing their water quality standards (WQS). States should include in their WQS the maximum number of years a pattern of cyanotoxin excursions can occur for the recreational use to remain supported.

The EPA does not recommend using a 10-day average concentration or a rolling average to determine an excursion, consistent with available toxicity information. States have flexibility in applying the 10-day assessment period. Some may choose to use pre-defined 10-day assessment periods for water bodies with a documented history of HAB occurrence or detection of elevated levels of cyanotoxins. Another approach is to begin the 10-day assessment period upon observation of a visible bloom. However, only considering the presence of visible blooms can miss episodes of elevated toxins (Raymond 2016). States are encouraged to consider the application of the frequency and duration components to capture elevated toxin concentrations, which may or may not coincide with the general proliferation of total cyanobacteria at high densities. More information on implementation of these values as criteria is provided in technical support materials.

6.4 Frequency and Duration for Swimming Advisory

Local and state governments can use swimming advisories to provide information to recreators on their potential exposure to cyanobacteria and their toxins. Some local and state governments currently post notifications for swimmers, in the form of advisories or warnings, when a cyanobacterial bloom is reported in recreational waters or when cyanotoxin levels exceed advisory thresholds. Table B-2 in Appendix B summarizes currently available information on state cyanotoxin-related guidelines and associated actions, including the issuance of swimming advisories.

The EPA recommends that the magnitude of the swimming advisory value not be exceeded on any single day, to provide timely information for people visiting beaches. The EPA also recommends that any exceedance of the recommended magnitude result in a swimming advisory being issued until the toxin concentration falls below the recommended magnitude. By increasing the monitoring frequency at a site where a swimming advisory is issued, water resources managers may get a clearer understanding of the temporal and spatial nature of water quality that can be useful in making decisions that protect the recreational use. Increased monitoring can also help water managers decide when to remove an advisory. The EPA has published materials for recreational water body managers that describe communicating risk to the public about cyanotoxins in recreational water bodies, monitoring, and responding to HABs (U.S. EPA 2017).

6.5 Recommended Recreational Criteria and Swimming Advisory for Microcystins and Cylindrospermopsin

The magnitude, duration, and frequency are summarized in Table 6-1.

Table 6-1. Recreational Criteria or Swimming Advisory Recommendations for Microcystins and Cylindrospermopsin[a]

Application of Recommended Values	Microcystins			Cylindrospermopsin		
	Magnitude (µg/L)	Duration	Frequency	Magnitude (µg/L)	Duration	Frequency
Recreational Water Quality Criteria	8	1 in 10-day assessment period across a recreational season	More than 3 excursions in a recreational season, not to be exceeded in more than one year[b]	15	1 in 10-day assessment period across a recreational season	More than 3 excursions in a recreational season, not to be exceeded in more than one year[b]
Swimming Advisory		One day	Not to be exceeded		One day	Not to be exceeded

[a] These recommendations can apply independently within an advisory program or in WQS. States can choose to apply either or both toxin recommendations when evaluating excursions within and across recreational seasons.

[b] An excursion is defined as a 10-day assessment period with any toxin concentration higher than the criteria magnitude. When more than three excursions occur within a recreational season and that pattern reoccurs in more than one year, it is an indication the water quality has been or is becoming degraded and is not supporting its recreational use. **As a risk-management decision, states should include in their WQS an upper-bound frequency stating the number of years that pattern can reoccur and still support its recreational use.**

The recommended magnitude represents the concentration of microcystins or cylindrospermopsin that is not expected to result in adverse human health effects from short-term recreational exposure to the toxins via incidental ingestion while swimming, based on exposure to young children. The adverse health effects include liver toxicity (for microcystins) and kidney toxicity (for cylindrospermopsin) and could result from exposures to waters containing elevated levels of these toxins.

The **water quality criteria** developed by the EPA describe the magnitude, duration, and the frequency of occurrence of pollutants. HABs may be caused or exacerbated by human activities and elevated nutrient concentrations, but cyanotoxins differ from other pollutants as they are not typically discharged into a water body. The EPA developed recommended criteria for these cyanotoxins that provide a magnitude (8 µg/L microcystins or 15 µg/L cylindrospermopsin) and duration (not to be exceeded in more than three 10-day assessment periods over the course of a recreational season). The EPA expects states to make an explicit risk management decision regarding the frequency (i.e., the number of years this pattern of exceedances can occur in the waterbody) and still support its recreational use.

As a basis for issuing a **swimming advisory**, the EPA recommends a concentration of 8 µg/L microcystins or 15 µg/L cylindrospermopsin not be exceeded on a single day. This is consistent with the goal of a swimming advisory to provide prompt information to people who wish to use the water body for recreation.

7.0 EFFECTS CHARACTERIZATION

7.1 Enhanced Susceptibility

Based on the available studies in animals, individuals with liver or kidney disease may be more susceptible to health effects than the general population as the detoxification mechanisms in the liver and impaired excretory mechanisms in the kidney may be compromised. Data from an episode in a dialysis clinic in Caruaru, Brazil, where microcystins (and possibly cylindrospermopsin) were not removed by treatment of dialysis water, identify dialysis patients as a population of potential concern in cases where the drinking water source was contaminated with cyanotoxins.

The data on red blood cell acanthocytes observed in animal studies of cylindrospermopsin suggest that individuals that suffer from anemia (e.g., hemolytic or iron-deficiency) might be a potentially sensitive population. Several rare genetic defects such as abetalipoproteinemia (i.e., a rare autosomal recessive disorder that interferes with the normal absorption of fat and fat-soluble vitamins from food) and hypobetalipoproteinemia are associated with abnormal red blood cell acanthocytes, which appears to result from a defect in expression of hepatic apoprotein B-100, a component of serum low-density lipoprotein complexes (Kane and Havel 1989). Individuals with either condition might be sensitive to exposure to cylindrospermopsin.

Available animal data are not sufficient to determine if there is a definitive difference in the response of males versus females following oral exposure to microcystins. Fawell et al. (1999) observed a slight difference between male and female mice in body weight and serum proteins, but no sex-related differences in liver pathology. Available animal data are not sufficient to determine if there is a definitive difference in the response of males versus females following oral exposure to cylindrospermopsin.

7.2 Recreational Exposure Duration

Recreational exposure data available in the literature are expressed in two primary ways: 1) the volume of water incidentally ingested during recreation (e.g., L/hr), and 2) the duration of the recreational activity (e.g., minutes of recreation per day). A daily incidental ingestion rate distribution was developed by combining these two distributions (for more information see Appendix E). The 90th percentile of the daily incidental ingestion rate distribution for children (see section 7.3) was selected for the derivation of the criteria and swimming advisories, consistent with the 2000 Human Health Methodology.

The EPA identified the following sources of data on the duration of the recreational activity: the EPA's EFH (2011); Schets et al. (2011); and DeFlorio-Barker et al. (2017) (DFB study). See Table 7-1a and Table 7-1b for summary overviews of these studies. One major difference between the studies is in the unit of exposure, reported in minutes per day in one study and minutes per swimming event in the two other studies.

Table 7-1a. Durations of Recreational Exposures in Minutes per Day

Reference	Recreational Environment	Age Group (Years Old)	Sample Size	Mean	Units
U.S. EPA *Exposure Factors Handbook* (2011)	In Outdoor Pool or Spa	1 to 4	9	85.6	minutes per day
		5 to 11	15	164.2	
		12 to 17	5	97.0	
		18 to 64	44	117.6	
		> 64	10	78.9	

Table 7-1b. Durations of Recreational Exposures in Minutes per Swimming Event[a]

Reference	Recreational Environment	Age Group (Years Old)	Sample Size	Mean	Units
Schets et al. (2011)	Freshwater	< 15 Years	1,689	79.0	minutes per event
		16+	4,123	54.0	
	Swimming Pool	< 15	1,689	81.0	minutes per event
		16+	4,123	67.5	
DeFlorio-Barker et al. (2017)	Freshwater	< 1	171	56	minutes per event
		1 to 3	1,061	66.7	
		4 to 7	1,738[b]	88.5	
		8 to 12	2,136[c]	92.9	
		13 to 18	1,855	64	
		19 to 34	5,478	45.4	
		35+	8,058	47	
	Marine	< 1	350	60.5	
		1 to 3	2,687	79.1	
		4 to 7	4,260	107.8	
		8 to 12	5,398	121.4	
		13 to 18	4,021	102	
		19 to 34	10,786	68.2	
		35+	19,745	66.9	

[a] Additional information is needed to translate minutes per event to minutes per day.

[b] Number of children ages 4–7 reported to have contact with water: 1,562.

[c] Number of children ages 8–12 reported to have contact with water: 1,901.

The EPA considered these three studies and selected the EFH for use in deriving the criteria and swimming advisories primarily because the EFH dataset represents exposures in minutes per day. Other datasets measured the duration of recreational exposure on an event basis, which require assumptions about how many recreational events occur per day to create the relevant distribution. The EPA conducted analyses comparing these datasets, as described below to evaluate the differences in the distributions given differences in sample size, and evaluated differences given different assumptions of number of events per day.

The EFH (U.S. EPA 2011) lists time spent per 24 hours in an outdoor spa or pool for different age groups (including children five to 11 years old). The EPA acknowledges that the reported sample size

for this study is small (n = 15) for the five-to-11-year-old group. Schets et al. (2011) demonstrate that time spent in swimming pools is similar to time spent in freshwater and therefore EPA concluded that these data are representative of recreational exposure in freshwater. The EFH also presents data for minutes spent "outdoors at a pool/river/lake." The EPA did not select these data as it is uncertain if this is time spent in the water, or total time "at" the location.

Schets et al. (2011) investigated swimming durations in freshwater, marine water, and pools. They surveyed 8,000 adults, 1,924 of whom also provided estimates for their eldest child (< 15 years of age) and found that children spend, on average, 25 minutes longer swimming in freshwater compared to adults. Schets et al. (2011) reported similar mean duration times between swimming pools and freshwater locations for children less than 15 years old (average of 81 and 79 minutes per event, respectively; upper 95 percent CI: 200 and 270 minutes per event, respectively).

The DFB study (DeFlorio-Barker et al., 2017) compiled self-reported swimming durations from epidemiological study surveys from 12 beaches in which participants were asked to estimate, in minutes, the total time they spent in the water. Parents or guardians were responsible for answering survey questions assessing exposures such as getting water in the mouth or swallowing water, on behalf of their minor children. The study results represent 2,136 children ages eight to 12 years and 1,738 children ages four to seven recreating in freshwater. Marine recreators spent more time in the water compared with freshwater recreators. The authors suggest that behaviors may have been influenced by the warmer water at most of the marine sites (California and Gulf Coast) compared with the freshwater sites in the Great Lakes.

Although not represented in Table 7-1 a or b, the EPA's OPP uses a different approach to estimate chemical exposures for children during pool swimming, for use in its SWIMODEL (U.S. EPA 2003). This model simulates short-term exposure using a high-end estimate of exposure-time per event to represent a maximum, one-time exposure. It also simulates intermediate/long-term exposure using a shorter event duration to represent an average of maximum and minimum exposures over time. Among competitive children swimmers, the short-term exposure duration used by the SWIMODEL is one hour per day for children ages six to 10 and two hours per day for children ages 11 to 15 years based on a survey of swim coaches (U.S. EPA 2003). The competitive swimming scenario (e.g., children swimming laps) is appropriate for conducting risk assessments of exposure to swimming pool chemicals. However, it is less relevant to children's recreational activities in lakes or rivers and therefore was not used in this assessment.

7.2.1 Comparison of Duration of Exposure Distributions

Because the DFB study has a much larger sample size compared to the study results reported in EPA's EFH, the EPA conducted a statistical analysis to compare the distributions of duration of exposure. Because the DFB study age groupings and the EFH age groupings do not exactly align, the EPA compared the four-to-seven and the eight-to-12 age groups from the DFB study with the five-to-11 age group presented in the EFH. Both studies include self-reported data, which are prone to recall bias. Adult recollection of their children's time spent in the water is also uncertain. However, there is no reason to believe there would be differential recall bias between the studies.

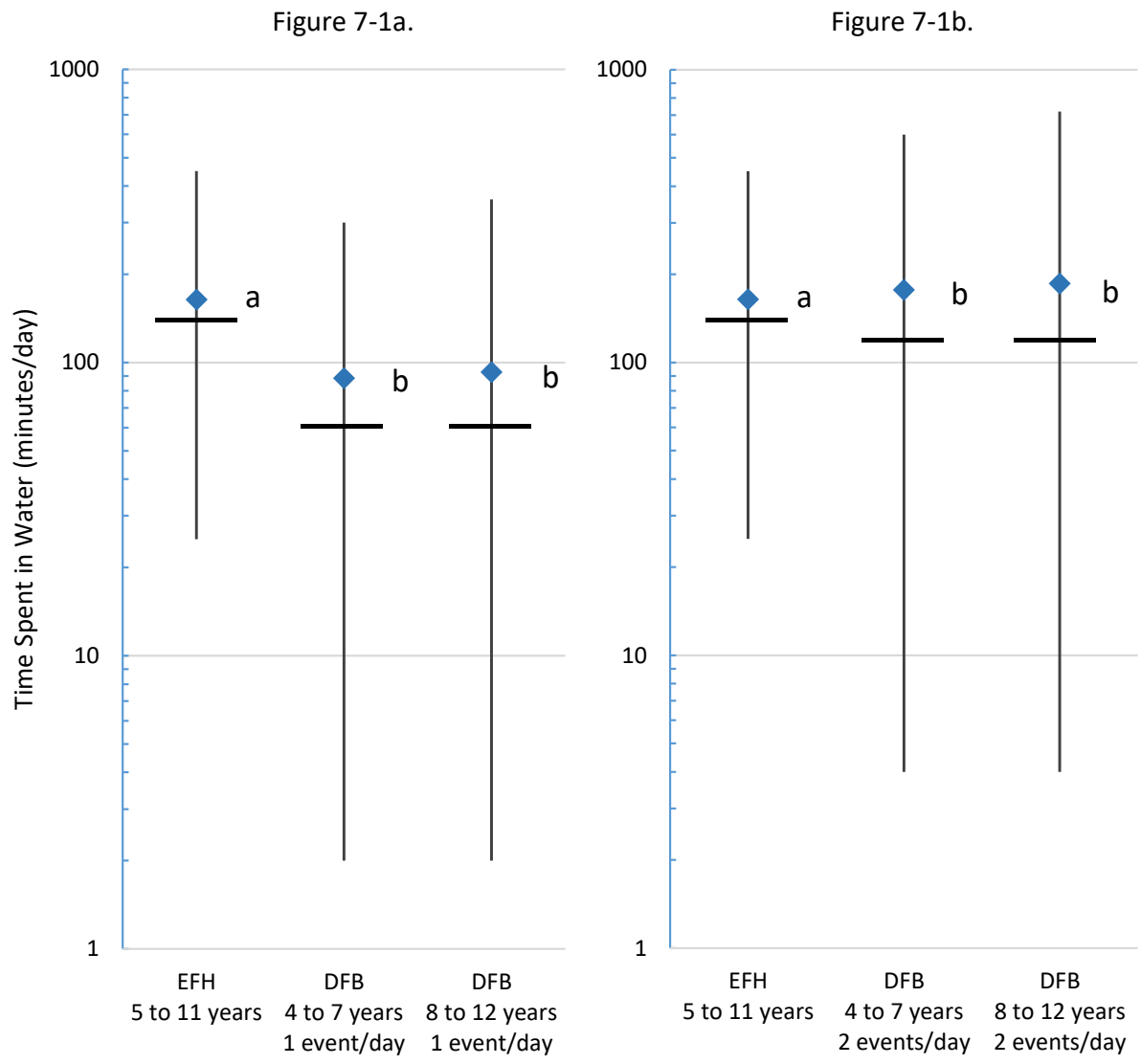
Table 7-2 shows the parameters used to create distributions for EFH and DFB studies. The EPA used assumptions of one swimming event per day and two events per day to translate the DFB duration from minutes per event to minutes per day for two different age groups. The EPA assumed the underlying distributions of exposure durations were log-normal. The observed mean and standard deviations in

Table 7-2 defined the parameters of the underlying log-normal distributions. The standard deviations take into consideration the numbers of samples, and therefore address differences in numbers of participants in the EFH and DFB studies. A large number (1 million) of samples were drawn from each log-normal distribution defined using these parameters. The distributions were truncated to reflect the observed maximum and minimum values in the EFH and DFB studies for the age groups of interest. Figures 7-1 a and b show the five resulting distributions: the EFH distribution and the DFB distributions assuming one (Figure 7-1a) and two (Figure 7-1b) events per day.

Table 7-2. Parameters Used to Fit Recreation Duration Distributions in Freshwater

Parameter Source	Age Group (sample size)	Mean (min/day)	Standard deviation	Minimum (min/day)	Maximum (min/day)
EPA 2011 EFH (minutes/day)	5 to 11 years (n = 15)	164.2	103.97	25	450
DFB 2017 (minutes/day, assuming one event/day)	4 to 7 years (n = 1,562)	88.5 (1 event)	62.8	2	300
DFB 2017 (minutes/day, assuming two events/day)		177 (2 events)	125.6	4	600
DFB 2017 minutes/day, assuming one event/day)	8 to 12 years (n = 1,901)	92.9 (1 event)	64.7	2	360
DFB 2017 minutes/day, assuming two event/day)		185.8 (2 events)	129.4	4	720

Figure 7-1 a and b. Comparison of Children's Duration of Time Spent Recreating



Comparison of children's time spent in water between EPA's 2011 *Exposure Factors Handbook* (five to 11 years old) (EFH; U.S. EPA 2011) and the DeFlorio-Barker study (DFB) (four to seven and eight to 12 years old) (DeFlorio-Barker et al. 2017) assuming one swimming event per day (Figure 7-1a) or two swimming events per day (Figure 7-1b) for the DFB data. The range of each distribution is represented by the vertical solid line, the short horizontal line indicates the median, and blue diamonds represent the mean. Letters beside the means denote significant differences of the means.

The EPA conducted two statistical tests to compare these distributions; one based on the means of the distributions and the other based on the full distributions. The full duration distribution, not the mean, in combination with the distribution of volume ingested per hour, was used to calculate the daily incidental ingestion rate. The EPA also explored how these comparisons change when one assumes that children engage in one or two swimming events per day (e.g., those who swam, took a break, and then re-entered the water at a later point in the day). The changes in the parameters are shown in Table 7-2.

For the comparison of the means, the EPA used a two-tailed t-test with unequal variances. The mean of the EFH is statistically different from both the DFB age group means (p-value < 0.001) for both one and two events per day. The means of the two DFB age groups are not statistically different from each other

(p-value = 0.08) assuming both one event and two events per day. Statistical differences between the means are denoted by letters (a and b) in Figure 7-1. Assuming two events per day for the DFB studies, the means for both DFB study age groups are significantly higher ($p < 0.001$) than the EFH mean. The larger sample size available in the DFB study results in a narrower confidence interval around the mean time spent in water, compared to the 95 percent CI for the mean used in the EFH.

For the comparison of the distributions, the EPA used the Kruskal-Wallis test. Results show that the EFH distribution is not statistically significantly different from either DFB age group distributions (p-value = 0.499, assuming one event per day; p-value = 0.498, assuming two events per day).

The EPA concluded that because the EFH and DFB distributions are not significantly different, the EFH dataset is the most appropriate for deriving criteria and swimming advisory values as it does not require additional assumptions about the number of swimming events that occur per day. The 90th percentile incidental ingestion rates are shown in Table 7-3 below for the EFH distribution and for the DFB distributions. The resulting 90th percentiles of daily incidental ingestion rate are also shown. The 90th percentile of daily ingestion rate based on the EFH distribution most closely corresponds to the 90th percentile of daily ingestion rate using the DFB dataset when two swimming events per day are assumed.

Table 7-3. Calculated Daily Incidental Ingestion Rates Based on EFH and DFB Datasets

Volume per Hour Data Source	Event Duration Data Source	Age Group (years)	Events per Day (if assumed)	90th Percentile Daily Ingestion Rate (L/day)
Recreational AWQC Appendix E full dataset (L/hr)	EPA *Exposure Factors Handbook* (2011)[a] (hr/day)	5 to 11	not needed	0.21
	DeFlorio-Barker et al. (2017) (DFB) (hr/event)	4 to 7	1	0.11
			2	0.23
		8 to 12	1	0.12
			2	0.24

[a] This distribution was used in the derivation of the criteria and recreational swimming advisories.

7.3 Evaluation of Health Protective Values for Different Lifestages

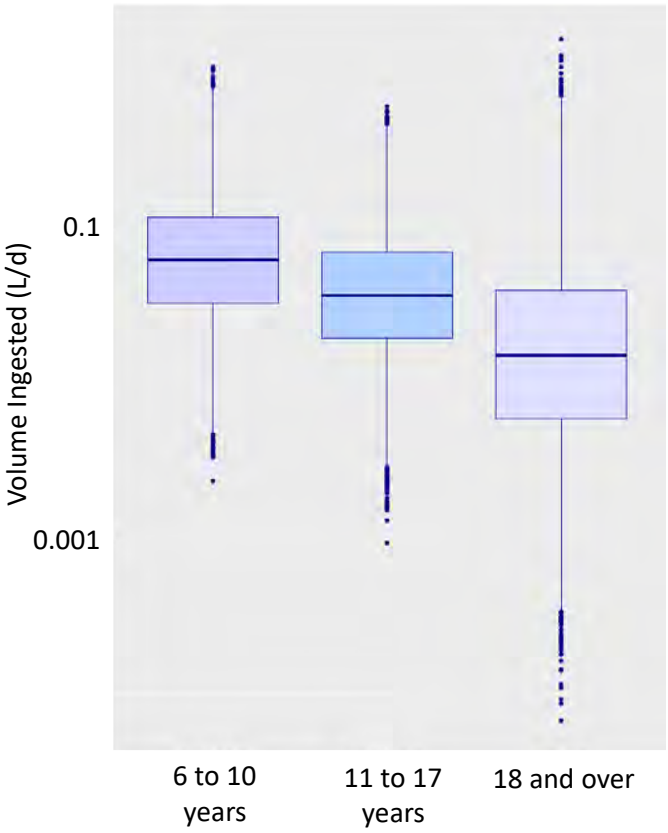
The EPA compiled and evaluated available information for various life stages before selecting children ages six to 10 years as the basis for the recreational criteria values or swimming advisory. This section discusses potential health protective values for children and adults (section 7.3.1) and focuses on exposures of younger children (less than six years) (section 7.3.2).

7.3.1 Consideration of Multiple Lifestages

The EPA used the Appendix E and the Dufour et al. (2017) dataset provided in U.S. EPA (2018a) to generate the box and whisker plots shown in Figure 7-2 for three life stages (children six to 10 years, children 11 to 17 years, and adults 18 years or older). The Appendix E Dufour data for volume ingested per swimming event was normalized to one hour. Each participant's volume ingested was adjusted to one hour based on the length of time that participant reported being in the water. The EPA converted volume of water ingested from L/event to L/hour, then used the swimming duration per day from the EPA's 2011 EFH (hours/day). The distributions were assumed to be log-normal and the plot is

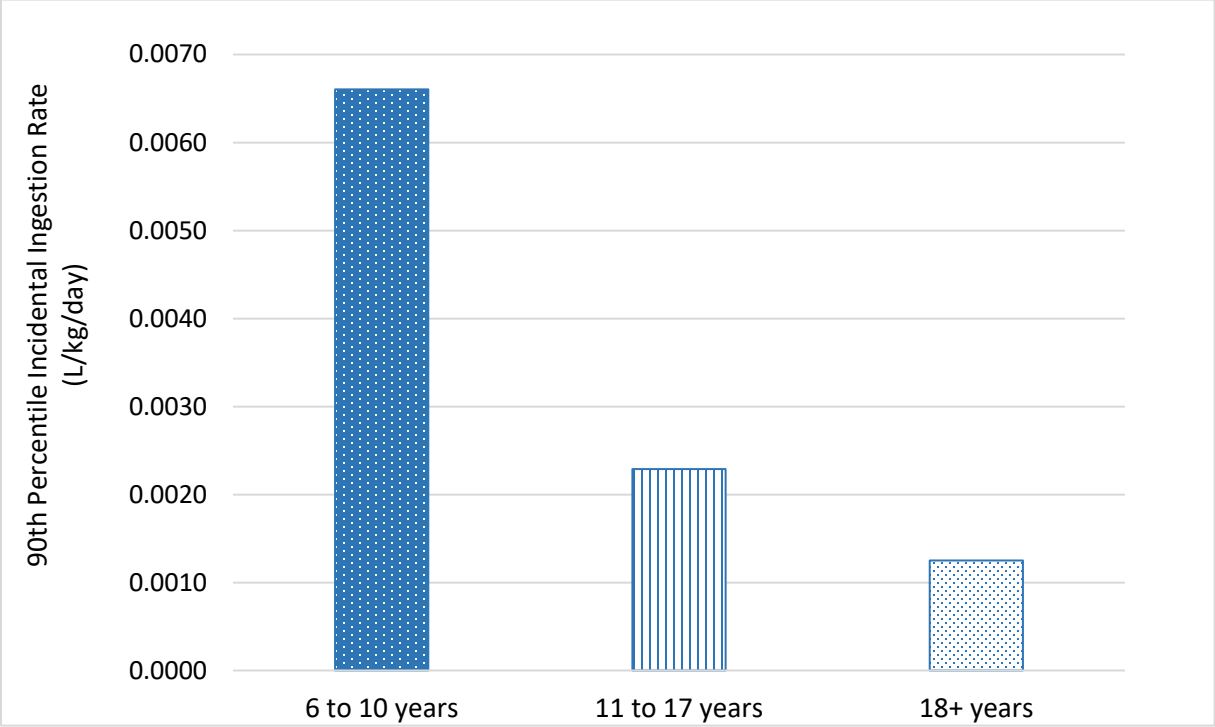
visualized in log space. The EPA used the Appendix E Dufour data on ingestion rate (shown in Figure 7-2) and the body weight estimates from the EPA's (2011) EFH (kg) to calculate the ingestion normalized by body weight (L/kg/day) shown in Figure 7-3.

Figure 7-2. Incidental Ingestion During Recreational Activity Based on Age (Appendix E)



In this box plot, the horizontal line the middle of the box is the median (Q2). The length of the box is the interquartile range (IQR) or the 25th percentile to the 50th percentile. The upper whisker vertical line extends to the greatest value less than or equal to Q3+1.5*IQR; the lower whisker extends to the smaller value less than or equal to Q1-1.5*IQR. The dots represent extreme values that are either greater than the upper whisker or lower than the lower whisker.

Figure 7-3. Comparison of Children and Adults Incidental Ingestion Rate During Recreational Activity Adjusted for Body Weight



Body weight varies by age. Table 8-1 in the EPA's EFH (U.S. EPA 2011) reported recommended statistics based on the 1999–2006 National Health and Nutrition Examination Survey. Table 7-4 shows the mean body weight for the age groups compared in this section (U.S. EPA 2011).

Table 7-4. Mean Body Weight by Age Group Based on U.S. EPA (2011)

Age Group	Body Weight (kg)
Children 6 to 10 years	31.8
Children 11 to 17 years	56.8
Adults 18 to 64	80

The EPA estimated recreational health protective values for these three different age groups for microcystins and cylindrospermopsin to demonstrate the variability due to body weight, recreational

water incidental ingestion, and exposure duration by lifestage. Inputs for these calculations are in Table 7-5.

Table 7-5. Inputs for Calculation of Protective Values for Microcystins and Cylindrospermopsin

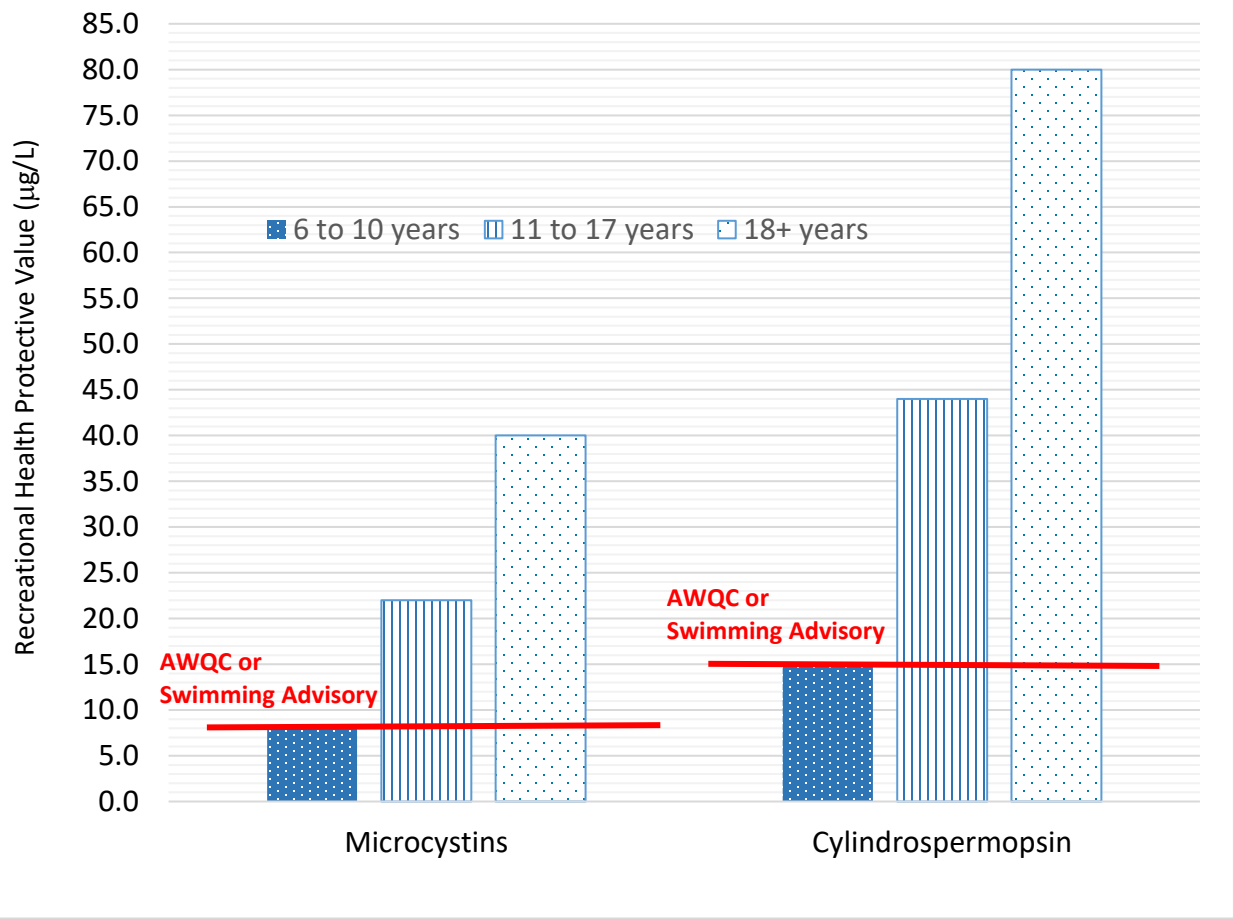
Age Group	Ingestion Rate [a] (L/day)	Body Weight [b] (kg)
Children 6 to 10 years	0.21	31.8
Children 11 to 17 years	0.13	56.8
Adults 18+ years	0.10	80.0

[a] Value is 90th percentile of the combined distribution (i.e., ingestion and duration data combined); see Appendix E.

[b] For children age 6 to 10 years, the mean body weight for the 6-to-10-year age group (31.8 kg) was used. For 11 to 17 years, the mean body weight for the 11- to 15-year-old age group (56.8 kg) was used because it was the closest age group available from the EPA's *Exposure Factors Handbook* (U.S. EPA 2011). For adults 18+ the mean body weight for the 21+ year age group (80 kg) was used (U.S. EPA 2011).

As illustrated in Figure 7-4, the AWQC and swimming advisories the EPA calculated to be protective of children ages six to 10 years are also protective of older children and adults.

Figure 7-4. Comparison of Calculated Recreational Health Protective Values for Microcystins and Cylindrospermopsin for Children, Older Children, and Adults



7.3.2 Exposure Factors for Children Younger Than Six Years Old

In the calculation of the cyanotoxin values reported in section 6, the EPA utilized exposure parameters reported in the EFH (U.S. EPA 2011) and peer-reviewed study data (study design presented in Dufour et al. 2017; data analyzed in Appendix E; U.S. EPA 2018a). The available incidental ingestion volume and exposure duration values from the Appendix E and the EPA's EFH (U.S. EPA 2011), respectively, were limited to specific age ranges. For incidental ingestion, the data reported were limited to children six years old and older because the Dufour et al. (2017) study design did not include children younger than six years. The EPA's EFH (U.S. EPA 2011) provided a mean recreational exposure duration for children ages one to four years (1.4 hour/day). This duration is shorter than the mean duration for children ages five to less than 11 years (2.7 hour/day). Values for exposure duration were not available for children younger than one year.

The EPA found one other study that characterized incidental ingestion for children. Schets et al. (2011) reported incidental ingestion volumes and durations of recreational events for children ages zero to < 15 years. However, the study did not further divide this cohort into younger children and older children. The incidental ingestion data for children < 15 years represent parental estimates of volumes of freshwater incidentally ingested by their children, which is a different methodological approach compared to the more quantitative approach used by Dufour et al. (2017). The exposure durations were also parental estimates.

The EPA calculated the 90th percentile incidental ingestion rate per day for children younger than six years old in order to compare the daily ingestion rate (L/day) between children six to 10 years and those younger than six years. The daily ingestion rate (0.21 L/day) used to derive the recreational criteria was calculated by combining the distributions for incidental ingestion and exposure duration via a probabilistic (Monte Carlo) analysis (described in section 4.2.3.1). The daily ingestion rate for children younger than six years old (0.11 L/day) was a mixed-age estimate calculated by dividing the 90th percentile for incidental ingestion for children age six to 10 years (0.077 L/hour; see Appendix E) by the mean exposure duration for children one to four years (1.4 hour/day; U.S. EPA 2011). The daily ingestion rate for children younger than six years old is lower than for children six to 10 years old. This calculation was also performed using data from Schets et al. (2011) and resulted in a daily ingestion rate of 0.1 L/day.[6] The EPA evaluated the effect of using parameter values for children younger than six years by including an age-specific body weight and the mixed-age estimate for the daily ingestion rate (L/day) parameters. Table 7-6 shows a comparison of the microcystins magnitude for the two different age groups, children ages six to 10 years and children ages one to less than six years.

The estimates for children younger than six years have large uncertainties given the lack of measured incidental ingestion data specifically for this age group. Information on exposure durations for children less than one-year-old is also lacking. Because exposure durations are greatest for five- to 11-year-olds, the EPA concluded that calculating the ingestion rate using a higher duration was protective of children younger than six years old. Research designed to fill this data gap could be helpful for characterizing the risks to children younger than six years old. Specifically, data to better characterize the volume of water ingested during recreational events would enhance EPA's confidence that the criteria values are protective of children younger than six years old.

[6] This number was calculated as follows: 0.07 L/hour (90th percentile ingestion volume for age zero to less than 15 years from Schets et al. (2011)) divided by 1.4 hr/d (mean for children one to four years from U.S. EPA 2011).

Table 7-6. Microcystins Magnitude Comparison Between Children Six to 10 and Children One to Less Than Six Years Old

Age Group	RfD (µg/kg/day)	Body Weight (kg)	Ingestion Rate (L/day)	Magnitude (µg/L)	Magnitude (µg/L) Rounded
6 to 10 years	0.05	31.8	0.21	7.57	8
1 to < 6 years	0.05	15.6 [a]	0.11	7.09	7

[a] This value is the weighted mean of the age groups one to less than two years, two to less than three years, three to less than six years (U.S. EPA 2011).

7.4 Other Recreational Exposure Pathways

The EPA selected primary contact activities and incidental ingestion of water as the primary exposure pathway for derivation of the recreational criteria and swimming advisories. Inhalation and dermal toxicity data were not available; however, there are limited available data to estimate inhalation and dermal exposure. The EPA conducted analyses to compare inhalation and dermal exposure to incidental ingestion of the cyanotoxins while recreating. Section 7.4.1 compares recreational ingestion and inhalation exposures to microcystins. Similarly, section 7.4.2 compares recreational ingestion and dermal exposure. Section 7.7 briefly discusses tribal considerations. Further research is needed to better understand the toxicity from inhalation and dermal exposure to cyanotoxins. The EPA describes the screening analyses in this section because sufficient data to quantify toxicity via these routes were not available.

7.4.1 Inhalation of Cyanotoxins

Volatilization of microcystins and cylindrospermopsin from water to air is not expected due to their size and charges. Both cyanotoxins are rather large molecules compared to volatile chemicals. Microcystin-associated acid groups are charged at the pH of normal surface waters. Cylindrospermopsin features both negative and positive changes and, like other zwitterions, do not volatilize significantly into the air from water (Butler et al. 2012).

According to Wood and Dietrich (2011), waterborne cyanotoxins can be aerosolized through a bubble-bursting process, in which the cyanobacteria and cyanotoxins are ejected and carried into the air where they can bind to particulate matter. Microcystins that are free or bound to particulate matter in air can be deposited into the deepest bronchiolar or alveolar cavities; air borne cyanobacterial cells from aerosolized water droplets would likely be deposited in the upper respiratory tract (Wood and Dietrich 2011).

The EPA identified field studies that measured recreators' exposure levels to aerosols containing microcystins from lakes with blooms containing microcystin-producing *Microcystis aeruginosa*. The studies found low inhalation exposures. In one study, Backer et al. (2008) used personal air samplers in a three-day study of recreational activities in a lake with a cyanobacterial bloom, either carried by the study participant or placed on the participant's boat. The microcystin concentrations in air ranged from below the limit of detection (0.0037 ng/m^3) to 0.456 ng/m^3. Backer et al. (2010) also detected microcystins in ambient air for one day, at one lake, and only from the shoreline sampler. The average air concentration was 0.052 ng/m^3. Backer et al. (2010) also collected 44 personal air samples, which ranged from the limit of detection (0.1 ng/m^3) to 0.4 ng/m^3. The study identified no associations between health effects and microcystin concentrations from inhalation exposure from activities that included swimming, water skiing, Jet Skiing, or boating. The authors noted that the daily mean microcystin

concentrations in personal air samples did not correlate with the concentrations of *Microcystis aeruginosa* cells, dissolved microcystins, or total microcystins in the sampled lake water.

In another study by Backer et al. (2010), the lakes had a wider range of concentrations of microcystins (< 10 to > 500 µg/L). The study authors measured microcystins exposure via personal air samplers, nasal swabs, and blood samples for individuals whose activities included swimming, boating, tubing/wakeboarding, riding watercraft, wading, and fishing at the lakes. They found low microcystin levels in personal air samplers below the limit of detection (0.1 ng/m^3) to 2.89 ng/m^3 and also in nasal swabs below the limit of detection (0.1 ng) to 5 ng. The average aerosolized microcystin concentration was approximately 0.3 ng/m^3. Based on the nasal swab data, the investigators estimated on average that the adults inhaled 0.8 ng of microcystins. Microcystin concentration in the water-soluble plasma fraction of the study subjects was also below the limit of detection (1 µg/L). The investigators cautioned that microcystin might be bound to a protein component in the blood or sequestered in liver tissue.

Wood and Dietrich (2011) studied Lake Rotorua (New Zealand) when it was experiencing a dense bloom of microcystin-producing *Microcystis*. The authors measured a maximum microcystin concentration in the water of 2,140 µg/L and air concentrations from 0.0003 to 0.0018 ng/m^3.

Cheng et al. (2007) used high volume and personal air samplers to measure microcystins in the air at a lake with a cyanobacterial bloom. The authors measured low microcystin concentrations in the water (approximately 1 µg/L) and air concentrations ranging from below the detection limit (0.02 ng/m^3) to 0.08 ng/m^3.

The EPA performed a screening analysis to characterize potential relative exposures. The EPA analyzed the relative potential dose of the cyanotoxins via inhalation exposure compared to oral ingestion to evaluate if recreational criteria values or swimming advisories based on ingestion could be protective of the other exposure routes. Although the recreational use is primary contact recreation, such as swimming, data are available for secondary contact activities such as Jet Skiing or boating and white-capped wave, bubble-bursting action, which can result in cyanotoxins becoming aerosols (microscopic liquid or solid particles suspended in air).

Using the information from Cheng et al. (2007) and inhalation exposure parameters provided in the EPA's EFH (2011), the EPA compared the estimated microcystin ingested dose to the inhaled dose. The first step in this comparative screening analysis was to calculate the incidental ingestion dose using the following equation:

$$\text{Ingestion dose (ng/day)} = \text{Ingestion rate} \times \text{Concentration}_{\text{water}}$$

Where:

Ingestion rate = 90th percentile incidental ingestion rate based on combined distributions of incidental ingestion (Appendix E) and recreational duration (U.S. EPA 2011) (L/day)

$\text{Concentration}_{\text{water}}$ = assumed concentration in water (1,000 ng/L from Cheng et al. (2007)) (ng/L)

The parameters used in the calculation of the estimated ingestion dose for each age group are presented in Table 7-7.

Table 7-7. Ingestion Parameters and Estimated Ingestion Dose for Screening-level Comparative Inhalation Exposure Analysis

Age Group	Ingestion Rate (L/day) [a]	Concentration in Water (ng/L) [b]	Ingestion Dose (ng/day)
Children	0.21	1000	210
Adults	0.10	1000	100

[a] Daily recreational incidental ingestion rate calculated in combined distribution analysis for children and adults as described in section 4.2.3.1.

[b] Cheng et al. (2007) measured 0.08 ng/m^3 in air near surface waters with a concentration of 1 µg/L microcystins. This concentration in water was assumed as part of this analysis because Cheng et al. (2007) provided aerosolized levels given a specific concentration in water.

The second step in the comparative screening analysis was to estimate the inhaled dose using the following equation:

$$\text{Inhalation dose (ng/day)} = \text{Inhalation rate} \times \text{Inhalation duration} \times \text{Concentration}_{air}$$

Where:

Inhalation rate = inhalation rate from the EPA's EFH (U.S. EPA 2011; Table 6-2) (m^3/min)

Inhalation duration = inhalation exposure duration from the EPA's EFH (U.S. EPA 2011; Table 16-20) (minutes/day)

$\text{Concentration}_{air}$ = concentration in air (0.08 ng/m^3) assumed from Cheng et al. (2007) (ng/m^3)

The inhalation exposure parameters the EPA used in this equation and the resulting estimated inhaled dose are listed in Table 7-8. The EPA selected inhalation rates for children and adults from the EPA's EFH (U.S. EPA 2011). For this conservative comparative analysis, the EPA selected the highest 95th percentile short-term, moderate intensity activity level inhalation rate—the volume of air inhaled per minute (m^3/minute)—listed for children and adults in EPA's EFH Table 6-2 "Recommended Short-

Table 7-8. Inhalation Exposure Parameters and Estimated Inhaled Dose

Age Group	Inhalation Rate (m^3/min)[a]	Duration of Inhalation Exposure per Day (minutes/day)[b]	Daily Inhalation Rate Adjusted for Duration of Exposure (m^3/day)	Concentration in Air (ng/m^3)[c]	Estimated Inhalation Dose (ng/day)
Children	0.037	560	21	0.08	1.7
Adults	0.04	511	20	0.08	1.6

[a] The EPA's *Exposure Factors Handbook* (EFH; U.S. EPA 2011) did not report recommended short-term, moderate intensity activity level inhalation rate values for children or adults in aggregate; used highest inhalation rate listed for children and adult age groups for this conservative screen. For children, it was the age group 16 to < 21 years, and for adults, it was 51 to < 61 years.

[b] Values are the longest 90th percentile duration reported for child and adult age groups in the EPA's EFH (U.S. EPA 2011) from Table 16-20 "Time Spent (minutes/day) in Selected Outdoor Locations, Doers Only, Outdoors at a Pool/River/Lake." The child and adult age groups with the longest durations spent near or in the water were children 1 to 4 years old and adults 18 to 64 years old.

[c] Cheng et al. (2007) measured 0.08 ng/m^3 in air near surface waters with a concentration of 1 mg/L microcystins.

Term Exposure Values for Inhalation (males and females combined)." The child and adult age groups with the highest of these inhalation rates were 16 to < 21 years and 51 to < 61 years, respectively.

To estimate the amount inhaled in a day, the EPA multiplied the inhalation rates for children and adults by an estimated daily inhalation exposure duration for each of these age groups. The EPA estimated daily inhalation exposure duration using a different dataset from the set it used for the incidental ingestion analysis (described in section 4.2.3.1). This was because people do not need to enter the water to be exposed via inhalation, they only need to be *near* or *at* the water. In contrast, recreators who incidentally ingest water while swimming must be *in* the water.

The EPA's EFH (U.S. EPA 2011) provides in Table 16-20 the time spent (in minutes/day) outdoors at a pool/river/lake. The EPA estimated inhalation exposure duration using the number of minutes per day spent outdoors at a pool/river/lake (U.S. EPA 2011). The EPA selected the longest 90th percentile duration values reported for child and adult age groups. The child and adult age groups with the longest times spent outdoors at a pool/river/lake were children one to four years old and adults 18 to 64 years old.

A comparison of the EPA's EFH data provided for time spent outdoors at a pool/river/lake and time spent in the water indicates that all age groups spent more time at a pool/river/lake than they spend in a pool/spa (U.S. EPA 2011). Consistent with the trend that children have longer durations of recreation in water than adults, children's time spent near recreational waters was greater than adults. The children's age group exposure patterns differed between the datasets. The data suggest younger children (one to four years) spend more time at recreational waters compared to school-aged children (five years and older), but children five to 11 years old spend more time in the water compared to other children (U.S. EPA 2011).

It is reasonable that younger children spend more time engaged in activities *at* a pool/river/lake compared to time spent recreating in recreational waters. The EPA selected this dataset to characterize inhalation exposure because younger children can spend more time playing on a beach, where they can be exposed to aerosolized cyanotoxins, than in the water where incidental ingestion can be the primary route of exposure.

The final step for this comparative screening analysis was to compare the ingestion and inhalation doses. The results are presented in Table 7-9. Using conservative assumptions for inhalation rates and inhalation exposure duration and comparing with daily incidental ingestion rates, the ingested dose is estimated to be higher than the estimated inhaled dose for children and adults. This analysis is for screening only and is highly uncertain. Further research is needed to better understand the toxicity from inhalation exposure to cyanotoxins.

Table 7-9. Results of Screening Analysis Comparing Ingestion and Inhalation Doses

Age Group	Ingestion Dose (ng/day)	Inhalation Dose (ng/day)
Children	210	1.7
Adults	100	1.6

[a] Calculations used unrounded parameters; results slightly differ with rounded values.

This analysis supports the conclusion that the inhaled dose can be much less than the incidental ingestion dose while recreating. The studies conducted by Backer et al. (2008, 2010) found low microcystin levels in aerosols above lakes with low or high microcystin concentrations and did not detect microcystin levels in the blood of study participants. In an animal study, no clinical signs or

effects on body or organ weights were observed after exposure to microcystin-LR aerosol (Benson et al. 2005). The EPA did not conduct a similar analysis for cylindrospermopsin because published measured air concentration data for this cyanotoxin were not available.

The California Environmental Protection Agency (CalEPA) came to a similar conclusion for water skiers (Butler et al. 2012). They cited Cheng et al. (2007) and noted that their results showed that a liter of water contains 700,000 to 800,000 times the amount of cyanotoxins as in a cubic meter of air. CalEPA calculated that this concentration is equivalent to 1.3 to 1.4 µL aerosolized microcystins/m^3. Compared to the ingestion assumptions used for swimmers in the calculation of their recreational guideline (i.e., 50 mL/hour), CalEPA calculated that a water skier would have to inhale at least 35,000 m^3/hour while skiing to achieve a dose equal to the swimmer, which is 17,000 times the inhalation rate of a marathon runner. CalEPA concluded that a water skier would not inhale enough aerosol to receive a dose similar to what a swimmer gets from ingestion.

Another comparison considers spray exposures from personal watercraft and boat spray. Sinclair et al. (2016) modeled a water-spray exposure scenario and observed much lower exposures than those resulting from swimming or limited contact recreational activities reported in the previous study. Thus, the EPA expects that the comparison above based on exposure from secondary contact recreation is protective of primary contact recreation. Sinclair et al. (2016) also measured urinary concentrations of cyanuric acid after 26 participants' exposure to spray in a simulated 10-minute car wash situation. Subjects wore a protective coverall with hood, vinyl gloves, waterproof footwear, and safety glasses to ensure that only their face and mouths were exposed. The estimated median and 90th percentile ingestion volumes were 0.18 and 1.89 mL, respectively. Converted to a duration of one hour, the amounts would be 1.08 mL and 11.3 mL, which are much lower than the incidental ingestion intakes per hour.

7.4.2 Dermal Absorption

The EPA did not find any peer-reviewed measured data for microcystins or cylindrospermopsin dermal absorption. The EPA's *Dermal Exposure Assessment: A Summary of EPA Approaches* (U.S. EPA 2007) states that to get through the skin, a chemical must dissolve into the stratum corneum, which is a stabilized lipid barrier; therefore, lipid solubility is required initially (U.S. EPA 2007).

The EPA performed a comparative screening analysis to estimate the potential dermal absorbed dose of microcystins and compare it to the incidentally ingested dose. The first step in this comparative screening analysis was to calculate the incidental ingestion dose using the following equation:

$$\text{Ingestion dose} = \text{Ingestion rate} \times \text{Concentration}_{\text{water}}$$

Where:

Ingestion rate = 90th percentile incidental ingestion rate based on combined distributions of incidental ingestion (Appendix E) and recreational duration (EFH; U.S. EPA 2011) (L/day)

$\text{Concentration}_{\text{water}}$ = concentration in water assumed as the health protective value the EPA derived in this document for microcystins (mg/L)

The parameters used in the calculation of the estimated ingestion dose are presented in Table 7-10.

Table 7-10. Ingestion Parameters and Estimated Ingestion Dose for Screening-level Comparative Dermal Absorption Exposure Analysis

Ingestion Rate (L/day)[a]	Chemical Concentration in Water (mg/L)[b]	Ingestion Dose (mg/day)
0.21	0.008	0.002

[a] Daily recreational incidental ingestion rate calculated in combined distribution analysis for children and adults as described in section 4.2.3.1.

[b] Concentration in water assumed to be the health protective value for microcystins the EPA derived in this document, converted to mg/L.

To estimate the potential dermal absorbed dose, the EPA used exposure equations in its *Risk Assessment Guidance for Superfund* (U.S. EPA 2004). The first step was to use chemical-specific octanol-water partition coefficient and molecular weight values to estimate dermal permeability, a parameter needed for the equation to estimate dermally absorbed dose. Octanol-water partition coefficients are available for four microcystins, including microcystin-LR. Ward and Codd (1999) estimated the log octanol-water partition coefficients of microcystin-LR, -LY, -LW and -LF using high performance liquid chromatography (HPLC) as 2.16, 2.92, 3.46, and 3.56, respectively. The EPA could not estimate cylindrospermopsin dermal absorption due to the lack of these lipophilicity parameters.

The equation to estimate skin permeability coefficient from U.S. EPA (2004) is:

$$\text{Log } K_p = -2.80 + 0.66 \times \log K_{ow} - 0.0056 \times MW$$

Where:

K_p = dermal permeability coefficient of compound in water (cm/hour)

K_{ow} = octanol-water partition coefficient from Ward and Codd (1999) (dimensionless)

MW = molecular weight (g/mole)

The chemical-specific dermal exposure parameters used to estimate skin permeability are listed in Table 7-11.

Table 7-11. Parameters Used to Estimate Skin Permeability of Microcystins

Microcystin Congener	Log K_{ow}[a]	Molecular Weight (g/mole)	Skin Permeability Coefficient (Log K_p)	Skin Permeability Coefficient (K_p) (cm/hour)
Microcystin-LR	2.16	995.17	-6.95	1.1×10^{-7}
Microcystin-LY	2.92	1002.16	-6.48	3.3×10^{-7}
Microcystin-LW	3.46	1025.2	-6.26	5.5×10^{-7}
Microcystin-LF	3.56	986.16	-5.97	1.1×10^{-6}

[a] Ward and Codd (1999)

The equation to estimate dermal absorbed dose for highly ionized organic chemicals from U.S. EPA (2004) is:

$$\text{Dermal absorbed dose} = K_p \times \text{Concentration}_{water} \times t$$

Where:

Dermal absorbed dose	=	dermal absorbed dose per event (mg/cm^2-event)
K_p	=	dermal permeability coefficient of compound in water (cm/hour)
$\text{Concentration}_{water}$	=	chemical concentration in water (mg/cm^3)
t	=	event duration (hour/event)

The exposure parameters and estimated microcystins absorbed dose based on these calculations are presented in Table 7-12.

Table 7-12. Dermal Absorption Exposure Parameters and Estimated Dermal Absorbed Dose

Microcystin Congener	Chemical Conc. in Water (mg/cm^3)[a]	Event Duration[b] (hour/event) (mean for 5- to 11-year-olds)	Dermal Absorbed Dose per Event (mg/cm^2-event)	Total Body Surface Area (cm^2)[c]	Dermal Absorbed Dose per Event (mg/event)
Microcystin-LR	8×10^{-6}	2.7	2.4×10^{-12}	14,800	3.6×10^{-8}
Microcystin-LY			7.1×10^{-12}		1.0×10^{-7}
Microcystin-LW			1.2×10^{-11}		1.8×10^{-7}
Microcystin-LF			2.3×10^{-11}		3.4×10^{-7}

[a] Concentration in water assumed to be the health protective value for microcystins the EPA derived in this document, converted to mg/cm^3.

[b] Event duration is defined as time spent per day in outdoor pool or spa at home as reported in the EPA's EFH (U.S. EPA 2011).

[c] Value is 95th percentile Children 6 to 10 years from U.S. EPA (2011), converted to cm^2.

The final step for this comparative screening analysis was to compare the ingestion and dermal absorbed doses. The results are presented in Table 7-13. The estimated ingested dose is higher than the estimated dermal absorbed dose for children. This assessment is highly uncertain. Further research is needed to better understand the toxicity from dermal exposure to cyanotoxins.

CalEPA also concluded dermal absorption of microcystins and cylindrospermopsin while swimming is not expected to be significant due to the large size and charged nature of these molecules (Butler et al. 2012). CalEPA eliminated the dermal absorption pathway from its risk assessment of microcystins and cylindrospermopsin citing evidence that similarly large molecules such as antibiotics have not been able to be formulated in a way to penetrate the skin (Butler et al. 2012). A U.S. Army-contracted in vitro study by Kemppainen et al. (1990) measured microcystin dermal penetration in 48 hours through excised human abdominal skin and found 0.9 (± 0.3) percent of the total dose in water penetrated through the skin; however, this study has not been peer reviewed.

Table 7-13. Results of Screening Analysis Comparing Ingestion and Dermal Absorbed Doses

Microcystin Congener	Ingestion Dose (mg/day)	Dermal Absorbed Dose[a] (mg/event)
Microcystin-LR	0.002	3.6×10^{-8}
Microcystin-LY	0.002	1.0×10^{-7}
Microcystin-LW	0.002	1.8×10^{-7}
Microcystin-LF	0.002	3.4×10^{-7}

[a] Calculations used unrounded parameters; results slightly differ with rounded values.

7.5 Cyanobacterial Cells

Cyanobacteria are associated with two distinct types of stressors, as described in the conceptual model, section 4.1. The first type of stressor are the toxins (microcystins and cylindrospermopsin) produced by the cyanobacteria. Section 3 of this document discusses the nature of these stressors and section 5 discusses related health effects endpoints. These stressors are the basis of the recreational criteria and swimming advisories. The second type of stressor is cyanobacterial cells. At this time, available data are insufficient to develop quantitative recreational values for total cyanobacterial cell density related to inflammatory health endpoints. However, various state and international agencies use total cyanobacterial cell densities in decision-making to determine water quality and to post recreational warnings to the public.

Exposure to cyanobacteria cells in ambient waters is associated with numerous inflammatory health endpoints, including: rashes, respiratory and GI distress, and ear and eye irritation. These effects can be the result of direct contact with bioactive compounds in the cyanobacteria (also referred to as "endotoxins"), or by contact with cyanobacteria-associated microbial commensals via dermal, oral, or inhalation exposure routes (Eiler and Bertilsson 2004; Gademann and Portmann 2008). Section 7.5.1 and Appendix D provide more information about the health effects associated with exposure to cyanobacteria cells based on the scientific literature and related uncertainties. Section 7.5.2 presents information about the use of total cyanobacteria, or other biomass metrics, as an indicator of potential hazard associated with cells or cyanotoxins. Gene-based enumeration methods, satellite remote sensing and uncertainties related to use of cells as indicators are also described. Section 7.5.3 discusses guidelines that use total cyanobacterial cell density as an indicator for toxin presence, quantification of toxigenic cells, and an approach providing cell density estimates related to the recommended 304(a) cyanotoxin criteria.

7.5.1 Health Effects Associated with Cyanobacterial Cells and Uncertainties

Various health studies, described in more detail in Appendix D, relate recreational exposure to increasing densities of cyanobacterial cells with increased incidence of specific health endpoints that can be described as acute inflammatory or allergenic reactions. The EPA identified epidemiological studies, clinical studies, and recreational water outbreak reports in searches of the publicly available and peer-reviewed scientific literature that characterize the human health effects associated with recreating in surface waters where cyanobacteria were present (see Appendix D).

The epidemiological studies provide evidence for statistically significant associations between cyanobacterial cell densities and possible inflammatory or allergenic health endpoints:

- Pilotto et al. (1997) reported a significant association with the occurrence of one or more symptoms, such as skin rashes, eye irritation, ear irritation, gastrointestinal distress, fever and respiratory symptoms, and exposure to greater than 5,000 cells/mL for more than one hour. In discussing the significance of the trend of increasing symptom occurrence and with the 5,000 cells/mL cut point, Pilotto et al. (1997) specifically suggested that the 20,000 cell/mL threshold might be too high to be adequately protective of recreators.

- Stewart et al. (2006d) found a significant increase in the inflammatory health effects associated with recreators exposed to > 100,000 total cyanobacteria/mL or a total cyanobacterial surface area > 12 mm^2/mL.

- Lévesque et al. (2014) observed a significant increase in GI symptoms associated with recreational contact. The increase in GI symptoms was significant in the > 20,000-cells/mL and > 100,000-cells/mL categories, and the positive trend for increasing illness with increased total cyanobacterial cell densities also was significant at p-value = 0.001.

- Lin et al. (2015) reported significant associations between respiratory symptoms and exposure to the 25th to 75th percentile range of cyanobacterial cells excluding picocyanobacteria (range 37–237 cells/mL) and between reported respiratory, rash, and earache symptoms and exposure to the highest quartile (range 237–1,461 cells/mL). The 1,461-cells/mL value was the highest cell density observed in that study (Lin et al. 2015).

- Lévesque et al. (2016) reported a significant trend of increasing of GI illness in recreators associated with exposure to the concentration of endotoxins. The authors noted a positive correlation between endotoxin concentrations and total cyanobacterial counts. Relative risks for GI illness were higher for families that also received drinking water from the lakes studied or from wells under the influence of surface water contamination. There was no relationship between GI illness and exposure to *E. coli*. Relative risks also increased for recreators engaged in full (e.g., swimming, water skiing, diving, etc.) or limited (e.g., fishing, use of watercraft) contact recreation and adjustment for the level of exposure did not alter the health relationship.

The variability in the reported epidemiological associations in these studies in both the range of cyanobacterial cell densities reported and specific symptomologies characterized limited identification of a discrete cyanobacterial cell density value associated with a consistent level of effect. Some researchers have suggested that the lack of a described dose-response characterizing cell-related inflammatory health effects could suggest a "threshold" rather than a specific dose-response relationship (Cochrane et al. 2015; Stewart et al. 2006b). Allergy is an example of a threshold mechanism, meaning that there is a level of exposure (i.e., a threshold value) below which the development of sensitization and the elicitation of an allergic reaction will not occur. Defining accurate numerical values for threshold exposure levels is difficult due to lack of validated methods and uncertainties about the mechanism of sensitization (Cochrane et al. 2015).

Scientists investigating the health effects posed by cyanobacteria have pointed out factors that contribute to the epidemiological variability observed and uncertainties in determining what level of cyanobacterial cells result in a specific level of inflammatory responses. For example:

- There are differing cyanobacterial community composition and proportions of the more allergenic, non-cyanotoxin-producing strains relative to the cyanotoxin-producing strains at each

site. Researchers have reported non-toxin-producing strains can be more allergenic compared to toxin-producing strains (Torokne et al. 2001).

- There is variability in sensitivity in the study populations.
- There are differences among the specific sites studied.
- The limited size of some studies could have affected the ability to detect significantly increased rates of illness in individual symptom categories (Pilotto et al. 1997; Stewart et al. 2006b). Small sample size diminishes the statistical power of the study and the ability to detect an association if one exists (Rothman et al. 2008).
- The incomplete characterization or consideration of frank or opportunistic pathogens that could co-occur with cyanobacteria in ambient waters also complicates conclusions related to the etiologic agent of the reported symptoms (Lévesque et al. 2014; Lin et al. 2015; Pilotto et al. 1997; Stewart et al. 2006b).

The number of cells in freshwater reported to be statistically-associated with a significant increase in inflammatory endpoints ranged from 5,000 to 100,000 cells per mL. The EPA concluded that, although significant associations with adverse health effects occur across a wide range of cyanobacterial cell densities, the EPA cannot derive the CWA section 304(a) criteria based on total cyanobacterial cell density at this time. There is considerable uncertainty and variability associated with the epidemiological results that did not identify consistent effects at similar cell densities and available data do not support a consistent quantitative dose-response relationship.

Additional research is needed to better describe the health effects associated with exposure to cyanobacteria with more precision using consistent health symptomologies in context with the community of cyanobacteria present (e.g., population of toxigenic versus non-toxin-producing cyanobacteria, shifts in community profile during the study, etc.) and other factors that influence the proliferation of cyanobacteria. Based on currently available science, inflammatory illnesses are significantly increased at values above 100,000 cyanobacterial cells per mL. Guideline values currently in use (see sections 2.1 and 7.5.3) that are within the 5,000 to 100,000 cell density range can find supporting scientific evidence in the peer-reviewed literature described above and in Appendix D.

7.5.2 Cyanobacteria Biomass Measurements as Indicators of Hazard

Under certain conditions, cyanobacteria possessing the toxin synthesis genes, also referred to as toxigenic cyanobacteria, begin producing cyanotoxins. Toxigenic cyanobacteria are a functional subgroup of the total cyanobacterial population that may be present in a water body and the proportion of toxigenic cells present can vary geographically and over time. Numerous biotic and abiotic factors can influence not only the dominance of cyanobacteria within the overall phytoplankton community, but also the proportion of toxigenic cyanobacteria relative to non-toxin-producing cyanobacteria (Davis et al. 2009; Hyenstrand et al. 1998; McCarthy et al. 2009; Neilan et al. 2013; Gobler et al. 2016). Multiple species of cyanobacteria are capable of producing the same toxin, such as the microcystins, which can pose a risk to human and animal health (Crawford et al. 2017). Although scientists have observed a generalized relationship between total cyanobacteria density or chlorophyll *a* and cyanotoxin concentration, these relationships are affected by the dominance of the toxin-producing cyanobacteria within the overall cyanobacterial community (Zhang et al. 2014; Loftin et al. 2016b).

Total cyanobacterial cell biomass, described by cell densities or other metrics, such as chlorophyll *a*, can function as a measure of the ecological health of a water body and as an indicator of potential public health hazards, such as inflammatory reactions from exposure to cells and adverse health effects associated with the presence of cyanotoxins. The extent, frequency, persistence, and severity of cyanobacteria proliferation can indicate the eutrophic status of a water body (Yuan and Pollard 2015). Surface water enrichment with nitrogen, notably reduced forms of nitrogen, and phosphorus have been linked to cyanobacteria becoming the dominant phytoplankton (Beaulieu et al. 2013; Glibert et al. 2016; Paerl 2008; Watson et al. 1997). Proliferating cyanobacterial biomass can result in an increased potential for toxins being produced (Pearl et al. 2001; Otten et al. 2012).

Although there can be large variation in the number of toxigenic cyanobacteria present relative to non-toxigenic cyanobacteria in any given body of water, measures of the total cyanobacterial biomass, such as cell counts, chlorophyll, or even visual assessments, can be used effectively in decision-making as early warnings of potential HAB-associated hazards (Loftin et al. 2016b). Pacheco et al. (2016) stated that these measurements can be good indicators of the potential risk of cyanotoxin exposure and useful when access to more sophisticated approaches, resources, or expertise may be limiting. Measurements of total cyanobacteria may also be particularly useful in waters with a history of HAB occurrence and the presence of elevated cyanotoxins.

7.5.2.1 Remote Sensing Techniques for Estimating Cyanotoxins

New and innovative methods, such as remote sensing techniques using satellite imagery, coupled with quantitative analysis to identify cyanobacterial blooms are of increasing interest to states. To date, these techniques cannot yet detect cyanotoxins, but they can quantify cyanobacterial densities in water bodies, an indicator of potential for cyanotoxin presence. Satellite measures of chlorophyll *a*, phycocyanin, or both are used to estimate cyanobacterial cell density based on validated algorithms that quantify relationships between these parameters and in situ measurements of cell density. For example, Stumpf (2014) and Wynne et al. (2010) readily detected by satellite areas of high *Microcystis* densities in larger freshwater bodies, such as Lake Erie.

U.S. EPA has collaborated since 2015 with the National Aeronautics and Space Administration (NASA), NOAA, and the USGS on the Cyanobacteria Assessment Network (CyAN) project. This project is developing the capability to detect and quantify total cyanobacterial blooms and related water quality of U.S. lakes and estuaries using satellite data records (U.S. EPA 2018b). This includes improving interpretation of satellite data and refining algorithms across satellite platforms. CyAN defined an approach for identifying lakes that can be spatially resolved (i.e., visually separated) with satellite imagery given differences in pixel resolutions, a method to quantify frequency of bloom occurrence in recreational freshwater sites, and a method for evaluating changes in the spatial extent of cyanobacterial blooms over time to support state-level assessments (Clark et al. 2017; Urquhart et al. 2017). CyAN has developed a mobile application that makes its processed satellite data more widely available. In 2017, the application was made available to state agencies for beta testing (U.S. EPA 2018b). A CyAN project that compares satellite-based estimations of total cyanobacterial cell density data from monitoring programs in eight states in the eastern United States found that satellite information provided robust estimates for freshwater lakes greater than 100 hectares when the cell densities less than 109,000 cells/mL and above 1 million cells/mL (Lunetta et al. 2015). The estimates were less on target for intermediate densities (i.e., between 110,000 and 1 million cells). The authors attributed this lower performance to the gap in taxonomic information needed to facilitate conversions between cell count and cell volumes (Lunetta et al. 2015).

Challenges remain for using remote sensing for cyanotoxin detection and mapping and Stumpf et al. (2016) identify these and a strategy for resolving them. The challenges they note include the lack of a steady relationship between the indicator pigments (i.e., chlorophyll *a* or phycocyanin) and cyanotoxins. These relationships may be valid for several weeks but start to vary over longer time periods due to changes in the amount of cyanotoxin produced as a function of cyanobacterial biomass. Strategic collection of pigment and toxin measurements will improve the application of remote sensing and associated models. The Ocean Land Colour Imager on the Sentinel-3 satellite, launched in 2016, will help address this need and improve data availability for most medium to large lakes around the world.

Given the inherent spatial uncertainty in the distribution of blooms and the potential issues with use of the appropriate satellite product, more attention should be given to the use of field measurements of reflectance to parameterize derivative-based pigment models (Tomlinson et al. 2016). This approach will help standardize processing of the satellite data to consistent reflectance-based products. Standardization is a factor in pigment and cyanotoxin measurement that will also require closer scrutiny. Propagation of known measurement error and uncertainty into the models will establish confidence levels for a variety of applications besides toxin maps. Improving strategies for collecting pigment measurement with toxin measurement will allow a better understanding and use of remote sensing to inform monitoring of toxins in lakes.

7.5.2.2 Molecular Methods for Estimating Cyanotoxins

Scientists have applied newer methods of quantifying microbes in environmental matrices, which increases understanding of bloom dynamics and functional subgroups of cyanobacteria, such as the toxigenic cells (Davis et al. 2009). The use of gene-based enumeration methods allows the quantification of cyanobacteria that contain specific gene sequences for toxin synthesis—without which a cell cannot produce the toxin. When toxigenic cyanobacteria are characterized with these tools, they have been shown to be better predictors of subsequent increases in toxin concentrations than with other traditional enumeration methods.

More recently, the use of gene-based quantification methods has helped to shed light on the community dynamics within a bloom, understand some of the factors that trigger toxic blooms, and provide faster and less expensive measurements of potential bloom toxicity compared to ELISA- and LC/MS/MS-based methodologies. Researchers have shown that microcystins and cylindrospermopsin are produced by non-ribosome-associated peptide synthetases (Dittmann et al. 1997; Moreira et al. 2013). The microcystin synthetase complex is encoded by 10 *mcy* genes (*mcyA* to *mcyJ*) (Neilan et al. 2013). Studies have characterized the abundance of various *mcy* genes in ambient waters (Pacheco et al. 2016; Qiu et al. 2013). The cylindrospermopsin synthetase gene cluster, *cyr*, is not as well characterized, but has been studied in multiple cylindrospermopsin-producing cyanobacteria (Neilan et al. 2013). Other researchers have used qPCR methods to characterize the relative abundance of total cyanobacteria, *Cylindrospermopsis raciborskii* and cylindrospermopsin synthase in lake water (Moreira et al. 2011). Selected examples of monitoring studies using gene-based approaches are described below.

- Davis et al. (2009) characterized toxic and nontoxic strains of *Microcystis* by quantifying the *mcyD* (toxigenic strains) and the 16S rDNA genes (all *Microcystis*) in four lakes in the northeastern United States over a two-year period. At all sites, toxigenic *Microcystis* were a better predictor of microcystin concentrations compared to total cyanobacteria, total *Microcystis*, chlorophyll *a*, or other environmental factors. Gene copies of *mcyD* were significantly correlated with microcystin concentrations in every lake studied (Davis et al. 2009).

- HABs in lakes and reservoirs are prevalent in Alberta, Canada, and are affected predominantly by elevated microcystins (Alberta Health 2014). Multiple Canadian governmental departments and public health laboratories in Alberta conducted a monitoring and advisory program for cyanobacteria at beaches. Among the findings were: microcystin-producing cyanobacteria species were dominant in most lakes with blooms peaking in late August to September, microcystin concentrations exceeding Canadian guidelines were not consistently associated with elevated total cyanobacterial cell densities in most cases, and the *mcyE* gene measured by qPCR was a good predictor for cyanobacterial blooms in some lakes (Alberta Health 2014).

- In response to the 2014 Lake Erie HAB event that contaminated the drinking water of Toledo, Ohio, the EPA revised the monitoring requirements for Ohio public water systems. Included in those requirements are testing for the *mycE* gene. If > 5 *myc*E genes/µL are detected in raw water samples, public water systems must monitor for microcystins (Ohio EPA 2017). Ohio is currently testing qPCR methods for total cyanobacteria (16s rDNA) and toxigenic cyanobacteria such as microcystin (*mcy*E gene) and saxitoxin (*sxt*A gene) producers. Ohio's HAB response strategy for recreational waters (Ohio EPA 2017) includes qPCR assessment for cyanotoxin-production genes as an option for cyanobacterial screening. If the qPCR testing indicates an abundance of toxigenic cyanobacteria, additional analysis for the toxin is recommended (Ohio EPA 2017).

- In Lake Champlain, in the northeastern United States, Fortin et al. (2015) applied qPCR-based methods and high-throughput sequencing to evaluate the effect of physico-chemical parameters and nutrients on the dynamics of cyanobacterial community. The researchers observed that total cyanobacteria were correlated with microcystin concentrations (Fortin et al. 2015). They also showed a significant correlation between the microcystin concentrations, the abundance of the *mcyD* gene, and the abundance of *Microcystis* 16S rDNA gene copies. Previous work had shown that *Microcystis* were the predominant microcystin producer present in the same water body (Ngwa et al. 2014).

- Pacheco et al. (2016) reviewed studies examining relationships between the prevalence of microcystin synthetase genes and microcystin concentration, and between chlorophyll *a* or cell density and microcystin concentration. While many studies included in the review did show a correlation for both comparisons, some did not. A lack of correlation between the synthetase genes and microcystin concentration was reported in studies that: (1) extracted the particulate-associated microcystins only; (2) included waters with very low concentrations of total microcystins (e.g., < 0.5 µg/L); or (3) in one study, monitored lakes at a single fixed point in the pelagic zone at the deepest site in each lake using depth-integrated water samples representing the entire photic zone (Beversdorf et al. 2015a; Pacheco et al. 2016). For studies not reporting a correlation between chlorophyll *a* or cell density and toxin concentration, only particulate-associated microcystin was analyzed or a very low concentration (e.g., < 0.05 µg/L) of total microcystins was observed (Pacheco et al. 2016). Zhang et al. (2014), one of the studies included in the Pacheco et al. (2016) review, characterized cylindrospermopsin- and microcystin-producing genotypes in the Macau reservoir, China, and found high cylindrospermopsin concentrations correlated to the prevalence of the *pks* gene ($r^2 = 0.95$, p-value < 0.01) and that *Cylindrospermopsis* dominated the cyanobacterial population in the reservoir studied.

- Crawford et al. (2017) applied an integrated monitoring approach including microscopic cyanobacteria identification, multiplex qPCR for toxin genes, and toxin analysis to assess

potential risks and inform bloom management decisions in a HAB event on the Murray River, Australia, in 2016. The qPCR results showed that cylindrospermopsin and saxitoxin genes were present, but were below the level of quantification. No microcystin genes were detected. The qPCR results were corroborated with the lack of detection of any cylindrospermopsin, microcystin, or saxitoxin (Crawford et al. 2017).

7.5.2.3 Uncertainties in Using Cyanobacterial Cells as Indicators

While cell density and pigment measurements can be useful for early detection of cyanobacterial proliferation and informative for bloom monitoring, these approaches may not be sufficiently accurate to predict risk from cyanotoxins (Pacheco et al. 2016). Uncertainties related to the use of total cyanobacteria in decision-making related to toxin concentrations should be considered.

1. Toxigenic cell densities can be a better indicator of the potential of a bloom to produce cyanotoxins compared to measures of total cyanobacterial biomass.

The amount of toxin produced by a toxigenic cyanobacterial cell and the relative abundance of toxigenic strains relative to non-toxigenic ones can vary considerably and be affected by environmental factors (Gobler et al. 2016). Gene-based quantification of toxigenic cyanobacteria can be beneficial for decision-making for HAB management approaches (Lee et al. 2015; Crawford et al. 2017). Davis et al. (2009) observed that quantifying toxigenic *Microcystis* was a better predictor of in situ microcystin levels than other surrogates, such as total cyanobacteria and chlorophyll *a*. The use of qPCR to characterize temporal and spatial variations in the abundance of toxigenic strains can identify the capability of a bloom to produce toxins, and hence the potential for recreator exposure to toxins, including perhaps prior to the hazardous condition occurring (Pacheco et al. 2016).

The importance of the toxigenic cyanobacterial cells has been recognized by the WHO and previously discussed in section 2.1. Based on toxigenic *Microcystis*, approximately 20 µg microcystins per L could be expected, but other species, such as *Planktothrix*, can contain higher microcystin concentrations in a cell compared to *Microcystis* (Fastner et al. 1999). Thus, the WHO commented that microcystin concentrations could be much higher (e.g., 50–100 µg/L) if species with high microcystin content dominate a bloom (WHO 2003a).

2. Total cyanobacteria can be informative as an indicator for the presence of toxins if toxigenic species are abundant or the dominant members of the cyanobacterial community.

Evidence from prior monitoring may demonstrate toxigenic strains tend to dominate blooms in a water body or that a prior bloom had increased densities of toxigenic species occurring in conjunction with elevated toxins. Studies showing good correlation between increased cell densities or other parameters linked to cell proliferation and elevated toxin concentrations can also show the bloom is dominated by toxin-producing species (Rinta-Kanto et al. 2009; Zhang et al. 2014; Pacheco et al. 2016). In one study on Lake Erie over multiple seasons, Rinta-Kanto et al. (2009) observed a positive correlation between the abundance of cyanobacterial and *Microcystis* gene copies and the number of microcystin synthetase genes. *Microcystis* were a strong contributor to the concentration of microcystins in Lake Erie and the relative abundance of *Microcystis* cells was correlated with microcystin concentrations (Rinta-Kanto et al. 2009). Lack of correlation can occur when toxigenic cell density is low or undetectable (Crawford et al. 2017) or low concentrations of toxin are recorded (Rinta-Kanto et al. 2009) and in such cases measures of total cyanobacteria are not good indicators of toxins.

3. The proliferation of toxigenic cells and the timing of the presence of elevated toxin concentrations may or may not coincide with the visible proliferation of a HAB.

Decisions to issue recreational water warnings/advisories, or initiate monitoring for cyanotoxins based on total cyanobacteria once a bloom is observed (i.e., green, discolored water, or scum formation/accumulation associated with high densities of cells) may overlook situations where extracellular toxins are present. Cells may accumulate in locations different from where the bloom originated (e.g., by wind or wave action, or both, or be transported downstream). A cell density of 40,000 cells/mL is lower than what might be typically associated with a visible bloom (WHO 2003a). Decision points contingent on visually confirmed blooms may miss or delay the identification of the hazardous condition associated with exposure to elevated cyanotoxins, especially in water bodies with a previous history of HAB events or toxin detections and the downstream waters potentially affected by the HAB.

Davis et al. (2009, 2010) observed bloom dominance shift between toxigenic strains and non-toxigenic strains over the course of a summer. Spatial and temporal dynamics in cyanobacterial population succession is noted in other seasonal studies (Sabart et al. 2010; Otten et al. 2012, Beversdorf et al. 2015b; Fortin et al. 2015; Chen et al. 2017). Ha et al. (2009) observed similar seasonal variations in both the gene copies of microcystin synthetase genes and for total cyanobacteria gene copies, although the cyanobacterial community was consistently dominated by microcystin-producing cells throughout the study.

7.5.3 Use of Cyanobacteria Cell Densities in Guidelines

7.5.3.1 Cyanobacteria Cell Guidelines

A number of states and international agencies include both total cyanobacteria and toxigenic cyanobacteria density guidelines to account for both inflammatory- and toxin-associated health endpoints. Cyanobacterial cell densities used by states and local health departments to provide guidance to recreators on water quality are presented elsewhere in this document (see Table 2-3 for a list of states with cyanobacterial cell density guidelines; see Appendix B for state guidelines and associated actions).

As discussed in section 2.1, the 35 states that currently have HAB-related guidelines include different approaches and guideline levels (see Table 2-3). Seven states have guideline levels that address toxin-producing cyanobacteria as a proportion of the total cyanobacterial population or include a toxin-specific cyanobacteria cell density (California, Idaho, Maryland, New York, New Hampshire, Oregon, and Virginia). The Karuk Tribe, located in California, developed cell-based values for posting cyanotoxin public health warnings for the tribe's recreational waters (Kann 2014).

As described in section 2.1 of this document, the WHO (2003a) guideline value development was informed by results from a review conducted by Chorus and Bartram (1999) and a prospective epidemiology study by Pilotto et al. (1997), which evaluated health effects after recreational exposure to cyanobacteria and reported associations between cyanobacterial cell densities and health. The WHO recommended three tiers of guideline values describing an increasing scale of potential adverse health effects and "between the chiefly irritative symptoms caused by unknown cyanobacterial substances and the potentially more severe hazard of exposure to high concentrations of cyanotoxins, particularly microcystins."

- The lowest tier of guideline values (< 20,000 cyanobacterial cells per ml; < 10 µg/L chlorophyll *a*) was mainly associated with a significant increase in irritative or allergenic effects (the inflammatory health endpoints). The WHO, using conservative assumptions, also estimated that microcystin concentrations of 2 to 4 µg/L, and possibly up to 10 µg/L, may be expected at a cell density of 20,000 cells/mL where microcystin producers dominate.

- The second tier (20,000 to 100,000 cyanobacterial cells per ml; 10 to 50 µg/L chlorophyll *a*), describing a moderate probability of adverse health effects from cyanotoxins was informed by (1) modifying the value for the WHO drinking water guideline for microcystin-LR for a recreational exposure scenario and (2) translating microcystin concentrations to cell densities based on the average microcystin content of *Microcystis* cells. The WHO, using conservative assumptions, also estimated that 100,000 cyanobacterial cells/mL could correspond to 20 µg microcystins/L if a bloom consists of *Microcystis* and has an average microcystin content of 0.2 pg/cell.

- At the third tier (> 100,000 cells per mL; > 50 µg/L chlorophyll *a*) "there is the potential for some frequently occurring species (i.e., *Microcystis*) to form scums," which can "increase risks for bathers and others involved in body-contact water sports." The high probability of adverse health effects category is associated with the elevated potential for exposure to cyanotoxins and the potential for severe health outcomes. "The presence of cyanobacterial scum in swimming areas represents the highest risk of adverse health effects due to abundant evidence for potentially severe health outcomes associated with these scums."

- Very high densities of cells occurring in scums (e.g., > 10 million cells/mL or > 5,000 µg/L chlorophyll *a*) can be associated with very high concentrations of toxin.

The Australian National Health and Medical Research Council (NHMRC) published a two-tiered guideline for managing cyanobacteria in recreational water (NHMRC 2008). Tier one includes numeric targets for microcystins based on children's recreational exposures and a toxigenic cell density for *Microcystis aeruginosa*. The NHMRC recommends a secondary guideline for the protection from health hazards associated with high densities of non-toxigenic cyanobacteria consistent with the WHO cyanobacterial cell density recommendations for the moderate probability of health effects. NHMRC used the epidemiological results published by Stewart et al. (2006b) to inform the derivation of the Australian total cyanobacteria guideline number. Stewart et al. (2006b) found a significant increase in the inflammatory health effects associated with recreators exposed to >100,000 total cyanobacteria/mL or a total cyanobacterial surface area > 12 mm^2/mL. Because different cyanobacteria species can have different sizes, the surface area estimate of biomass can take those size differences into account (e.g., 1,000 very big cells versus 1,000 very small cells). NHMRC converted the cell surface reported by Stewart et al. (2006b) to an equivalent biovolume and rounded that value to 10 mm^3/L. This biovolume guideline value applies when toxigenic cyanobacteria are absent in a bloom (NHMRC 2008)

NHMRC calculated a child-based total microcystin concentration of 9.4 µg/L, rounded to 10 µg/L (NHMRC 2008). The authors then converted the toxin concentration to an equivalent toxigenic cell density (50,000 *Microcystis aeruginosa*/mL) using the microcystin cell quota value (0.2 pg/cell). To account for the potential hazard posed by other microcystin-producing cyanobacteria, the cell density was converted into a biovolume equivalent (4 mm^3/L). Other species have different cell sizes, so the biovolume measurement allows comparisons with the other known toxin-producing cyanobacteria that may be present. The biovolume equivalent applies to the total of all cyanobacteria where a known toxin producer is dominant (NHMRC 2008).

7.5.3.2 Amount of Toxin per Cell

Toxigenic cyanobacteria produce cyanotoxins that can accumulate inside the cells or be released to the water column. The amounts of toxin produced by a toxigenic cyanobacterium is also referred to as "cell quota." There is variability in the estimate of cyanotoxin concentrations associated with cell density, in part because a bloom can contain both the toxigenic and non-toxin-producing strains of the same species and cyanobacterial community differences between locations could affect the level of cyanotoxin that is present. Thus, it is important to understand the abundance of toxigenic cyanobacteria in a water body. As discussed above, characterizing the abundance of toxin genes can be a better predictor of toxin produced than can calculations based on a toxin cell quota. The WHO's microcystin estimates at the different risk levels were based on converting the recommended total cyanobacterial cell density using a *Microcystis* cell quota value for microcystins (0.2 pg/cell) derived from a laboratory study conducted by Mole et al. (1997) reporting an average microcystin cell quota in laboratory cultures of 0.2 pg/cell (range: 0.07–0.3 pg/cell) (Fitzgerald et al. 1999), but other species and strains of microcystin producers could result in much higher water-column microcystin concentrations given the same cell density (WHO 2003a).

The EPA searched the published peer-reviewed scientific literature for information on the amount of microcystin and cylindrospermopsin produced by or contained in a cell to inform the development of toxigenic cell densities equivalent to the recommended criteria concentrations. Appendix G presents the details related to the search strategy, reference prioritization and search results. The search resulted in the collation of multiple studies reporting cell quotas for microcystin and cylindrospermopsin in multiple genera of cyanobacteria. Laboratory-based culture studies with numerous clones of *Microcystis aeruginosa*, *Cylindrospermopsis raciborskii, Planktothrix agardhii*, and *Planktothrix rubescens* were also found. Many of these references also included either biomass-toxin conversions or graphic data that would support conversion factors from cyanobacterial cell density (expressed in a variety of units including: cells/L, biovolume (μm^3/L), and chlorophyll *a*/L) to toxin concentrations for these species. Aggregated data are presented in Table 7-12. Table G-3 in Appendix G provides additional detail on the studies identified containing cell quota information.

To facilitate a comparison of this information with the value used by the WHO, the EPA organized the reported cell quota information by toxin and by genus (Table 7-14). Within each row, the study type, quantification method, reported means and ranges, and references to the original study are included. Not every study reported a mean, median, maximum, or minimum, so each row represents a collation of the values reported. Ranges of reported cell quotas were large. For example, for all microcystin-producing genera, reported cell quotas ranged from 0 to 4.3 pg/cell and the reported range of the means were 0.015 to 0.58 pg/cell. For *Microcystis*, the mean of the means, for seven studies published between 2008 and 2013, was 0.15 pg/cell. This value is similar to the 0.2 pg/cell value used by the WHO and provides additional evidence that this conversion factor is supported by multiple scientific studies. For the genus *Planktothrix*, the studies identified by the EPA do not suggest that this genus produces much higher amount of microcystin compared to *Microcystis*. However, the EPA's literature search focused on more recently published data and the *Planktothrix* values in the summary table come from only two recent studies that may have not characterized toxin production under optimal conditions. Based on the data presented in Table 7-14, the EPA concluded that the microcystin cell quota used by the WHO is supported. The caveat expressed by the WHO (i.e., cell quota values can be variable within and between species of microcystin-producing cyanobacteria) is also substantiated by the EPA's literature search results. The EPA included the 0.2 pg/cell value in the calculation of a toxigenic cell density for

microcystin-producing cyanobacteria equivalent to the recommended toxin magnitude (see section 7.5.3.3).

The EPA also collated similar information for cylindrospermopsin cell quotas. As with other aspects of cylindrospermopsin, less information was available, but multiple field and laboratory studies reporting the mass of toxin per cell were identified. The range of cylindrospermopsin cell quotas (0.0028–14.6 pg/cell in *Cylindrospermopsis*) was larger than for microcystins, as was the range of reported means (0.0028–0.17 pg/cell). The highest value (14.6 pg/cell) was reported from a field study (see Table G-1). The highest value reported in a laboratory study was 0.17 pg/cell. The mean value for all studies was 0.047 pg/cell (n = 10) and for field studies (n = 2) was 0.023 pg/cell. Given the few number of field studies, large uncertainties exist with how representative the mean is of the central tendency of the range. Less information was identified for *Aphanizomenon*, another well-known cylindrospermopsin producer. To have a similar confidence level in the cylindrospermopsin cell quota data compared to microcystins, additional data and an improved sense of the central tendency within the reported ranges is needed. At present, the EPA is not sufficiently confident in the cylindrospermopsin cell quota database to estimate a toxigenic cell density specific for cylindrospermopsin.

Table 7-14. Aggregated Cell Quota Summary Data for Selected Microcystin and Cylindrospermopsin-producing Genera

Toxin Genus	Quantification Method[a]; Study Type[b]	Range of Means[c]	Mean[c,d]	Median of Means[c]	Minimum; Maximum[c,e]	References
Microcystins						
All microcystin-producing genera	Mass per cell; Field and lab	0.015 pg/cell – 0.58 pg/cell	0.11 pg/cell	0.091 pg/cell	0 pg/cell – 4.3 pg/cell	Orr and Jones (1998); Jähnichen et al. (2001); Wiedner et al. (2003); Akcaalan et al. (2006); Jähnichen et al. (2007); Briand et al. (2008); Fahnenstiel et al. (2008); Vasconcelos et al. (2011); Sitoki et al. (2012); Tao et al. (2012); Wood et al. (2012); Cires et al. (2013); Sabart et al. (2013); Wang et al. (2013); Pineda-Mendoza et al. (2014); Chia et al. (2016); Wei et al. (2016)
Microcystis	Mass per cell; Field and lab	0.015 pg/cell – 0.58 pg/cell	0.11 pg/cell	0.072 pg/cell	0 pg/cell – 4.3 pg/cell	Orr and Jones (1998); Jähnichen et al. (2001); Wiedner et al. (2003); Jähnichen et al. (2007); Fahnenstiel et al. (2008); Vasconcelos et al. (2011); Sitoki et al. (2012); Tao et al. (2012); Wood et al. (2012); Cires et al. (2013); Sabart et al. (2013); Wang et al. (2013); Pineda-Mendoza et al. (2014); Chia et al. (2016); Wei et al. (2016)
	Mass per cell; Field	0.015 pg/cell – 0.58 pg/cell	0.15 pg/cell	0.075 pg/cell	0 pg/cell; 4.19 pg/cell	Fahnenstiel et al. (2008); Vasconcelos et al. (2011); Sitoki et al. (2012); Tao et al. (2012); Cires et al. (2013); Sabart et al. (2013); Wang et al. (2013)
Planktothrix	Mass per cell; Field and lab	0.076 pg/cell – 0.24 pg/cell	0.12 pg/cell	0.10 pg/cell	0.076 pg/cell; 0.24 pg/cell[e]	Akcaalan et al. (2006); Briand et al. (2008);
	Mass per cell; Field	0.091 pg/cell – 0.24 pg/cell	0.16 pg/cell	0.16 pg/cell	0.091 pg/cell; 0.24 pg/cell[e]	Akcaalan et al. (2006); Briand et al. (2008)
Fisherella	Mass per biomass; Lab	N/A	N/A	N/A	43 µg/g	Cires et al. (2014)

Toxin Genus	Quantification Method[a]; Study Type[b]	Range of Means[c]	Mean[c,d]	Median of Means[c]	Minimum; Maximum[c,e]	References
Cylindrospermopsin						
Aphanizomenon	Mass per biomass; Field and lab	N/A	N/A	N/A	7,390 µg/g; 9,330 µg/g	Yilmaz et al. (2008)
Cylindrospermopsis	Mass per cell; Field and lab	0.0028 pg/cell – 0.17 pg/cell	0.047 pg/cell	0.027 pg/cell	0.0028 pg/cell[e]; 14.6 pg/cell	Hawkins et al. (2001); Orr et al. (2010); Carneiro et al. (2013); Mohamed and Al-Shehri (2013); Davis et al. (2014); Pierangelini et al. (2015); Willis et al. (2015); Willis et al. (2016); Yang et al. (2016a)
	Mass per cell; Field	0.023 pg/cell	0.023 pg/cell	N/A	0.006 pg/cell; 14.6 pg/cell	Orr et al. (2010); Mohamed and Al-Shehri (2013)
	Mass per cell; Lab	0.0028 pg/cell – 0.17 pg/cell	0.052 pg/cell	0.031 pg/cell	0.0028 pg/cell[e]; 0.17 pg/cell[e]	Hawkins et al. (2001); Carneiro et al. (2013); Davis et al. (2014); Pierangelini et al. (2015); Willis et al. (2015); Willis et al. (2016); Yang et al. (2016a)
	Mass per biovolume; Lab	N/A	N/A	N/A	416 fg/µm^3; 447 fg/µm^3	Pierangelini et al. (2015)

fg = femtogram; pg = picogram; µg = microgram; N/A = not available.

[a] Various methods were used to quantify toxin quotas and quota values were presented in different forms, including toxin mass per cyanobacterial cell and toxin mass per cyanobacterial biomass.

[b] Studies were conducted in two different settings: the field (i.e., environmental) or a laboratory.

[c] Study authors reported data using multiple measurement units. When possible, the EPA converted data to the standard units of pg per cell. The EPA did not identify appropriate conversion factors that would allow genus-specific conversion of quotas described in mass per biomass to mass per cell.

[d] Shows single reported mean if only one study was available or average of reported means.

[e] If reported toxin quota means from one study were the lowest or highest toxin quotas reported within a genus, then these values were listed as the minimum or maximum values, respectively, to better reflect the range of toxin quota values.

[f] *Cylindrospermopsis* is now known as *Raphidiopsis.*

Challenges with collating this information include the variable conditions under which the studies characterized toxin quotas and the various ways the toxin quota data were reported. Conditions under which the toxin quotas were studied include laboratory and field conditions, different environmental and collection-based strains included in the study, and the different environmental conditions existing at the various locations where the field studies were conducted. For the latter, information on some of the external factors affecting toxin production is summarized above to help demonstrate the complex interactions that affect not just if the toxin is produced, but also how much toxin can be produced. The various ways that toxin cell quotas were reported include: toxin mass per cell, toxin mass per unit biomass, and toxin mass per unit biovolume. When possible, the EPA converted the cell quota information into pg per cell to enable a straightforward comparison to the WHO value.

7.5.3.3 Toxigenic Cyanobacteria Value Associated with Recommended Microcystins Criteria/Swimming Advisory

As discussed in section 7.5.3.2 the abundance of toxigenic cells in a water body affects the amount of cyanotoxin produced. The number of toxigenic cyanobacteria relative to the number of total cyanobacteria can vary in time and space. Quantifying the abundance of toxigenic cyanobacteria is a better predictor of potential toxin production compared to total cyanobacteria. Below, the EPA presents a similar approach to that used by the WHO to calculate a cyanobacterial cell density corresponding to recommended criteria/ swimming advisory value for microcystins. Because more data are available for microcystins compared to cylindrospermopsin, this calculation is based on microcystins only.

$$\text{Cyanobacterial cell density (CCD)} = \frac{\text{Ambient cyanotoxin concentration (ACC)}}{\text{Cell toxin amount (CTA)}}$$

Where:

CCD	=	calculated toxigenic cell density associated with a specific toxin concentration
ACC	=	specific toxin concentration target in ambient water (e.g., AWQC value)
CTA	=	amount of toxin produced in a cyanobacterial cell

For the microcystins-producing cyanobacteria (e.g., *Microcystis*):

ACC	=	8 µg/L; recommended recreational criteria value for microcystins
CTA	=	0.2 pg/cell; reported mean concentration of microcystin in a cell of microcystin-producing cyanobacteria

Adding in the conversion factors to convert units, the equation is:

$$\text{CCD} = \frac{\text{ACC (µg/L)} \times 10^6 \text{ pg/µg}}{\text{CTA (0.2 pg/cell)}} \times \frac{\text{L}}{1000 \text{ mL}}$$

Adding in the values,

$$\text{CCD} = \frac{8\ \mu\text{g/L} \times 10^6\ \text{pg/}\mu\text{g}}{0.2\ \text{pg/cell}} \times \frac{1\ \text{L}}{1000\ \text{mL}} = 40{,}000\ \text{cells/mL}$$

Thus, a toxigenic microcystin-producing cell density of 40,000 cells/mL has the potential to result in a microcystin concentration of 8 µg/L.

7.6 Other Sources of Microcystins and Cylindrospermopsin

Although the EPA is not including other sources of cyanotoxins in this recreational exposure scenario, the Agency has included summary information on potential sources of cyanotoxins, such as drinking water, ground water, fish, shellfish, dietary supplements, air, soil, and sediments. Exposure to cyanotoxins in finished drinking water is characterized in the Drinking Water Health Advisories (U.S. EPA 2015a, 2015b). States may wish to consider these other sources of cyanotoxins in their public health approach.

7.6.1 Drinking Water

The occurrence of cyanotoxins in drinking water depends on their levels in the raw source water and the effectiveness of treatment methods for removing cyanobacteria and cyanotoxins during the production of drinking water. The EPA has provided *Recommendations for Public Water Systems to Manage Cyanotoxins in Drinking Water* to assist public drinking water systems (PWSs) that choose to develop system-specific plans for evaluating their source waters for vulnerability to contamination by microcystins and cylindrospermopsin (U.S. EPA 2015e). Cyanotoxin management plan templates, water treatment optimization, and a communications tool box are also available on the EPA's Cyanotoxins in Drinking Water website (U.S. EPA 2015e).

The American Water Works Association Research Foundation (AWWARF) conducted a study on the occurrence of cyanobacterial toxins in source and treated drinking waters from 24 public water systems in the United States and Canada in 1996–1998 (AWWARF 2001). Of 677 samples tested, microcystins were found in 80 percent (539) of the waters sampled, including source and treated waters. Only two samples of finished drinking water were above 1 µg/L. A survey conducted in 2000 in Florida (Burns 2008) reported that microcystins were the most commonly found toxin in pre- and post-treated drinking water. Finished water concentrations ranged from below detection levels to 12.5 µg/L.

During the summer of 2003, a survey was conducted to test for microcystins in 33 U.S. drinking water treatment plants in the northeastern and midwestern United States (Haddix et al. 2007). Microcystins were detected at low levels ranging from undetectable (< 0.15 µg/L) to 0.36 µg/L in all 77 finished water samples.

In August 2014, the city of Toledo, Ohio, issued a do-not-drink or -boil advisory to nearly 500,000 customers in response to the presence of total microcystins in the city's finished drinking water at levels up to 2.50 µg/L. The presence of the toxins was due to a cyanobacterial bloom near Toledo's drinking water intake located on Lake Erie. The advisory was lifted two days later, after treatment adjustments led to the reduction of the cyanotoxin concentrations to concentrations below the WHO guideline value of 1 µg/L in all samples from the treatment plant and distribution system.

During the late spring and early summer of 2018, both microcystins and cylindrospermopsin were found in the finished drinking water of Salem, Oregon (Novak Consulting Group 2018). Salem's finished drinking water source is the North Santiam River, which is fed by Detroit Lake, a reservoir located southeast of the city. In late May 2018, the State of Oregon issued a recreational advisory for cyanotoxins for Detroit Lake. Less than a week later, the City of Salem issued a do not drink advisory due to the presence of levels of microcystins and cylindrospermopsin in drinking water exceeding health advisories. The drinking water advisory was lifted in the beginning of July based on many consecutive days of finished water results being below health advisory levels.

The EPA has published Drinking Water Health Advisories to address microcystins and cylindrospermopsin in drinking water (U.S. EPA 2015a, 2015b).

7.6.2 Ground Water

Only very limited data are available on microcystins in ground water and no monitoring data were identified for cylindrospermopsin. A study reported microcystins in ground water from a well located near the shore of Lake Chaohu, in China (also known as Chao Lake), which contained high microcystin concentrations (Yang et al. 2016b). Therefore, under certain conditions, ground water hydraulically connected to surface water has the potential to be contaminated by cyanotoxins.

7.6.3 Fish and Shellfish

Fish and shellfish living in waters affected by a cyanobacterial bloom may accumulate cyanotoxins in their muscle tissue and internal organs (Gibble et al. 2016; Kinnear 2010). Some authors have found that microcystins accumulate less in the edible parts of aquatic organisms, such as muscle (Deblois et al. 2011; Gutiérrez-Praena et al. 2013; Song et al. 2009; Vareli et al. 2012; Wilson et al. 2008; Xie et al. 2005; Zimba et al. 2006). Cylindrospermopsin has also been found in fish and shellfish exposed for longer periods of time to a cyanobacterial bloom (Funari and Testai 2008; Ibelings and Chorus 2007; Kinnear 2010; Saker and Eaglesham 1999). For additional information on occurrence of microcystins and cylindrospermopsin in fish and shellfish, please see the Health Advisory document published (U.S. EPA 2015a, 2015b).

7.6.4 Dietary Supplements

Extracts from *Arthrospira* (Spirulina) and *Aphanizomenon flos-aquae* have been used as dietary blue-green algae supplements (BGAS) (Funari and Testai 2008). These supplements are reported to have beneficial health effects including supporting weight loss, and increasing alertness, energy and mood elevation for people suffering from depression (Jensen et al. 2001). A study suggested that BGAS could be contaminated with microcystins ranging from 1 μg/g up to 35 μg/g (Dietrich and Hoeger 2005). In two separate studies, Heussner et al. (2012) and Roy-Lachapelle et al. (2017) both analyzed 18 different commercially available BGAS for the presence of cyanotoxins. Heussner et al. (2012) reported that all products containing *Dolichospermum flos-aquae* (formerly *Aphanizomenon flos-aquae*) tested positive for microcystins at levels ≤ 1 μg microcystin-LR equivalents/g dry weight. Cylindrospermopsin was not found in any of the supplements. Roy-Lachapelle et al. (2017) reported that of the 14 products containing *Spirulina*, three contained total microcystins at levels ≤ 1 μg/g. All four products containing *Dolichospermum flos-aquae* tested positive for total microcystins ranging from 0.8 μg/g to 8.2 μg/g using the Adda oxidation method and from 0.52 μg/g to 5.8 μg/g using the sums of microcystins standards. Cylindrospermopsin was not found in any of the supplements.

7.6.5 Ambient Air

Four studies provide air concentration data for cyanotoxins indicating that recreational surface waters with toxigenic cyanobacterial blooms can result in aerosolized cyanotoxins (Backer et al. 2008, 2010; Wood and Dietrich 2011; Cheng et al. 2007). These studies are summarized in section 7.4.1.

7.6.6 Soils and Sediments

Microcystins can adsorb onto naturally suspended solids and dried crusts of cyanobacteria. Cyanotoxins can precipitate out of the water column and reside in sediments for months (Falconer 1998; Han et al. 2012; Wu et al. 2012). In sediments, cylindrospermopsin adsorbs to organic carbon, with little adsorption observed in sandy and silt sediments (Klitzke et al. 2011). The low adsorption of cylindrospermopsin in sediments/silts with low levels of organic carbon reduces the opportunity for microbial degradation.

Maghsoudi et al. (2015) tested adsorption of cyanotoxins onto three fractionated sediment particles, clay-silt (< 75 μm), fine sand (75–315 um) and coarse sand (315–2000 μm) and found that adsorption capacity of coarse sand fraction for all the tested cyanotoxins was less than four percent of the clay-silt fraction. They found that highest adsorption for cylindrospermopsin, microcystin-LW, and microcystin-LF were 73, 57, and 55 percent, respectively, and occurred within two hours. Desorption experiments demonstrated that less than nine percent of cyanotoxins desorbed from sediment within 96 hours.

Song et al. (2015) found that a statistically significant part of the variability of the microcystin concentration in the sediments could be explained by a combination of variables in the water column, such as total microcystins in the water, cyanobacterial biomass in water, pH, and temperature.

7.7 Tribal Considerations

The EPA considered alternative exposure scenarios tribal communities might have, given their cultural practices. Native American food foraging customs or cultural or religious ceremonies can put them into primary or secondary contact with cyanotoxins. Primary contact ceremonial use may include the use of a surface water body for religious or traditional purposes by members of a tribe, involving immersion and intentional or incidental ingestion of water (Eastman 2007).

It is uncertain whether these activities would lead to cyanotoxin exposures higher than the primary recreational contact assumptions for incidental ingestion and exposure duration used in this assessment.

7.8 Livestock and Pet Concerns

The earliest observations of adverse effects of cyanobacterial exposure to animals include the rapid death of stock animals in Australia in 1878 (Francis 1878). Since then, numerous cases of mammal and bird deaths have been documented (Backer et al. 2015; Hilborn and Beasley 2015). These cases were reported throughout the 20th century on all continents except Antarctica (Stewart et al. 2008). The impacts of cyanotoxins on domestic and companion animals are likely under-recognized because many cases are misdiagnosed, few cases are biochemically confirmed, and even fewer are reported in the scientific literature or to animal health systems (Zaias et al. 2010).

Livestock and pets potentially can be exposed to higher concentrations of cyanotoxins, or have increased exposure to cyanotoxins than humans because they are known to consume cyanobacterial scum and mats

and drink cyanobacteria-contaminated water (Backer et al. 2013). Dogs are also at risk, as they may lick cyanobacterial cells from their fur after swimming in a water body with an ongoing bloom (CDC 2017a). Mats and scums can represent thousand-fold to million-fold concentrations of cyanobacterial cell populations, and published microcystin concentrations have ranged up to 24 mg microcystins/L from scum material (Chorus and Bartram 1999). Common signs of HAB cyanotoxin poisonings in pets include repeated vomiting, diarrhea, loss of appetite, abdominal swelling, stumbling, seizures, convulsions, disorientation, inactivity, or skin rashes and hives, and in extreme cases collapse and sudden death (CDC 2017a; New York Sea Grant 2014; Trevino-Garrison et al. 2015). Although reports of livestock deaths are uncommon, in extreme cases, death can occur minutes after drinking from a contaminated water source. Acute symptoms of cyanotoxin poisoning can include loss of appetite, weakness, staggering, or inflammation of the muzzle, ear, or udder. Higher levels of cyanotoxins can lead to severe liver damage, the development of jaundice, and severe photosensitization. Often livestock or pets that recover from these ailments can then suffer from chronic failure to thrive (Australia Department of Economic Development Jobs Transport and Resources 2013; Robinson and Alex 1987).

The Centers for Disease Control and Prevention (CDC) provides multiple resources, such as frequently asked questions (FAQs), Veterinarian Cards, and Animal Safety Alerts, to help educate the public of the dangers associated with cyanotoxin exposure to pets (CDC 2017a, 2017b, 2017c). The CDC suggests that pet owners prevent their animals from playing in or drinking scummy water. If a dog has been swimming in scummy water, the CDC recommends rinsing them off immediately to prevent the dog from licking cyanobacteria off their fur (CDC 2017b).

The CDC recommends that pet owners contact a veterinarian if their animal shows the following symptoms of cyanotoxin poisoning: loss of appetite, loss of energy, vomiting, stumbling and falling, foaming at the mouth, diarrhea, convulsions, excessive drooling, tremors and seizures or any other unexplained sickness after being contacted with water (CDC 2017c). While there have been no HAB-associated human deaths in the United States, there have been many pet deaths (especially dogs) due to cyanotoxin exposure via swimming and ingesting contaminated waters. Overall, CDC encourages the public to follow the phrase “when in doubt, its best to keep out” (CDC 2017a).

The One Health Concept acknowledges a connection between human, animal, and environmental health, suggesting that HAB-associated animal illnesses and deaths could serve as predictors of potential HAB-associated risks in humans (CDC 2017d). Following this concept, the CDC created a voluntary reporting system called the One Health Harmful Algal Bloom System (OHHABS) (CDC 2017d). While there are other reporting systems that capture aggregate information on human illnesses or outbreaks, such as the National Outbreak Reporting System (NORS), OHHABS expands reporting to include HAB-associated environmental data, animal case data, and human case data (CDC 2017d). By collecting this information, the goal of OHHABS is to better understand HABs and HAB-associated illnesses. Members of the public can report HABs and cases of HAB-related human or animal illness by contacting local or state public health agencies (CDC 2017d).

The New York State Department of Health (NYSDOH) applied the One Health approach to implement a pilot surveillance system of HAB-related illnesses in 2015. During this pilot period, three dogs were reported to have GI symptoms after exposure to HABs in recreational water; one of these cases was also associated with a human case (Figgatt et al. 2017).

7.8.1 States and Animal HAB Guidelines

A few states have guideline levels specific to the protection of animals from cyanotoxin poisoning (Appendix H). California calculated cattle and dog action levels for the cyanotoxins microcystin and cylindrospermopsin (Butler et al. 2012). California first calculated an RfD (mg/kg body weight/day) for domestic animals for each of the cyanotoxins, based on laboratory studies. For both dogs and cattle, California estimated drinking water ingestion rates (L/kg body weight/day) based on two publications by the National Research Council, *Nutrient Requirements for Beef Cattle and Nutrient Requirements for Dogs and Cats*, and applied an UF of three to account for preferential consumption of cyanobacteria. To determine action levels (acute action level of 100 µg/L for microcystins and 200 µg/L for cylindrospermopsin), California divided the domestic animal RfD for each cyanotoxin by the final water and cyanobacterial biomass intake exposure levels calculated for cattle and dogs, and performed a unit conversion, providing a cyanotoxin concentration that would result in exposure at the RfD level or below. The state performed these calculations for an acute (lethal) and a subchronic scenario.

Oregon followed a similar approach to California's to calculate dog-specific guideline values for the cyanotoxins cylindrospermopsin, microcystin, anatoxin-a, and saxitoxin (Oregon Health Authority 2018). Oregon estimated tolerable daily intake (TDI) values for humans (µg/kg body weight/day) for each of the cyanotoxins, and applied these values to dogs (Farrer et al. 2015). Using California's dog-specific exposure estimate (L/kg body weight/day), Oregon divided the human TDI by the dog-specific ingestion rate to determine its guideline values (0.2 µg/L for microcystin and 0.4 µg/L for cylindrospermopsin).

Grayson County in Texas estimated the quantity of water that would result in a potentially lethal dose of microcystin and cylindrospermopsin for small and large dogs. Using advisory levels of 20 ppb for microcystin and cylindrospermopsin, the county calculated the volume of water that would result in a lethal or near-lethal dose of cyanotoxin by extrapolating the results of mouse studies to 10- and 80-pound dogs. This estimate does not include additional dose amounts that could be ingested by a dog while self-grooming cyanobacteria scum off its fur (Lillis et al. 2012).

At Presque Isle State Park in Pennsylvania, a HABs task force (a partnership of six agencies and organizations) monitors for microcystin and cylindrospermopsin at multiple locations on Lake Erie within the park. Some of the locations monitored include designated dog beaches. Warning signs are posted specifically for dog owners when microcystin levels are detected above 0.2 µg/L (Schnars personal communication 2017; Best personal communication 2017).

Other states mention animal poisoning in their guideline documents but do not give guideline values specific to livestock or companion animals. For example, Utah and Washington report that animal illness or death can be reason to issue or accelerate a HAB advisory warning (Hardy and Washington State Department of Health 2008; Utah Department of Environmental Quality and Department of Health 2017). Ohio includes pets in their public health advisory at threshold levels of 6 µg/L for microcystin and 5 µg/L for cylindrospermopsin; however, Ohio issues the disclaimer that thresholds used are protective of human exposure and may or may not be protective of animals such as dogs or livestock (Ohio EPA 2016). Several other states including Connecticut, Idaho, Kansas, Massachusetts, Nebraska, Vermont, and Virginia provide information via pamphlets and state websites warning about harm to pets or other animals or post about harm to animals in their beach warnings and advisory signage (CDPH 2017; CDEEP 2017; IDEQ 2015; Kansas Department of Health and Environment 2016; Massachusetts Bureau of Environmental Health 2015; Nebraska Department of Environmental Quality and Nebraska Department of Health and Human Services: Division of Public Health 2018; Vermont Department of Health 2015; Virginia Department of Health 2012).

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APPENDIX A. INTERNATIONAL RECREATIONAL WATER GUIDELINES FOR CYANOTOXINS AND CYANOBACTERIA

Jurisdiction	Recreational Water Guideline Level	Recommended Action
Australia[a]	Cyanobacteria (total): $\geq$ 10 mm^3/L (where known toxins are not present)	Red level action mode; level 2 guideline: • Immediately notify health authorities for advice on health risk. • Make toxicity assessment or toxin measurement of water if this has not already been done. • Health authorities warn of risk to public health (i.e., the authorities make a health risk assessment considering toxin monitoring data, sample type and variability).
	Cyanobacteria (total): $\geq$ 4 mm^3/L (where a known toxin producer is dominant in the total biovolume)	Red level action mode; level 1 guideline: • Immediately notify health authorities for advice on health risk. • Make toxicity assessment or toxin measurement of water if this has not already been done. • Health authorities warn of risk to public health (i.e., the authorities make a health risk assessment considering toxin monitoring data, sample type and variability).
	Cyanobacteria (total): $\geq$ 0.4 to < 10 mm^3/L (where known toxin producers are not present)	Amber level alert mode: • Increase sampling frequency to twice weekly where toxigenic species are dominant within the alert level definition (i.e., total biovolume). • Monitor weekly or fortnightly where other types are dominant. • Make regular visual inspections of water surface for scums. • Decide on requirement for toxicity assessment or toxin monitoring.
	Cyanobacteria (total): $\geq$ 0.4 to <4 mm^3/L (where a known toxin producer is dominant in the total biovolume)	Amber level alert mode: • Increase sampling frequency to twice weekly where toxigenic species are dominant within the alert level definition (i.e., total biovolume). • Monitor weekly or fortnightly where other types are dominant. • Make regular visual inspections of water surface for scums. • Decide on requirement for toxicity assessment or toxin monitoring.
	Cyanobacteria (total): $\geq$ 0.04 to <0.4 mm^3/L	Green level surveillance mode: • Weekly sampling and cell counts at representative locations in the water body where known toxigenic species are present; or • Fortnightly for other types including regular visual inspection of water surface for scums.
	Cyanobacterial scums consistently present	Red level action mode; level 2 guideline: • Immediately notify health authorities for advice on health risk. • Make toxicity assessment or toxin measurement of water if this has not already been done. • Health authorities warn of risk to public health (i.e., the authorities make a health risk assessment considering toxin monitoring data, sample type and variability).

Jurisdiction	Recreational Water Guideline Level	Recommended Action
	Microcystins (total): ≥ 10 µg/L	Red level action mode; level 1 guideline: • Immediately notify health authorities for advice on health risk. • Make toxicity assessment or toxin measurement of water if this has not already been done. • Health authorities warn of risk to public health (i.e., the authorities make a health risk assessment considering toxin monitoring data, sample type and variability).
	Microcystis aeruginosa (total): ≥ 50,000 cells/ml	Red level action mode; level 1 guideline: • Immediately notify health authorities for advice on health risk. • Make toxicity assessment or toxin measurement of water if this has not already been done. • Health authorities warn of risk to public health (i.e., the authorities make a health risk assessment considering toxin monitoring data, sample type and variability).
	Microcystis aeruginosa (total): ≥ 5,000 to < 50,000 cells/ml	Amber level alert mode: • Increase sampling frequency to twice weekly where toxigenic species are dominant within the alert level definition (i.e., total biovolume). • Monitor weekly or fortnightly where other types are dominant. • Make regular visual inspections of water surface for scums. • Decide on requirement for toxicity assessment or toxin monitoring
	Microcystis aeruginosa (total): ≥ 500 to < 5,000 cells/ml	Green level surveillance mode: • Weekly sampling and cell counts at representative locations in the water body where known toxigenic species are present; or • Fortnightly for other types including regular visual inspection of water surface for scums.
Canada[d]	Cyanobacteria (total): ≥ 100,000 cells/ml	Issue swimming advisory.
	Detection of a cyanobacterial bloom	Issue beach closure.
	Microcystins (total): ≥ 20 µg/L (expressed as microcystin-LR)	Issue swimming advisory.
Cuba[c]	Any report of toxic effect in humans or animals	Action (in red): as for "Alert," but with increased actions for public communication.
	Benthic mats: < 40 percent coverage of surfaces with any cyanobacteria; > 20 percent with toxicogenic cyanobacteria; > 50 percent with potentially toxicogenic cyanobacteria	Alert: increased sampling (weekly and more sites); daily inspection; notification to public health unit and local managers; report to local government; warning of the public.

Jurisdiction	Recreational Water Guideline Level	Recommended Action
	(particularly where they are visibly detaching and accumulating in scum)	
	Cyanobacteria: < 500 cells/ml	Monthly visual inspection.
	Cyanobacteria: ≥ 1 of the species known as potentially toxic	Alert: increased sampling (weekly and more sites); daily inspection; notification to public health unit and local managers; report to local government; warning of the public.
	Phytoplankton cells: ≥ 20,000 to < 100,000 cells/ml, > 50 percent of cells cyanobacteria	Alert: increased sampling (weekly and more sites); daily inspection; notification to public health unit and local managers; report to local government; warning of the public.
	Phytoplankton: > 0 to < 1,500 cells/ml	Monthly visual inspection and sampling at least four months per year.
	Scum consistently present; confirmed bloom persistence	Action (in red): as for "Alert," but with increased actions for public communication.
Czech Republic[c]	Cells: > 100,000 cells/ml	Second warning level: closure for public recreation.
	Cells: > 20,000 cells/ml	First warning level (not otherwise specified).
Denmark[c]	Chlorophyll *a*: > 50 µg/L, dominated by cyanobacteria	Relevant authorities are informed and decide when and how the public should be informed; warnings include signs, media, and contact to local user groups such as kindergartens, scouts, water sports clubs.
	Visible surface scum	Relevant authorities are informed and decide when and how the public should be informed; warnings include signs, media, and contact to local user groups such as kindergartens, scouts, water sports clubs.
European Union[f]	Cyanobacterial proliferation (occurrence)	When cyanobacterial proliferation occurs and a health risk has been identified or presumed, adequate management measures shall be taken immediately to prevent exposure, including information to the public.
	Cyanobacterial proliferation (potential for)	Appropriate monitoring shall be carried out to enable timely identification of health risks.
Finland[c]	Algae (includes cyanobacteria): detected	Level 1: Possibly microscopic examination and even toxin analysis if there is a specific cause such as very popular beach or reports of adverse health effects or animal deaths.

Jurisdiction	Recreational Water Guideline Level	Recommended Action
	Algae (includes cyanobacteria): high amount	Level 2: Preferably microscopical examination; toxin analysis; warning of the public is compulsory.
	Algae (includes cyanobacteria): very high amount	Level 3: Preferably microscopical examination; toxin analysis; warning of the public is compulsory.
France[c]	Bloom, scum, change in water color	Microscopy examination. If cyanobacteria are absent: no further action. If present: counting and genus identification.
	Cyanobacteria: < 20,000 cells/ml (± 20 percent)	Active daily monitoring. Counting at least on a weekly basis. Normal recreational activity at the site.
	Cyanobacteria: > 100,000 cells/ml (± 20 percent)	Bathing and recreational activities are restricted. Public is informed.
	Cyanobacteria: ≥ 20,000 to < 100,000 cells/ml (± 20 percent)	Active daily monitoring. Counting on a weekly basis. Recreational activities are still allowed; the public is informed by posters on site.
	Microcystins: 25 µg/L (± 5 percent)	If microcystins < 25 µg/L bathing and recreational activities are restricted. If microcystins > 25 µg/L bathing is banned and recreational activities are restricted. In either case, public is informed.
	Visible scum or foam in recreational or bathing area	All water activities in this area are prohibited. Restrictions do not necessarily apply to the whole recreational site. Other areas without scum may still be open.
Germany[c]	Secchi Disk reading > 1 m AND biovolume: < 1 mm^3/L	Monitor further cyanobacterial development.
	Secchi Disk reading > 1 m AND biovolume: ≥ 1 mm^3/L	Publish warnings, discourage bathing, consider temporary closure.
	Secchi Disk reading > 1 m AND chlorophyll *a* (with dominance by cyanobacteria): < 40 µg/L	Monitor further cyanobacterial development.
	Secchi Disk reading > 1 m AND chlorophyll *a* (with dominance by cyanobacteria): ≥ 40 µg/L	Publish warnings, discourage bathing, consider temporary closure.

Jurisdiction	Recreational Water Guideline Level	Recommended Action
	Secchi Disk reading > 1 m AND microcystins: < 10 µg/L	Monitor further cyanobacterial development.
	Secchi Disk reading > 1 m AND microcystins: ≥ 10 µg/L	Publish warnings, discourage bathing, consider temporary closure.
	Visible heavy scums and/or microcystins: > 100 µg/L	Publish warnings, discourage bathing, temporary closure is recommended.
Hungary[c]	Cell count: ≥ 50,000 to < 100,000 cells/ml	No recommended actions listed, water body classification: Acceptable.
	Cell count: < 20,000 cells/ml	No recommended actions listed, water body classification: Excellent.
	Cell count: ≥ 20,000 to < 50,000 cells/ml	No recommended actions listed, water body classification: Good.
	Cell count: ≥ 100,000 cells/ml	No recommended actions listed, water body classification: Unacceptable.
	Chlorophyll *a* (with dominance by cyanobacteria): < 10 µg/L	No recommended actions listed, water body classification: Excellent.
	Chlorophyll *a* (with dominance by cyanobacteria): ≥ 10 to < 25 µg/L	No recommended actions listed, water body classification: Good.
	Chlorophyll *a* (with dominance by cyanobacteria): ≥ 25 to < 50 µg/L	No recommended actions listed, water body classification: Acceptable.
	Chlorophyll *a* (with dominance by cyanobacteria): ≥ 50 µg/L	No recommended actions listed, water body classification: Unacceptable.
	Microcystins: ≥ 4 to < 10 µg/L	No recommended actions listed, water body classification: Good.
	Microcystins: ≥ 10 to < 20 µg/L	No recommended actions listed, water body classification: Acceptable.
	Microcystins: < 4 µg/L	No recommended actions listed, water body classification: Excellent.
	Microcystins: ≥ 20 µg/L	No recommended actions listed, water body classification: Unacceptable.

Jurisdiction	Recreational Water Guideline Level	Recommended Action
Italy[i]	Cyanobacterial cell count for cyanotoxin-producing species other than microcystins (e.g., cylindrospermopsin, anatoxin-a) > 100,000 cells/ml (± 20 percent)	Emergency phase: weekly sampling and intensified visual inspection; quantification of all identified cyanotoxins; health surveillance; temporary bans on bathing and removal of scums from water and shoreline in addition to alert phase management measure.
	Cylindrospermopsin and anatoxin-a > 20 µg/L	Emergency phase: weekly sampling and intensified visual inspection; quantification of all identified cyanotoxins; health surveillance; temporary bans on bathing and removal of scums from water and shoreline in addition to alert phase management measures.
	Microcystin-LR: > 20 µg/L equivalents	Emergency phase: weekly sampling and intensified visual inspection; quantification of all identified cyanotoxins; health surveillance; temporary bans on bathing and removal of scums from water and shoreline in addition to alert phase management measures.
	Total cyanobacterial cell count > 20,000 cells/ml (± 20 percent) AND microcystin-LR < 20 µg/L equivalents	Alert phase: weekly sampling and visual inspection every 2 days; assessment of bloom extent and stretches of coastline affected; identify presence of cyanotoxins other than microcystins (when relevant); management measures put in place to inform citizens and prevent hazardous exposures using informative and warning panels/signs at waterfront and/or at beach access points, newsletters, brochures, publications on regional and national websites, local information systems, social network, and a Ministry toll-free number.
	Transparency ≥ 1 m AND total phosphorus < 20 µg/L	Routine phase 1: monthly sampling.
	Transparency ≥ 1 m AND total phosphorus > 20 µg/L AND total cyanobacterial cell count ≤ 2,000 cells/ml	Routine phase 2: monthly sampling and weekly visual inspection.
	Transparency ≤ 1 m AND total phosphorus > 20 µg/L AND total cyanobacterial cell count > 2,000 to < 20,000 cells/ml (± 20 percent)	Routine phase 3: fortnightly sampling and weekly visual inspection.
	Visible surface scum	Emergency phase: weekly sampling and intensified visual inspection; quantification of all identified cyanotoxins; health surveillance; temporary bans on bathing and removal of scums from water and shoreline in addition to alert phase management measures.
Netherlands[c]	Biovolume (cyanobacterial cell count): > 0 to < 2.5 mm^3/L	Surveillance level: continue fortnightly monitoring

Jurisdiction	Recreational Water Guideline Level	Recommended Action
	Biovolume (cyanobacterial cell count): > 15 mm^3/L (if 80 percent dominance of microcystin producers and microcystins < 20 µg/L, revert to Alert Level 1).	Alert level 2: weekly monitoring and advice against bathing (by public authority): "You are advised not to bathe in this water;" prohibition by local authority is possible.
	Biovolume (cyanobacterial cell count): ≥ 2.5 to ≤ 15 mm^3/L	Alert level 1: weekly monitoring and issue warning (by site operator) for duration of that week: "Toxic blue-green algae. Risk of skin irritation or intestinal problems." In case of daily site inspection, reevaluate the warning on a daily basis.
	Chlorophyll *a*: > 0 to < 12.5 µg/L	Surveillance level: continue fortnightly monitoring.
	Chlorophyll *a*: > 75 µg/L	Alert level 2: weekly monitoring and advice against bathing (by public authority): "You are advised not to bathe in this water;" prohibition by local authority is possible.
	Chlorophyll *a*: ≥ 12.5 to ≤ 75 µg/L	Alert level 1: weekly monitoring and issue warning (by site operator) for duration of that week: "Toxic blue-green algae. Risk of skin irritation or intestinal problems." In case of daily site inspection, reevaluate the warning on a daily basis.
	Surface scum: category 1	Surveillance level: continue fortnightly monitoring.
	Surface scum: category 2	Alert level 1: weekly monitoring and issue warning (by site operator) for duration of that week: "Toxic blue-green algae. Risk of skin irritation or intestinal problems." In case of daily site inspection, reevaluate the warning on a daily basis.
	Surface scum: category 3	Alert level 2: weekly monitoring and advice against bathing (by public authority): "You are advised not to bathe in this water"; prohibition by local authority is possible.
New Zealand[h]	Cyanobacteria (benthic): 20−50 percent coverage of potentially toxigenic cyanobacteria attached to substrate	Alert (amber mode): • Notify the public health unit. • Increase sampling to weekly. • Recommend erecting an information sign. • Consider increasing the number of survey sites. • If toxigenic cyanobacteria dominate the samples, testing for cyanotoxins is advised. If cyanotoxins are detected in mats or water samples, consult the testing laboratory to determine if levels are hazardous.
	Cyanobacteria (benthic): greater than 50 percent coverage of potentially toxigenic cyanobacteria attached to substrate	Action (red mode) situation 1: • Immediately notify the public health unit. • If potentially toxic taxa are present (see Table 2) then consider testing samples for cyanotoxins. • Notify the public of the potential risk to health.

Jurisdiction	Recreational Water Guideline Level	Recommended Action
	Cyanobacteria (benthic): Up to 20 percent coverage of potentially toxigenic cyanobacteria attached to substrate	Surveillance (green mode): • Undertake fortnightly surveys between spring and autumn at representative locations in the water body where known mat proliferations occur and where there is recreational use.
	Cyanobacteria (benthic): up to 50 percent where potentially toxigenic cyanobacteria are visibly detaching from the substrate, accumulating as scums along the river's edge or becoming exposed on the river's edge as the river level drops.	Action (red mode) situation 2: • Immediately notify the public health unit. • If potentially toxic taxa are present (see Table 2) then consider testing samples for cyanotoxins. • Notify the public of the potential risk to health.
	Cyanobacteria (total): < 0.5 mm^3/L (biovolume equivalent of the combined total of all cyanobacteria)	Surveillance (green mode): • Undertake weekly or fortnightly visual inspection and sampling of water bodies where cyanobacteria are known to proliferate between spring and autumn.
	Cyanobacteria (total): $\leq$ 500 cells/ml	Surveillance (green mode): • Undertake weekly or fortnightly visual inspection and sampling of water bodies where cyanobacteria are known to proliferate between spring and autumn.
	Cyanobacteria (total): $\geq$ 1.8 mm^3/L (biovolume equivalent of potentially toxic cyanobacteria)	Action (red mode) situation 1: • Continue monitoring as for alert (amber mode). • If potentially toxic taxa are present (see Table 1), then consider testing samples for cyanotoxins • Notify the public of a potential risk to health.
	Cyanobacteria (total): $\geq$ 0.5 to < 1.8 mm^3/L (biovolume equivalent of potentially toxic cyanobacteria)	Alert (amber mode): • Increase sampling frequency to at least weekly. • Notify the public health unit. • Multiple sites should be inspected and sampled.
	Cyanobacteria (total): $\geq$ 0.5 to < 10 mm^3/L (total biovolume of all cyanobacterial material where the cyanobacterial population has been tested and shown not to contain known toxins)	Alert (amber mode): • Increase sampling frequency to at least weekly. • Notify the public health unit. • Multiple sites should be inspected and sampled.
	Cyanobacteria (total): $\geq$ 10 mm^3/L (total biovolume of all cyanobacterial material	Action (red mode) situation 2:

Jurisdiction	Recreational Water Guideline Level	Recommended Action
	where the cyanobacterial population has been tested and shown not to contain known toxins)	• Continue monitoring as for alert (amber mode). • If potentially toxic taxa are present (see Table 1), then consider testing samples for cyanotoxins. • Notify the public of a potential risk to health.
	Cyanobacterial scums consistently present for more than several days in a row	Action (red mode) situation 3: • Continue monitoring as for alert (amber mode). • If potentially toxic taxa are present (see Table 1), then consider testing samples for cyanotoxins. • Notify the public of a potential risk to health.
	Microcystins (total): ≥ 12 µg/L	Action (red mode) situation 1: • Continue monitoring as for alert (amber mode). • If potentially toxic taxa are present (see Table 1), then consider testing samples for cyanotoxins. • Notify the public of a potential risk to health.
Poland[c]	Visible blooms	Sampling of bathing sites not less than 4 times per season (the interval between sampling does not exceed one month), including responses to cyanobacteria if blooms are observed.
Scotland[e]	Chlorophyll *a*: ≥ 10 µg/L with dominance of cyanobacteria	1. Watch for scum or conditions conducive to scums. 2. Discourage bathing and further investigate hazard. 3. Post on-site risk advisory signs. 4. Inform relevant authorities.
	Cyanobacteria: ≥ 20,000 cells /ml	1. Watch for scum or conditions conducive to scums. 2. Discourage bathing and further investigate hazard. 3. Post on-site risk advisory signs. 4. Inform relevant authorities.
	Cyanobacterial scum formation in bathing areas	1. Immediate action to control contact with scums; possible prohibition of swimming and other water contact activities. 2. Public health follow-up investigation. 3. Inform public and relevant authorities.
Singapore[c]	Chlorophyll *a*: ≤ 50 µg/L (of 95 percent of a 3-year rolling period)	Status of the sites reviewed annually. If the assessment is that the water body is unsuitable for primary water contact activities, the public is notified.
Spain[c]	Cyanobacteria proliferation potential (High, Medium, Low)	Criteria for assessment of health risk and response are set locally; some health authorities use WHO scheme, others include further risk parameters (such as number of users, type of use); temporary closure has occasionally occurred based on the abundance of cyanobacteria.

Jurisdiction	Recreational Water Guideline Level	Recommended Action
Turkey[c]	Cells: < 20,000 cells/ml	Level 1: recreational activities are allowed to continue and users are informed by posters on site. Monitoring (sampling, counting and species identification) should be done fortnightly.
	Cells: 20,000–100,000 cells/ml	Level 2: At > 20 000 cells/mL, microcystins are analyzed. If microcystin-LR equivalents >25 µg/L, immediate action to inform relevant authorities and public. Discourage users from swimming and other water contact activities by advisory signs on site.
	Chlorophyll *a* (if dominated by cyanobacteria): < 10 µg/L	Level 1: recreational activities are allowed to continue and users are informed by posters on site. Monitoring (sampling, counting and species identification) should be done fortnightly.
	Microcystin-LR: < 10 µg/L equivalents	Level 1: recreational activities are allowed to continue and users are informed by posters on site. Monitoring (sampling, counting and species identification) should be done fortnightly.
	Microcystin-LR: > 25 µg/L equivalents	Level 2: At > 20,000 cells/mL, microcystins are analyzed. If microcystin-LR equivalents >25 µg/L, immediate action to inform relevant authorities and public. Discourage users from swimming and other water contact activities by advisory signs on site.
	Visible scum in bathing area	Level 3: all activities in the water may be prohibited.
World Health Organization (WHO)[b,g]	Chlorophyll *a*: 10 µg/L with dominance of cyanobacteria	Low risk: post on-site advisory signs, inform relevant authorities.
	Chlorophyll *a*: 50 µg/L with dominance of cyanobacteria	Moderate risk: watch for scums or conditions conducive to scums, discourages swimming and further investigate hazard, post on-site risk advisory signs, inform relevant authorities.
	Cyanobacteria: 100,000 cells/ml	Moderate risk: watch for scums or conditions conducive to scums, discourages swimming and further investigate hazard, post on-site risk advisory signs, inform relevant authorities.
	Cyanobacteria: 20,000 cells/ml	Low risk: post on-site advisory signs, inform relevant authorities.
	Cyanobacterial scum formation in areas where whole-body contact and/or risk of ingestions/aspiration occur	High risk: immediate action to control contact with scums, possible prohibition of swimming and other water contact activities, public health follow-up investigation, inform public and relevant authorities.

[a] Australian Government National Health and Medical Research Council (2008). Guidelines for Managing Risk in Recreational Water.
[b] Chorus, I. and Bartram, J. (eds.) (1999). Toxic cyanobacteria in water: A guide to public health significance, monitoring and management. E. and F.N. Spon, Chapman, and Hall, London, United Kingdom.
[c] Federal Environment Agency (Germany) (2012). Current approaches to Cyanotoxin risk assessment, risk management and regulations in different countries.
[d] Health Canada (2012). Guidelines for Canadian Recreational Water Quality, Third Edition. Water, Air and Climate Change Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario. (Catalogue No H129-15/2012E).
[e] Scottish Government Health and Social Care Directorates Blue-Green Algae Working Group (2012). Cyanobacteria (Blue-Green Algae) in Inland and Inshore Waters:

Assessment and Minimization of Risks to Public Health.

[f] European Parliament and the Council of the European Union (2006). Directive 2006/7/EC of the European Parliament and of the Council of 15 February 2006 concerning the management of bathing water quality and repealing Directive 76/160/EEC.

[g] WHO (World Health Organization) (2003). Guidelines for Safe Recreational Water Environments: Volume 1: Coastal and Fresh Waters. World Health Organization.

[h] Wood, S; Hamilton, D; Safi, K; Williamson, W. (2008). New Zealand Guidelines for Cyanobacteria in Recreational Fresh Waters: Interim Guidelines. New Zealand Ministry for the Environment and Ministry of Health.

[i] Funari, W; Manganelli, M; Buratti, FM; Testai, E. (2017). Cyanobacteria blooms in water: Italian guidelines to assess and manage the risk associated to bathing and recreational activities. *Science of the Total Environment, 598*, 867-880.

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APPENDIX B. STATE RECREATIONAL WATER GUIDELINES FOR CYANOTOXINS AND CYANOBACTERIA

EPA compiled the information presented in this appendix based on searches of state websites for publicly available information regarding guidelines or action levels for cyanotoxins and cyanobacteria. Online searches for state guidance were conducted in 2015, 2016, and 2018. Direct personal communication of state guidelines and state public comments on the draft AWQC revealed some updated information.

Table B-1. Summary Counts of State Recreational Water Guidelines for Cyanotoxins and Cyanobacteria by Type and Scope of Guidelines

Recreational Water Guideline Type and Scope	Number of States and List of States	Additional Information
Quantitative guidelines for cyanobacteria only	5 states: Arizona, Idaho, Maine, New Hampshire, Wisconsin	Measurements for these criteria include cyanobacterial cell densities, proportion of toxigenic cyanobacteria, chlorophyll concentration, and Secchi disk depth measurements.
Quantitative guidelines for cyanotoxins only	4 states: Illinois, Iowa, Nebraska, Ohio	State guidelines address four cyanotoxins in order from most to least common: microcystins (24 states) anatoxin-a (11 states) cylindrospermopsin (9 states) saxitoxin (5 states)
Quantitative guidelines for cyanotoxins and either quantitative or qualitative guidelines for cyanobacteria	20 states: California, Colorado, Connecticut, Indiana, Kansas, Kentucky, Maryland, Massachusetts, Michigan, New Jersey, New York, North Dakota, Oklahoma, Oregon, Pennsylvania, Rhode Island, Utah, Vermont, Virginia, Washington	
Qualitative guidelines only	6 states: Delaware, Florida, Missouri, Montana, North Carolina, West Virginia	Examples include: presence of surface scum visible discoloration presence of potentially toxic algae presence/absence test for microcystins
Guidelines under development	4 states: Arkansas, Georgia, Minnesota, Wyoming	

Note: The EPA found that Texas and North Carolina published guidelines in the past, but the guidelines are no longer found on their websites. Missouri is in the process of developing quantitative thresholds.

Table B-2. State Recreational Water Quality Guideline for Cyanotoxins and Cyanobacteria Sorted by Type

State	Recreational Water Guideline Level	Recommended Action	Reference
States with Guidelines Based on Cyanobacteria Only			
Arizona	Blue-green algae (mean value based on a minimum of two sample events within one peak season): 20,000 cells/ml and Chlorophyll *a* result (mean value based on a minimum of two sample events within one peak season) in target range	Violation of the Narrative Nutrient Standard.	Arizona Department of Environmental Quality (2008). Narrative Nutrient Standard Implementation Procedures for Lakes and Reservoirs. http://www.azdeq.gov/environ/water/standards/download/draft_nutrient.pdf. Last Accessed: 11/27/2018.
Idaho	*Microcystis* or *Planktothrix*: >40,000 cells/ml	Public health advisory posting by Public Health District in conjunction with water body operator.	IDEQ (Idaho Department of Environmental Quality) (2015). Blue-Green Algae Bloom Response Plan: Final. http://www.healthandwelfare.idaho.gov/Portals/0/Health/EnvironmentalHealth/Idaho%20Blue-Green%20Algae%20Response%20PlanFinal.pdf. Last Accessed: 11/27/2018.
	Sum of all potentially toxigenic taxa: ≥ 100,000 cells/ml	Public health advisory posting by Public Health District in conjunction with water body operator.	
	Visible surface scum that is associated with toxigenic species	Public health advisory posting by Public Health District in conjunction with water body management agency.	

State	Recreational Water Guideline Level	Recommended Action	Reference
Maine	Secchi disk reading < 2 meters caused by algae	Body of water considered impaired, but still safe to swim.	Maine Department of Environmental Protection (2013). Reports of Algal Blooms. http://www.maine.gov/dep/water/lakes/repbloom.html. Last Accessed: 11/27/2018.
New Hampshire	Cyanobacteria: > 50 percent of total cell counts from toxigenic cyanobacteria OR the cyanobacteria cell count is greater than 70,000 cells per ml of water	Post beach advisory.	New Hampshire Department of Environmental Services (2014). Beach Advisories. http://des.nh.gov/organization/divisions/water/wmb/beaches/advisories.htm. Last Accessed: 11/27/2018.
Wisconsin	Cyanobacteria: > 100,000 cells/ml	Post health advisory and possible beach closure.	Wisconsin Department of Natural Resources (2012). Draft Blue-Green Algae Section of 303 (d) Report – 7/3/2012: Harmful Algal Blooms. http://dnr.wi.gov/lakes/bluegreenalgae/documents/HarmfulAlgalBloomsvs2.pdf. Last Accessed: 11/27/2018. Wisconsin Department of Health Services (2016). Harmful Algal Blooms Toolkit: A Planning Guide for Public Health and Emergency Response Professionals. https://www.dhs.wisconsin.gov/publications/p0/p00853.pdf. Last Accessed: 11/27/2018.
	Visible scum layer	Post health advisory and possible beach closure.	Werner M, and Masnado R (2014). Guidance for Local Health Departments: Cyanobacteria and Human Health. http://city.milwaukee.gov/ImageLibrary/Groups/healthAuthors/DCP/PDFs/CyanobacteriaLHD.pdf. Last Accessed: 11/27/2018. Wisconsin Department of Health Services (2016). Harmful Algal Blooms Toolkit: A Planning Guide for Public Health and

State	Recreational Water Guideline Level	Recommended Action	Reference
			Emergency Response Professionals. https://www.dhs.wisconsin.gov/publications/p0/p00853.pdf. Last Accessed: 11/27/2018.
States with Guidelines Based on Cyanotoxin(s) Only			
Illinois	Microcystin-LR: > 10 µg/L	Appropriate lake management personnel and Illinois EPA staff will be notified; follow-up monitoring by the Illinois EPA may occur as professional judgment dictates and staff, laboratory, and financial resources allow.	Illinois Environmental Protection Agency (2013). 2013 Statewide Harmful Algal Bloom Program. https://www2.illinois.gov/epa/topics/water-quality/monitoring/algal-bloom/Pages/2013-program.aspx. Last Accessed: 11/27/2018. Illinois Environmental Protection Agency (2018). Blue-Green Algae and Harmful Algal Blooms. https://www2.illinois.gov/epa/topics/water-quality/monitoring/algal-bloom/Pages/default.aspx. Last Accessed: 12/5/2018.
Iowa	Microcystin: ≥ 20 µg/L	Warnings are posted at state park beaches.	Iowa Environmental Council (2018). Toxic Blue-Green Algae: A Threat to Iowa Beachgoers. http://www.iaenvironment.org/our-work/clean-water-and-land-stewardship/swimming-advisories. Last Accessed: 11/27/2018.
Nebraska	Microcystin: ≥ 20 µg/L	Health alert; signs posted advising public to use caution; affected swimming beaches will be closed; boating and other recreational activities will be allowed, but public advised to use caution and avoid prolonged exposure to the water.	Nebraska Department of Environmental Quality and Nebraska Department of Health and Human Services: Division of Public Health (2018). Fact Sheet: Precautions and facts regarding toxic algae at Nebraska Lakes.

State	Recreational Water Guideline Level	Recommended Action	Reference
			http://deq.ne.gov/NDEQProg.nsf/OnWeb/ENV042607. Last Accessed: 5/10/2019.
Ohio	Anatoxin-a: 300 µg/L	Issue no contact advisory.	Ohio EPA (2016). State of Ohio Harmful Algal Bloom Response Strategy For Recreational Waters. http://epa.ohio.gov/portals/35/hab/HABResponseStrategy.pdf. Last Accessed: 11/27/2018.
	Anatoxin-a: 80 µg/L	Issue recreational public health advisory.	
	Cylindrospermopsin: 20 µg/L	Issue no contact advisory.	
	Cylindrospermopsin: 5 µg/L	Issue recreational public health advisory.	
	Microcystins: 20 µg/L	Issue no contact advisory.	
	Microcystins: 6 µg/L	Issue recreational public health advisory.	
	Saxitoxin: 0.8 µg/L	Issue recreational public health advisory.	
	Saxitoxin: 3 µg/L	Issue no contact advisory.	
States with Guidelines Based on Cyanobacteria and Cyanotoxin(s)			
California	Anatoxin-a: detection using an analytical method that detects <1 µg/L	Caution trigger level: increase monitoring and post caution sign warning people to stay away from scum and warning people to keep pets and livestock away from water and scum.	Butler N, Carlisle J, Kaley KB, and Linville R (2012). Toxicological Summary and Suggested Action Levels to Reduce Potential Adverse Health Effects of Six Cyanotoxins. http://www.waterboards.ca.gov/water_issues/programs/peer_review/docs/calif_cyanotoxins/cyanotoxins053112.pdf. Last Accessed: 11/27/2018. Cyanobacteria Harmful Algal Bloom Network (2016a). Appendix to the CCHAB Preliminary Changes to the Statewide Voluntary Guidance of CyanoHABs in Recreational Waters.
	Anatoxin-a: 20 µg/L	Warning tier 1: post warning sign stating that swimming is not recommended and that pets and livestock should be kept away from the water.	
	Anatoxin-a: 90 µg/L	Danger tier 2: post sign stating that there is a present danger and that people, pets and livestock should stay out of the water and away from water spray.	
	Cylindrospermopsin: 1 µg/L	Caution trigger level: increase monitoring and post caution sign warning people to stay away from scum and warning people to keep pets and livestock away from water and scum.	

State	Recreational Water Guideline Level	Recommended Action	Reference
	Cylindrospermopsin: 4 µg/L	Warning tier 1: post warning sign stating that swimming is not recommended and that pets and livestock should be kept away from the water.	http://www.mywaterquality.ca.gov/monitoring_council/cyanohab_network/docs/appendix_a.pdf. Last Accessed: 11/27/2018. Cyanobacteria Harmful Algal Bloom Network (2016b). Table 1: CyanoHAB trigger levels for human health. http://www.mywaterquality.ca.gov/monitoring_council/cyanohab_network/docs/triggers.pdf. Last Accessed: 11/27/2018.
	Cylindrospermopsin: 17 µg/L	Danger tier 2: post sign stating that there is a present danger and that people, pets and livestock should stay out of the water and away from water spray.	
	Microcystins: 0.8 µg/L	Caution trigger level: increase monitoring and post caution sign warning people to stay away from scum and warning people to keep pets and livestock away from water and scum.	
	Microcystins: 6 µg/L	Warning tier 1: post warning sign stating that swimming is not recommended and that pets and livestock should be kept away from the water.	
	Microcystins: 20 µg/L	Danger tier 2: post sign stating that there is a present danger and that people, pets and livestock should stay out of the water and away from water spray.	
	Site-specific indicators of cyanobacteria (e.g., blooms, scums, mats)	Caution trigger level: increase monitoring and post caution sign warning people to stay away from scum and warning people to keep pets and livestock away from water and scum.	
	Toxin-producing cyanobacteria: 4,000 cells/ml	Caution trigger level: increase monitoring and post caution sign warning people to stay away from scum and warning people to keep pets and livestock away from water and scum.	
Colorado	Anatoxin-a: ≥ 7 µg/L	Issue toxic algae caution: a. post sign with “caution” language. b. perform routine testing for toxin levels. bi. if test results are below caution thresholds, test at	Colorado Department of Public Health and Environment. Algae bloom risk-management toolkit for recreational waters.

State	Recreational Water Guideline Level	Recommended Action	Reference
		least once per week until algae visually subsides. bii. if test results are above caution thresholds, test at least twice per week until toxin levels are below caution thresholds for two consecutive tests. c. notify drinking water providers and county health department if toxin levels exceed the caution thresholds. d. toxic algae caution ends when there is no visual evidence of algae and toxin levels are non-detectable for two consecutive weeks. di. notify drinking water providers and county health department that bloom has ended. dii. remove "caution" sign.	https://www.colorado.gov/pacific/cdphe/harmful-algae-blooms . Last Accessed: 11/27/2018
	Cylindrospermopsin: ≥ 7 µg/L	Issue toxic algae caution: a. post sign with "caution" language. b. perform routine testing for toxin levels. bi. if test results are below caution thresholds, test at least once per week until algae visually subsides. bii. if test results are above caution thresholds, test at least twice per week until toxin levels are below caution thresholds for two consecutive tests. c. notify drinking water providers and county health department if toxin levels exceed the caution thresholds. d. toxic algae caution ends when there is no visual evidence of algae and toxin levels are non-detectable for two consecutive weeks. di. notify drinking water providers and county health department that bloom has ended. dii. remove "caution" sign.	
	Microcystin-LR: ≥ 10 µg/L and < 20 µg/L	Issue toxic algae caution: a. post sign with "caution" language. b. perform routine testing for toxin levels. bi. if test results are below caution thresholds, test at least once per week until algae visually subsides.	

State	Recreational Water Guideline Level	Recommended Action	Reference
		bii. if test results are above caution thresholds, test at least twice per week until toxin levels are below caution thresholds for two consecutive tests. c. notify drinking water providers and county health department if toxin levels exceed the caution thresholds. d. toxic algae caution ends when there is no visual evidence of algae and toxin levels are non-detectable for two consecutive weeks. di. notify drinking water providers and county health department that bloom has ended. dii. remove "caution" sign.	
	Microcystin-LR: ≥ 20 µg/L	Issue toxic algae warning: a. immediately post sign with "warning" language. b. take necessary steps to prevent contact with water in affected area for humans and pets. c. notify drinking water providers and county health department if toxin levels exceed warning thresholds. d. test at least twice per week until toxin levels are below warning thresholds for two consecutive tests. e. posting can be reduced to "caution" language when microcystin test results drop below the warning threshold and no new human illness or pet deaths have been reported for two consecutive weeks.	
	Saxitoxin: ≥ 4 µg/L	Issue toxic algae caution: a. post sign with "caution" language. b. perform routine testing for toxin levels. bi. if test results are below caution thresholds, test at least once per week until algae visually subsides. bii. if test results are above caution thresholds, test at least twice per week until toxin levels are below caution thresholds for two consecutive tests. c. notify drinking water providers and county health department if toxin levels exceed the caution thresholds.	

State	Recreational Water Guideline Level	Recommended Action	Reference
		d. toxic algae caution ends when there is no visual evidence of algae and toxin levels are non-detectable for two consecutive weeks. di. notify drinking water providers and county health department that bloom has ended. dii. remove "caution" sign.	
	Potentially toxic algae are visible	Issue toxic algae caution: a. post sign with "caution" language. b. perform routine testing for toxin levels. bi. if test results are below caution thresholds, test at least once per week until algae visually subsides. bii. if test results are above caution thresholds, test at least twice per week until toxin levels are below caution thresholds for two consecutive tests. c. notify drinking water providers and county health department if toxin levels exceed the caution thresholds. d. toxic algae caution ends when there is no visual evidence of algae and toxin levels are non-detectable for two consecutive weeks. di. notify drinking water providers and county health department that bloom has ended. dii. remove "caution" sign.	
Connecticut	Visual rank category 2: cyanobacteria present in low numbers; there are visible small accumulations but water is generally clear; OR blue-green algae cells > 20,000 cells/ml and < 100,000 cells/ml	Notify Connecticut Department of Public Health (CT DPH), Connecticut Department of Energy and Environmental Protection (CT DEEP); increase regular visual surveillance until conditions change; consider cautionary postings at public access points.	Connecticut Department of Public Health and Connecticut Department of Energy and Environmental Protection (CDPH and CDEEP) (2017). Guidance to Local Health Departments for Blue-Green Algae Blooms in Recreational Freshwaters. http://www.ct.gov/deep/lib/deep/water/water_quality_management/monitoringpubs/bluegreenalgaeblooms_guidanceforlhds_2017version.pdf. Last Accessed: 11/27/2018.
	Visual rank category 3: cyanobacteria present in high numbers; scums may or may not be present; water is discolored throughout; large areas affected; color assists to rule out sediment and other algae; OR blue-green algae cells > 100,000 cells/ml	Update/inform CTDPH and CTDEEP and expand risk communication efforts; collect samples for analysis and/or increase frequency of visual assessment; POSTED BEACH CLOSURE: if public has beach access, alert water users that a blue-green	

State	Recreational Water Guideline Level	Recommended Action	Reference
		algae bloom is present; POSTED ADVISORY: at other impacted access points.	Connecticut Department of Energy and Environmental Protection (CDEEP). (2017). Comment Letter Regarding Human Health Recreational Ambient Water Quality Criteria and/or Swimming Advisories for Microcystins and Cylindrospermopsin. March 20, 2017. Docket No. EPA-HQ-OW-2016-0715. https://www.regulations.gov/docket?D=EPA-HQ-OW-2016-0715. Last accessed: 11/27/2018.
	Anatoxin-a: 80 µg/L	Issue recreation advisory.	
Indiana	Blue-green algae: 100,000 cells/ml	Issue recreation advisory.	Indiana Department of Environmental Management (2018). Blue-Green Algae: Indiana Reservoir and Lake Update. http://www.in.gov/idem/algae/. Last Accessed: 11/27/2018.
	Cylindrospermopsin: 8 µg/L	Issue recreation advisory.	
	Microcystin-LR: 20 µg/L	Close beaches.	
	Microcystin-LR: 4 µg/L	Issue recreation advisory.	
	Cyanobacteria: ≥ 10,000,000 cells/ml	Recommended that all in-lake recreation cease and that picnic, camping and other public land activities adjacent to affected waters be closed.	
Kansas	Cyanobacteria: ≥ 250,000 cells/ml	Issue public health warning.	Kansas Department of Health and Environment (2015). Guidelines for Addressing Harmful Algal Blooms in Kansas Recreational Waters. http://www.kdheks.gov/algae-illness/download/HAB_policy.pdf. Last Accessed: 11/27/2018. Kansas Department of Health and Environment (2015). Harmful Algal Blooms (HABs): KDHE Agency Response Plan. http://www.kdheks.gov/algae-illness/download/HAB_response_plan.pdf. Last Accessed: 11/27/2018.
	Cyanobacteria: ≥ 80,000 and < 250,000 cells/ml	Issue public health watch.	
	Microcystin: ≥ 2,000 µg/L	Recommended that all in-lake recreation cease and that picnic, camping and other public land activities adjacent to affected waters be closed.	
	Microcystin: ≥ 20 µg/L	Issue public health warning.	
	Microcystin: ≥ 4 and < 20 µg/L	Issue public health watch.	
	Blue-green algae: > 100,000 cells/ml	Issue an HAB advisory.	

State	Recreational Water Guideline Level	Recommended Action	Reference
Kentucky	Microcystins: > 20 µg/L	Issue recreational use advisory.	Kentucky Department for Environmental Protection (2014). Harmful Algal Blooms: Background. http://water.ky.gov/waterquality/Documents/HAB_FACTs/HAB%20Background%20Fact%20Sheet.pdf. Last Accessed: 11/27/2018.
	Microcystis aeruginosa or other potential microcystin-producing blue-green algae > 40,000 cells/ml, and samples contain microcystins: > 10 ppb	Put up signs advising public of health risk, notify local press (through joint DHMH, DNR, MDE press release) and coordinate with local health department, place advisory information on DNR web site (Eyes on the Bay), Maryland Healthy Beaches web site if a swimming beach is affected, or other local web site. MDE will initiate emergency closure to shellfish harvesting if warranted, and coordinate with DNR Natural Resource Police.	Commonwealth of Kentucky: Department for Environmental Protection Division of Water (2015). Harmful Algal Blooms. http://water.ky.gov/waterquality/pages/HABS.aspx. Last Accessed: 11/27/2018.
Maryland	Presence of potentially toxic algae	Issue algae bloom beach alert.	Wazniak C personal communication. (2016). Regarding Maryland Department of Natural Resources Harmful Algal Bloom (HAB) Monitoring and Management SOP. Sent via email correspondence from Catherine Wazniak, Program Manager at the MD DNR, on February 22, 2016, to John Ravenscroft, U.S. EPA. Maryland Department of Natural Resources (2014). Harmful Algal Bloom Management in the Chesapeake and Coastal Bays. http://dnr.maryland.gov/waters/bay/Documents/HAB_Management.pdf. Last Accessed: 11/27/2018.
	Blue-green algae: > 50,000 cells/ml	Toxin testing of lysed cells should be done to ensure that guideline of 14 ppb is not exceeded.	

State	Recreational Water Guideline Level	Recommended Action	Reference
Massachusetts	Blue-green algae: > 70,000 cells/ml	Post an advisory against contact with the water.	Massachusetts Bureau of Environmental Health (2015). MDPH Guidelines for Cyanobacteria in Freshwater Recreational Water Bodies in Massachusetts. Boston, Massachusetts. http://www.mass.gov/eohhs/docs/dph/environmental/exposure/protocol-cyanobacteria.pdf. Last Accessed: 11/27/2018. Massachusetts Department of Public Health (2008). MDPH guidelines for cyanobacteria in freshwater recreational water bodies in Massachusetts. http://www.mass.gov/eohhs/docs/dph/environmental/exposure/protocol-cyanobacteria.pdf. Last Accessed: 11/27/2018.
	Microcystins: ≥ 14 µg/L	Post an advisory against contact with the water.	
	Visible cyanobacteria scum or mat is evident	MDPH recommends an immediate posting by the local health department, state agency, or relevant authority to advise against contact with the water body.	
	Microcystin: ≥20 micrograms per liter (µg/L)	Not reported.	
Michigan	Other algal toxins are at or above appropriate guidelines that have been reviewed by MDEQ-WRD	Not reported. Post advisory.	Michigan Department of Environmental Quality (2018). Algae (Harmful Algal Blooms) website http://www.michigan.gov/deq/0,4561,7-135-3313_3681_3686_3728-383630--,00.html. Last Accessed: 11/27/2018. Kohlhepp (2015) Harmful Algal Bloom Monitoring and Assessment in Michigan Waters. Michigan Department of Environmental Quality Water Resources Division. MI/DEQ/WRD-15/013. http://www.michigan.gov/documents/deq/wrd-swas-algae-HABsummary_551207_7.pdf. Last Accessed: 03/6/2018.
	Chlorophyll *a*: >30 µg/L and visible surface accumulations/scum are present, or cells are visible throughout the water column		
	Microcystins (as total including –LR and other detectable congeners): 3 µg/L		
New Jersey	Cylindrospermopsin: 8 µg/L	Post advisory.	

State	Recreational Water Guideline Level	Recommended Action	Reference
	Anatoxin-a: 27 µg/L	Post advisory.	New Jersey Department of Environmental Protection (2017). Cyanobacterial Harmful Algal Bloom (HAB) Freshwater Recreational Response Strategy. http://www.state.nj.us/dep/wms/bfbm/NJHABResponseStrategy.pdf. Last Accessed: 11/27/2018
	Cyanobacterial cell count: ≥ 20,000 cells/ml	Post advisory.	
	Visual indication of a bloom – receipt of a bloom report or digital photograph	Suspicious Bloom: DEC HABs Program staff determine if a bloom is Suspicious and whether collection of a sample is feasible or warranted.	
New York	Blue-green chlorophyll levels: ≥ 25 µg/L; OR Microscopic confirmation that majority of sample is cyanobacteria and present in bloom-like densities; OR only in absence of the previous criteria being met: microcystin ≥ 4 µg/L but less than 20 µg/L and accompanied by ancillary evidence of the presence or recent history of a bloom	Confirmed Bloom: Signs have been developed by NY State Department of Health for use at regulated swimming beaches when Local Health Department personnel or beach operators close beaches. Online summer notification provides weekly update on the number of HABs locations in New York is included in MakingWaves, the DEC email subscription.	New York State Department of Environmental Conservation (2017). Harmful Algal Blooms (HABs) Program Guide. http://www.dec.ny.gov/docs/water_pdf/habsprogramguide.pdf. Last Accessed: 11/27/2018.
	Microcystin ≥ 20 µg/L (shoreline samples only); OR microcystin ≥ 10 µg/L (open water samples only); OR known risk of exposure to anatoxin or another cyanotoxin, based on consult between DEC HABS Program and NYSDOH staff	Confirmed with High Toxins Bloom: Signs have been developed by NY State Department of Health for use at regulated swimming beaches when Local Health Department personnel or beach operators close beaches. Online summer notification provides weekly update on the number of HABs locations in New York is included in MakingWaves, the DEC email subscription.	
	Blue-green algae bloom is present AND microcystin-LR: < 10 µg/L	Issue advisory.	
North Dakota	Blue-green algae bloom is present over a significant portion of the lake AND microcystin-LR: ≥ 10 µg/L	Issue warning.	North Dakota Department of Health: Division of Water Quality (2016). Blue-green algae advisories and warnings.
	Cyanobacteria: 100,000 cell/ml	Issue advisory.	

State	Recreational Water Guideline Level	Recommended Action	Reference
			http://www.ndhealth.gov/wq/sw/habs/defaulthabs.htm. Last Accessed: 11/27/2018.
Oklahoma	Microcystin: > 20 µg/L	Issue advisory.	Oklahoma Legislature (2012). SB 259 Bill Summary. http://webserver1.lsb.state.ok.us/CF/2011-12%20SUPPORT%20DOCUMENTS/BILLSUM/House/SB259%20ccr%20a%20billsum.doc. Last Accessed: 11/27/2018.
	Anatoxin-a: ≥ 20 µg/L	Issue public health advisory.	
Oregon	Cylindrospermopsin: ≥ 20 µg/L	Issue public health advisory.	Oregon Health Authority (2018). Oregon Harmful Algae Bloom Surveillance (HABS) Program Public Health Advisory Guidelines: Harmful Algae Blooms in Freshwater Bodies. https://www.oregon.gov/oha/ph/HealthyEnvironments/Recreation/HarmfulAlgaeBlooms/Documents/HABPublicHealthAdvisoryGuidelines.pdf. Last Accessed: 11/27/2018.
	Microcystin: ≥ 10 µg/L	Issue public health advisory.	
	Microcystis: > 40,000 cells/ml	Issue public health advisory.	
	Planktothrix: > 40,000 cells/ml	Issue public health advisory.	
	Saxitoxin: ≥ 10 µg/L	Issue public health advisory.	
	Toxigenic species: > 100,000 cells/ml	Issue public health advisory.	
	Visible scum with documentation and testing	Issue public health advisory.	
	Microcystin: > 6 µg/L	Recreational Public Health Advisory.	
Pennsylvania	Microcystin: > 20 µg/L	Recreational No Contact Advisory.	Pennsylvania Department of Environmental Protection (2014). Lake Erie Harmful Algal Bloom Monitoring and Response Strategy for Recreational Waters. https://seagrant.psu.edu/sites/default/files/PA%20Lake%20Erie%20Harmful%20Algal%20Bloom%20Response%20Strategy%20For%20Recreational%20Waters%20-%202nd%20Draft.pdf. Last Accessed: 11/27/2018.
	Cylindrospermopsin: > 5 µg/L	Recreational Public Health Advisory.	
	Cylindrospermopsin: > 20 µg/L	Recreational No Contact Advisory.	
	Anatoxin-a: > 80 µg/L	Recreational Public Health Advisory.	
	Anatoxin-a: > 300 µg/L	Recreational No Contact Advisory.	
	Saxitoxin: > 0.8 µg/L	Recreational Public Health Advisory.	
	Saxitoxin: > 3 µg/L	Recreational No Contact Advisory.	

State	Recreational Water Guideline Level	Recommended Action	Reference
	HAB verified by visual observation	Recreational no contact advisory.	
	Cyanobacteria: > 70,000 cells/ml	Issue health advisory.	
Rhode Island	Microcystin-LR: ≥ 14 µg/L	Issue health advisory.	Rhode Island Department of Environmental Management, and Rhode Island Department of Health (2013). Cyanobacteria Related Public Health Advisories in Rhode Island. http://www.health.ri.gov/publications/datareports/2013CyanobacteriaBloomsInRhodeIsland.pdf. Last Accessed: 11/27/2018.
	Visible cyanobacteria scum or mat	Issue health advisory.	
	Anatoxin-a: detection 90 µg/L	Tier 2: Warning: Issue WARNING advisory, Post WARNING signs, sampling recommended weekly.	
Utah	Anatoxin-a: > 90 µg/L	Tier 3: Danger: Issue DANGER advisory, Post DANGER signs, consider CLOSURE, sampling recommended at least weekly.	Utah Department of Environmental Quality and Department of Health (2017). Utah HAB Guidance Summary. http://health.utah.gov/enviroepi/appletree/HAB/HAB_Guidance_Summary_2017.pdf . Last Accessed: 11/27/2018.
	Cyanobacteria: 20,000 – 10,000,000 cells/ml	Tier 2: Warning: Issue WARNING advisory, Post WARNING signs, sampling recommended weekly.	
	Cyanobacteria: >10,000,000 cells/ml	Tier 3: Danger: Issue DANGER advisory, Post DANGER signs, consider CLOSURE, sampling recommended at least weekly.	
	Microcystin: 4 – 2,000 µg/L	Tier 2: Warning: Issue WARNING advisory, Post WARNING signs, sampling recommended weekly.	
	Microcystin: > 2,000 µg/L	Tier 3: Danger: Issue DANGER advisory, Post DANGER signs, consider CLOSURE, sampling recommended at least weekly.	
	Cylindrospermopsin: > 8 µg/L	Tier 2 or 3: Consult with Utah Department of Environmental Quality and Utah Department of Health as needed on this issue.	
	Reports of animal illnesses or death	Tier 2: Warning: Issue WARNING advisory, Post WARNING signs, sampling recommended weekly.	

State	Recreational Water Guideline Level	Recommended Action	Reference
	Reports of human illness	Tier 3: Danger: Issue DANGER advisory, Post DANGER signs, consider CLOSURE, sampling recommended at least weekly.	
	Anatoxin-a: ≥ 10 µg/L	Close recreational beaches.	
Vermont	Cylindrospermopsin: ≥ 10 µg/L	Close recreational beaches.	Vermont Department of Health (2015). Cyanobacteria (Blue-green Algae) Guidance for Vermont Communities. http://www.healthvermont.gov/sites/default/files/documents/2016/12/ENV_RW_CyanobacteriaGuidance.pdf. Last Accessed: 11/27/2018.
	Microcystin-LR (equivalents): ≥ 6 µg/L	Close recreational beaches.	
	Visible known blue-green algae bloom/scum or an unknown, potentially blue-green algae (i.e., not pollen), bloom/scum	Close recreational beaches.	
	Blue-green algal "scum" or "mats" on water surface	Immediate public notification to avoid all recreational water contact where bloom is present; continue weekly sampling.	
Virginia	Microcystin: > 6 µg/L	Immediate public notification to avoid all recreational water contact where bloom is present; continue weekly sampling.	Virginia Department of Health (Division of Environmental Epidemiology) (2012). Virginia Recreational Water Guidance for Microcystin and *Microcystis* Blooms: Provisional Guidance. http://www.vdh.virginia.gov/content/uploads/sites/12/2016/02/VDHMicrocystisGuidance.pdf. Last Accessed: 11/27/2018.
	Microcystis: > 100,000 cells /ml	Immediate public notification to avoid all recreational water contact where bloom is present; continue weekly sampling.	
	Microcystis: 20,000 to 100,000 cells/ml	Notify public through press release and/or signage; advise people and pet owners that harmful algae are present; initiate weekly water sampling.	
	Microcystis: 5,000 to < 20,000 cells/ml	Local agency notification; initiate bi-weekly water sampling.	
	Anatoxin-a: 1 µg/L	Tier 2: local health posts WARNING sign; local health takes additional site-specific steps; minimum weekly sampling. In addition, if history of high toxicity, or reports of illness, pet death than tier 3: local health posts DANGER sign; lake closed.	

State	Recreational Water Guideline Level	Recommended Action	Reference
Washington	Bloom is forming or a bloom scum is visible (toxic algae may be present); toxin levels do not exceed thresholds	Tier 1: local health posts CAUTION sign; samples taken and sent for toxicity tests; weekly sampling until bloom dissipates.	Hardy J, and Washington State Department of Health (2008). Washington State Recreational Guidance for Microcystins (Provisional) and Anatoxin-a (Interim/Provisional). http://www.doh.wa.gov/Portals/1/Documents/4400/334-177-recguide.pdf. Last Accessed: 11/27/2018.
	Cylindrospermopsin: 4.5 µg/L	Tier 2: local health posts WARNING sign; local health takes additional site-specific steps; minimum weekly sampling. In addition, if history of high toxicity, or reports of illness, pet death than tier 3: local health posts DANGER sign; lake closed.	Hardy J, and Washington State Department of Health (2011). Washington State Provisional Recreational Guidance for Cylindrospermopsin and Saxitoxin. http://www.doh.wa.gov/portals/1/documents/4400/332-118-cylindrosax%20report.pdf. Last Accessed: 11/27/2018.
	Microcystins: 6 µg/L	Tier 2: local health posts WARNING sign; local health takes additional site-specific steps; minimum weekly sampling. In addition, if history of high toxicity, or reports of illness, pet death than tier 3: local health posts DANGER sign; lake closed.	
	Saxitoxin: 75 µg/L	Tier 2: local health posts WARNING sign; local health takes additional site-specific steps; minimum weekly sampling. In addition, if history of high toxicity, or reports of illness, pet death than tier 3: local health posts DANGER sign; lake closed.	Hardy J, and Washington State Department of Health (2008). Washington State Recreational Guidance for Microcystins (Provisional) and Anatoxin-a (Interim/Provisional). http://www.doh.wa.gov/Portals/1/Documents/4400/334-177-recguide.pdf. Last Accessed: 11/27/2018.
	Saxitoxin: 75 µg/L	Tier 2: local health posts WARNING sign; local health takes additional site-specific steps; minimum weekly sampling. In addition, if history of high	Hardy J, and Washington State Department of Health (2011). Washington State Provisional Recreational Guidance for Cylindrospermopsin and Saxitoxin.

State	Recreational Water Guideline Level	Recommended Action	Reference
		toxicity, or reports of illness, pet death than tier 3: local health posts DANGER sign; lake closed.	http://www.doh.wa.gov/portals/1/documents/4400/332-118-cylindrosax%20report.pdf. Last Accessed: 11/27/2018.
States with Qualitative Guidelines Only			
Delaware	Thick green, white, or red scum on surface of pond	Post water advisory signs.	Delaware Department of Natural Resources and Environmental Control: Division of Water. Blue-Green Algae in Delaware. (2016). http://www.dnrec.delaware.gov/wr/INFORMATION/OTHERINFO/Pages/Blue-GreenAlgae.aspx. Last Accessed: 11/27/2018.
Florida	Cyanobacteria bloom	Issue health advisory; post warning signs.	Florida Department of Environmental Protection (2019). Freshwater Algal Blooms: Frequently Asked Questions. https://floridadep.gov/sites/default/files/freshwater-algal-bloom-faqs_2019.pdf Last Accessed: 5/10/2019.
Missouri	Microcystins: presence (test strip range 0 to 10 ng/ml) Cylindrospermopsin: presence (test strip range 0 to 10 ng/ml) Anatoxin-a: presence (test strip range 0 to 2.5 ng/ml)	Missouri has a multi-agency proactive approach to address events which can result in the decision to temporary close swim beaches and post notices regarding the bloom around the lake to protect the citizens of Missouri from the health risk posed by exposure to a HAB. Information is also released to through the news media and social media to quickly share the possible health risk with the largest audience possible.	Missouri Department of Natural Resources (2017) Qualitative screening of algal toxins in drinking water and recreational waters using strip test by Abraxas, Inc. https://dnr.mo.gov/env/docs/mdnresp360.pdf. Last Accessed: 11/27/2018. Missouri Department of Natural Resources (2018) Harmful Algal Blooms and Blue-Green Algae. Website https://dnr.mo.gov/env/cyanobacteria.htm. and http://ephtn.dhss.mo.gov/EPHTN_Data_Portal/pdf/success-stories/MO-Blue-Green-

State	Recreational Water Guideline Level	Recommended Action	Reference
			Algae-Task-Force-Establishment.pdf Last Accessed: 11/27/2018.
Montana	Reservoirs that seem stagnated and harbor large quantities of algae	The Montana Department of Environmental Quality advises people to avoid swimming in ponds, lakes, or reservoirs.	State of Montana Newsroom (2015). DEQ Issues Advisory on Blue-Green Algae Blooms: Ponds, Lakes, and Reservoirs Most Often Affected. http://news.mt.gov/Home/ArtMID/24469/ArticleID/1564/DEQ-Issues-Advisory-on-Blue-Green-Algae-Blooms. Last Accessed: 11/27/2018.
North Carolina	Visible discoloration or surface scum	Microcystin testing.	North Carolina Health and Human Services: Division of Public Health (2014). Occupational and Environmental Epidemiology: Cyanobacteria (Blue-green Algae). http://epi.publichealth.nc.gov/oee/a_z/algae.html. Last Accessed: 11/27/2018.
West Virginia	Blue-green algal blooms observed and monitored	Issue public health advisory.	West Virginia Department of Health and Human Resources (2015). DHHR Continuing to Monitor Blue-Green Algal Blooms on the Ohio River: Residents Advised to Adhere to Public Health Advisory. http://www.dhhr.wv.gov/News/2015/Pages/DHHR-Continuing-to-Monitor-Blue-Green-Algal-Blooms-on-the-Ohio-River%3B-Residents-Advised-to-Adhere-to-Public-Health-Advisory.aspx. Last Accessed: 11/27/2018.
States with Guidelines Under Development			
Arkansas	TBD	TBD	Arkansas Beautiful Buffalo River Action Committee (2018). https://bbrac.arkansas.gov/pdfs/201701205

State	Recreational Water Guideline Level	Recommended Action	Reference
			-arkansas-harmful-algal-bloom-(habs)-workgroup.pdf. Last Accessed: 11/27/2018.
Georgia	TBD	TBD	Georgia Department of Public Health (2018). https://www.gachd.org/programs-services/environmental-health-2/beach_water_testing/. Last Accessed: 03/6/2018.
Minnesota	TBD	TBD	Minnesota Department of Health (2015). Toxicological Summary for: Microcystin-LR. http://www.health.state.mn.us/divs/eh/risk/guidance/gw/microcystin.pdf. Last Accessed: 11/27/2018.
Wyoming	TBD	TBD	Wyoming Department of Environmental Quality (2018). Harmful Algal Bloom Website. http://deq.wyoming.gov/wqd/nutrient-pollution/resources/harmful-algal-blooms/. Last Accessed: 11/27/2018.

Note: Alabama, Alaska, Hawaii, Louisiana, Mississippi, Nevada, New Mexico, South Carolina, South Dakota, Tennessee, and Texas did not have guidelines available online. Missouri is in the process of developing quantitative thresholds.

APPENDIX C. LITERATURE SEARCH DOCUMENTATION

The recreational ambient water quality criteria (AWQC) or swimming advisories document for microcystins and cylindrospermopsin relied significantly on information identified, reviewed, and synthesized in the EPA's *Health Effects Support Document for the Cyanobacterial Toxin Microcystins*, *Heath Effects Support Document for the Cyanobacterial Toxin Cylindrospermopsin*, *Drinking Water Health Advisory for the Cyanobacterial Microcystin Toxins*, and *Drinking Water Health Advisory for the Cyanobacterial Toxin Cylindrospermopsin* (U.S. EPA 2015c, 2015d, 2015a, 2015b). The EPA conducted supplemental literature searches to answer additional questions related to recreational exposures, exposure factors, and to identify new health data.

For the Health Effects Support Documents (HESDs), the EPA conducted a comprehensive literature search from January 2013 to May 2014 using Toxicology Literature Online (TOXLINE), PubMed, and Google Scholar. The EPA assembled available information on occurrence; environmental fate; mechanisms of toxicity; acute, short-term, subchronic, and chronic toxicity and cancer in humans and animals; and toxicokinetics and exposure. For a detailed description of the literature review search and strategy, see the HESDs for microcystins and cylindrospermopsin (U.S. EPA 2015c, 2015d).

The EPA conducted supplemental literature searches in September 2015 to capture references published since the completion of the HESDs' literature searches and to account for the recreational exposure scenario. The specific questions investigated include:

1. What levels of anatoxin-a, cylindrospermopsin, or microcystins are humans—of all ages, including children—exposed to through recreational use (activities) in freshwaters or marine waters from incidental ingestion, inhalation, and dermal exposure routes?
2 What health effects information for humans or animals exposed to cylindrospermopsin or microcystins (through ingestion, inhalation, and dermal exposure routes) has been published since the health effects literature searches were conducted for the EPA's 2015 HESDs for cylindrospermopsin and microcystins?
3. What recreational water use safety levels or criteria have been set for microcystins or cylindrospermopsin by states or international governments, and how did they derive them?
4. What new information, if any, is available regarding how aquatic recreational exposure ingestion rates in children differ among age groups between zero and 18 years?
5. What incidents of companion animal (e.g., dogs, horses) or livestock poisonings, including mortality or adverse health effects, due to exposure to cyanotoxins in freshwaters, marine waters, or beaches have occurred in the past 15 years? Specifically, when and where did these incidents occur, to which cyanotoxin were the animals exposed, how were they exposed, and what were the weights and breeds of the affected animal(s)?

The EPA implemented a unique literature search strategy to address each research question. Trial searches were conducted, and results were evaluated to refine the search strategies (e.g., to reduce retrieval of citations unrelated to the research questions). The search strings were refined to improve the relevancy of the results. The literature search strategies implemented for each research question are subsequently detailed.

Research Question 1: What levels of anatoxin-a, cylindrospermopsin, or microcystins are humans—of all ages, including children—exposed to through recreational use (activities) in freshwaters or marine waters, from incidental ingestion, inhalation, and dermal exposure routes?

The EPA searched the bibliographic databases, PubMed and Web of Science (WoS), to identify candidate journal article literature relevant to human exposure to anatoxin-a, cylindrospermopsin, or microcystins through recreational activities. PubMed and WoS contain peer-reviewed journal abstracts and articles on various biological, medical, public health, and chemical topics. The WoS search string differs slightly from the PubMed search string due to how the search engines treat search terms with more than one word. Both search strings are presented below.

Results

The searches returned 321 journal articles after removing duplicates between PubMed and WoS results. Based on a screening review of each article's title and abstract, the EPA retrieved nine articles that appeared to be studies that measured, reviewed, or estimated human recreational exposure to cyanotoxins.

PubMed Search:

("*A. lemmermannii Raphidiopsis mediterranea*" OR *Anabaena flos-aquae* OR *flos-aquae* OR anatoxin-a OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia*)

AND

("boogie board" OR "boogie boarding" OR "jet ski" OR "jet skier" OR "jet skiers" OR "jet skiing" OR "water ski" OR "water skier" OR "water skiers" OR "water skiing" OR aerosol OR boat OR boating OR boats OR bodyboard OR bodyboarding OR canoe OR canoeing OR canoes OR capsize OR capsized OR dermal OR inhalation OR inhale OR kayak OR kayaker OR kayaking OR kayaks OR kneeboard OR kneeboarding OR paddle OR paddling OR raft OR rafting OR rafts OR recreation OR recreational OR rowing OR skin OR surf OR surfer OR surfing OR swim OR swimmer OR swimmers OR swimming OR tubing OR wading OR wakeboarding OR wakeboard)

AND

("marine water" OR "surface water" OR beach OR beaches OR estuaries OR estuarine OR estuary OR "fresh water" OR freshwater OR lake OR lakes OR ocean OR oceans OR pond OR ponds OR reservoir OR reservoirs OR river OR rivers OR sea OR stream OR streams OR water)

Filters: English
Date search was conducted: 10/9/2015
Publication dates searched: 1/1/1995 – 10/9/2015

Web of Science Search:

("*lemmermannii Raphidiopsis mediterranea*" OR *Anabaena flos-aquae* OR *flos-aquae* OR anatoxin OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*"

OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia*)

AND

("boogie board" OR "boogie boarding" OR "jet ski" OR "jet skier" OR "jet skiers" OR "jet skiing" OR "water ski" OR "water skier" OR "water skiers" OR "water skiing" OR aerosol OR boat OR boating OR boats OR bodyboard OR bodyboarding OR canoe OR canoeing OR canoes OR capsize OR capsized OR dermal OR inhalation OR inhale OR kayak OR kayaker OR kayaking OR kayaks OR kneeboard OR kneeboarding OR paddle OR paddling OR raft OR rafting OR rafts OR recreation OR recreational OR rowing OR skin OR surf OR surfer OR surfing OR swim OR swimmer OR swimmers OR swimming OR tubing OR wading OR wakeboarding OR wakeboard)

AND

("marine water" OR "surface water" OR beach OR Beaches OR estuaries OR estuarine OR estuary OR "fresh water" OR freshwater OR lake OR lakes OR ocean OR oceans OR pond OR ponds OR reservoir OR reservoirs OR river OR rivers OR sea OR stream OR streams OR water)

Filters: English
Date search was conducted: 10/9/2015
Publication dates searched: 1/1/1995–10/9/2015

C.1 Research Question 2: What health effects information for humans or animals exposed to microcystins, cylindrospermopsin, or anatoxin-a (through ingestion, inhalation, and dermal exposure routes) has been published since the health effects literature searches were conducted for the EPA's 2015 HESDs for Cylindrospermopsin and Microcystins?

The EPA searched PubMed and WoS to identify candidate journal article literature relevant to health effects associated with exposure to anatoxin-a, cylindrospermopsin, or microcystins. The WoS search string differs slightly from the PubMed search string due to how the search engines treat search terms with more than one word. Both search strings are presented below.

Results

The searches returned 1,000 journal articles after removing duplicates between PubMed and WoS results. Based on a screening review of each article's title and abstract, the EPA retrieved 40 articles that appeared to be prospective human epidemiological studies (n = 1), ecological human epidemiologic studies (n = 2), reviews of human health effects (n = 4), in vivo animal studies (n = 30), or reviews of in vivo animal studies (n = 3).

PubMed Search:

("*A. lemmermannii Raphidiopsis mediterranea*" OR *Anabaena flos-aquae* OR *flos-aquae* OR anatoxin-a OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia*)

AND

("non cancer" OR "blurred vision" OR "cell damage" OR "cellular damage" OR "health effect" OR "health endpoint" OR "health outcome" OR "health risk" OR "loss of protein" OR "loss of water" OR "micronucleated binucleate cell" OR abdominal pain OR ache OR acute OR alanine aminotransferase OR allergic OR allergies OR allergy OR aspartate aminotransferase OR blister OR blistered OR blisters OR cancer OR carcinogen OR carcinogenic OR carcinogens OR chronic OR clinical OR cough OR dermal OR detoxification OR detoxify OR develop OR development OR developmental OR dialysis OR diarrhea OR disease OR DNA OR dyspnea OR electrolyte OR emergency room OR enzyme OR enzymes OR epidemiologic OR epidemiological OR epidemiology OR epilepsy OR epileptic OR epithelium OR eye OR failure OR fever OR gastrointestinal OR genetox OR genotoxic OR glutamyltransferase OR head OR hematologic OR hematological OR hepatic OR histopathologic OR histopathological OR histpathology OR hospital OR hospitalizations OR hospitals OR hospitalization OR ill OR illness OR illnesses OR intoxicate OR intoxicated OR irritate OR irritated OR kidney OR larynx OR lesion OR lesions OR liver OR lung OR lymph OR lymph nodes OR lymphatic OR metabolic OR metabolism OR mucosa OR mutate OR mutated OR mutation OR mutations OR nausea OR necrosis OR neonatal OR neonate OR neonates OR neoplasm OR neurologic OR neurological OR noncancer OR oral OR organ OR pain OR placenta OR pneumonia OR polymorphism OR polymorphisms OR prenatal OR red blood cell OR renal OR reproduction OR respiratory OR seizure OR sick OR sickness OR skin OR stomach OR subacute OR subchronic OR symptom OR symptoms OR teratogen OR teratogenic OR teratogens OR throat OR toxic OR toxicity OR trachea OR tumor OR tumors OR urinary OR urine OR vomit OR vomiting OR conjugate OR conjugated OR diagnose OR diagnosis OR diagnosed OR diagnoses)

Filters: English
Date search was conducted: 10/9/2015
Publication dates searched: 1/1/2014–10/9/2015

Web of Science Search:

("*lemmermannii Raphidiopsis mediterranea*" OR *Anabaena flos-aquae* OR *flos-aquae* OR anatoxin OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia*)

AND

("non cancer" OR "blurred vision" OR "cell damage" OR "cellular damage" OR "health effect" OR "health endpoint" OR "health outcome" OR "health risk" OR "micronucleated binucleate cell" OR abdominal pain OR ache OR acute OR alanine aminotransferase OR allergic OR allergies OR allergy OR aspartate aminotransferase OR blister OR blistered OR blisters OR cancer OR carcinogen OR carcinogenic OR carcinogens OR chronic OR clinical OR cough OR dermal OR detoxification OR detoxify OR develop OR development OR developmental OR dialysis OR diarrhea OR disease OR DNA OR dyspnea OR electrolyte OR emergency room OR enzyme OR enzymes OR epidemiologic OR epidemiological OR epidemiology OR epilepsy OR epileptic OR epithelium OR eye OR failure OR fever OR gastrointestinal OR genetox OR genotoxic OR glutamyltransferase OR head OR hematologic OR hematological OR hepatic OR histopathologic OR histopathological OR histpathology OR hospital OR hospitalizations OR hospitals OR hospitalization OR ill OR illness OR illnesses OR intoxicate OR intoxicated OR irritate OR irritated OR kidney OR larynx OR lesion OR lesions OR liver OR lung OR lymph OR lymph nodes OR lymphatic OR metabolic OR metabolism OR mucosa OR mutate OR mutated OR mutation OR mutations OR nausea OR necrosis OR neonatal OR neonate OR neonates OR

neoplasm OR neurologic OR neurological OR noncancer OR oral OR organ OR pain OR placenta OR pneumonia OR polymorphism OR polymorphisms OR prenatal OR red blood cell OR renal OR reproduction OR respiratory OR seizure OR sick OR sickness OR skin OR stomach OR subacute OR subchronic OR symptom OR symptoms OR teratogen OR teratogenic OR teratogens OR throat OR toxic OR toxicity OR trachea OR tumor OR tumors OR urinary OR urine OR vomit OR vomiting OR conjugate OR conjugated OR diagnose OR diagnosis OR diagnosed OR diagnoses)

Filters: English
Date search was conducted: 10/9/2015
Publication dates searched: 1/1/2014–10/9/2015

WoS research areas searched: Environmental Sciences Ecology OR Marine Freshwater Biology OR Toxicology OR Pharmacology Pharmacy OR Public Environmental Occupational Health OR Microbiology OR Immunology OR Biotechnology Applied Microbiology OR Biochemistry Molecular Biology OR Research Experimental Medicine OR Water Resources OR Infectious Disease OR Science Technology Other Topics OR Life Sciences Biomedicine Other Topics OR Gastroenterology Hepatology OR Pediatrics.

C.2 Research Question 3: What recreational water use safety levels or criteria have been set for microcystins or cylindrospermopsin by states or international governments and how did they derive them?

To identify state-level recreational guidelines for cyanobacteria and cyanotoxins, the EPA searched the websites of state-level departments of public health, environmental health, and natural resources for all 50 U.S. states. If relevant recreational guidelines were not found by searching state-level websites, the EPA conducted Google searches of the internet using state names, key terms for cyanobacteria and cyanotoxins (e.g., harmful algal bloom, blue-green algae, microcystin, cylindrospermopsin), and key terms for guidelines (e.g., advisory, guidance, guideline, standard, regulation). For international governments, the EPA used the 2012 report, *Current Approaches to Cyanotoxin Risk Assessment, Risk Management and Regulations in Different Countries*, by Dr. Ingrid Chorus, Federal Environment Agency, Germany, to identify international government recreational safety levels for cyanobacteria and cyanotoxins. In addition, the EPA implemented the same search strategy as used for U.S. states to identify updated international recreational guidelines or guideline levels not featured in the 2012 report by Dr. Ingrid Chorus.

C.3 Research Question 4: What new information, if any, is available regarding how aquatic recreational exposure ingestion rates in children differ among age groups between zero and 18 years?

Search of Bibliographic Databases

The EPA searched PubMed, WoS, and Google Scholar to identify literature that has cited, or is similar (based on terms identified in the titles and abstracts) to, the studies that provide water ingestion data for swimmers or during water recreational activities in the EPA's (2011) *Exposure Factors Handbook* (EFH) (i.e., Dorevitch et al. 2011; Dufour et al. (2006); Schets et al. 2011). The PubMed and WoS searches were conducted on 10/9/2015, the publication dates searched were 1/1/2011 to 10/9/2015, and an English filter was applied. The Google Scholar search was conducted on 10/9/2015 and could not be limited by year or language.

Results

Together all three searches returned 341 journal articles. Duplicates were removed between PubMed and WoS, but this total might include duplicates between Google Scholar results and WoS/PubMed results. Based on a screening review of each article's title and abstract, the EPA retrieved five articles, four of which were published between 2013 and 2015 and appeared to measure or estimate incidental water ingestion. The EPA also retrieved one 2012 study that assessed duration of non-swimming recreational water exposure by using novel time lapse photography technology.

Google Search of Internet

In addition, the EPA conducted a Google search of the internet focused on specified URL domains (listed in Table C-1) to identify candidate gray literature (e.g., state, federal, or international government reports or guidance). The Google search string is presented below. The Google search of the internet could not be limited by year or language.

Table C-1. Internet URL Domains Searched for Research Question 4

Organization	URL Domain
U.S. Government	.gov .us
All U.S. States	Google Custom Search Engine
Centers for Disease Control and Prevention, including Agency for Toxic Substances and Disease Registry	cdc.gov
Australia, including Australian Department of Health	gov.au
Canada, including Health Canada	gc.ca
European Union, including • European Chemicals Agency • European Commissions on Environment, Public Health, Food, and Health and Consumers	europa.eu
Public Health England	hpa.org.uk
United Kingdom	gov.uk
Germany	.de
Education websites	.edu
HERA (Human and Environmental Risk Assessment) Project	heraproject.com
World Health Organization	who.int

Results

The Google search returned 390 results after removing duplicates. Based on a preliminary screen of each result, the EPA retrieved two documents which appeared to either derive or cited an incidental ingestion rate while recreating which had not previously been identified during the literature search process.

Google Search of Internet (conducted separately for each URL domain listed in Table C-1)

(pool OR swim OR swimmer OR swimmers OR swimming OR recreation OR recreational)

AND

(adolescents OR boys OR child OR children OR girls OR kids OR teenagers)

AND

("activity-related ingestion" OR "incidental ingestion" OR "activity-related ingestion" OR "ingestion of water" OR "water ingestion")

AND

rate

AND

inurl:.

Filters: None
Date search was conducted: 10/9/2015
Dates searched: Not specified
Web browser: Internet Explorer

C.4 Research Question 5: What incidents of companion animal (e.g., dogs, horses) or livestock poisonings, including mortality or adverse health effects, due to exposure to cyanotoxins in freshwaters, marine water, or beaches have occurred in the past 15 years? Specifically, when and where did these incidents occur, to which cyanotoxin were the animals exposed, how were they exposed, and what were the weights and breeds of the affected animal(s)?

The EPA searched PubMed, WoS, and Agricola to identify candidate journal article literature relevant to companion animal or livestock poisoning due to exposures to cyanobacterial cells, anatoxin-a, cylindrospermopsin, or microcystins. The EPA first searched PubMed and WoS with a focus on dogs. The EPA conducted two additional searches in PubMed, WoS, and Agricola focused on livestock, and on cats and birds. The search strings for each search iteration are presented below.

Results

The number of journal articles returned by the three searches is provided in Table C-2. Based on a screening review of the article's title and abstract, the EPA retrieved five of the 35 journal articles retrieved during the search focused on dogs. These five articles appeared to provide information about an incident of cyanotoxin exposure to an animal where the authors confirm that the animal was exposed to a cyanotoxin by either measuring the concentration of cyanotoxin found in the animal or by sampling the body of water to which the animal had contact.

Table C-2. Number of Journal Articles Returned by Three Search Strategies for Research Question 5

Search Strategy Focus	Number of Results Returned from PubMed, WoS, and Agricola Searches
Dogs	35[a]
Livestock	100
Cats and birds	169[b]

[a] Search conducted in PubMed and WoS only.

[b] Duplicates between PubMed/WoS results and Agricola results were not removed. Therefore, the cats and birds search might include duplicates between Agricola results and PubMed/WoS results.

C.4.1 Search Strategy Focused on Dogs

PubMed Search

("*A. lemmermannii Raphidiopsis mediterranea*" OR *flos-aquae* OR anatoxin-a OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia* OR Cyanobacteria OR cyanotoxin OR Cyanotoxins OR "harmful algae" OR "harmful algal bloom" OR blue green algae)

AND

("health effect" OR "health endpoint" OR "health outcome" OR dead OR death OR deaths OR died OR disease OR diseased OR diseases OR exposed OR exposure OR ill OR illness OR illnesses OR infect OR infected OR infection OR infections OR morbidity OR mortality OR poison OR poisoned OR poisoning OR poisonings OR sick OR sickness OR toxic OR toxicity OR diagnose OR diagnosis OR diagnosed OR diagnoses)

AND

(canine OR canines OR dog OR dogs OR "*Canis lupus familiaris*" OR "*Canis familiaris*")

Filters: English
Date search was conducted: 10/5/2015
Publication dates searched: 1/1/2012–10/5/2015

Web of Science Search

("*lemmermannii Raphidiopsis mediterranea*" OR *flos-aquae* OR anatoxin OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia* OR Cyanobacteria OR cyanotoxin OR Cyanotoxins OR "harmful algae" OR "harmful algal bloom" OR blue green algae)

AND

("health effect" OR "health endpoint" OR "health outcome" OR dead OR death OR deaths OR died OR disease OR diseased OR diseases OR exposed OR exposure OR ill OR illness OR illnesses OR infect OR infected OR infection OR infections OR morbidity OR mortality OR poison OR poisoned OR poisoning OR poisonings OR sick OR sickness OR toxic OR toxicity OR diagnose OR diagnosis OR diagnosed OR diagnoses)

AND

(canine OR canines OR dog OR dogs OR "*Canis lupus familiaris*" OR "*Canis familiaris*")

Filters: English
Date search was conducted: 10/5/2015
Publication dates searched: 1/1/2012–10/5/2015

C.4.2 Search Strategy Focused on Livestock

PubMed and Agricola Searches

("*A. lemmermannii Raphidiopsis mediterranea*" OR *flos-aquae* OR anatoxin-a OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborski*i" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatori*a OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia* OR *Cyanobacteria* OR *cyanotoxin* OR *Cyanotoxins* OR "harmful algae" OR "harmful algal bloom" OR blue green algae)

AND

("health effect" OR "health endpoint" OR "health outcome" OR dead OR death OR deaths OR died OR disease OR diseased OR diseases OR exposed OR exposure OR ill OR illness OR illnesses OR infect OR infected OR infection OR infections OR morbidity OR mortality OR poison OR poisoned OR poisoning OR poisonings OR sick OR sickness OR toxic OR toxicity OR diagnose OR diagnosis OR diagnosed OR diagnoses)

AND

(alpaca OR alpacas OR bronco OR broncos OR buffalo OR bull OR bulls OR cattle OR colt OR colts OR cow OR cows OR bovine OR bison OR oxen OR donkey OR donkeys OR duck OR ducks OR equine OR ewe OR ewes OR fillies OR filly OR foal OR foals OR gelding OR geldings OR heifer OR heifers OR horse OR horses OR lamb OR lambs OR livestock OR llama OR llamas OR mare OR mares OR mule OR mules OR mustang OR mustangs OR ponies OR pony OR ram OR rams OR sheep OR stallion OR stallions OR steer OR pig OR pigs OR piglet OR piglets)

Filters: English
Date search was conducted: 11/25/2015
Publication dates searched: 1/1/2012–11/25/2015

Web of Science Search:

("*lemmermannii Raphidiopsis mediterranea*" OR *flos-aquae* OR anatoxin OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR

Microcystis OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia* OR Cyanobacteria OR cyanotoxin OR Cyanotoxins OR "harmful algae" OR "harmful algal bloom" OR blue green algae)

AND

("health effect" OR "health endpoint" OR "health outcome" OR dead OR death OR deaths OR died OR disease OR diseased OR diseases OR exposed OR exposure OR ill OR illness OR illnesses OR infect OR infected OR infection OR infections OR morbidity OR mortality OR poison OR poisoned OR poisoning OR poisonings OR sick OR sickness OR toxic OR toxicity OR diagnose OR diagnosis OR diagnosed OR diagnoses)

AND

(alpaca OR alpacas OR bronco OR broncos OR buffalo OR bull OR bulls OR cattle OR colt OR colts OR cow OR cows OR bovine OR bison OR oxen OR donkey OR donkeys OR duck OR ducks OR equine OR ewe OR ewes OR fillies OR filly OR foal OR foals OR gelding OR geldings OR heifer OR heifers OR horse OR horses OR lamb OR lambs OR livestock OR llama OR llamas OR mare OR mares OR mule OR mules OR mustang OR mustangs OR ponies OR pony OR ram OR rams OR sheep OR stallion OR stallions OR steer OR pig OR pigs OR piglet OR piglets)

Filters: English
Date search was conducted: 11/25/2015
Publication dates searched: 1/1/2012–11/25/2015

C.4.3 Search Strategy Focused on Cats and Birds

PubMed and Agricola Searches

("*A. lemmermannii Raphidiopsis mediterranea*" OR *flos-aquae* OR anatoxin-a OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia* OR *Cyanobacteria* OR cyanotoxin OR Cyanotoxins OR "harmful algae" OR "harmful algal bloom" OR blue green algae)

AND

("health effect" OR "health endpoint" OR "health outcome" OR dead OR death OR deaths OR died OR disease OR diseased OR diseases OR exposed OR exposure OR ill OR illness OR illnesses OR infect OR infected OR infection OR infections OR morbidity OR mortality OR poison OR poisoned OR poisoning OR poisonings OR sick OR sickness OR toxic OR toxicity OR diagnose OR diagnosis OR diagnosed OR diagnoses)

AND

(feline OR felines OR cat OR cats OR kitten OR kittens OR "*F. Catus*" OR "*Felis Catus*" OR bird OR birds OR avian OR waterfowl)

Filters: English
Date search was conducted: 2/1/2016
Publication dates searched: 1/1/2012–2/1/2016

Web of Science Search

("*lemmermannii Raphidiopsis mediterranea*" OR *flos-aquae* OR anatoxin OR *Aphanizomenon* OR cylindrospermopsin OR "*C. raciborskii*" OR *Cuspidothrix* OR *Cylindrospermopsis* OR *Cylindrospermum* OR "*Cylindrospermopsis raciborskii*" OR *Dolichospermum* OR "*M. aeruginosa*" OR *Microcystis* OR microcystin OR microcystins OR *Oscillatoria* OR *Planktothrix* OR *Phormidium* OR *Tychonema* OR *Woronichinia* OR Cyanobacteria OR cyanotoxin OR Cyanotoxins OR "harmful algae" OR "harmful algal bloom" OR blue green algae)

AND

("health effect" OR "health endpoint" OR "health outcome" OR dead OR death OR deaths OR died OR disease OR diseased OR diseases OR exposed OR exposure OR ill OR illness OR illnesses OR infect OR infected OR infection OR infections OR morbidity OR mortality OR poison OR poisoned OR poisoning OR poisonings OR sick OR sickness OR toxic OR toxicity OR diagnose OR diagnosis OR diagnosed OR diagnoses)

AND

(feline OR felines OR cat OR cats OR kitten OR kittens OR "*F. Catus*" OR "*Felis Catus*" OR bird OR birds OR avian OR waterfowl)

Filters: English
Date search was conducted: 2/1/2016
Publication dates searched: 1/1/2012–2/1/2016

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APPENDIX D. REVIEW OF THE STATE OF THE SCIENCE ON CYANOBACTERIAL CELLS HEALTH EFFECTS

D.1 Introduction

This appendix provides information gathered and reviewed to determine the state of the science on health effects from cyanobacterial cells. The EPA conducted literature searches to identify studies relevant to the health effects from cyanobacterial cells. Detailed information on the design and implementation of these searchers is provided in Appendix C. Results from these literature searches were reviewed for relevance to cyanobacterial cell exposures and health effects.

D.1.1 Animal Studies

Cyanobacterial cells cause allergenicity and irritation in animals, independent of whether the cyanobacterial cells produce toxin. Three animal studies (Shirai et al. 1986; Stewart et al. 2006c; Torokne et al. 2001) demonstrated hypersensitivity reactions and dermal and eye irritation in several species that did not correlate with microcystin content. Although the number of studies is limited and different species were evaluated in each study, these studies provide evidence to support hypersensitivity reactions in animals from exposure to cyanobacteria when cyanotoxins are not present (Shirai et al. 1986; Torokne et al. 2001) and when they are (Stewart et al. 2006c).

Cyanobacteria bloom samples collected from five different lakes or ponds were tested for allergenic and irritative effects in guinea pigs and rabbits, respectively (Torokne et al. 2001). The microcystin content (presumed to be total LR, RR, and YR) ranged from not detected to 2.21 mg/g. To determine sensitization, guinea pigs were initiated with an intradermal injection of freeze-dried cyanobacteria followed seven days later by topical application at the injection site. Sensitization was moderate to strong in 30–67 percent of guinea pigs and did not correlate with microcystin content. The *Aphanizomenon ovalisporum* sample (a non-toxin-producing strain) sensitized 91 percent of the animals and was the strongest allergen. Skin irritation tests in albino rabbits showed slight or negligible irritation, except for *Aphanizomenon ovalisporum*, which showed moderate irritation. The eye irritation evaluation in rabbits was positive for four of the five samples containing *Microcystis*.

Shirai et al. (1986) reported that C3H/HeJ mice, immunized intraperitoneal with either sonicated or live cells from a *Microcystis* water bloom, developed delayed-type hypersensitivity when challenged two weeks later with a subcutaneous injection sonicated *Microcystis* cells. A positive reaction, as assessed by footpad swelling, was seen in mice immunized with either live cells or sonicated cells. Both toxic and nontoxic *Microcystis* cells induced delayed-type hypersensitivity in this mouse study. Because this strain of mouse is unresponsive to lipopolysaccharide (LPS), the footpad delayed-type hypersensitivity was not related to LPS, thus, the antigenic component of the sonicated cyanobacterial cells is not known.

Stewart et al. (2006c) conducted a mouse ear swelling test in which cylindrospermopsin and *Cylindrospermopsis raciborskii* solutions generated irritation of the abdominal skin exposed during induction (two percent w/v lysed cell solution containing 73 µg/mL cylindrospermopsin). Subsequent dermal exposures to the *Cylindrospermopsis raciborskii* solution produced hypersensitivity reactions ($p = 0.001$). The cyanobacteria *Microcystis aeruginosa* and *Anabaena circinalis* elicited no responses in this test.

Two of the cyanobacterial cell studies in animals found that rodents became sensitized after exposure and subsequent challenge to non-toxin strains (Shirai et al. 1986; Torokne et al. 2001). Torokne et al. (2001) found that a nontoxic strain was more sensitizing and irritating than the toxic strains evaluated. These experiments support the conclusion that there is no relationship between the cyanotoxin content and the allergenic effect of cyanobacteria.

D.1.2 Clinical and Laboratory Human Studies

Several types of studies and reports provide information on associations between cyanobacteria exposure and health effects. Clinical and in vitro studies (Bernstein et al. 2011; Geh et al. 2015; Pilotto et al. 2004; Stewart et al. 2006a) have been able to assess associations between cyanobacteria exposure and human health effects including dermal and allergenic reactions. Three clinical studies assessed dermal exposure to cyanobacterial cells using skin-patch or skin-prick testing in humans (Bernstein et al. 2011; Pilotto et al. 2004; Stewart et al. 2006a). Some of the exposed individuals showed mild irritation or allergenicity. No statistically significant dose-response relationships were found between skin irritation and increasing cyanobacterial cell concentrations. The allergenicity study suggests that cyanobacteria are allergenic, particularly among people with chronic rhinitis (Bernstein et al. 2011).

Skin-patch testing in humans was performed by Pilotto et al. (2004) with laboratory-grown cylindrospermopsin-producing *Cylindrospermopsis raciborskii* cells, both whole and lysed, which were applied using adhesive patches at concentrations ranging from < 5,000 to 200,000 cells/mL to the skin of 50 adult volunteers. After 24 hours, patches were removed and evaluation of the erythematous reactions were graded. Analysis of participants' reactions to patches treated with whole cells showed an odds ratio (OR) of 2.13 and a 95 percent confidence interval (CI) of 1.79–4.21 ($p < 0.001$). Lysed cells patch analysis showed an OR of 3.41 and a 95 percent CI of 2.00–5.84 ($p < 0.001$). No statistically significant increase or dose-response between skin reactions and increasing cell concentrations for either patches (whole or lysed) was observed. Subjects had skin reactions to the cylindrospermopsin, and positive control patches more frequently than to the negative control patches. The mean percentage of subjects with a reaction was 20 percent (95 percent CI: 15–31 percent). When subjects reacting to negative controls (39) were excluded, the mean percentage was 11 percent (95 percent CI: 6–18 percent). Evaluation of erythematous reactions showed that mild irritations (grade 2) were resolved in all cases within 24 to 72 hours.

Stewart et al. (2006a) conducted a skin-patch test with 39 volunteers (20 dermatology outpatients; 19 controls) who were exposed to six cyanobacterial suspensions, including toxigenic species, nontoxigenic species, mixed suspensions, and two cyanobacterial LPS extracts. All cyanobacterial suspensions of lyophilized cells were tested at three concentrations, 0.25 percent w/v, 0.05 percent w/v, 0.005 percent w/v, and the estimated doses of cyanotoxins were 2.4 ng/kg cylindrospermopsin and 2.6 ng/kg microcystins. Only one subject showed significant responses to cyanobacterial suspensions, specifically to two suspensions of cyanobacterial cells: *Cylindrospermopsis raciborskii* and mixed *Microcystis aeruginosa* and *Cylindrospermopsis raciborskii*, both of which contained one or more cyanotoxins. This subject showed no evidence of any dose-response effect in the dermal reactions. None of the participants reacted to the cyanobacterial LPS extracts, which ranged from 260 ppb to 31 ppm. This small clinical study demonstrated that dermal hypersensitivity reactions to cyanobacteria exposure occur infrequently, and further research into risk factors for predisposition to this type reaction could be beneficial.

Bernstein et al. (2011) studied skin sensitization to nontoxic extracts of *Microcystis aeruginosa* in 259 patients with chronic rhinitis over two years. Patients were evaluated with aeroallergen skin testing and

skin-prick testing. The authors found that 86 percent of the subjects had positive skin-prick tests to *Microcystis aeruginosa*, and that patients with existing allergic rhinitis were more likely to have reactions and sensitization to cyanobacteria than the controls (non-atopic health subjects). This study indicated that cyanobacterial allergenicity is associated with the nontoxic portion of the cyanobacteria.

Geh et al. (2015) studied the immunogenicity of extracts of toxic and nontoxic strains of *Microcystis aeruginosa* in patient sera (18 patients with chronic rhinitis and three non-atopic healthy subjects as documented in Bernstein et al. 2011). Enzyme Linked Immunosorbent Assay (ELISA) test was used to test IgE-specific reactivity, and gel electrophoresis, followed by immunoblot and mass spectrometry, was done to identify the relevant sensitizing peptides. The authors found an increase in specific IgE in those patients tested with the nontoxic *Microcystis aeruginosa* extract than the extract from the toxic strain. After pre-incubation of the nontoxic extract with various concentrations of microcystin, the authors found that phycocyanin and the core-membrane linker peptide were responsible for the release of β-hexosaminidase in rat basophil leukemia cells. The authors concluded that non-toxin-producing strains of cyanobacteria are more allergenic than toxin-producing strains in allergic patients, and that the toxin may have an inhibitory effect on the allergenicity of the cyanobacterial cells.

Facciponte et al. (2018) used polymerase chain reaction (PCR) to detect aerosolized cyanobacteria inhaled into the human respiratory tract. They found cyanobacteria at high frequencies in the upper respiratory tract (92.2 percent) and central airway (79.3 percent) of the study subjects (n = 77). The findings suggests that humans inhale aerosolized cyanobacteria, which can remain in the nostrils and the lungs.

D.1.3 Epidemiological Studies, Case Reports, and Outbreaks

Among the epidemiological studies discussed here, some identified significant associations between cyanobacteria exposure and a range of health outcomes including dermal, eye/ear, gastrointestinal (GI), and respiratory effects. Several of these studies also measured one or more cyanotoxins and found no association between cyanotoxin occurrence or exposure and health effects. Additional evidence from outbreak and case reports provides support for health effects associated with cyanobacteria exposure. The studies vary in study design, methods used, size of study population, cyanobacterial species evaluated, health effects identified, and cyanobacterial cell densities associated with human health effects. Therefore, substantial uncertainty remains regarding the associations between cyanobacterial cell exposure and human health effects. Overall, these studies provide evidence of statistically significant associations between cyanobacterial cell exposure and human health effects even in the absence of cyanotoxins. However, the reported associations between cyanobacterial cell densities and health outcomes are not consistent.

Eight epidemiological studies evaluated short-term health effects associated with recreational exposure to cyanobacterial blooms (El Saadi et al. 1995; Lévesque et al. 2014; Lin et al. 2015; Philipp 1992; Philipp and Bates 1992; Philipp et al. 1992; Pilotto et al. 1997; Stewart et al. 2006d). See Table D-1 for a summary list of these studies. The health outcomes evaluated included dermal, GI, respiratory, and other acute effects, such as eye or ear symptoms. Seven studies evaluated recreational exposure to freshwater cyanobacteria, and one evaluated exposure to marine water cyanobacteria (Lin et al. 2015). Two studies included field sites in the continental United States or Canada (Lévesque et al. 2014; Stewart et al. 2006d), three occurred in the United Kingdom (Philipp 1992; Philipp and Bates 1992; Philipp et al. 1992), and three were conducted in subtropical and tropical regions in Australia (El Saadi

et al. 1995; Pilotto et al. 1997) and Puerto Rico (Lin et al. 2015). These epidemiological studies are discussed below in chronological order.

Table D-1. Cyanobacteria Epidemiological Studies Summary

Reference	Study Design, n, and Location	Cyanobacteria Identified	Cyanotoxins Measured	Health Association[a]	Lowest Significant Cyanobacterial Cell Density (cells/mL)
Philipp (1992)	Cross-sectional n = 246 United Kingdom (Hampshire)	*Microcystis* sp., *Gleotrichia* sp.	–	No statistically significant health associations	No quantitative cyanobacterial cell densities provided
Philipp and Bates (1992)	Cross-sectional n = 382 United Kingdom (Somerset)	*Microcystis* sp., *Gleotrichia* sp.	–	No statistically significant health associations	No quantitative cyanobacterial cell densities provided
Philipp et al. (1992)	Cross-sectional n = 246 United Kingdom (Lincolnshire, South Yorkshire)	*Oscillatoria* sp., *Aphanizomenon* sp., *Aphanothece* sp., *Merismopedia* sp.	–	No statistically significant health associations	No quantitative cyanobacterial cell densities provided
El Saadi et al. (1995)	Case-control n cases = 102 GI, 86 dermatological n controls = 132 Australia (South Australia)	*Anabaena* sp., *Aphanizomenon* sp., *Planktothrix* sp., *Anabaena circinalis*, *Microcystis aeruginosa*	–	No statistically significant health associations	No quantitative cyanobacterial cell densities provided
Pilotto et al. (1997)	Cross-sectional n = 295 exposed n = 43 unexposed Australia (South Australia, New South Wales, Victoria)	*Microcystis aeruginosa*, *Microcystis* sp., *Anabaena* sp., *Aphanizormenon* sp., *Nodularia spumigena*	Hepatotoxins detected by mouse bioassay	Significant positive association between combined symptoms (GI, dermal, respiratory, fever, eye or ear irritation) and cyanobacteria	> 5,000
Stewart et al. (2006d)	Cohort (prospective) n = 1,331 Australia (Queensland, New South Wales) and Florida	Cyanobacteria identified, species not specified	Microcystins detected by HPLC with photodiode array detection or ELISA; cylindro-spermopsin and anatoxin-a detected by HPLC-MS/MS; saxitoxins not detected by HPLC with fluorescence detection	Significant positive association between respiratory symptoms and cyanobacteria Significant positive association between combined symptoms (GI, dermal, respiratory, fever, eye or ear irritation) and cyanobacteria	> 100,000[b]

Reference	Study Design, n, and Location	Cyanobacteria Identified	Cyanotoxins Measured	Health Association[a]	Lowest Significant Cyanobacterial Cell Density (cells/mL)
Lévesque et al. (2014)	Cohort (prospective) n = 466 Canada (Quebec)	Cyanobacteria identified, species not specified	Microcystins detected by ELISA	Significant positive association between GI symptoms with fever and cyanobacteria	20,000–100,000
Lin et al. (2015)[c]	Cohort (prospective) n = 15,726 Puerto Rico (Boquerón)	Cyanophyte filament, *Pseudanabaena* sp., Picocyanophyte, *Synechococcus* sp., *Synechocystis* sp., Cyanophyte cell pair, *Phormidium* sp., *Lyngbya* sp., *Trichodesmium* sp., *Aphanothece* sp., *Johannesbaptistia* sp., *Komvophoron* sp., Cyanophyte colony, Cyanophyte unicell sphere	Lyngbyatoxin-a and debromo-aplysiatoxin measured but not detected by HPLC-MS	Significant positive association between respiratory illness and cyanobacteria other than picocyanobacteria	36.7–237.4
				significant positive association between rash and cyanobacteria other than picocyanobacteria	> 237.4

sp. = unspecified species of the genus; HPLC = high performance liquid chromatography; MS = mass spectrometry; MS/MS = tandem mass spectroscopy

[a] Includes only significant associations between recreational cyanobacteria exposure and health effects.

[b] Values were converted from cyanobacterial cell surface area (> 12.0 mm^2/mL) to cyanobacterial cell density (> 100,000 cells/mL) using conversions in NHMRC (2008). Relationship between biomass and cyanobacterial cell density can vary by species and cell size (Lawton et al. 1999; Stewart et al. 2006d).

[c] Lin et al. (2015) evaluated picocyanobacteria and cyanobacteria other than picocyanobacteria separately.

Three cross-sectional studies were conducted by Philipp et al. (Philipp 1992; Philipp and Bates 1992; Philipp et al. 1992) to evaluate health effects related to exposure to cyanobacteria from recreational activities including sailing, windsurfing, and fishing in water bodies in the United Kingdom. Questionnaires were administered to participants who visited one of six inland lakes to evaluate exposure and morbidity (including dermal, eye/ear, GI, and respiratory symptoms). Several species of cyanobacteria were identified and, in some cases, cyanobacterial levels exceeded the National Rivers Authority threshold for "potential to cause harm." Only minor morbidity was identified among recreators, and no statistically significant associations between cyanobacteria exposure and morbidity were identified.

El Saadi et al. (1995) conducted a case-control study in Australia to evaluate exposure to river water with detectable levels of cyanobacteria and GI and dermatological symptoms evaluated by a medical practitioner. This river was used as a source for drinking water, domestic water, and recreational water. The authors found no significant association between recreational exposure to river water with cyanobacteria and GI or dermatological symptoms. Cyanotoxins were not measured, but species of cyanobacteria were present that were capable of producing cyanotoxins.

These four studies (El Saadi et al. 1995; Philipp 1992; Philipp and Bates 1992; Philipp et al. 1992) provided no quantitative data on cyanobacterial cell densities. Therefore, they could not help inform determination of a quantitative level associated (or not associated) with health effects.

Four more recent epidemiological studies assessed the association between exposure to recreational waters containing cyanobacteria and human health and provide quantitative density data for cyanobacterial cells (Lévesque et al. 2014; Lin et al. 2015; Pilotto et al. 1997; Stewart et al. 2006d). These studies reported at least one statistically significant association between exposure to cyanobacteria and human health outcomes, including GI illness (Lévesque et al. 2014), respiratory symptoms (Lin et al. 2015; Stewart et al. 2006d), dermal symptoms (Lin et al. 2015), or combined symptomology (GI, dermal, respiratory, and other symptoms) (Pilotto et al. 1997; Stewart et al. 2006d). These associations were linked to a range of densities of cyanobacterial cells from as low as > 5,000 cells/mL (Pilotto et al. 1997) to as high as 100,000 cells/mL (analogous to ≥ 12 mm^2/mL (NHMRC 2008; Stewart et al. 2006d). In contrast to the studies that examined all cyanobacteria, Lin et al. (2015) evaluated picocyanobacteria, larger cyanobacterial cells, and total phytoplankton, and reported health effects associated with 37–1,461 cells/mL for cyanobacteria other than picocyanobacteria.

Pilotto et al. (1997) investigated the health effects from recreational exposures (including jet-skiing, water skiing, swimming, and windsurfing) to cyanobacteria in Australia. The study included 852 participants, 777 who had water contact and were considered exposed, and 75 not exposed. There were 338 recreators (295 exposed, 43 not exposed) after exclusion of those who experienced symptoms or had recreational exposure in the five days prior to the initial interview at the water recreation site (the *after exclusion* study group). Health outcomes evaluated included diarrhea, vomiting, flu-like symptoms (e.g., cough), skin rashes, mouth ulcers, fevers, or eye or ear infections. Water samples were collected for evaluation of cyanobacterial cell counts, hepatotoxins, and neurotoxins.

In the *after exclusion* study group, when all symptoms were combined, the authors found a significant trend of increasing symptom occurrence with duration of exposure at seven days post-exposure (p-value for trend =0.03). Similarly, in the *after exclusion* study group there was a significant trend of increasing symptom occurrence with increasing cyanobacterial cell count (p-value for trend = 0.04). To account for the combined effect of duration of exposure and cyanobacterial cell density, unexposed participants were compared with those exposed for up to 60 minutes and for more than 60 minutes to water with up to 5,000 cells/mL and to water with more than 5,000 cells/mL. For the *after exclusion* study group, a significant trend of increasing symptom occurrence with increasing levels of exposure was identified (p-value for trend = 0.004). In addition, participants with recreational exposure for more than 60 minutes to cyanobacterial densities above 5,000 cells/mL had a significantly higher symptom occurrence rate at seven days post-exposure than unexposed participants (OR = 3.44, CI: 1.09–10.82). In this study, the significant trends observed in the *after exclusion* study group were not observed when all participants were included.

Pilotto et al. (1997) reported toxicity data collected by the Australia Water Quality Center. Presence or absence of particulate (intracellular) hepatotoxins in concentrated surface water phytoplankton samples was measured by mouse bioassay. The authors reported that hepatotoxins were identified at one site on two separate interview days and at three sites for one day each. No evidence of neurotoxins was detected. They reported that no significant association was found between the presence of hepatotoxins and symptom occurrence at two and seven days after exposure. Data and analysis methods were not provided. The authors point out that trends were observed at seven days and not at two days after exposure and this might suggest a delayed rather than an immediate allergic response. The authors also stated they could not rule out other causative factors, such as other microorganisms, that could co-occur with cyanobacteria. The results from this study informed the recommendations made by WHO in *Guidelines for Safe Recreational Water Environments* (WHO 2003).

Stewart et al. (2006d) conducted a prospective cohort study to investigate the incidence of acute symptoms in individuals exposed to cyanobacteria via recreational activities in lakes and rivers in Australia and Florida. This study included 1311 recreators with any water contact-related activity (e.g., swimming, boat entry/egress). Cyanobacterial cell densities were characterized in terms of cell surface area rather than cell counts (to normalize for cell size differences among different species). Authors evaluated incidence of acute symptoms in recreators exposed to low, medium and high levels of cyanobacteria.

Study subjects were asked to complete a self-administered questionnaire before leaving for the day after enrollment and to submit to a telephone follow-up interview. The questionnaire and follow-up interview forms gathered information on various acute illnesses, their onset and severity. Respiratory symptoms among study participants in the high recreational exposure group (total cyanobacterial cell surface area > 12 mm^2/mL on day of recreation) were significantly greater compared to participants in the low recreational exposure group (< 2.4 mm^2/mL) (adjusted OR = 2.1, 95 percent CI: 1.1–4.0). Respiratory symptoms were defined as difficulty breathing, dry cough, productive cough, runny nose, unusual sneezing, sore throat, or wheezy breathing. Reports of any symptom among study participants in the high exposure group were significantly greater compared to reports among study participants in the low recreational exposure group (adjusted OR = 1.7, 95 percent CI: 1.0–2.9). However, when subjects with recent prior recreational water exposure were excluded the result remained positive but not significant (adjusted OR = 1.6, 95 percent CI: 0.8–3.2). A dose-response relationship between increased cyanobacterial biomass and increased symptom reporting was not identified. The authors speculated that the pattern in their data could be due to a threshold effect. No other significant associations with health effects were identified.

For water samples that contained potentially toxic cyanobacteria, Stewart et al. (2006d) measured cyanotoxins including microcystins, saxitoxins, cylindrospermopsin and anatoxin-a by HPLC or HPLC-MS/MS methods. Cyanotoxins were infrequently identified and only at low levels. Microcystins were detected on two occasions (1 and 12 μg/L). Cylindrospermopsin was found on seven occasions (ranging from 1 to 2 μg/L). Anatoxin-a was identified on a single recruitment day at a concentration of 1 μg/L. A statistically significant increase in symptom reporting was found to be associated with anatoxin-a exposure, but the number of exposed subjects was very low (n =18). No relationship between fecal indicator bacteria (fecal coliforms) and symptoms was identified.

Lévesque et al. (2014) conducted a prospective study of health effects including GI, respiratory, dermal, eye/ear, and other symptoms associated with cyanobacteria and microcystin exposure at three lakes in Canada (Quebec), one of which was a local supply of drinking water. The study evaluated acute symptoms in humans (466 subjects included in analysis) living in proximity to lakes affected by blooms and analyzed recreational exposure (full and limited contact) and drinking water exposure scenarios for both cyanobacterial cells and microcystins.

More severe GI symptoms, defined as diarrhea, vomiting, nausea and fever, or abdominal cramps and fever, were associated with recreational contact (full and limited) and cyanobacteria. For the more severe GI symptoms, the adjusted relative risk (RR) increased with cyanobacterial cell counts providing evidence of a dose-response relationship (p-value for trend = 0.001, < 20,000 cells/mL: RR = 1.52, 95 percent CI: 0.65–3.51; 20,000–100,000 cells/mL: RR = 2.71, 95 percent CI: 1.02–7.16; > 100,000 cells/mL: RR = 3.28, 95 percent CI: 1.69–6.37). No evidence of a dose-response relationship for cyanobacterial cell counts and the less severe GI symptoms was found. No relationship was observed between duration of contact or head immersion and risk of GI symptoms. A significant increase for both

the less and the more severe GI symptoms was found with contact in the more highly impacted lakes (median cell densities 20,001–21,485 cells/mL), but not in the less impacted lake (median 1,032 cells/mL). No relationship was observed between microcystin concentrations and risk of GI symptoms. No significant associations between recreational exposures to cyanobacteria and health effects other than GI effects were identified.

To evaluate possible co-exposures, authors measured microcystin concentrations and *E. coli* as a fecal indicator. Lévesque et al. (2014) measured particulate (intracellular) and dissolved microcystins by ELISA and found that microcystin concentrations varied by lake and by sample location (littoral versus limnetic). Microcystins were detected in all three lakes. At Lake William the median values were below the limit of detection at littoral and limnetic stations, with maximum values of 0.63 µg/L and 0.02 µg/L, respectively. At Lake Roxton littoral stations, the median concentration was 0.23 µg/L (range: 0.008 µg/L–108.8 µg/L) and at limnetic stations the median was 0.12 µg/L (range: 0.04 µg/L–1.12 µg/L). The Mallets Bay littoral stations had a median of 0.70 µg/L (range: under limit of detection – 773 µg/L) and the limnetic stations had a median of 0.35 µg/L (range: 0.001 µg/L–125 µg/L).

Lévesque et al. (2014) reported that as a whole the microcystin concentrations during contact were relatively low (first tertile: < 0.0012 µg/L; second tertile: 0.0012–0.2456 µg/L; third tertile: > 0.2456 µg/L). Symptoms were examined in relation to recreational and drinking water exposure to cyanobacteria and microcystins. Only GI symptoms were associated with recreational contact. The highest microcystin concentration at which an episode of GI symptoms was reported was 7.65 µg/L. There was no significant increase in adjusted RR of GI symptoms with recreational exposure to more than 1 µg/L microcystins. Adjusted RR (adjusted for gender, gastrointestinal (GI) symptoms reported in the two weeks prior to data collection, residence's source of drinking water) for GI illness without fever and GI illness with fever were 1.06 (95 percent CI=0.32–3.52) and 1.48 (95 percent CI = 0.41–5.23), respectively. There were significant increases in adjusted RR of several symptoms in participants who received their drinking water from a source contaminated by cyanobacteria (muscle pain, GI illness, skin, and ear symptoms).

Lévesque et al. (2014) found that the geometric mean of *E. coli* at the three lakes ranged from 0 to 145 CFU per 100 mL, and there was no association between GI illness and *E. coli* levels. The authors noted that GI symptoms could have other causes, such as *Aeromonas* infections; however, the symptoms were not related to fecal contamination as measured by culturable *E. coli.* They also noted that people avoided full recreational contact during blooms and more people engaged in limited contact recreation at higher cell counts. This observation explains the counterintuitive finding that participants with limited contact exposure (fishing, watercraft without direct water contact) had higher likelihood of symptom reporting compared to participants with full contact.

A follow-up analysis (Lévesque et al. 2016) characterized the same health data as Lévesque et al. (2014) to evaluate the relationship of bacterial endotoxin (e.g., LPS) concentration to GI symptoms. Endotoxin concentrations were slightly correlated with cyanobacterial counts (polychoric correlation coefficient = 0.57). The highest tertile of endotoxin concentration (> 48 endotoxin units/mL) was significantly associated with GI illness both with and without fever (GI illness without fever RR = 2.87, CI: 1.62–5.08; GI illness with fever RR = 3.11, CI: 1.56–6.22). Adjustment to the level of cyanobacteria did not alter the relationship between endotoxin and GI illness and authors hypothesize that other Gram negative bacteria might play a role in the relationship between endotoxin levels and GI illness as has been suggested in a previous study (Berg et al. 2011). Authors note that they stored filtered water samples at −80 ºC for several months prior to conducting endotoxin testing and that another study

(O'Toole et al. 2009) showed a 44 percent mean decline in the concentration of endotoxins in samples stored at -80 ºC for several weeks compared to samples stored at 4 ºC for 24 hours. Lévesque et al. (2016) caution that concentrations reported could be underestimated and should be interpreted on an ordinal basis. Two other studies conducting endotoxin testing on frozen samples found concentrations of a similar magnitude as this study (Berg et al. 2011; Rapala et al. (2002).

Lin et al. (2015) conducted a prospective study based on data collected in 2009 at Boquerón, Puerto Rico for 26 study days involving 15,726 enrollees to examine the association between phytoplankton cell counts and illness among beachgoers. Three categories of phytoplankton were evaluated: picocyanobacteria, cyanobacteria other than picocyanobacteria, and total phytoplankton. The analysis compared people exposed at phytoplankton cell count levels > 25th percentile (e.g., 25th to 75th percentile, > 75th percentile) to people exposed at levels < 25th percentile (range of cyanobacteria other than picocyanobacteria: < 37 to 1461 cells/mL).

The Lin et al. (2015) study reported significant associations between recreational exposure to cyanobacteria other than picocyanobacteria and respiratory symptoms, rash, and earache. For the other symptoms measured, including eye irritation, no significant associations were observed. More specifically, cyanobacterial (other than picocyanobacterial) densities of 37 to 237 cells/mL (> 25th to < 75th percentile) and densities ≥ 237 cells/mL (≥ 75th percentile) were associated with increased respiratory symptoms (> 25th to < 75th percentile, OR = 1.30, 95 percent CI: 1.08–1.56; ≥ 75th percentile, OR = 1.37, 95 percent CI: 1.12–1.67) in study participants who reported body immersion. Respiratory symptom occurrence was defined as any two of the following: sore throat, cough, runny nose, cold, or fever. Cyanobacterial (other than picocyanobacterial) densities >237 cells/mL were associated with rash (OR = 1.32, 95 percent CI = 1.05–1.66) and earache (OR = 1.75, 95 percent CI: 1.09–2.82). Study participants who reported head submersion or swallowing of water showed no relationship between recreational exposures to cyanobacteria (other than picocyanobacteria) and respiratory symptoms. There was no association between recreational exposures to cyanobacteria (other than picocyanobacteria) and respiratory symptoms in study participants who reported head submersion or swallowing of water. A statistically significant association between cyanobacterial cell exposure (other than picocyanobacterial cell exposure) and all health effects combined was also observed.

Lin et al. (2015) measured the dermatoxins, debromoaplysiatoxin, and lyngbyatoxin, using HPLC-mass spectrometry and did not detect levels above the limit of detection of 1.0 ppb. Authors reported that debromoaplysiatoxin and lyngbyatoxin-a are photolabile and are unlikely to persist in the water column (Moikeha and Chu 1971). They noted that the health effects identified in this study were consistent with previous blooms of *Lyngbya majuscula*, which can produce these toxins, though *Lyngbya* only comprised three percent of total planktonic cyanobacteria (other than picocyanobacteria). It is also possible that the cyanobacterial cells or associated contaminants could be having direct health effects as cyanotoxins levels were below the limit of detection.

To evaluate possible co-exposures, some studies measured cyanotoxins and fecal indicators. Lin et al. (2015), Lévesque et al. (2014), Pilotto et al. (1997), and Stewart et al. (2006d) measured one or more cyanotoxins or total hepatotoxins. In some cases, cyanotoxin levels were below the limit of detection. To determine if study participants possibly were exposed to fecal contamination, three of the studies (Lévesque et al. 2014; Lin et al. 2015; Stewart et al. 2006d) measured bacterial fecal indicators at some study locations and times. Of the studies that measured bacterial fecal indicators, none found an association between bacterial fecal indicators and health effects. Of these studies, the only one with data

available for viral fecal indicators or concentrations of waterborne pathogens was Lin et al. (2015) provided in Wade et al. (2010) and Soller et al. (2016).

In summary, although four studies identified significant associations between cyanobacteria exposure and health effects, the type of health effect identified varied. One study reported a significant association between GI illness and exposure to cyanobacteria (Lévesque et al. 2014). Stewart et al. (2006d) and Lin et al. (2015) identified statistically significant associations between cyanobacterial cell exposure and respiratory effects. Lin et al. (2015) also found a statistically significant association between earache and cyanobacterial densities. Both Pilotto et al. (1997) and Stewart et al. (2006d) found statistically significant associations between cyanobacterial cell exposure and all symptoms combined. The three cross-sectional studies conducted in the United Kingdom in 1990 found no statistically significant associations, although some minor elevated morbidity was observed in exposed individuals (Philipp 1992; Philipp and Bates 1992; Philipp et al. 1992). Another 1992 case-control epidemiological study in Australia found no statistically significant symptoms for exposed recreators (El Saadi et al. 1995).

The Centers for Disease Control and Prevention (CDC) has collected information on illness outbreaks associated with HABs, which commonly involve cyanobacteria. This information includes human health effects and water-sampling results voluntarily reported to the Waterborne Disease Outbreak Surveillance System via the National Outbreak Reporting System and the Harmful Algal Bloom-related Illness Surveillance System. CDC published summary information on HAB-associated outbreaks from recreational exposures focusing on 2009–2010 with limited additional information available for outbreaks that occurred in 2001, 2004, and 2011–2012 (Dziuban et al. 2006; Hilborn et al. 2014; Hlavsa et al. 2014; Yoder et al. 2004). CDC defines a recreational water-associated outbreak as the occurrence of similar illnesses in two or more persons, epidemiologically linked by location and time of exposure to recreational water or recreational water-associated chemicals volatilized into the air surrounding the water.

The 2009–2010 reporting cycle was notable, as almost half (46 percent) the recreational water outbreaks reported to CDC were associated with HABs (Hilborn et al. 2014). Three of the outbreaks confirmed the presence of cyanobacteria, and four confirmed the presence of microcystins at levels greater than 20 μg/L. GI and dermatologic symptoms were the most commonly reported symptom categories associated with HAB-related outbreaks in freshwater (Dziuban et al. 2006; Hilborn et al. 2014; Hlavsa et al. 2014; Yoder et al. 2004). For the cyanobacteria-associated outbreaks with reported symptom counts, the most common symptoms reported were GI related, including vomiting, diarrhea, and nausea (estimated to be > 40 percent). The second most frequent outbreak symptom reported was skin rash (> 27 percent cases reported). Fever, earache, skin irritation, and headache were the next most frequently reported symptoms (11 percent, nine percent, and nine percent of cases reported, respectively).

During 2009 and 2010 in the United States, 11 outbreaks of illness associated with HABs were reported to CDC, all occurring in freshwater lakes and reported via the National Outbreak Reporting System (NORS) and the Harmful Algal Bloom-related Illness Surveillance System (HABISS). Hilborn et al. (2014) analyzed the HAB outbreak data from 2009–2010 and found the 11 outbreaks affected at least 61 persons, resulting in two hospitalizations, and included GI, dermatologic, respiratory, neurologic, and other symptoms. Sixty-six percent of case patients were individuals aged one to 19 years (n = 38 of 58 total) and 35 percent were aged nine years or younger (n = 20). In addition, in a cyanobacteria-associated outbreak in 2001, 42 children were affected. Outbreak data are typically limited in scope and thought to represent an underreporting of the "true" occurrence of illness in a population, but available

information suggests that children may share a disproportional share of the health burden associated with recreational exposures to cyanobacterial HABs.

Dziuban et al. (2006) and Walker et al. (2008) reported on outbreaks in Nebraska. Dziuban et al. (2006) described two 2004 cyanobacteria-associated outbreaks in which 22 cases of illness were reported from exposure to Nebraska lakes. The predominant illnesses in both outbreaks included dermatitis and gastroenteritis, and individuals who sought medical care showed a combination of rashes, diarrhea, cramps, nausea, vomiting, and fevers. Walker et al. (2008) also reported about a Nebraska outbreak. Levels of total microcystins at the east swimming beach of Pawnee Lake exceeded 15 ppb on July 12, 2004, and a health alert was issued. However, heavy public use of Pawnee Lake occurred that weekend and more than 50 calls were received from the public, complaining about symptoms such as skin rashes, lesions, blisters, vomiting, headaches, and diarrhea after swimming or water skiing in Pawnee Lake (Walker et al., 2008).

D.2 Mode of Action

Few mechanistic investigations have been completed on how exposure to cyanobacterial cells might lead to inflammatory response. Torokne et al. (2001) evaluated the sensitization and irritation potential of *Microcystis*, *Anabaena*, *Cylindrospermopsis*, and *Aphanizomenon* bloom and strain samples and found no correlation between the cyanotoxin content and allergenicity. For example, the nontoxic *Aphanizomenon* was the most allergenic sample, more allergenic than the most toxic cyanobacterial cells they studied, *Microcystis aeruginosa*. Stewart et al. (2006e) concluded that cutaneous effects strongly suggest allergic reactions, and symptoms such as rhinitis, conjunctivitis, asthma, and urticaria (or hives) also indicate immediate hypersensitivity responses, which are probably explained by a cascade action of pro-inflammatory cytokines.

Bernstein et al. (2011) suggested that the allergenic structure of cyanobacteria might be associated with a non-toxin-producing part of the organism. Building on this conclusion, Geh et al. (2015) conducted a series of experiments to identify the cyanobacteria allergen(s) responsible for sensitization. Study participants were given skin-prick tests with extracts from nontoxic *Microcystis aeruginosa* strains. Serum from these individuals was collected from a subset of 15 patients who elicited strong skin test responses to *Microcystis aeruginosa* and from three healthy control subjects. The lysate from nontoxic *Microcystis aeruginosa* strains was significantly ($p < 0.01$) more immunoreactive than the lysate from the toxin-producing strains, which suggests that the nontoxic strain was more allergenic than the toxic strain. They found, however, that IgE binds to *Microcystis aeruginosa* peptides present in lysates of both the toxic and nontoxic strains. Geh et al. (2015) also performed a β-hexosaminidase release assay, as a surrogate assay for measuring histamine release, to identify functional activity of the *Microcystis aeruginosa* extracts using rat basophil leukemia cells. The authors concluded that the same allergen is present in toxic and nontoxic *Microcystis aeruginosa* lysates, but suggest the toxic *Microcystis aeruginosa* lysate might contain an endogenous inhibitor that prevents IgE from effectively binding to the specific allergen. The further analysis by Geh et al. (2015) of the sera of individuals exposed to nontoxic *Microcystis aeruginosa* lysate indicated that either linker core-membrane peptide or phycocyanin, or both, are potentially responsible for *Microcystis aeruginosa* allergenicity.

Epidemiological studies and case reports suggest respiratory effects that could be consistent with an allergic or hay fever type reaction (Giannuzzi et al. 2011; Stewart et al. 2006e). Inhalation exposure to bacterial endotoxins (i.e., a toxin that is part of the cyanobacterial cell as opposed to exotoxins such as microcystins and cylindrospermopsin) has been found to be associated with pulmonary disease,

including asthma, chronic obstructive airway disease, and emphysema (Stewart et al. 2006b). A recent review of the structure and effects of cyanobacterial LPS suggested that it could act as an antagonist of the TLR4 receptor and inhibit the inflammatory-response pathway (Durai et al. 2015).

Stewart et al. (2006e) also noted that, although symptoms and time to onset can be disparate, several reports described:

> "a collective group of symptoms resembling immediate or Type-I hypersensitivity reactions. Immediate hypersensitivity reactions are commonly associated with atopy, which is the familial tendency to react to naturally occurring antigens, mostly proteins, through an IgE-mediated process. Atopy frequently manifests as a spectrum of diseases, e.g., seasonal rhinitis, conjunctivitis, asthma, and urticaria."

Documentation of this type of respiratory response is consistent with results from Geh et al. (2015) and further supports that immune system response follows exposure to cyanobacteria.

In older literature, cyanobacterial LPS was suspected as being a cause of inflammatory response because this cell structure, also found in many Gram negative bacterial species, has been observed to initiate acute inflammatory responses in mammals that are typical of a host reaction to tissue injury or infection (Stewart et al. 2006b). The Stewart et al. (2006e) review, however, found evidence to support this mechanism lacking. Although all cyanobacteria contain the pigment phycocyanin, not all species of cyanobacteria have shown dermal reactions. Also, some species of cyanobacteria produce toxins that are known dermal irritants (e.g., lyngbyatoxin-a). Pilotto et al. (2004), however, found that 20–24 percent of the study participants exposed to cyanobacterial cells via skin patches for 24 hours showed dermal reactions to cyanobacteria species, both whole and lysed cells.

Stewart et al. (2006b) noted that the effects of microcystin- and cylindrospermopsin-producing bacteria on the GI tract could suggest that cyanotoxins and LPS from the cyanobacteria or other bacteria residing in the gut might cross a gut mucosal barrier that has been disrupted and enhance the adverse effects of cyanotoxins.

An aquatic invertebrate study using brine shrimp (*Artemia salina, Daphnia magna*, and *Daphnia galeata*) to determine the toxicity of microcystin and cylindrospermopsin in combination with cyanobacterial LPS found that pre-exposure to LPS increased the lethal concentration (LC_{50}) of cylindrospermopsin eight-fold (Lindsay et al. 2006). The authors concluded that the decrease in susceptibility to cylindrospermopsin was due to the effects of LPS on detoxification enzyme pathways; LPS decreased toxic metabolites of cylindrospermopsin by suppressing the invertebrate cytochrome P450 system, thus decreasing toxicity.

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APPENDIX E. INCIDENTAL INGESTION EXPOSURE FACTOR COMBINED DISTRIBUTION ANALYSIS

This appendix describes in detail the approach used to derive the value for ingestion rate in units of liters per day. The ingestion rate is used in the derivation of the recommended cyanotoxin values in this document.

To arrive at liters of ingestion per day, the EPA combined data on liters of ingestion per hour and the number of hours spent in the water per day. Both of these parameters were represented as log-normal distributions. The sources of the data were:

- Recreational water ingestion per hour – The lead author of Dufour et al. (2017) provided the EPA's Office of Water, Health and Ecological Criteria Division with the raw data collected and analyzed in the study, which included mL of water ingested during a swimming event. Each participant in the study also reported the length of time they spent in the water. The ingestion per event was normalized to one hour for each participant and converted to liters to arrive at liters ingested per hour. The mean and standard deviation were calculated for different age groups (6 to 10, 11 to 17, 18 years and up, and all ages). See Table E-1 below for summary statistics for this parameter. Subsequent to the EPA's analysis, Dufour et al. posted their raw dataset on data.gov (U.S. EPA 2018). There are few minor variations in the dataset analyzed here and the posted dataset (i.e., the posted dataset included an additional adult participant's results, specified time spent in the water as 45 minutes for two participants, rounded ingestion volumes of 0.5 up to 1, and indicated a higher ingestion volume for one adult woman). The EPA performed a sensitivity analysis to see if these differences impacted the results and found no significant effect. The very slight differences were within the rounding to the third decimal. No differences were observed between the datasets for the results of the combined distribution analysis for the six- to 10-year age group.

- Duration of swimming per day – the EPA's 2011 *Exposure Factors Handbook* (EFH; Table 16-20). Time Spent (minutes/day) in Selected Outdoor Locations, Doers Only, At Home in the Outdoor Pool or Spa). Table E-2 below shows the summary statistics provided by the EPA's EFH.

Table E-1. Parameters Used to Fit Ingestion Distributions

Ingestion Rate (L/hour)				
Age Group (sample size)	**Mean[a]**	**Standard deviation**	**Minimum**	**Maximum**
6 to 10 (child) (n = 66)	0.03745	0.03355	0.00033	0.20000
11 to 17 (child) (n = 170)	0.03996	0.04377	0.00067	0.26800
18+ (adult) (n = 312)	0.02811	0.04960	0.00012	0.36800
All (6 to 50+) (n = 549)	0.03290	0.04643	0.00012	0.36800

[a] Arithmetic mean based on raw data provided by the Dufour et al. (2017) study authors. The ingestion rates for age groups children (6 to 10), teens (11 to 15), and adults (16 and over) were reported as geometric means in Dufour et al. (2017).

Table E-2. Parameters Used to Fit Recreation Duration Distributions

EPA 2011 EFH (Excerpt from Table 16-20) (minutes/day)					
Age Group (sample size)	**Mean**	**Standard deviation**	**Median (50th percentile)**	**Minimum**	**Maximum**
1 to 4 (n = 9)	85.6	86.3	60	15	255
5 to 11 (n = 15)	164.2	103.97	140	25	450
12 to 17 (n = 5)	97	53.8	100	40	180
18 to 64 (n = 44)	117.6	112.7	83	4	450
> 64 (n = 10)	78.9	85.3	53	1	258

R (open source programming language) was used to perform the calculations described in this appendix. The annotated R code is shown below, following a summary of what calculations were performed and assumptions.

The water ingestion rate per hour data from Dufour were used to compute an arithmetic mean and standard deviation, which are in turn used to compute the log geometric mean (GM) and log geometric standard deviation (GSD) using a mathematical conversion formula. The log GM and log GSD are used as distributional parameters to generate 10,000 random samples representing water ingestion rates per hour of recreational activity (L/hour).

The mean and standard deviation of the number of recreational hours spent in the water per day are reported as summary statistics in the EFH 2011, and are used to compute the log GM and log GSD using a mathematical conversion formula. The log GM and log GSD are used as distributional parameters to generate 10,000 random samples representing water ingestion rates per hour of recreational activity (hour/day).

The two component distributions are assumed to be statistically independent of each other and are multiplied to generate a combined distribution with 10,000 values for the ingestion rate of water per day of recreational activity in L/day. Summary statistics, including the mean, standard deviation, and point estimates of various percentiles, are then computed from the combined distribution. The EPA chose the 90th percentile point estimate for children six to 10 (0.21 L/day) to calculate the recommended cyanotoxin values.

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R Code

```
#Cyanotoxin recAWQC WA

# This script is to combine distributions for water ingestion rate (L/hr) and recreational exposure duration (hr/day) to develop a distribution for ingestion/day (L/day) and to generate a histogram of this combined distribution

# The first distribution is the incidental ingestion rate per hour from the Dufour dataset

# The second distribution is the recreational exposure duration (hr/day) from the EPA 2011 Exposure Factors Handbook Table 16-20. Time Spent (minutes/day) in Selected Outdoor Locations, Doers Only, At Home in the Outdoor Pool or Spa

# Both distributions are assumed to be log-normal


#####Read required libraries and set simulation sample size ##################


rm(list=ls()) # Remove all current R objects from memory


library(truncnorm) #import library for truncated normal distribution


nsamp = 1000000 # specify number of samples in monte-carlo analysis


set.seed(1984756) # set seed for analysis replicability


#########################################################


# The combined distribution function (cdist) assumes a log-normal distribution for ingestion rate (L/hour) and a log-normal distribution for exposure duration (hr/d)

#....using the mean and sd as parameter inputs. This function is called in later sections of the code for each age group analysis.


cdist<-function(nsamp,mean_dur,sd_dur,min_dur,max_dur,mean_ing,sd_ing,min_ing,max_ing){
```

```
n<-nsamp # number of samples to be drawn

#transform mean and sd of duration

sd_dur_ln<-sqrt(log((sd_dur/mean_dur)^2+1)) # standard deviation of duration in log space
mean_dur_ln<-log(mean_dur)-((sd_dur_ln^2)/2) # mean of duration in log space
min_dur_ln<-log(min_dur) # minimum duration in log space
max_dur_ln<-log(max_dur)

#transform mean and sd of ingestion rate

sd_ing_ln<-sqrt(log((sd_ing/mean_ing)^2+1))
mean_ing_ln<-log(mean_ing)-((sd_ing_ln^2)/2)
min_ing_ln<- -10^10
max_ing_ln<-log(max_ing)

# draw n samples from the truncated ingestion rate distribution in L/hr

ingperhr_ln_trunc<-exp(rtruncnorm(n=n, a=min_ing_ln, b=max_ing_ln, mean=mean_ing_ln,
sd=sd_ing_ln)) #truncated log normal distribution

# draw n samples from the truncated duration distribution (hr/d)

duration_hr_ln_trunc<-exp(rtruncnorm(n=n, a=min_dur_ln, b=max_dur_ln, mean=mean_dur_ln,
sd=sd_dur_ln))

# compute n samples for the combined ingestion rate per day distribution (L/d)

ingperday<-ingperhr_ln_trunc*duration_hr_ln_trunc #combine distributions
```

```
  print(summary(ingperday)) # print summary statistics of the combined distribution

  print(quantile(ingperday, probs=0.90)) # print 90th percentile of the combined distribution

  #Generate histogram

  hist(ingperday,xlab="Ingestion rate (L/day)",ylab="Probability", main ="Truncated hybrid distribution fit", xlim=c(0, 2.0), ylim=c(0, 1))
  h=hist(ingperday)
  h$density=h$counts/sum(h$counts)
  plot(h,xlab="Ingestion rate (L/day)",ylab="Probability", main ="Log-normal distribution fit", xlim=c(0, 1), ylim=c(0, 0.99), xaxp=c(0,1.5,15), freq=FALSE)

}

#########################################################

#I. Analysis for 6 to 10 age group

# These values are from 2011 EFH table 16-20 for ages 5 to 11.

mean_dur_min=164.2
sd_dur_min=103.97
min_dur_min=25
max_dur_min=450

# Convert exposure data from the EPA's EFH from min/day to hr/day

mean_dur<-mean_dur_min/60 #mean exposure duration hr/day
```

```
sd_dur<-sd_dur_min/60 #sd exposure duration hr/day

min_dur<-min_dur_min/60 #minimum exposure duration hr/day

max_dur<-max_dur_min/60 #maximum exposure duration hr/day


# These ingestion rate values are computed from the Dufour dataset


mean_ing<- 0.03745 # mean ingestion rate in L/hr

sd_ing<-0.03355 # sd ingestion rate in L/hr

min_ing<-0.00033 # minimum ingestion rate in L/hr

max_ing<-0.20000 # maximum ingestion rate in L/hr


cdist(nsamp,mean_dur,sd_dur,min_dur,max_dur,mean_ing,sd_ing,min_ing,max_ing) # call combined
distribution function



##########################################################


#II. Analysis for 11 to 17 age group


# These values are from 2011 EFH table 16-20 for age 12 to 17

mean_dur_min=97

sd_dur_min=53.81

med_dur_min=100

min_dur_min=40

max_dur_min=180


# Convert exposure data from the EPA's EFH from min/day to hr/day


mean_dur<-mean_dur_min/60 #mean exposure duration hr/day
```

```
sd_dur<-sd_dur_min/60 #sd exposure duration hr/day
med_dur<-med_dur_min/60 #median exposure duration hr/day
min_dur<-min_dur_min/60 #minimum exposure duration hr/day
max_dur<-max_dur_min/60 #maximum exposure duration hr/day

# These ingestion rate values are computed from the Dufour dataset

mean_ing<-0.03996 # mean ingestion rate in L/hr
sd_ing<-0.04377 # sd ingestion rate in L/hr
min_ing<-0.00067 # minimum ingestion rate in L/hr
max_ing<-0.26800 # maximum ingestion rate in L/hr

cdist(nsamp,mean_dur,sd_dur,min_dur,max_dur,mean_ing,sd_ing,min_ing,max_ing) # call combined distribution function

#########################################################

#III. Analysis for 18+ age group
# Combine exposure duration data for 18 to 64 and for >64 age groups from 2011 EFH table 16-20.

mean_dur_min=(117.61+78.9)/2
sd_dur_min=sqrt((112.72^2+85.32^2)/2)
min_dur_min=1
max_dur_min=450

# Convert exposure data from the EPA's EFH from min/day to hr/day

mean_dur<-mean_dur_min/60 #mean exposure duration hr/day
```

```
sd_dur<-sd_dur_min/60 #sd exposure duration hr/day

min_dur<-min_dur_min/60 #minimum exposure duration hr/day

max_dur<-max_dur_min/60 #maximum exposure duration hr/day


# These ingestion rate values are computed from the Dufour dataset


mean_ing<-0.02811 # mean ingestion rate in L/hr

sd_ing<-0.04960 # sd ingestion rate in L/hr

min_ing<-0.00012 # minimum ingestion rate in L/hr

max_ing<-0.36800 # maximum ingestion rate in L/hr


cdist(nsamp,mean_dur,sd_dur,min_dur,max_dur,mean_ing,sd_ing,min_ing,max_ing) # call combined
distribution function



##########################################################


# IV. Analysis for all age groups (including 1-4 yo)


# Combine exposure duration data for all age groups (1 to 4, 5 to 11, 12 to 17, 18 to 64, >64) from 2011
EFH table 16-20.


mean_dur_min=(85.56+164.2+97+117.61+78.9)/5

sd_dur_min=103.71 # SD reported in EFH for all ages

min_dur_min=1

max_dur_min=450


# Convert exposure duration data from min/day to hr/day


mean_dur<-mean_dur_min/60 #mean exposure duration hr/day
```

```
sd_dur<-sd_dur_min/60 #sd exposure duration hr/day

min_dur<-min_dur_min/60 #minimum exposure duration hr/day

max_dur<-max_dur_min/60 #maximum exposure duration hr/day


# These ingestion rate values are computed from the Dufour dataset


mean_ing<- 0.03290 # mean ingestion rate in L/hr

sd_ing<- 0.04643 # sd ingestion rate in L/hr

min_ing<-0.00012 # minimum ingestion rate in L/hr

max_ing<-0.36800 # maximum ingestion rate in L/hr


cdist(nsamp,mean_dur,sd_dur,min_dur,max_dur,mean_ing,sd_ing,min_ing,max_ing) # call combined
distribution function
```

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APPENDIX F. INGESTION STUDIES

The EPA reviewed seven studies to evaluate recreation-associated incidental ingestion (DeFlorio-Barker et al. 2017; Dorevitch et al. 2011; Dufour et al. 2006, 2017; Schets et al. 2011; Schijven and de Roda Husman 2006; Suppes et al. 2014). Evans et al. 2006 was also reviewed, but is the same dataset as Dufour et al. (2017), so it is not included in the table. The EPA's approach for estimating incidental exposure while swimming used by the EPA's Office of Pesticide Programs (OPP) is also summarized below.

F.1 DeFlorio-Barker et al. (2017)

DeFlorio-Barker et al. (2017) combined ingestion data from Dufour et al. (2017) and time spent in the water data from 12 cohorts of epidemiological studies to estimate the volume of water ingested per swimming event. They calculated the ingested volume per minute (mL/minute) for each Dufour et al. (2017) study participant, using the mL ingested and the self-reported time spent in the water for each participant. The National Epidemiological and Environmental Assessment of Recreational Water Study and Southern California Coastal Water Research Project epidemiological studies included 68,685 recreators at four freshwater and eight marine beaches. The participants in these studies estimated how much time they spent in the water. DeFlorio-Barker et al. (2017) combined the mL/minute ingestion rate from Dufour et al. (2017) and the self-reported time spent in the water for the epidemiological study participants to calculate the volume of water ingested per event. The results of this study corroborate other studies that demonstrate that, on average, children have higher incidental ingestion than adults when recreating.

F.2 Dorevitch et al. (2011)

Dorevitch et al. (2011) evaluated incidental ingestion associated with multiple types of water contact activities in both surface water (canoeing, fishing, kayaking, motor boating, and rowing) and in pools (canoeing, fishing, kayaking, swimming, wading/splashing, and walking around the pool as a control). The surface water activities did not include swimming because the water body was designated for secondary contact recreation only. Volume of ingestion was self-reported via interviews (3,367 participants: 2,705 individuals recreating in the Chicago Area Waterway System (CAWS, surface water) and 662 individuals recreating at a public outdoor swimming pool). At the end of their exposure, participants self-reported whether they ingested water, and how much, during their recreational experience. The authors used a subset of the pool exposures to assess cyanuric acid in urine to determine the accuracy of the self-reported ingestion volumes. There was strong agreement between self-reported results and cyanuric acid measurement (none = 0.0014±0.008 L; drop to teaspoon = 0.0094±0.011 L; mouthful = 0.026±0.037 L).

The results indicate that the odds of ingesting a teaspoon or more of water are significantly higher among swimmers than among those who just immersed their head in a swimming pool or those who participated in the other, more limited contact activities on surface waters. More specifically, rowing, motor boating, fishing, wading/splashing, and non-capsizing kayaking and canoeing were found to be low-ingestion activities, resulting in 95 percent upper confidence limit ingestion volumes between 0.01 and 0.012 L/hour. Less than five percent of limited contact recreators on surface waters reported swallowing any water. The study authors considered those who capsized during canoeing or kayaking a "middle ingestion category," with mean incidental ingestions of 0.006 to 0.005 L/hour. Swimmers were the highest ingestion category, with a mean of 0.01 L/hour. Swimmers in a pool were more than

50 times as likely to report swallowing a teaspoon of water compared to people who canoed or kayaked in surface waters.

In surface water, participants ages six years and above incidentally ingested the most water while canoeing and capsizing compared to any other activity assessed (median = 0.0036 L; mean = 0.006 L; Upper 95 percent CI: 0.0199 L). Kayaking and capsizing in surface water resulted in nearly as high incidental ingestion (mean = 0.005 L; Upper 95 percent CI: 0.0165). In swimming pool water, participants ages six and above incidentally ingested the most water while swimming compared to any other activity assessed (median = 0.006 L; mean = 0.01 L; Upper 95 percent CI: 0.0348 L). Duration of activities was not reported, so the ingestion volumes are on a per event basis.

F.3 Dufour et al. (2006)

The EPA's *Exposure Factors Handbook* (EFH) (2011) presents values for incidental ingestion while recreating values citing Dufour et al. (2006). Dufour et al. (2006) measured the incidental ingestion of water while participants were swimming in a pool and found that children under the age of 18 ingested higher volumes of water while swimming than adults. The 2006 study design instructed participants to swim for at least 45 minutes, so the time the participants spent in the water is probably not representative of preferred or regular patterns for recreation duration and the actual duration was not recorded. Both studies reported higher ingestion among children compared to adults. The values presented in the EFH adjusted the Dufour et al. (2006) data from a per event basis to an hourly ingestion rate. The EFH recommends using the 97th percentile ingestion rate for children and the maximum reported value for adults because the dataset is limited (U.S. EPA 2011).

F.4 Evans et al. (2006)

Evans et al. (2006) presented results from an observational study of incidental water ingestion during recreational swimming activities using the same methodology as the Dufour et al. (2006) pilot study. This study characterized ingestion volumes for younger children verses older children and adults. Evans et al. (2006) reported higher ingestion volumes for younger children. Although study results were presented at a conference, they were not published, so the EPA did not cite this publication in the derivation of the recommended cyanotoxin values. However, Dufour et al. (2017) includes the data reported by Evans et al. (2006).

F.5 Schets et al. (2011)

A study in the Netherlands by Schets et al. (2011) used questionnaires to collect estimates of water swallowed while swimming/bathing in freshwater, marine water, and swimming pools. Of the 8,000 adults who completed the questionnaire, 1,924 also provided estimates for their eldest child (< 15 years of age). The participants estimated the amount of water they or their children swallowed while swimming. Participants chose between four categories of water volumes: (1) no water or only a few drops; (2) one to two mouthfuls (a shot glass); (3) three to five mouthfuls (coffee cup); and (4) six to eight mouthfuls (soda glass). Schets et al. (2011) also conducted a series of experiments to measure the amount of water that corresponded to a mouthful of water and converted the data in the four response categories to volumes of water ingested per event. Adult men swallowed, on average 0.030 L/hour and women swallowed 0.020 L/hour, with somewhat greater ingestion in marine waters than in freshwater or a swimming pool. In fresh and marine waters children swallowed about the same as adults, and in swimming pools they ingested more than adults, on average, 0.038 L/hour compared with 0.030 and 0.021 for males and females, respectively (Schets et al. 2011). The EPA made the assumption that

exposure in a swimming pool is roughly equivalent to exposure in fresh and marine waters. Schets (2011) supports that assumption, although it is a somewhat more conservative assumption for children. However, when bodyweight is taken into account the greater exposure to children versus adults becomes clear. Additional research would be helpful to clarify uncertainty in differences in ingestion from different types of waters.

F.6 Schijven and de Roda Husman (2006)

Schijven and de Roda Husman (2006) studied sport and occupational diver incidental ingestion. The types of water studied for occupational divers (n = 37 divers) were open sea and coastal marine water, and freshwater. For sport divers (n = 483 divers), the types of water considered were open sea and coastal marine water, fresh recreational water, canals and rivers, city canals, and swimming pools. The divers were asked to estimate how much water they swallowed in terms of: none, few drops, shot glass, coffee cup, or soda glass. The authors translated the description of volumes from the questionnaires into average volumes. Occupational divers reported incidentally ingesting more water per dive in marine water (mean: 0.0098 L/dive; maximum: 0.1 L/dive) compared to freshwater (mean: 0.0057 L/dive; maximum: 0.025 L/dive). Sports divers wearing an ordinary diving mask reported incidentally ingesting the most water per dive in swimming pools (mean: 0.02 L/dive; maximum: 0.19 L/dive), followed by recreational freshwater (mean: 0.013 L/dive; maximum: 0.19 L/dive) and coastal marine water (mean: 0.0099 L/dive; maximum: 0.19 L/dive). Sports divers wearing a full face mask reported incidentally ingesting less water than sports divers wearing an ordinary diving mask. The mean ingestion rates in freshwater ranged from 0.0015 to 0.019 L/hour, with the highest mean being for adult recreational divers wearing an ordinary diving mask and the lowest mean for adult recreational divers wearing a full face mask. The mean ingestion rates in marine water ranged from 0.0005 to 0.014 L/hour, with the highest mean being for adult recreational divers wearing an ordinary diving mask and the lowest mean for adult recreational divers wearing a full face mask. The age of the divers was not included in the study report. Occupational divers dived on average 60–95 minutes and sport divers dived on average 42–52 minutes per dive.

F.7 Suppes et al. (2014)

Suppes et al. (2014) used a similar measurement method as Dufour et al. (2006, 2017), (i.e., using cyanuric acid as an indicator of pool water ingestion) to evaluate the rate of water ingested by 16 children ages five to 17 years. They and found that children on average ingested pool water at a higher rate than adult participants. Total time in water, quantified by viewing videos, was used to adjust pool water ingestion volumes to obtain rates. After adjustments for false-positive measurements were applied, the mean rate at which adults ingested water was 0.0035 L/hour with range 0–0.051 L. The mean rate at which children ingested water was 0.026 L/hour with range 0.0009–0.106 L/hour.

F.8 U.S. EPA (2003)

Additional estimates of incidental water ingestion rates while swimming in pools have been identified by the EPA's OPP. OPP calculated people's exposures to pool chemicals while they swim using its Swimmers Exposure Assessment Model (SWIMODEL) (U.S. EPA 2003). SWIMODEL uses incidental ingestion values for children that are twice the values used for adults. Incidental ingestion rates among adults while swimming competitively and noncompetitively are 0.0125 L/hour and 0.025 L/hour, respectively. The model assumes an incidental ingestion rate of 0.050 L/hour for children ages seven to 10 years and 11 to 14 years while swimming noncompetitively. The 0.050 L/hour value is the value used

in the EPA OPP's Standard Operating Procedures (U.S. EPA 2000) and is based on recommendations from EPA's Risk Assessment Guidance for Superfund, Part A (U.S. EPA 1989, 2000, 2003).

F.9 Summary

Although these studies used different methodologies and have limitations with respect to reporting information for different age group categories, their results show a similar pattern compared to Dufour et al. (2006, 2017): children ingest water at a higher rate while swimming than adults. Dufour et al. (2017) and Dufour et al. (2006) identified mean ingestion rates for children of 0.037–0.040 and 0.049 L/hour, respectively, and adult rates of 0.028 and 0.021 L/hour, respectively. Depending on water type, Schets et al. (2011) found a mean ingestion volume for children aged zero to 14 years of 0.028–0.038L/hour for children and 0.020–0.036 L/hr for males and females. The most pronounced differences were for swimming pools, where children ingested at a higher rate (0.038 L/hour) than adults (males: 0.030 L/hour; females: 0.021 L/hour). Dorevitch et al. (2011) reported ingestion rates while swimming for all ages of 0.010L/hour. Suppes et al. (2014) reported an adjusted mean ingestion rate of 0.026 L/hour for children and a rate of 0.0035 L/hour for adults.

Table F-1 includes: sample size, measurement methodology, the maximum values or the upper confidence intervals (CI) for the mean ingestion per event, time spent in the water (mean or range), and the mean ingestion volume normalized to one hour (or range if a range of durations were reported). This information supports comparison of the studies and help with understanding the range of different recreational exposures from activities.

The column with normalized ingestion (mL/hour) was populated using the following methods:

- Dufour et al. (2017) – The EPA used the individual data points from this dataset. Each participant's volume ingested was adjusted to one hour based on the length of time that participant reported being in the water.

- Dufour et al. (2006) – The EPA assumed that all swimming events were 45 minutes in duration. The values reported in Table F-1 are the same as the values in EPA's EFH (2011).

- DeFlorio-Barker et al. (2017) – Normalized data are not included in Table F-1 because the authors used the Dufour et al. (2017) rate in their modeling, so including the normalized data would be duplicative of Dufour et al. (2017).

- Dorevitch et al. (2011) – Study authors included normalized values in the study publication.

- Schets et al. (2011) – The EPA used the mean duration values provided in the publication to calculate the normalized value for each age and activity category.

- Suppes et al. (2014) – Study authors reported volume per hour.

- Schijven and de Roda Husman (2006) – The EPA used the range of duration values provided in the publication to calculate the normalized value for each activity category.

F.10 References

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Table F-1. Studies of Incidental Ingestion Volumes While Recreating

Reference	Study Sample Size	Measurement Methodology	Water Type, Recreational Activity	Age Group[a] (Years Old)	Mean Ingestion (Maximum Value) (mL/event)	Mean Duration of Event (minutes)	Normalized Ingestion (mL/hour)
Dataset from Dufour (data collection methods reported in Dufour et al. (2017))[b]	>500	Cyanuric acid was measured in pool water and urine samples, and ingestion rate was calculated based on duration of swimming event	Swimming pool, Swimming	6 to 10	59.8 (245)	each participant reported a duration[c]	37
				11 to 17	35.6 (267)	each participant reported a duration[c]	40
				18+	23.7 (279)	each participant reported a duration[c]	28
				All ages (6+)	31.7 (279)	each participant reported a duration[c]	33
Dufour et al. (2006)	53	Cyanuric acid was measured in pool water and urine samples	Swimming pool, Swimming	6 to ≤ 18	37 (NR)	≥ 45	49
				18+	16 (NR)	≥ 45	21
				All ages (6+)	32 (NR)	≥ 45	43
DeFlorio-Barker et al. (2017)	12 cohorts totaling 68,685	Estimates of amount of water swallowed were self-reported	Freshwater	6 to 10	58.9 (142)[d]	(NR)	–
				11 to 17	55.5 (140) [d]	(NR)	–
				18+	21.9 (46.7)[d]	(NR)	–
			Marine Water	6 to 10	74.4 (180)[d]	(NR)	–
				11 to 17	75.6 (186.7)[d]	(NR)	–
				18+	32.4 (72) [d]	(NR)	–
Dorevitch et al. (2011)	3,367	Estimates of amount of water swallowed were self-reported	Surface water, Canoeing/capsizing	All ages (6+)	6 (19.9)[e]	No duration constraints	–
			Surface water, Kayaking/capsizing	All ages (6+)	5 (16.5)[e]	No duration constraints	–
		Estimates of amount of water swallowed were self-reported; cyanuric acid was measured in urine in a subset of participants	Swimming pool, Swimming	All ages (6+)	10 (34.8)[e]	60	10
			Swimming pool, Canoeing/capsizing	All ages (6+)	6.6 (22.4)[e]	60	6.6
			Swimming pool, Kayaking/capsizing	All ages (6+)	7.9 (7.9)[e]	60	7.9

Reference	Study Sample Size	Measurement Methodology	Water Type, Recreational Activity	Age Group[a] (Years Old)	Mean Ingestion (Maximum Value) (mL/event)	Mean Duration of Event (minutes)	Normalized Ingestion (mL/hour)
Schets et al. (2011)	9,924 (1,924 of which were children)	Descriptive estimates of the amount of water swallowed were self-reported by participants or parents of participants, and estimates were converted to volumes	Freshwater, Swimming	0 to 14	37 (170)[e]	79	28
				15+, males	27 (140)[e]	54	30
				15+, females	18 (86)[e]	54	20
			Marine water, Swimming	0 to 14	31 (140)[e]	65	29
				15+, males	27 (140)[e]	45	36
				15+, females	18 (90)[e]	41	26
			Swimming pool, Swimming	0 to 14	51 (200)[e]	81	38
				15+, males	34 (170)[e]	68	30
				15+, females	23 (110)[e]	67	21
Suppes et al. (2014)	38	Cyanuric acid was measured and total time in water was quantified using videos to adjust ingestion volumes to rates; authors adjusted ingestion volumes to correct for potential false-positive measurements from cyanuric acid carry-over between sample injections	Swimming pool, Swimming (adjusted)	5 to 17	26 (106)	60[f]	26
				18+	4 (51)	60[f]	3.5
				All ages (5+)	14 (106)	60[f]	14
			Swimming pool, Swimming (unadjusted)	5-17	59 (225)	60[f]	59
				18+	9 (NR)	60[f]	9
				All ages (5+)	32 (NR)	60[f]	32
Schijven and de Roda Husman (2006)	517	Descriptive estimates of the amount of water swallowed were self-reported, and estimates were converted to volumes	Freshwater, Recreational diving w/ordinary diving mask	Adults	13 (190)	42 to 52	15 to 19
			Freshwater, Recreational diving w/full face mask	Adults	1.3 (15)	42 to 52	1.5 to 1.9
			Freshwater, Occupational diving	Adults	5.7 (25)	60 to 95	4 to 6
			Marine Water (coastal), Recreational diving w/ordinary diving mask	Adults	9.9 (190)	42 to 52	11 to 14

Reference	Study Sample Size	Measurement Methodology	Water Type, Recreational Activity	Age Group[a] (Years Old)	Mean Ingestion (Maximum Value) (mL/event)	Mean Duration of Event (minutes)	Normalized Ingestion (mL/hour)
			Marine water (coastal), Recreational diving w/full face mask	Adults	1.3 (15)	42 to 52	1.5 to 1.9
			Marine Water (open sea), Recreational diving w/ordinary diving mask	Adults	7.7 (100)	42 to 52	9 to 11
			Marine water (open sea), Recreational diving w/full face mask	Adults	0.43 (2.8)	42 to 52	0.5 to 0.6
			Marine Water (coastal and open sea combined), Recreational diving w/ordinary diving mask	Adults	9.0 (190)	42 to 52	10 to 13
			Marine water (coastal and open sea combined), Occupational diving	Adults	9.8 (100)	60 to 95	6 to 10
			Swimming pool, Recreational diving w/ordinary diving mask	Adults	20 (190)	42 to 52	23 to 29
			Swimming pool, Recreational diving w/full face mask	Adults	13 (190)	42 to 52	15 to 19

[a] Age group ranges reflect the age groupings reported in the study. In some cases the authors did not separate data by different age groups among children or between adults and children.

[b] The values shown are arithmetic means calculated from the Dufour dataset. The Dufour et al. (2017) publication reported ingestion volumes as geometric means for children (6 to 10 years), teens (11 to 15 years), and adults (16 years and over).

[c] Each participant's volume ingested was adjusted to one hour based on the length of time that participant reported being in the water.

[d] No maximum values are reported in the study; 90th provided percentile in parentheses.

[e] No maximum values are reported in the study; upper limit of the CI is provided.

[f] Swimming duration was reported as ≥ 45 minutes, however authors derived and reported only hourly ingestion per event.

APPENDIX G. INFORMATION ON CELLULAR CYANOTOXIN AMOUNTS

The information in the tables in this appendix was generated from a brief survey of the peer-reviewed and published scientific literature. This survey was not a formal systematic literature search and was conducted to evaluate the availability of data needed to calculate a cyanobacterial cell density potentially associated with a specific cyanotoxin concentration.

The information in Tables G-3 and G-4 was generated from both a brief survey and a standardized search of the peer-reviewed and published scientific literature. The purpose of these searches was to evaluate the availability of cyanotoxin quota data (i.e., cyanotoxin content per cyanobacterial cell or per unit biomass, for microcystins and cylindrospermopsin) needed to calculate a cyanobacterial cell density potentially associated with a specific cyanotoxin concentration.

The EPA conducted a brief initial survey of the available peer-reviewed and published scientific literature in December 2016 and identified 29 studies with data on cellular toxin amounts. After reviewing the available data, a formal literature search was conducted. The purpose of this literature search and screening was to identify literature relevant to answering the following research question: What cyanotoxin cell quota data (i.e., cyanotoxin content per cyanobacterial cell or per unit biomass, for microcystins and cylindrospermopsin) are available in the peer-reviewed literature?

Search terms were identified with support from a subject matter expert and library science professionals and included genera of known microcystins or cylindrospermopsin producers, names of the toxins of interest, and keywords that could indicate that quota data were reported. The search was conducted in PubMed and results were limited to articles published in English from 1987 to March 2017. A summary of the literature search results is provided in Table G-1.

Table G-1. Summary of Cyanotoxin Cell Quota Data Literature Search Results

Database	Results	Notes/Limits
PubMed	253	1987 to present; English
Web of Science	472	1987 to present; English
Total Unique	**485**	

The EPA developed search strategies for each database. Both search strategies included the same set of keywords but varied in how these keywords were strung together. The Web of Science search strategy also included limits, a feature not characteristic of a search strategy conducted using PubMed. The search strategies are provided below.

PubMed

Date of Search: 3/01/2017
Date Limit: 1987 to present
Language = English

Set	PubMed Search Strategy
1	(Anabaena[tiab] OR Anabaena[mh] OR Anabaena-flos-aquae[tiab] OR Anabaenopsis[tiab] OR Aphanizomenon[tiab] OR Aphanizomenon[mh] OR C.-raciborskii[tiab] OR Chrysosporum-ovalisporum[tiab] OR Cuspidothrix[tiab] OR Cylindrospermopsis[tiab] OR Cylindrospermopsis[mh] OR Cylindrospermopsis-raciborskii[tiab] OR Cylindrospermum[tiab] OR Dolichospermum[tiab] OR Fischerella[tiab] OR Gloeotrichia[tiab] OR Lyngbya[tiab] OR

Set	PubMed Search Strategy
	M.-aeruginosa[tiab] OR Microcystis[tiab] OR Microcystis[mh] OR Microcystis-aeruginosa[tiab] OR Nostoc[tiab] OR Nostoc[mh] OR Oscillatoria[tiab] OR Oscillatoria[mh] OR Phormidium[tiab] OR Planktothrix[tiab] OR Sphaerospermopsis[tiab] OR Synechococcus[tiab] OR Synechococcus[mh])
2	AND (microcystin[tiab] OR microcystins[tiab] OR microcystins[mh] OR cylindrospermopsin[tiab] OR cylindrospermopsin[Supplementary Concept])
3	AND (quota[tiab] OR cell-content[tiab] OR cellular-concentration[tiab] OR cyanotoxin-content[tiab] OR intracellular-content[tiab] OR intracellular-concentration[tiab] OR toxin-content[tiab] OR microcystin-content[tiab] OR microcystin-LR-content[tiab] OR MC-content[tiab] OR MCYST-content[tiab] OR MC-LR-content[tiab] OR intracellular-microcystin[tiab] OR intracellular-MC[tiab] OR microcystin-production[tiab] OR microcystin-LR-production[tiab] OR microcystins-production[tiab] OR MC-production[tiab] OR MCYST-production[tiab] OR MC-LR-production[tiab] OR CYN-content[tiab] OR particulate-CYN[tiab] OR cylindrospermopsin-production[tiab])

Web of Science
Date of Search: 3/01/2017
Date Limit: 1987 to present
Language = English
All terms searched in Topic (Title, Abstract, and Keywords)

Set	Web of Science Search Strategy
1	(Anabaena OR Anabaena-flos-aquae OR Anabaenopsis OR Aphanizomenon OR C.-raciborskii OR Chrysosporum-ovalisporum OR Cuspidothrix OR Cylindrospermopsis OR Cylindrospermopsis-raciborskii OR Cylindrospermum OR Dolichospermum OR Fischerella OR Gloeotrichia OR Lyngbya OR M.-aeruginosa OR Microcystis OR Microcystis-aeruginosa OR Nostoc OR Oscillatoria OR Phormidium OR Planktothrix OR Sphaerospermopsis OR Synechococcus)
2	AND (microcystin OR microcystins OR cylindrospermopsin)
3	AND (microcystin-RR-content OR MC-RR-content OR particulate-microcystin OR particulate-MC OR cylindrospermopsin-content OR intracellular-CYN OR quota OR cell-content OR cellular-concentration OR cyanotoxin-content OR intracellular-content OR intracellular-concentration OR toxin-content OR microcystin-content OR microcystin-LR-content OR MC-content OR MCYST-content OR MC-LR-content OR intracellular-microcystin OR intracellular-MC OR microcystin-production OR microcystin-LR-production OR microcystins-production OR MC-production OR MCYST-production OR MC-LR-production OR CYN-content OR particulate-CYN OR cylindrospermopsin-production)
Limits	AND Research Areas: (AGRICULTURE OR OCEANOGRAPHY OR ENVIRONMENTAL SCIENCES ECOLOGY OR PHARMACOLOGY PHARMACY OR EVOLUTIONARY BIOLOGY OR BIOCHEMISTRY MOLECULAR BIOLOGY OR FISHERIES OR PLANT SCIENCES OR BIODIVERSITY CONSERVATION OR PUBLIC ENVIRONMENTAL OCCUPATIONAL HEALTH OR RESEARCH EXPERIMENTAL MEDICINE OR BIOTECHNOLOGY APPLIED MICROBIOLOGY OR SCIENCE TECHNOLOGY OTHER TOPICS OR CELL BIOLOGY OR CHEMISTRY OR LIFE SCIENCES BIOMEDICINE OTHER TOPICS OR TOXICOLOGY OR MARINE FRESHWATER BIOLOGY OR WATER RESOURCES OR METEOROLOGY ATMOSPHERIC SCIENCES OR ZOOLOGY OR MICROBIOLOGY)

The EPA conducted title and abstract screening of the 253 search results (generated from both database searches) and classified them as "relevant," "maybe relevant," or "not relevant." Titles were considered "relevant" if the title or abstract included mention of cell quota data for microcystins or

cylindrospermopsin or if the title or abstract indicated that the study had quantitative information on cyanobacterial cell density and microcystins or cylindrospermopsin concentration and therefore may contain sufficient data to calculate a quota. Titles were considered "maybe relevant" if the title or abstract indicated the article might have information relevant to the research question. Title and abstract did not specifically include the term "quota" but indicated that it may have had quantitative information on cyanobacterial cell density and microcystins or cylindrospermopsin concentration or if cyanobacterial cells were only quantified by molecular methods such as PCR and toxin concentrations were measured. Titles were considered "not relevant" if the title/abstract did not appear to have information about microcystins or cylindrospermopsin quotas or densities/concentrations, if the study was a spiked cyanotoxin experiment (meaning cyanotoxins were added, not produced by cyanobacteria present), or if the study was not a peer-reviewed article, book, or government document.

The EPA prioritized the studies to facilitate the review. Prioritization yielded a high number of studies classified as "relevant" or "maybe relevant." Relevant studies were further prioritized for each cyanotoxin of interest based on date of publication. The approach for prioritization is presented in Table G-2. A full text review was conducted on Priority 1 studies only.

Table G-2. Summary of Study Prioritization

Toxin	Priority 1 Classification Criteria	Priority 2 Classification Criteria	Priority 3 Classification Criteria
Microcystins	Classified as relevant based on title/abstract screening; Did not use PCR quantification or evaluate benthic cyanobacteria; Identified predominant species without statistical analysis; Published in last 5 years; and Field study or study with both field and laboratory component.	Studies that use only PCR for quantification of cyanobacteria; and All laboratory studies (internal or external forcing, mitigation studies, studies evaluating non-nutrient pollutants).	Methods studies; and Studies on benthic cyanobacteria.
Cylindrospermopsin	Classified as relevant based on title/abstract screening; Did not use PCR quantification or evaluate benthic cyanobacteria; Identified predominant species without statistical analysis; Published in last 10 years; and Field study or laboratory study.	Studies that use only PCR for quantification of cyanobacteria.	Methods studies; and Studies on benthic cyanobacteria.

Extracted data from studies meeting the criteria for "Priority 1" are presented below in Table G-3 and are further summarized in Table G-4. Relevant quota data were extracted from both the text and figures in "Priority 1" studies. All figures were digitized using GraphPad Digitizer software, as appropriate. All extracted data from text and figures underwent primary and secondary review for quality assurance purposes.

The EPA's primary interest when reviewing the data was to identify the amount of toxin per cyanobacterial cell when toxin was present in a sample. In the environment, it is possible for cyanobacterial cells to be present with no toxin being produced (e.g., the cyanobacteria are a non-toxin-producing strain or environmental conditions do not support toxin production). The EPA only included quota data where toxin was detected.

The studies included in Table G-3 vary in methods used, conditions evaluated, and presentation of data. Typically, complete, raw data were unavailable. The EPA made choices regarding selection, presentation, aggregation, and conversion of data to develop the necessary standardization required for comparing and analyzing these data. Specifically, if quota values were from the same sample at a single location, the average and range were recorded; results from different sampling locations were recorded separately; and multiple mean quota values within the same study were recorded separately (note that separate mean values could be reported for different sampling sites or species within the same genera).

The EPA found that study authors report toxin quota data in various forms, including but not limited to toxin mass per cyanobacterial cell, toxin mass per cyanobacterial biomass, and toxin mass per cyanobacterial biovolume. Scientific measurement units vary among studies. The EPA presents the cell quota data in Table G-3 in the units reported by the study authors (i.e., without conversion to standard units). However, when possible, the EPA converted data to a standard set of units, picograms (pg) per cell, in Table G-4 so that data could be summarized and compared. The EPA did not identify appropriate conversion factors that would allow genus-specific conversion of quotas described in mass per biovolume to mass per cell or mass per biomass to mass per cell. The EPA considered converting biovolume quotas using methods cited in the Australian national guidelines (Australian Government National Health and Medical Research Council, 2008) and Ackaalan (2006), but ultimately decided that the number of uncertainties associated with these methods were too great. Thus, data with unique units are summarized separately in Table G-3, Table G-4, and Table 7-14.

Within Table G-3 and Table G-4, the EPA categorized studies as either "field" or "lab." Field studies include studies where environmental samples were collected and analyzed for cell quota data without additional manipulation of growing conditions. In some studies, environmental samples were taken to a laboratory where growing conditions were optimized or manipulated to determine cyanotoxin cell quota. These studies were categorized as laboratory studies. Other laboratory studies analyzed cell quota in laboratory strains that were not collected in the environment for the purpose of the analysis. For laboratory studies, only control data were extracted. In laboratory studies where there was no true control the conditions closest to ambient conditions were selected (e.g., multiple conditions were tested and none was the clear control, all data were included).

While the traditional definition of toxin quota refers to the intracellular amount of toxin, some studies presented the total toxin present normalized by the cell density or the extracellular toxin normalized by cell density as a quota. In other cases, methods for calculation of the quota were not very clear. If a quota value was presented (i.e., intracellular toxin per cell) this was recorded. If this value was not available or was not clearly described, was recorded as presented by the study authors and assumed to be intracellular or the total amount of toxin per cell. Extracellular toxin per cell was not recorded. The EPA recognizes that the exclusion of extracellular toxin data could lead to an underestimation of the amount of toxin per cell, in particular for cylindrospermopsin as *Cylindrospermopsis* has been shown to constitutively produce the toxin, which can stay inside the cells during log phase growth and accumulate externally upon entering the stationary phase (Davis et al. 2014; Burford et al. 2016). Researchers have also demonstrated that cylindrospermopsin production can be excreted in response to phosphorus limitation and induce other cells to excrete alkaline phosphatase to the water body resulting in a phosphorus scavenging effect (Bar-Yosef et al. 2010).

Some field studies identified the presence of cyanotoxins and multiple cyanobacterial genera including more than one potential toxin producer with no clear predominant toxin-producing species. Table G-3 only includes cell quota values from field studies where there was a clear predominant toxin-producing genera. In these instances, the study was grouped with the predominant toxin-producing genera. In mixed samples with multiple cyanobacteria and no predominant toxin-producing species, quota data were not included. The EPA recognizes that this approach presents a possible limitation to conclusions on toxin quota as studies conducted under non-bloom conditions were excluded. Predominant species are easier to identify when there is a bloom, however, traditional microscopic identification of cyanobacteria does not distinguish between toxigenic and non-toxigenic strains. The proportion of toxigenic cells within a cyanobacterial community and the copy number of the *mcyD* gene per cell can vary significantly, both affected by environmental parameters (Davis et al. 2009).

Table G-3 includes cell quota data for microcystin and cylindrospermopsin-producing genera. For each study, data are provided, where available, on the genus and species of the cyanobacteria, the site where the sample was collected or the clone used to estimate cellular toxin for, the type of study (i.e., field or laboratory), and the reported toxin quota data. Notes relevant to each study are reported in the final column of the table, when appropriate.

Relevant toxin data include the mean toxin quota per cell, the median toxin quota per cell, the minimum toxin quota per cell, or the maximum toxin quota per cell. These data are reported where available and not all data points were reported in each study. Data are presented using the units of measure reported by the study authors.

Table G-3. Cell Quota Data for Microcystin and Cylindrospermopsin-Producing Genera

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
Microcystin	*Microcystis* spp.	Grangent Reservoir, France	Field	Mean: 0.576 pg/cell Min: 0.042 pg/cell Max: 4.19 pg/cell	Sabart et al. (2013)	Data digitized from Figure 6b; The authors report cell quotas for different size ranges of *Microcystis aeruginosa* cells and these values represent the minimum and maximum for all sizes; Mean calculated using all cell quota data reported at all time points for all sizes; Study provides highest reported mean for *Microcystis* spp. mass per cell, field and field and lab combined
	Microcystis spp.	Lake Victoria, Kenya	Field	Mean: 17 fg/cell Median: 553 fg/cell	Sitoki et al. (2012)	Sixteen *Microcystis* strains identified
	Microcystis spp.	Lake Taihu, China	Field	Mean: 0.015 pg/cell Min: 0 pg/cell Max: 0.159 pg/cell	Wang et al. (2013)	Data digitized from Figure 4a,b; Mean calculated using all cell quota data reported at all time points for all colony sizes; Study provides minimum cell quota value and lowest reported mean for *Microcystis* spp. mass per cell, field, and field and lab combined
	Microcystis spp.	Dapugang River, Lake Taihu, China	Field	Cell quota data not presented	Xue et al. (2016)	
	Microcystis spp.	Umia River, Galicia, Spain	Field	Max: 570 µg/g biomass	Alvarez et al. (2016)	Mixed bloom: *Microcystis aeruginosa*, *Scenedesmus* spp., *Kirchneriella* spp.; unclear which is predominant
	Microcystis spp.	Lake Taihu, China	Field	Mean: 640.59 µg/g biomass	Wei et al. (2016)	Data digitized from Figure 4a,b; Only microcystin-L-R

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
				Min: 13.21 µg/g biomass Max: 1389.13 µg/g biomass		congener reported; Mean calculated using cell quota data for all time points; Study provides mean, minimum, and maximum cell quota values for *Microcystis* spp. mass per biomass, field
	Microcystis spp.	FACHB-905	Lab	Mean: 20.25 fg/cell Min: 17.05 fg/cell Max: 28.47 fg/cell	Wei et al. (2016)	Data digitized from Figure 1D and Figure 2D; Mean calculated using cell quota data for all time points
	Microcystis aeruginosa	Lake Huron, United States	Field	Mean: 140 fg/cell Mix: 10 fg/cell Max: 350 fg/cell	Fahnenstiel et al. (2008)	Study provides highest reported mean, maximum, and minimum cell quota values for *Microcystis aeruginosa* mass per cell, field, and field and lab combined
	Microcystis aeruginosa	Aguieira reservoir, Portugal	Field	Mean: 0.12 fg/cell Mix: 0.07 fg/cell Max: 0.22 fg/cell	Vasconcelos et al. (2011)	Data digitized from Figure 5; *Microcystis aeruginosa* was dominant microcystins producer; Mean calculated using all cell quota data for all yearly time points
	Microcystis aeruginosa	Lake Erie, United States	Field	Mean: 3.34 µg/mg biomass Min: 1.37 µg/mg biomass	Horst et al. (2014)	Data digitized from Figure 3 and Figure 6; Study provides mean and maximum cell quota value for *Microcystis aeruginosa* mass per biomass, field
	Microcystis aeruginosa	Hartbeespoort Dam, South Africa	Field	Min: 0.14 µg/g biomass Max: 268 µg/g biomass	Mbukwa and Mamba (2012)	Study provides minimum cell quota value for *Microcystis aeruginosa* mass per biomass, field
	Microcystis aeruginosa	BCCUSP232	Lab	Mean: 18.84 fg/cell Min: 15.07 fg/cell	Chia et al. (2016)	Data digitized from Figure 4b; Study provides lowest reported mean for

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
				Max: 22.61 fg/cell		*Microcystis aeruginosa* mass per cell, lab, field and lab combined and the minimum cell quota value for mass per cell, lab
	Microcystis aeruginosa	Model was used to simulate cyanobacteria	Lab	Mean: 91.5 fg/cell	Jähnichen et al. (2001)	Model used cell quota data reported by Long et al. (2001), Orr and Jones (1998), Jahnichen et al. (2001), and Watanabe et al. (1989); Study provides highest reported mean for *Microcystis aeruginosa* mass per cell, lab
	Microcystis aeruginosa	Model was used to simulate cyanobacteria	Lab	Min: 18 fg/cell Max: 23.7 fg/cell	Jähnichen et al. (2007)	Microcystins cell quota data reported in the presence of sodium and potassium, respectively; Study provides minimum cell quota value for *Microcystis aeruginosa* mass per cell, lab
	Microcystis aeruginosa	MASH01 non-axenic	Lab	Mean: 84.7 fg/cell Min: 41.53 fg/cell Max: 165.89 fg/cell	Orr and Jones (1998)	Data digitized from Figure 5; Mean calculated using quota data presented for each treatment
	Microcystis aeruginosa	MASH01-A19	Lab	Mean: 93.92 fg/cell Min: 46.58 fg/cell Max: 138.47 fg/cell	Orr and Jones (1998)	Data digitized from Figure 5; Mean calculated using quota data presented for each treatment; Study provides highest reported mean and maximum cell quota value for *Microcystis aeruginosa* mass per cell, lab
	Microcystis aeruginosa	PCC 7806	Lab	Min: 34.5 fg/cell Max: 81.4 fg/cell	Wiedner et al. (2003)	Mean quota value not reported, however data could be digitized from Figure 1B to calculate a mean

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
	Microcystis aeruginosa	Lake Rotura, New Zealand	Lab	Mean: 0.064 pg/cell Min: 0.017 pg/cell Max: 0.134 pg/cell	Wood et al. (2012)	Data digitized from Figure 1B; Mean calculated using cell quota data from all time points; Study provides minimum cell quota value for *Microcystis aeruginosa* mass per cell, lab
	Microcystis aeruginosa	Ontario, Canada	Lab	Min: 40.3 fg/cell Max: 62.4 fg/cell	Pineda-Mendoza et al. (2014)	The range of quota data presented was assumed to be the minimum and maximum values
	Microcystis aeruginosa	New Mexico, United States	Lab	Min: 34.5 fg/cell Max: 136.3 fg/cell	Pineda-Mendoza et al. (2014)	The range of quota data presented was assumed to be the minimum and maximum values
	Microcystis aeruginosa	Umia River, Galicia, Spain	Lab	Mean: 11 µg/g biomass	Alvarez et al. (2016)	Study provides lowest mean and minimum cell quota value for *Microcystis* spp. mass per cell, field, and field and lab combined
	Microcystis aeruginosa	Dayet Afourgah lake, Morocco	Lab	Max: 688.4 µg/g biomass	Douma et al. (2017)	Maximum reported as total microcystins content
	Microcystis aeruginosa	Aguelmam Azigza lake, Morocco	Lab	Max: 699 µg/g biomass	Douma et al. (2017)	Maximum reported as total microcystins content
	Microcystis aeruginosa	Aguelmam Azigza lake, Morocco	Lab	Max: 859.6 µg/g biomass	Douma et al. (2017)	Maximum reported as total microcystins content
	Microcystis aeruginosa	Lake Erie, United States	Lab	Mean: 2.44 µg/mg biomass	Horst et al. (2014)	Data digitized from Figure 5; Study provides highest mean and maximum cell quota value for *Microcystis aeruginosa* mass per biomass, lab
	Microcystis aeruginosa, M. flos-aquae, M. novacekii	Cogotas, Spain	Field	Min: 1.2 pg/cell Max: 4.3 pg/cell	Cires et al. (2013)	Data digitized from Figure 1; Study provides maximum cell quota value for *Microcystis* spp. Mass per

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
						cell, field, and field and lab combined
	Microcystis aeruginosa, M. flos-aquae, M. novacekii	Valmayor, Spain	Field	Min: 3.4 pg/cell Max: 4.1 pg/cell	Cires et al. (2013)	Data digitized from Figure 1; Study provides maximum cell quota value for *Microcystis* spp. mass per cell, field, and field and lab combined
	Microcystis aeruginosa, M. flos-aquae, M. viridis, M. wesenbergii	Lake Taihu, China	Field	Mean: 0.027 pg/cell Min: 0.001 pg/cell Max: 0.087 pg/cell	Tao et al. (2012)	Data digitized from Figure 2c; Mean calculated using all cell quota data for all time points
	Fisherella	NQAIF311 from Queensland, Australia	Lab	Max: 43 µg/g biomass	Cires et al. (2014)	Data digitized from Figure 1
	Geitlerinema	Florida, United States	Field	Min: 0.02 µg/g biomass Max: 0.10 µg/g biomass	Gantar et al. (2009)	
	Geitlerinema	Florida, United States	Lab	Mean: 0.40 µg/g biomass Min: 0.15 µg/g biomass Max: 0.30 µg/g biomass	Gantar et al. (2009)	
	Leptolyngbya	Florida, United States	Field	Min: 0 µg/g biomass Max: 0.08 µg/g biomass	Gantar et al. (2009)	
	Leptolyngbya	FLK BBD1; Florida, United States	Lab	Mean: 0.10 µg/g biomass Min: 0.06 µg/g biomass Max: 0.20 µg/g biomass	Gantar et al. (2009)	
	Phormidium	Florida, United States	Field	Mean: 0.026 µg/g biomass	Gantar et al. (2009)	

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
	Planktothrix spp.	Occhito, Italy	Field	Median: 3.82 µg/mm^3 biovolume Min: 1.27 µg/mm^3 biovolume Max: 6.28 µg/mm^3 biovolume	Salmaso et al. (2014)	Data on minimum and maximum digitized from Figure 4a
	Planktothrix spp.	Pusiano, Italy	Field	Median: 0.59 µg/mm^3 biovolume Min: 0.37 µg/mm^3 biovolume Max: 0.87 µg/mm^3 biovolume	Salmaso et al. (2014)	Data on minimum and maximum digitized from Figure 4a
	Planktothrix spp.	Ledro, Italy	Field	Mean: 0.45 µg/mm^3 biovolume Min: 0.12 µg/mm^3 biovolume Max: 0.84 µg/mm^3 biovolume	Salmaso et al. (2014)	Data on minimum and maximum digitized from Figure 4a
	Planktothrix spp.	Garda, Italy	Field	Mean: 0.31 µg/mm^3 biovolume Min: 0 µg/mm^3 biovolume Max: 0.32 µg/mm^3 biovolume	Salmaso et al. (2014)	Data on minimum and maximum digitized from Figure 4a
	Planktothrix agardhii	Bassenwaithe Lake, England	Field	Mean: 91.2 fg/cell	Akcaalan et al. (2006)	
	Planktothrix agardhii	NIES 595	Lab	Mean: 75.6 fg/cell	Akcaalan et al. (2006)	
	Planktothrix rubescencs	Iznik Lake, Turkey	Field	Mean: 235.6 fg/cell	Akcaalan et al. (2006)	Study provides mean and maximum cell quota value for *Planktothrix rubescencs* mass per cell, field
	Planktothrix rubescencs	France	Field	Min: 0.13 pg/cell Max: 0.16 pg/cell	Briand et al. (2008)	Study provides maximum cell quota value for

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
						Planktothrix rubescencs mass per cell, field, and lab and the minimum cell quota value for mass per cell, field
	Planktothrix rubescencs	SL 03; Turkey	Lab	Mean: 103.9 fg/cell	Akcaalan et al. (2006)	Study provides lowest mean and minimum cell quota value for *Planktothrix rubescencs* mass per cell, lab, and field and lab combined
	Planktothrix rubescencs	Sapanca Lake, Turkey	Lab	Mean: 108.2 fg/cell	Akcaalan et al. (2006)	
	Pseudanabaena	Florida, United States	Field	Min: 0.02 µg/g biomass Max: 0.04 µg/g biomass	Gantar et al. (2009)	
	Spirulina	Florida, United States	Field	Mean: 0.12 µg/g biomass	Gantar et al. (2009)	
	Synechococcus	Florida, United States	Field	Min: 0.08 µg/g biomass Max: 0.27 µg/g biomass	Gantar et al. (2009)	
	Multiple genera including *Microcystis aeruginosa, Anabaenopsis*	Kiwah Island pond, South Carolina	Field		Greenfield et al. (2014)	Data available but were not digitized
	Multiple genera including *Microcystis* spp., *Anabaena* spp., and *Planktolyngbya* spp.	Lake Victoria, Tanzania	Field		Mbonde and Kurmayer (2015)	Data available but were not digitized
	Microcystis, Aphanomenizon, and others	Quebec lakes, Canada	Field		Monchamp et al. (2014)	Data available but were not digitized

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
	Multiple genera including *Microcystis* and *Anabaena*	Anzali wetland, Iran	Field		Rezaitabar et al. (2017)	Data available but were not digitized
	Multiple genera including *Microcystis* and *Anabaena*	Anzali wetland, Iran	Field		Rezaitabar et al. (2017)	Data available but were not digitized
	Multiple genera including *Microcystis, Dolichospermum,* others	Lake Chaohu, China	Field		Shang et al. (2015)	Data available but were not digitized
Cylindrospermopsin	*Aphanizomenon ovalisporum*	Florida, United States	Lab	Min: 7.39 µg/mg biomass Max: 9.33 µg/mg biomass	Yilmaz et al. (2008)	
	Cylindrospermopsis raciborskii[d]	Gazam Dam Lake, Saudi Arabia	Field	Min: 0.6 pg/cell Max: 14.6 pg/cell	Mohamed and Al-Shehri (2013)	Study provides maximum value for *Cylindrospermopsis raciborskii* mass per cell, field, and field and lab combined
	Cylindrospermopsis raciborskii	Queensland, Australia	Field	Mean: 23.12 fg/cell Median: 20.5 fg/cell Min: 5.9 fg/cell Max: 55.8 fg/cell	Orr et al. (2010)	Study provides minimum value for *Cylindrospermopsis raciborskii* mass per cell, field, and field and lab combined
	Cylindrospermopsis raciborskii	Queensland, Australia	Field	Median: 20.3 fg/cell Min: 10 fg/cell Max: 49.4 fg/cell	Orr et al. (2010)	
	Cylindrospermopsis raciborskii	CYP 030A; Australia	Lab	Min: 3.2 ng/10^6 cell Max: 5.7 ng/10^6 cell	Carneiro et al. (2013)	
	Cylindrospermopsis raciborskii	CYP 011K; Australia	Lab	Min: 12.1 ng/10^6 cell	Carneiro et al. (2013)	

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
				Max: 24.7 ng/10^6 cell		
	Cylindrospermopsis raciborskii	Queensland, Australia	Lab	Min: 13.4 fg/cell Max: 14.9 fg/cell	Davis et al. (2014)	
	Cylindrospermopsis raciborskii	New South Wales, Australia	Lab	Mean: 31 fg/cell Min: 12 fg/cell Max: 52 fg/cell	Hawkins et al. (2001)	
	Cylindrospermopsis raciborskii	Queensland, Australia	Lab	Min: 19 fg/cell Max: 26 fg/cell	Pierangelini et al. (2015)	
	Cylindrospermopsis raciborskii	CS-506; Queensland, Australia	Lab	Mean: 0.0028 pg/cell	Willis et al. (2015)	Study provides lowest mean value for *Cylindrospermopsis raciborskii* mass per cell, lab, and field and lab combined and minimum cell quota value for mass per cell, lab
	Cylindrospermopsis raciborskii	CS-506; Queensland, Australia	Lab	Mean: 0.018 pg/cell	Willis et al. (2015)	
	Cylindrospermopsis raciborskii	Lake Wivenhoe, Australia	Lab	Mean: 165.75 fg/cell	Willis et al. (2016)	Calculated mean based on data in Table 1; Study provides highest mean value for *Cylindrospermopsis raciborskii* mass per cell, lab, and field and lab combined and maximum cell quota value for mass per cell, lab
	Cylindrospermopsis raciborskii	CHAB3438, China	Lab	Mean: 43.76 fg/cell Min: 35.89 fg/cell Max: 52 fg/cell	Yang et al. (2016)	Data digitized from Figure 2; Mean calculated using quota data presented for each time point
	Cylindrospermopsis raciborskii	Queensland, Australia	Lab	Min: 416 fg/μm^3 biovolume Max: 447 fg/μm^3 biovolume	Pierangelini et al. (2015)	

Toxin	Genus/Species[a]	Site/Clone	Study Type[b]	Toxin Quota Data[c]	Reference	Notes
	Multiple genera including *Aphanizomenon*, *Anabaena*, *Nostocales*, and *Cylindrospermopsis*	Germany	Field		Rücker et al. (2007)	Data available but were not digitized
	Multiple genera including *Aphanizomenon*	Langer See, Germany	Field		Wiedner et al. (2008)	Data available but were not digitized

Abbreviations: *M.* = *Microcystis*; spp. = multiple species in the genus

[a] Both the genus and species are reported where available. In some studies, the genus was reported but the species was not reported. In other studies, multiple species were analyzed within a specific genus but the specific species were not identified. In both instances, studies were categorized as the genus name (e.g., *Microcystis*) spp. Separately, in some studies multiple genera were considered. In these studies, available toxin quota data were not digitized as they could not be used for comparison purposes. Only information about the studies are presented in this table with a note that data are available but were not digitized.

[b] Studies were conducted in two different settings: the field (i.e., environmental) or a laboratory. In some instances, field samples were subjected to optimized growth conditions in the laboratory. These studies were classified as laboratory; not as field studies.

[c] Toxin cell quota data were not converted and are reported in the measurement units used by the study authors. Significant figures were not normalized among the data points.

[d] The genus *Cylindrospermopsis* has been renamed to *Raphidiopsis*.

Table G-4 provides the first step in summarizing and grouping cell quota data for microcystin and cylindrospermopsin-producing genera. Studies presented in Table G-3 were grouped by genus and species when possible. Studies that looked at more than one species within a specific genus or that did not specify which species were considered within that genus were placed in a single group (e.g., *Microcystis* spp., *Planktothrix* spp.). Within each genus/species group, studies were further grouped based on their study type and the quantification method used in that study. For each study type and quantification method group, data were aggregated on the mean, minimum, and maximum cell quota values presented in each study included in that group. In Table G-4, the range of the means, arithmetic mean (of the means), median of the means, minimum cell quota value, and maximum cell quota value are reported for the studies included in that group. Note that studies were not identified in the literature search for all quantification methods and study types for all genus/species groups. The EPA converted data to a standard set of units, pg per cell, when possible. No other conversions were attempted. Additional information about the approach used to summarize the available cell quota data is provided in the footnotes accompanying the table.

Table G-4. Cell Quota Appendix Summary Data for Microcystin and Cylindrospermopsin-producing Genera

Toxin	Genus, Species	Quantification Method[a]; Study Type[b]	Range of Means[c,d]	Mean[c,d]	Median of Means[c,e]	Minimum; Maximum[c,f]	References
Microcystin	*Microcystis* spp.	Mass per cell; Field and lab	0.015–0.576 pg/cell	0.13 pg/cell	0.017 pg/cell	0 pg/cell; 4.30 pg/cell	Sitoki et al. (2012); Tao et al. (2012); Cires et al. (2013); Sabart et al. (2013); Wang et al. (2013); Wei et al. (2016)
		Mass per cell; Field	0.015–0.576 pg/cell	0.16 pg/cell	0.022 pg/cell	0 pg/cell; 4.30 pg/cell	Sitoki et al. (2012); Tao et al. (2012); Cires et al. (2013); Sabart et al. (2013); Wang et al. (2013)
		Mass per cell; Lab	0.020 pg/cell	0.02 pg/cell	N/A	0.017 pg/cell; 0.028 pg/cell	Wei et al. (2016)
		Mass per biomass; Field	640.59 µg/g biomass	640.59 µg/g biomass	N/A	13.21 µg/g; 1389.13 µg/g biomass	Alvarez et al. (2016); Wei et al. (2016)
	Microcystis aeruginosa	Mass per cell; Field and lab	0.02–0.14 pg/cell	0.09 pg/cell	0.09 pg/cell	0.01 pg/cell; 0.35 pg/cell	Orr and Jones (1998); Jähnichen et al. (2001); Wiedner et al. (2003); Jähnichen et al. (2007); Fahnenstiel et al. (2008); Vasconcelos et al. (2011); Wood et al. (2012); Pineda-Mendoza et al. (2014); Chia et al. (2016)
		Mass per cell; Field	0.12–0.14 pg/cell	0.13 pg/cell	0.13 pg/cell	0.01 pg/cell; 0.35 pg/cell	Fahnenstiel et al. (2008); Vasconcelos et al. (2011)
		Mass per cell; Lab	0.02–0.09 pg/cell	0.07 pg/cell	0.08 pg/cell	0.02 pg/cell; 0.17 pg/cell	Orr and Jones (1998); Jähnichen et

Toxin	Genus, Species	Quantification Method[a]; Study Type[b]	Range of Means[c,d]	Mean[c,d]	Median of Means[c,e]	Minimum; Maximum[c,f]	References
							al. (2001); Wiedner et al. (2003); Jähnichen et al. (2007); Wood et al. (2012); Pineda-Mendoza et al. (2014); Chia et al. (2016)
		Mass per biomass; Field	3,340 µg/g biomass	3,340 µg/g biomass	N/A	0.14 µg/g biomass; 3,340 µg/g biomass	Mbukwa and Mamba (2012); Horst et al. (2014)
		Mass per biomass; Lab	11–2,440 µg/g biomass	1225.5 µg/g biomass	1225.5 µg/g biomass	11 µg/g; 2,440 µg/g biomass	Horst et al. (2014); Alvarez et al. (2016); Douma et al. (2017)
	Fisherella	Mass per biomass; Lab	N/A	N/A	N/A	43 µg/g biomass	Gantar et al. (2009)
	Geitlerinema	Mass per biomass; Field	N/A	N/A	N/A	0.02 µg/g; 0.10 µg/g biomass	Gantar et al. (2009)
		Mass per biomass; Lab	0.40 µg/g biomass	0.40 µg/g biomass	N/A	0.15 µg/g; 0.40 µg/g biomass	Gantar et al. (2009)
	Leptolyngbya	Mass per biomass; Field	N/A	N/A	N/A	0 µg/g; 0.08 µg/g biomass	Gantar et al. (2009)
		Mass per biomass; Lab	0.10 µg/g biomass	0.10 µg/g biomass	N/A	0.06 µg/g; 0.20 µg/g biomass	Gantar et al. (2009)
	Phormidium	Mass per biomass; Lab	0.026 µg/g biomass	0.026 µg/g biomass	N/A	0.026 µg/g biomass	Gantar et al. (2009)
	Planktothrix spp.	Mass per biovolume; Field	N/A	N/A	N/A	0 µg/mm^3; 6.28 µg/mm^3 biomass	Salmaso et al. (2014)

Toxin	Genus, Species	Quantification Method[a]; Study Type[b]	Range of Means[c,d]	Mean[c,d]	Median of Means[c,e]	Minimum; Maximum[c,f]	References
	Planktothrix agardhii	Mass per cell; Field and lab	0.076–0.091 pg/cell	0.083 pg/cell	0.083 pg/cell	0.076 pg/cell; 0.091 pg/cell	Akcaalan et al. (2006)
		Mass per cell; Field	0.091 pg/cell	0.091 pg/cell	N/A	0.091 pg/cell	Akcaalan et al. (2006)
		Mass per cell; Lab	0.076 pg/cell	0.076 pg/cell	N/A	0.076 pg/cell	Akcaalan et al. (2006)
	Planktothrix rubescencs	Mass per cell; Field and lab	0.104–0.236 pg/cell	0.149 pg/cell	.108 pg/cell	0.104 pg/cell; 0.16 pg/cell	Akcaalan et al. (2006); Briand et al. (2008)
		Mass per cell; Field	0.236 pg/cell	0.236 pg/cell	N/A	0.13 pg/cell; 0.236 pg/cell	Akcaalan et al. (2006); Briand et al. (2008)
		Mass per cell; Lab	0.104–0.108 pg/cell	0.106 pg/cell	0.106 pg/cell	0.104 pg/cell; 0.108 pg/cell	Akcaalan et al. (2006)
	Pseudanabaena	Mass per biomass; Field	N/A	N/A	N/A	0.02 µg/g; 0.04 µg/g biomass	Gantar et al. (2009)
	Spirulina	Mass per biomass; Field	0.12 µg/g biomass	0.12 µg/g biomass	N/A	0.12 µg/g biomass	Gantar et al. (2009)
	Synechococcus	Mass per biomass; Field	N/A	N/A	N/A	0.08 µg/g; 0.27 µg/g biomass	Gantar et al. (2009)
Cylindrospermopsin	*Aphanizomenon ovalisporum*	Mass per biomass; Field and lab	N/A	N/A	N/A	7.39 µg/g ; 9.33 µg/mg biomass	Yilmaz et al. (2008)
	Cylindrospermopsis raciborskii[g]	Mass per cell; Field and lab	0.0028–0.17 pg/cell	0.05 pg/cell	0.03 pg/cell	0.006 pg/cell; 14.6 pg/cell	Orr et al. (2010); Mohamed and Al-Shehri (2013); Pierangelini et al. (2015); Willis et al. (2015); Yang et al. (2016a)

Toxin	Genus, Species	Quantification Method[a]; Study Type[b]	Range of Means[c,d]	Mean[c,d]	Median of Means[c,e]	Minimum; Maximum[c,f]	References
		Mass per cell; Field	0.023 pg/cell	0.023 pg/cell	N/A	0.006 pg/cell; 14.6 pg/cell	Orr et al. (2010); Mohamed and Al-Shehri (2013)
		Mass per cell; Lab	0.0028–0.17 pg/cell	0.057 pg/cell	0.031	0.0028 pg/cell; 0.17 pg/cell	Hawkins et al. (2001); Carneiro et al. (2013); Davis et al. (2014); Pierangelini et al. (2015); Willis et al. (2015); Willis et al. (2016); Yang et al. (2016)
		Mass per biovolume; Lab	N/A	N/A	N/A	416 fg/μm^3; 447 fg/μm^3	Pierangelini et al. (2015)

Acronyms and Abbreviations: fg = femtogram; pg = picogram; µg = microgram; N/A = not applicable.

[a] Various methods were used to quantify toxin quotas and quota values were presented in different forms, including toxin mass per cyanobacterial cell and toxin mass per cyanobacterial biomass.

[b] Studies were conducted in two different settings: the field (i.e., environmental) or a laboratory. In some instances, field samples were subjected to optimized growth conditions in the laboratory. These studies were classified as laboratory; not field.

[c] Study authors reported data using multiple measurement units. When possible, the EPA converted data to the standard units of pg per cell. The EPA did not identify appropriate conversion factors that would allow genus-specific conversion of quotas described in mass per biomass to mass per cell.

[d] Shows single reported mean if only one study was available or average of reported means.

[e] Median of means not calculated if only one mean value was available or if only minimum and/or maximum cell quota values were available.

[f] If reported toxin quota means from one study were the lowest or highest toxin quotas reported within a genus, then these values were listed as the minimum or maximum values, respectively, to better reflect the range of toxin quota values.

[g] The genus *Cylindrospermopsis* has recently been renamed to *Raphidiopsis*.

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APPENDIX H. TABLES OF STATE-ISSUED GUIDELINES SPECIFIC TO ANIMAL CYANOTOXIN POISONING

H.1 California

Table H-1. California Environmental Protection Agency (2012) Action levels for Selected Pet and Livestock Scenarios

	Microcystins[a]	Cylindrospermopsin	Media (units)
Subchronic water intake, dog[b]	2	10	water (µg/L)
Subchronic crust and mat intake, dog	0.01	0.04	crusts and mats (mg/kg dw)[c]
Acute water intake, dog[d]	100	200	water (µg/L)
Acute crust and mat intake, dog	0.5	0.5	crusts and mats (mg/kg dw)[c]
Subchronic water intake, cattle[e]	0.9	5	water (µg/L)
Subchronic crust and mat intake, cattle[e]	0.1	0.4	crusts and mats (mg/kg dw)[c]
Acute water intake, cattle[e]	50	60	water (µg/L)
Acute crust and mat intake, cattle[e]	5	5	crusts and mats (mg/kg dw)[c]

[a] Microcystins LA, LR, RR, and YR all had the same RfD so the action levels are the same.
[b] Subchronic refers to exposures over multiple days.
[c] Based on sample dry weight (dw).
[d] Acute refers to exposures in a single day.
[e] Based on small breed dairy cows because their potential exposure to cyanotoxins is greatest.

Table H-2. California Environmental Protection Agency (2012) Reference Doses and Acute and Subchronic Action Levels for Canine Exposure to Cyanotoxins in Drinking Water

	Microcystin	Cylindrospermopsin
Water consumption L/kg-day	0.085	0.085
Uncertainty factor (unitless)	3	3
Acute RfD[a] mg/kg/day	0.037	0.04
Acute action level µg/L	100	200
Subchronic RfD mg/kg/day	0.00064	0.0033
Subchronic action level µg/L	2	10

Reference:

Butler N, Carlisle J, Kaley KB, and Linville R (2012). Toxicological Summary and Suggested Action Levels to Reduce Potential Adverse Health Effects of Six Cyanotoxins. http://www.waterboards.ca.gov/water_issues/programs/peer_review/docs/calif_cyanotoxins/cyanotoxins053112.pdf. Last Accessed: 11/27/2018.

H.2 Indiana

Indiana has adopted guidance for cyanotoxins for dog exposures:

"A warning to dog owners using the Fort Harrison State Park Dog Park Lake will occur whenever any cyanotoxins are detected, and dogs will be prohibited from swimming at the values of 0.8 µg/L microcystin, any anatoxin-a detection, and 1.0 µg/L of cylindrospermopsin."

Reference:

Indiana Department of Environmental Management (2018). Blue-Green Algae: Indiana Reservoir and Lake Update.http://www.in.gov/idem/algae/. Last Accessed: 02/27/2018.

H.3 Oregon

Table H-3. Oregon Dog-specific Guideline Values for Cyanotoxins in Recreational Waters (µg/L)

	Microcystin	Cylindrospermopsin
Dog Guidance Value	0.2	0.4

Note: All dog-specific guideline values have been changed in this revision because California EPA's estimate of the amount of water an exercising dog consumes per kilogram body weight was updated in 2012 (from 0.168 to 0.255 L/kg-day). Current dog-specific guideline values are now consistent with the California EPA update. The dog-specific value for saxitoxins was further modified by application of an uncertainty factor to the dog-specific TDI for interspecies differences in sensitivity between humans (the species in the critical study) and dogs.

Reference:

Oregon Health Authority (2018). Oregon Harmful Algae Bloom Surveillance (HABS) Program Public Health Advisory Guidelines: Harmful Algae Blooms in Freshwater Bodies. https://www.oregon.gov/oha/ph/HealthyEnvironments/Recreation/HarmfulAlgaeBlooms/Documents/HABPublicHealthAdvisoryGuidelines.pdf.

H.4 Grayson County, Texas

Table H-4. Grayson County, Texas Microcystin Guidelines for Dogs

Quantity of Lake Water Ingested to Receive a Potentially Lethal Dose of Microcystin, Assuming that Mouse and Dog Toxic Responses are Equivalent

	Gallons of Water	Pounds of Water
10-pound dog	2.70	22.50
80-pound dog	21.57	180.00

Quantity of Lake Water Ingested to Receive a Potentially Lethal Dose of Microcystin, Assuming that Mouse and Dog Toxic Responses are Equivalent (at actual concentrations found in Grand Lake, Oklahoma, in June 2011) Highest measured microcystin concentration was 358 ppb.

	Gallons of Water	Pounds of Water
10-pound dog	0.15 (19.3 ounces)	1.26
80-pound dog	1.21	10.06

*This is not including additional dose amounts that could be ingested from a dog self-grooming algae scum off its fur.
**LD_{50} for microcystin-mouse used in calculations = 45 µg/kg
***20 ppb microcystin is algal toxin threshold for BGA Warning (condition red)

Quantity of Lake Water Ingested to Receive a Potentially Lethal Dose of Cylindrospermopsin, Assuming that Mouse and Dog Toxic Responses are Equivalent 20 ppb Cylindrospermopsin in Lake Water

	Gallons of Water	Pounds of Water
10-pound dog	263	2,200
80-pound dog	2,109	17,601

*This is not including additional dose amounts that could be ingested from a dog self-grooming algae scum off its fur.
**LD_{50} for cylindrospermopsin-mouse used in calculations = 4400 µg/kg
***20 ppb cylindrospermopsin is algal toxin threshold for BGA Warning (condition red)

Reference:

Lillis J, Ortez A, and Teel JH (2012). *Blue-Green Algae Response Strategy*. Sherman, Texas. http://www.co.grayson.tx.us/upload/page/0206/docs/Blue-Green_Algae_Response_Strategy.pdf. Last Accessed: 12/5/2018.

United States Environmental Protection Agency

Office of Water
EPA **823-D-19-002**
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Draft Technical Support Document:

Implementing the 2019 Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin

Notice: This technical support document contains several questions and answers relating to the Environmental Protection Agency's (EPA) *Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin*.[1] Microcystins and cylindrospermopsin are two types of cyanotoxins that are produced from photosynthetic bacteria, called cyanobacteria. Exposure to the microcystins and cylindrospermopsin can pose health risks to humans and animals. Symptoms reported after recreational exposure to cyanobacterial blooms included skin irritations, gastrointestinal illnesses, hepatomegaly and kidney damage.

The information in this document is primarily intended to support states, authorized tribes, and territories (collectively referred to as "states and authorized tribes") interested in adopting the recommended criteria into their state or tribal water quality standards (WQS) or using the recommended values as the basis for swimming advisories and related public notification purposes.[2] EPA envisions that if states or authorized tribes decide to use the values for issuing swimming advisories they might do so in a manner similar to their current recreational water advisory programs.

Chemical and physical factors in a waterbody can play a role in the composition and production of cyanotoxins associated with a toxigenic cyanobacterial bloom (such factors include, but are not limited to, levels of nitrogen and phosphorus, the availability of organic matter, turbidity, turbulence or flushing of a waterbody, light attenuation, water temperature and pH). The recommended criteria and swimming advisory values were derived based on data related to exposure factors, including the rate of incidental consumption of water while swimming and the duration of time spent recreating in water, and are recommended for waters designated for primary contact recreation. Some states and authorized tribes may experience other hazards associated with cyanobacterial blooms that are not addressed in this document (e.g., with respect to cytokines or other cyanotoxins), therefore a more holistic approach to monitoring waterbodies may be necessary to ensure public health protection.

Exposure to cyanobacteria and their toxins can also occur through non-recreational pathways such as consumption of cyanotoxin-contaminated drinking water and food (including fish), and during bathing or showering. This document does not address or provide recommendations for non-recreational exposures. Although there are no national level recommended criteria for cyanotoxins in public water supply designated uses, in 2015, EPA published health advisories for the cyanotoxins, microcystins and cylindrospermopsin in finished drinking water. For information related to the drinking water health advisories for microcystins and cylindrospermopsin, see https://www.epa.gov/ground-water-and-drinking-water/drinking-water-health-advisory-documents-cyanobacterial-toxins.

This document provides background information on the factors that contribute to cyanobacterial growth and toxin production; recommendations on how to monitor for cyanobacteria and two of their known cyanotoxins (microcystins and cylindrospermopsin) in waterbodies; and information on how to complete assessments, list impaired or threatened waters, and develop Total Maximum Daily Load (TMDLs), based

[1] USEPA (U.S. Environmental Protection Agency). 2019. *Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin 2019.* EPA 822-R-18-004. U.S. Environmental Protection Agency, Office of Water, Washington, DC. https://www.epa.gov/wqc/recommended-human-health-recreational-ambient-water-quality-criteria-or-swimming-advisories.

[2] When a bloom or the presence of cyanotoxins is confirmed, the recreational waterbody manager usually issues a public notification (either a swimming advisory or a closure of swimming areas) to raise awareness of the potential risks associated with contact with the cyanobacterial bloom or its toxins in recreational waters. Swimming advisories are recommendations to limit swimming or other recreational water-contact activities, due to an increased health risk from contact with or ingestion of the cyanobacteria or cyanotoxins; whereas, a closure notification or posting typically means that the waterbody is officially closed to the public. Decisions to post an advisory or closure for a waterbody or a beach are local decisions. Information about communicating swimming advisories or closures to the public, including sample templates, is available at Communicating about Cyanobacterial Blooms and Toxins in Recreational Waters (webpage).

on WQS that adhere to EPA's 2019 recommended criteria for total microcystins and cylindrospermopsin. While this document cites statutes and regulations that contain requirements applicable to these programs, it does not impose legally binding requirements on EPA, states, authorized tribes, other regulatory authorities, or the regulated community. EPA, states, authorized tribes and other decision makers may adopt approaches on a case-by-case basis that differ from those provided in this document, as appropriate and consistent with statutory and regulatory requirements. Also, EPA may update this document as new scientific and technical information becomes available. In addition to this document, EPA has prepared the following information to support states and authorized tribes in their efforts to monitor and respond to cyanobacterial blooms and cyanotoxins in recreational waters:

- Cyanobacterial Harmful Algal Blooms in Water (website): https://www.epa.gov/cyanohabs
- Monitoring and Responding to Cyanobacteria and Cyanotoxins in Recreational Waters
- Recommendations for Cyanobacteria and Cyanotoxin Monitoring in Recreational Waters

Table of Contents

General Questions about Cyanotoxin Water Quality Criteria

1. What are EPA's recommended recreational water quality criteria for total microcystins and cylindrospermopsin?

In 2019, EPA issued its recreational water quality criteria recommendations for the cyanotoxins, microcystins and cylindrospermopsin, reflecting the latest scientific knowledge, public comments, and external scientific peer review. The criteria are designed to protect the public from incidental exposure to harmful levels of these cyanotoxins while participating in water-contact activities where immersion and incidental ingestion of water are likely. Such activities include but are not limited to swimming, water skiing, tubing, skin diving, water play by children, or similar water-contact activities in all waterbodies designated for such recreational uses.[3] EPA issues such recommendations under the statutory authority of the Clean Water Act (CWA), Section 304(a).

EPA's recommended recreational water quality criteria for total microcystins and cylindrospermopsin consist of three components:

- *Magnitude* — The numeric expression of the maximum amount of a pollutant that may be present in a waterbody while also protecting the designated use(s) of the waterbody. In this case, the magnitude represents the concentration of total microcystins (8 µg/L) or cylindrospermopsin (15 µg/L) in the water column that is not expected to result in adverse human health effects from short-term recreational exposure to the cyanotoxins via incidental ingestion while recreating (e.g., swimming), based on children's exposure parameters.
- *Duration* — Duration is the period of time over which the magnitude is calculated. Exposure to recreational waters containing microcystins or cylindrospermopsin at or below the recommended magnitude concentrations over the short-term ten-day duration would not be expected to result in the adverse health effects discussed in health effects assessment (Section 5) of the criteria document.
- *Frequency* of excursion — The number of times the contaminant may be present above the recommended criteria concentration (magnitude) over the specified period (duration). For these criteria, the frequency corresponds to how often (within a single recreational season, *and across* multiple recreational seasons) the concentration of total microcystins or cylindrospermopsin in a waterbody may exceed the magnitude concentration and be protective of the designated use. The recommended frequency within a single recreational season is no more than three excursions. The number of years that a pattern of more than three excursions can occur *across* recreational seasons and still deemed to be protective of the designated use—referred to in Question 3 as the *recurrence frequency*—is a risk management decision, to be made by a state or authorized tribe. If a state or authorized tribe chooses to adopt the recommended criteria into its state or tribal WQS, EPA recommends that the state or authorized tribe include this recurrence frequency number in its WQS (see Table 1, note b).

EPA developed the following criteria values for total microcystins and cylindrospermopsin concentrations for states and authorized tribes to consider as the basis for human health protection in primary contact recreational waters (see Table 1 for a summary of the magnitude, duration and frequency).

[3] For information about the scope and applicability of recreational criteria, see EPA's 2012 *Recreational Water Quality Criteria*, (Section 2.0 Applicability and Scope). Office of Water 820-F-12-058. https://www.epa.gov/sites/production/files/2015-10/documents/rwqc2012.pdf

Table 1. Recreational Water Quality Criteria Recommendations for Microcystins and Cylindrospermopsin [a]

Total Microcystins Magnitude (µg/L)	Cylindrospermopsin Magnitude (µg/L)	Duration	Frequency
8	15	1 in 10-day assessment period across a recreational season	Not more than 3 excursions in a recreational season in more than one year [b]

[a] States and authorized tribes can choose to adopt one or both criteria recommendations.

[b] An excursion is defined as a 10-day assessment period with any toxin concentration higher than the criteria magnitude. When more than three excursions occur within a recreational season and that pattern reoccurs in more than one year, it is an indication the water quality has been or is becoming degraded and is not supporting its recreational use. As a risk management decision, states and authorized tribes should include in their WQS an upper-bound frequency stating the number of years that pattern can reoccur and still support its recreational use.

Cyanobacterial blooms can occur naturally but can be an uncommon event due to a convergence of climatic and other environmental factors that result in a single short-term bloom lasting days or a couple of weeks. In some cases, multiple blooms can occur in a single year. Alternatively, longer-term cyanobacterial blooms can occur regularly in some waters lasting for a few weeks, months, or possibly all year. Cyanobacterial blooms can occur while conditions conducive to cyanobacterial proliferation exist and limit the use of the waterbody for primary recreation. The criteria components listed in Table 1, above, can help to identify a short- or long-term temporal trend or a spatial distribution pattern of cyanotoxin excursions that can be used to evaluate a waterbody.

EPA recognizes there are multiple environmental factors that can cause variability in bloom formation and toxin production, and that some years may produce HABs that occur for long periods, or HABs of shorter duration that occur repeatedly throughout a single recreational season, but such events may not occur every year. Therefore, EPA concludes that it is appropriate to consider a pattern of multiple excursions within a recreational season as well as a pattern of excursions in multiple years (i.e., more than one year) when determining whether the use is attained. It is important to note that the years with multiple excursions do not have to be consecutive to indicate a water-quality problem. The upper bound frequency (e.g., one year out of three years) is a risk management decision that states and authorized tribes need to determine when developing their WQS. States and authorized tribes should include the number of years a pattern of cyanotoxin excursions can occur for the recreational use to be supported.

EPA recommends that if toxin concentrations are higher than the criterion magnitude in a sample collected during a ten-day assessment period, then that period should be considered an excursion from the recreational criteria. A short-term cyanobacterial bloom that does not reoccur can result in a small number of excursions of the criteria but is not expected to result in impairment of the recreational use. Such cyanobacterial blooms may result from conditions that occur naturally (e.g., as a result of unusually hot conditions), but not frequently. EPA recommends that when more than three excursions (an exceedance during the ten-day assessment period) occur within a recreational season *and* that pattern reoccurs in more than one year, it is an indication that the water quality is or may be becoming degraded such that the waterbody may no longer support the recreational use. Multiple excursions over a single recreation season indicates that a waterbody may not support a recreational use for a significant proportion of the season. EPA recommends that a concentration of 8 µg/L total microcystins or 15 µg/L cylindrospermopsin not be exceeded in more than three ten-day assessment periods over the course of a recreation season in more than one year. EPA does not recommend averaging sampling data to determine an excursion, because averaging does not give a clear picture of the pattern of cyanobacterial bloom formation and cyanotoxin exposure to the population using the waterbody for recreational purposes.

The number of years over which time an observed pattern (of three or more ten-day excursions during a recreational season) can occur across recreational seasons is a risk management decision that states and

authorized tribes should consider when developing their criteria. States and authorized tribes should include the number of years a pattern of cyanotoxin excursions can occur for the use to be supported in their WQS. Furthermore, states and authorized tribes may make different risk management decisions for different types of waterbodies. EPA recommends states and authorized tribes clarify these differences in their WQS. For more information about how the criteria were developed, see: https://www.epa.gov/wqc/recommended-human-health-recreational-ambient-water-quality-criteria-or-swimming-advisories.

2. How can EPA's recommended recreational water quality criteria values for total microcystins and cylindrospermopsin be used for swimming advisories?

State, tribal or local governments can use swimming advisories to provide information to recreators on their potential exposure to cyanobacteria and their toxins. EPA envisions that if states or authorized tribes decide to use the values as swimming advisory values, they can manage a cyanotoxin monitoring and advisory program in the same way as they manage any already existing recreational water advisory programs (i.e., those for *E. coli* or enterococci). States and authorized tribes may choose to apply either or both recommended magnitude values as the basis for swimming advisories (i.e., public notifications) for recreational waterbodies. For this purpose, EPA recommends that the magnitude (i.e., 8 µg/L total microcystins or 15 µg/L cylindrospermopsin) not be exceeded on any given day.

Table 2. Swimming Advisory Recommendations for Microcystins and Cylindrospermopsin[a]

Total Microcystins Magnitude (µg/L)	Cylindrospermopsin Magnitude (µg/L)	Duration	Frequency
8	15	One day	Not to be exceeded

EPA recognizes that some states and authorized tribes may handle swimming advisories through their health departments and not through their environmental quality departments. As a result, inter-departmental coordination may be helpful to implement an advisory program which can also serve to inform drinking water providers and water quality practitioners. EPA has provided an example Cyanobacteria Bloom Response Contact List on its website to help state or tribal employees consider who to contact in the event of a cyanobacterial bloom.

Swimming advisories can be used to provide information to recreators on their potential exposure to cyanobacteria and their toxins. Some local and state governments currently post notifications for swimmers, in the form of advisories or warnings, when a cyanobacterial bloom is reported in recreational waters or when cyanotoxin levels exceed swimming advisory thresholds.

If using the recommended values for public notification purposes (swimming advisory or beach closure), EPA recommends that the relevant authority (typically a local health department official or environmental protection agency) notify the public whenever a sample exceeds the recommended criteria concentration value. EPA also recommends that an exceedance of the recommended criteria concentration result in a swimming advisory being issued until subsequent sampling results show that the toxin concentration has fallen below the recommended magnitude value of the criteria. By increasing the monitoring frequency at a site where a swimming advisory is issued, recreational waterbody managers will get a clearer understanding of the temporal and spatial nature of toxins that can be useful in making management decisions to protect the recreational use, including when to remove an advisory.

EPA also recognizes that environmental conditions may change rapidly from one sample period to the next, depending on the frequency of samples taken. For an example of how the state of Ohio uses toxin concentration results to make swimming advisory decisions, see the Example 1 text box below. Appendix B of the criteria document summarizes available information on state recreational water guidelines for

cyanotoxins and cyanobacteria (see: https://www.epa.gov/wqc/recommended-human-health-recreational-ambient-water-quality-criteria-or-swimming-advisories.

Example 1: State of Ohio example of a tiered approach to public notifications

The Ohio Environmental Protection Agency collaborated with the Department of Natural Resources and the Department of Health to develop a tiered notification system which takes different actions based on different numeric thresholds for cyanotoxin concentrations in recreational waters. That is, the state takes various actions—such as posting information about harmful algal blooms (HABs), issuing a recreational public health advisory, or temporarily closing recreational waters through a no contact advisory—depending on the severity of the bloom event.

Toxin of concern	Informational sign postings about HABs at recreational waters	Recreational public health advisory	Elevated recreational public health advisory (e.g. no contact)
Microcystin-LR	< 6 µg/L	6 µg/L	20 µg/L
Cylindrospermopsin	< 5 µg/L	5 µg/L	20 µg/L

Once an advisory is posted, Ohio conducts a standard sampling protocol for cyanotoxins and lifts the advisory once two consecutive samples taken at least one week apart show cyanotoxins have decreased below the advisory threshold. For more information, see the *State of Ohio Harmful Algal Bloom Response Strategy for Recreational Waters* (Ohio EPA, 2016).

EPA has published materials for recreational waterbody managers responsible for monitoring and responding to cyanobacterial blooms. These materials include a communication toolbox with examples of public messages, press releases, and warning and posting signage that recreational waterbody managers may use to inform the public of increased health risks associated with exposure to cyanobacteria and their toxins. In addition, EPA has provided recommendations for public health officials or waterbody managers (or relevant state, local or tribal officials) to consider various water monitoring, sampling and testing methods to determine whether a cyanobacterial bloom is producing toxins, whether the bloom presents an increased risk to water-contact recreators and human health, and whether immediate actions should be taken to notify the public if a closure is recommended based on waterbody test results.

- Communicating about Cyanobacterial Blooms and Toxins in Recreational Waters
- Monitoring and Responding to Cyanobacteria and Cyanotoxins in Recreational Waters
- Recommendations for Cyanobacteria and Cyanotoxin Monitoring in Recreational Waters

3. What flexibilities do states and authorized tribes have when they choose to adopt recreational water quality criteria for total microcystins and cylindrospermopsin?

When states or authorized tribes choose to adopt EPA's *Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin*, states and authorized tribes have flexibility to accommodate specific water quality-related circumstances while meeting the requirements of the CWA and the WQS regulation. In addition to considering the Agency's national recommended water quality criteria when revising their WQS, states and authorized tribes may adopt, where appropriate, other scientifically defensible criteria that differ from EPA's recommendations. For example, states and authorized tribes can:

- *Define the length of the recreational season.* States and authorized tribes can adopt seasonal designated uses of a waterbody with respect to various 304(a) recommended criteria, including the recommended criteria for total microcystins and cylindrospermopsin. Therefore, for states

and authorized tribes that have adopted seasonal uses, the recommended cyanotoxin criteria would apply only to the primary contact recreation season. 40 CFR 131.10(f) specifies that states and authorized tribes "may adopt seasonal uses as an alternative to reclassifying a waterbody or segment thereof to uses requiring less stringent water quality criteria. If seasonal uses are adopted, water quality criteria should be adjusted to reflect the seasonal uses, however, such criteria shall not preclude the attainment and maintenance of a more protective use in another season." The length of a "recreational season" is an important consideration because states and authorized tribes would likely monitor the quality of their highest-priority recreational waters throughout the recreational season (for more on prioritizing waterbodies for assessments, see Question 6). For purposes of establishing seasonal WQS, a change to the recreational season constitutes a change to the state's or authorized tribe's designated use in their WQS under 40 CFR 131.10(f) and would need to be reviewed and approved by EPA pursuant to section 303(c) of the CWA. Because local health departments or departments of parks and recreation may define the recreational seasons for inland waterbodies, it is important for states and authorized tribes to coordinate with these local authorities when identifying the length of the state's or tribe's recreational season in WQS.

- *Define a recurrence frequency.* The criteria for total microcystins and cylindrospermopsin recommend that the magnitude values should not be exceeded in *more than three ten-day periods per recreational season in more than one year,* but the criteria do not specify an upper-bound number of years that pattern can occur across recreational seasons (i.e., a recurrence frequency). This provides states and authorized tribes the flexibility to define a recurrence frequency. For example, some states or authorized tribes might count the recurrence frequency over a rolling 5-year period while others choose to count over a rolling 10-year period.

  If a state or authorized tribe chooses to adopt the recommended criteria, EPA expects the state or tribe to also include this recurrence frequency in its WQS. A state or authorized tribe may achieve this with a written statement in its standards. An example written statement could say, *"The concentration of total microcystins shall not exceed 8 µg/L in more than three ten-day periods per recreational season, for more than one recreational season, over a 5-year period."*

- *Derive site-specific criteria elements*. States and authorized tribes may adopt EPA recommended criteria or may modify the criteria to fit their unique situation based on site-specific data and information. For example, a state or tribe may derive site-specific criteria based, in part, on information about the exposure variables among the population that uses the waterbody (e.g., age and incidental ingestion rates) or to allow for specific cultural or land use practices at or near the site. The site-specific criteria must be scientifically defensible and protective of the designated use of the state's or tribe's waterbodies and approved by EPA pursuant to section 303(c) of the CWA.

- *Revise the designated uses of waterbodies within their state*. If a state or authorized tribe finds that attaining one of the designated CWA section 101(a)(2) uses[4] in its standards is not feasible, the state or authorized tribe may remove or modify the designated use based on appropriate use attainability analysis and documentation subject to EPA's review and approval. Therefore, if the recreational criteria for total microcystins or cylindrospermopsin cannot be attained, the state or authorized tribe can consider whether to modify or remove the designated use.

[4] Uses that provide for the protection and propagation of fish, shellfish, and wildlife, and recreation in and on the water, as well as for the protection of human health when consuming fish, shellfish, and other aquatic life (e.g. recreation use, aquatic life use).

4. What is the relationship between cyanobacterial blooms and the recommended criteria?

Cyanobacteria, commonly referred to as blue-green algae, are photosynthetic bacteria that occur naturally in waters, including waters used for primary contact recreation. Under certain conditions, cyanobacteria may grow rapidly to form accumulations known as cyanobacterial blooms. When the proliferation of cells making up the bloom also contains toxin-producing strains of cyanobacteria, there is the potential to have elevated concentrations of the cyanotoxins, microcystins and cylindrospermopsin, present. Generally referred to as a harmful algal blooms (HABs), these toxin-producing cyanobacterial blooms can cause harm to animals, people, or the environment. Accumulations of cyanobacteria have been associated with outbreaks of inflammatory illness (e.g., rashes) unrelated to the production of cyanotoxins.

Some states have chosen to adopt a value based on cell density as a screening tool, with further tests for specific toxins after confirming that a bloom is present. EPA did not develop recommended criteria or swimming advisories based on cyanobacterial cell density; however, the Agency summarized available information on adverse effects that may result from exposure to cyanobacterial cells and estimated a cell density value corresponding to the toxin-based criteria magnitude. For more information on the health effects from exposure to cyanobacterial cells, go to:

- Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin (see section 7.5 Cyanobacterial Cells)

Exposure to the toxins, microcystins and cylindrospermopsin, can pose additional health risks to humans and animals. Symptoms reported after recreational exposure to cyanobacterial blooms (including microcystin-producing genera) included skin irritations, allergic reactions or gastrointestinal illnesses. Symptoms of exposure to cylindrospermopsin have been reported to include fever, headache, vomiting, bloody diarrhea, hepatomegaly and kidney damage with the loss of water, electrolytes and protein.

For more information on the health effects from exposure to microcystins and cylindrospermopsin, go to:

- Health Effects Support Document for the Cyanobacterial Toxin Microcystins
- Health Effects Support Document for the Cyanobacterial Toxin Cylindrospermopsin

The presence of cyanobacteria does not necessarily mean that cyanotoxins are being produced, and conversely, cyanotoxins can be present at levels above the criteria magnitude when accumulations of cyanobacteria were not observed.[5] Benthic cyanobacteria, occurring at the bottom of the waterbody, may not be visible from the surface, but may still produce toxins.

For information on identifying blooms and specific cyanobacteria, go to:

- USGS Field and Laboratory Guide to Freshwater Cyanobacteria Harmful Algal Blooms for Native American and Alaska Native Communities.

[5] Lahti K, Rapala J, Färdig M, Niemelä M, & Sivonen K (1997b). Persistence of cyanobacterial hepatotoxin, microcystin-LR in particulate material and dissolved in lake water. *Water Research*, 31(5), 1005-1012.
Zastepa A, Pick F, & Blais J (2014). Fate and persistence of particulate and dissolved microcystin-LA from Microcystis blooms. *Human and Ecological Risk Assessment: An International Journal, 20*(6), 1670-1686.

5. Why should I consider nutrient pollution if I am implementing criteria for cyanotoxins?

Nitrogen and phosphorus loadings, or nutrient pollution input into recreational waterbodies from agricultural, industrial, and urban sources can provide optimal conditions for cyanobacterial blooms and cyanotoxin production. Preventing nutrient input and reducing nitrogen and phosphorus levels in a waterbody can help to reduce the occurrence of cyanobacterial blooms or the levels of cyanotoxins in blooms containing toxin-producing strains of cyanobacteria.[6]

The relationships between nutrients and other physical, chemical or environmental conditions is complex and can present an added challenge to recreational waterbody managers responsible for monitoring and responding to cyanobacterial blooms. In addition to nutrient concentrations, factors such as the availability of organic matter, turbidity, turbulence or flushing of a waterbody, light attenuation, temperature and pH can play a role in the composition and cyanotoxin production associated with a cyanobacterial bloom.

The sources of nutrients present in waterbodies are both natural and anthropogenic. Soil and erosion of phosphorus-containing rocks are the most significant natural sources of the phosphorus in surface waters, while bed sediment resuspension can be the major source of phosphorus to the lower water column. Significant natural sources of nitrogen include fixation of nitrogen gas, N_2, and dry and wet deposition of nitrogen compounds from the atmosphere. Human-induced nutrient pollution comes from several sources:

1. Stormwater runoff (e.g., urban and rural) contains nitrogen and phosphorus from fertilizers (especially those applied in excess or before a rain storm), yard clippings, leaves, and pet wastes that are washed away to local waterbodies or conveyed through storm sewer systems.
2. Municipal wastewater discharges (including septic systems) process billions of gallons of wastewater every day. Municipal wastewater contains nitrogen and phosphorus from human waste, food, and certain soaps and detergents.
3. Industrial wastewater discharges from industrial facilities, such as food processing and fertilizer manufacturing facilities, are sources of nitrogen and phosphorus.
4. Agricultural practices, including concentrated animal feeding operations and row crops, are sources of nitrogen and phosphorus. Other sources include: animal waste, gaseous nitrogen-based compounds like ammonia and nitrogen oxides released to the atmosphere from ventilated production houses, manure storage structures, or fields where manure (or other fertilizers) is applied and re-deposited onto land and eventually washes into surface waters.

Reductions in nutrient pollution help to reduce the occurrence of cyanobacterial blooms and waterbody eutrophication. Studies have shown that excess nitrogen can stimulate the growth of the cyanobacteria, *Microcystis*.[7] Elevated concentrations of total and dissolved phosphorus and soluble phosphates and nitrates provide optimal conditions for the increased production of microcystins. Controlling phosphorus and nitrogen inputs could reduce the biomass of cyanobacteria in the system, and the frequency and concentration of microcystins produced.

The World Health Organization: Guidelines for Safe Recreational Water Environments (PDF) contains a

[6] For the results of a study on how the experimental limitation of nutrient supplies aided in the diminishing of a cyanobacterial bloom, see:
Pace, M. et al. Reversal of a cyanobacterial bloom in response to early warnings. *Proceedings of the National Academy of Science. USA* 114, 352–357; DOI:10.1073/pnas.1612424114 (2017).

[7] See Recommended Human Health Recreational Ambient Water Quality Criteria or Swimming Advisories for Microcystins and Cylindrospermopsin, Section 3.0 Nature of the Stressors.

chapter on algae and cyanobacteria in fresh water, which includes short- and long-term management options—including nutrient management and implementation of control and abatement technologies, among other practices—with the goal of preventing or reducing the occurrence of cyanobacterial blooms in recreational waters.

As a long-term strategy to address eutrophication and cyanobacteria blooms, states and authorized tribes may consider adopting numeric nutrient criteria into their WQS or, alternatively, develop a numeric target to implement a narrative nutrient criterion that has already been adopted into their WQS. Numeric nutrient criteria or targets are useful as efficient and effective tools to support water quality assessments, watershed protection or restoration, TMDL development and permitting programs, where applicable. Specifically, numeric nutrient criteria or targets sets the foundation for states and authorized tribes to develop environmental baselines, manage water quality more effectively, measure progress, and support broader implementation initiatives (such as water quality trading, best management practices (BMPs), land stewardship, wetlands protection, voluntary collaboration, and stormwater runoff control strategies). EPA stands ready to provide technical assistance to states and authorized tribes who are developing numeric nutrient criteria or targets for different waterbody types through EPA's N-STEPS program.[8]

EPA continues to provide scientific and technical assistance to states and authorized tribes who are working to reduce nutrient pollution as a means of reducing the occurrence of cyanobacterial blooms. For more information on what EPA is doing to help reduce nutrient pollution and for tools to assist states and authorized tribes, refer to:

- Preventing Eutrophication: Scientific Support for Dual Nutrient Criteria Factsheet (PDF)
- Managing Microcystin: Identifying National-Scale Thresholds for Total Nitrogen and Chlorophyll a
- Deriving Nutrient Targets to Prevent Excessive Cyanobacterial Densities in U.S. Lakes and Reservoirs (PDF)
- Renewed Call to Action to Reduce Nutrient Pollution and Support for Incremental Actions to Protect Water Quality and Public Health (September 2016 EPA Memo) (PDF)
- Nutrient Pollution Policy and Data
- Toolkit of Resources to Assist States with Adopting and Implementing Numeric Nutrient Criteria
- Water Quality Standards Handbook
- A Compilation of Cost Data Associated with the Impacts and Control of Nutrient Pollution (PDF)

Implementation Questions about Monitoring, Assessment and Listing

6. What information should states and authorized tribes consider when prioritizing which waterbodies to monitor based on risk of elevated levels of cyanotoxins?

Recognizing the potential risks of exposure to cyanotoxins, states and authorized tribes across the country are monitoring for cyanobacteria (as a potential precursor to blooms), microcystins, cylindrospermopsin, or other algal toxins. Some states and authorized tribes have reported microcystins data through the Water Quality Portal, an interagency website that provides public access to water quality data.[9] Reporting data on cyanobacterial blooms and toxin levels can improve transparency with the public and help to provide a regional or national understanding of where blooms are taking place.

[8] https://www.epa.gov/nutrient-policy-data/n-steps
[9] https://www.waterqualitydata.us/

States and authorized tribes are encouraged to upload microcystins data to the Water Quality Portal (see: https://www.waterqualitydata.us/).

EPA recommends that states and authorized tribes prioritize their waterbodies, for monitoring purposes, based on risk factors relevant to the likelihood of a cyanotoxin or HABs event and its relative impact to recreational users of each waterbody. Prioritizing which waterbodies to monitor can help to direct often limited resources where they will be most effective.

States and authorized tribes may also consider their capacity to include cyanobacterial indicators as part of a robust monitoring and assessment program necessary to evaluate the condition of their waterbodies. Some states and authorized tribes consider existing phytoplankton data or use satellite imagery as a screening tool to identify waters that may need additional sampling.[10] States and authorized tribes may also use field test kits to screen for waters with elevated toxins before collecting a sample. States and authorized tribes are encouraged to collaborate with a range of state, tribal and local agencies, waterbody managers and citizen science/volunteer monitoring organizations to monitor local waterbodies, leveraging the presence of local resources to collect and analyze samples in a timely and appropriate manner for risk management and response activities.

CWA practitioners, public health officials, and other waterbody managers may consider a variety of information when prioritizing which waterbodies to monitor for potential cyanotoxins. Some of the factors listed below may help state, tribal and local officials to identify which waterbodies are most vulnerable:

- the type of waterbody, and types and numbers of recreational users;
- past/historical occurrence of HABs and cyanotoxin production;
- seasonal patterns of cyanobacterial blooms (influenced by temperature and precipitation, among other factors);
- point and nonpoint sources of contamination (especially nutrients) in the waterbody and in the watershed;
- physical and hydrologic factors (e.g., depth, fetch,[11] light attenuation, availability of organic matter, turbidity, pH and nutrients);
- chlorophyll-a and phycocyanin[12] levels (e.g., cell densities);
- other water quality limitations or impairments; and,
- any other information gathered as part of source water assessments or sanitary surveys.

EPA has developed materials to assist recreational waterbody managers interested in monitoring for and responding to cyanobacteria and cyanotoxins in recreational waters. In addition, EPA has provided recommendations for various water sampling and testing methods to determine whether a cyanobacterial bloom is producing cyanotoxins, whether the bloom presents a risk to human health, and whether actions should be taken to notify the public and reduce public health risks from various recreational uses of a waterbody, including if a closure is recommended.

- Monitoring and Responding to Cyanobacteria and Cyanotoxins in Recreational Waters.

[10] For a list of state recreational water guidelines relating to cyanobacteria or cyanotoxins, see Appendix B of the criteria document, https://www.epa.gov/wqc/recommended-human-health-recreational-ambient-water-quality-criteria-or-swimming-advisories.

[11] Fetch refers to the area of a lake surface over which the wind blows in an essentially constant direction, thus generating waves. The term also is used as a synonym for fetch length, which is the horizontal distance over which wave-generating winds blow.

[12] Phycocyanin is a pigment present in cyanobacteria.

- Recommendations for Cyanobacteria and Cyanotoxin Monitoring in Recreational Waters (PDF).

The following additional sources of information may inform states and authorized tribes when taking a risk-based approach to prioritizing recreational waterbodies.

- Cyanobacteria Assessment Network, CyAN: EPA's Office of Research and Development, the National Aeronautics and Space Administration (NASA), the National Oceanic and Atmospheric Administration (NOAA) and United States Geological Survey (USGS) are collaborating to provide interactive maps based on satellite imagery and data on chlorophyll-a levels in large waterbodies nationwide.[13]
- Field and Laboratory Guide to Freshwater Cyanobacteria Harmful Algal Blooms for Native American and Alaska Native Communities: This 2015 guide, produced by the USGS, provides field images to help differentiate between cyanobacterial blooms (some of which produce toxins), non-toxic algal blooms, and floating plants that might be confused with algae.
- Water Quality Portal: The National Water Quality Monitoring Council, USGS and EPA sponsor the Water Quality Portal, an interagency website that provides public access to water quality data collected by over 400 federal, state, tribal, and local agencies.
- Other resources may be found in pages 5 through 7 of EPA's document, Recommendations for Cyanobacteria and Cyanotoxin Monitoring in Recreational Waters (PDF).

7. How frequently and over what time period should states and authorized tribes collect data on cyanotoxin levels?

States and authorized tribes have discretion and flexibility when establishing a monitoring program for cyanotoxins. For example, states and authorized tribes may prioritize which waterbodies they monitor and how frequently they monitor those waterbodies (for information on prioritization, see Question 6). Baseline monitoring could include routine monitoring (e.g., weekly monitoring over the course of a recreation season), responsive/episodic monitoring (e.g., initiating sampling in response to a physical bloom or reported potential negative health impact to animals or humans), or a combination of the two. EPA recommends that states and authorized tribes use their monitoring strategy or annual workplans to identify the monitoring approach that will be implemented for the recreational season, considering available resources and the potential use of tools, such as remote sensing and citizens' volunteer monitoring.

[13] Algorithms to assess bloom frequency and severity have been developed; however, current satellite resolution and sensing capabilities limit the ability to apply the CyAN assessment methods on a national scale. Satellites with the highest resolution, such as Landsat, can capture 62% of lakes greater than 1 ha (2.47 acres) and 95% of the lakes with public drinking water intake locations. Unfortunately, these satellites make less frequent flyovers and are not equipped with sensors capable of distinguishing CyanoHABs from other types of HABs. Lower resolution satellites capture less than 1% of waterbodies and only 33% of drinking water intakes but make more frequent flyovers and can identify CyanoHABs. For more information, see:

Urquhart et al. *A method for examining temporal changes in cyanobacterial harmful algal bloom spatial extent using satellite remote sensing*. Harmful Algae, 67 (2017), pp. 144-152. https://doi.org/10.1016/j.hal.2017.06.001

Clark et al. *Satellite monitoring of cyanobacterial harmful algal bloom frequency in recreational waters and drinking water sources*. Ecological Indicators, 80 (2017), pp. 84-95. https://doi.org/10.1016/j.ecolind.2017.04.046

Stumpf et al. Challenges for mapping cyanotoxin patterns from remote sensing of cyanobacteria. Harmful Algae, 54 (2016), pp. 160-173. https://doi.org/10.1016/j.hal.2016.01.005

Following initial detection and confirmation of microcystins or cylindrospermopsin in waterbodies designated for primary contact recreation, EPA recommends more frequent monitoring until the bloom subsides or the cyanotoxin levels no longer exceed the cyanotoxin criteria magnitude level. Once the cyanotoxins have subsided, EPA suggests that regular monitoring of prioritized waterbodies continue for the duration of the recreation season.

For more information about monitoring for cyanotoxins, see EPA document:

- Recommendations for Cyanobacteria and Cyanotoxin Monitoring in Recreational Waters

8. How should states and authorized tribes analyze and interpret cyanotoxin monitoring data and information to evaluate ambient conditions and recreational use support?

The recommended ten-day duration component of the criteria naturally translates into use of ten-day assessment periods over the course of a recreation season to evaluate ambient waterbody condition and recreational use attainment. EPA recognizes that some states and authorized tribes have routine daily, weekly, biweekly, or monthly sampling schedules in place for different water bodies and recreational areas. Monitoring at other waterbodies may be event driven in response to a bloom. EPA recommends more frequent sampling if a bloom is detected, particularly if it is documented to have toxin levels that exceed the criteria magnitude.

Weekly monitoring may provide two samples within the ten-day assessment period, or weekly monitoring may provide data for two consecutive ten-day assessment periods (depending on how the weeks and ten-day periods align). If one or more samples are collected within the ten-day assessment period that exceed the criteria magnitude, these are counted as a single excursion. Biweekly or monthly sampling schedules can also be considered with respect to ten-day assessment periods, recognizing that some states and authorized tribes may choose to use fixed, *a priori* intervals while others may begin a ten-day period with the first sample that exceeded the criteria magnitude.

States and authorized tribes have flexibility delineating the ten-day assessment periods within their recreational season. The Example 2 text box is provided to illustrate how exceedances can count towards excursions when assessing use attainment. For the application of these criteria, an exceedance is defined as an instance when the monitoring data indicates a concentration of microcystins or cylindrospermopsin that is higher than the recommended magnitude value, and an excursion is defined as an instance of one or more exceedances within a ten-day assessment period that corresponds to the duration. The calendar figures in Example 2 illustrate two ways states and authorized tribes could set up their ten-day assessment periods, either by blocking out ten-day periods before the start of the recreation season (Example 2A) or by marking ten-day assessment periods from the date a bloom is detected and an exceedance is confirmed (Example 2B).

Example 2: Examples of How a State or Tribe Might Count Exceedances and Excursions

The following examples illustrate two hypothetical scenarios describing the evaluation of exceedances and determination of excursions for the ten-day duration period. These scenarios are intended to demonstrate how the number of excursions can be counted within a given recreation season. The red X shown on the example calendars denote days where cyanotoxin monitoring results were above the recommended cyanotoxin magnitude (exceedances). The shaded boxes represent ten-day assessment periods. An assessment period with one or more exceedances is counted as an excursion.

Example 2A: Predetermined Monitoring Schedule

June

Sunday	Monday	Tuesday	Wednesday	Thursday	Friday	Saturday
1	2	3	4	5	6	7
8 X	9 X	10 X	11 X	12	13	14
15	16	17	18	19	20	21 X
22 X	23 X	24	25	26	27 X	28 X
29	30					

Example 2A. In this example, the state or authorized tribe establishes regularly-scheduled monitoring to begin on June 1, and on June 8 monitoring results showed an exceedance of the cyanotoxin magnitude. Subsequent monitoring showed exceedances June 9, 10 and 11, and later in the month on June 21, 22 and 23, and June 27 and 28. In this case, the nine exceedances equate to three excursions because the exceedances were detected across three separate ten-day assessment periods.

Example 2B: Responsive Ten-Day Periods

June

Sunday	Monday	Tuesday	Wednesday	Thursday	Friday	Saturday
1	2	3	4	5	6	7
8 X	9 X	10 X	11 X	12	13	14
15	16	17	18	19	20	21 X
22 X	23 X	24	25	26	27 X	28 X
29	30					

Example 2B. In this example, the state or authorized tribe monitors for cyanotoxins in response to a suspected bloom, once the monitoring data indicates a concentration of microcystins or cylindrospermopsin that is higher than the recommended magnitude. On June 8 a suspected bloom was confirmed and monitoring results showed an exceedance of the cyanotoxin magnitude. Subsequent monitoring showed exceedances June 9, 10 and 11. A second bloom was suspected later in the month and confirmed through additional monitoring showed exceedances on June 21, 22, 23, 27 and 28. In this case, the nine exceedances equate to two excursions because the exceedances were detected across two separate ten-day periods.

9. What should states and authorized tribes consider when selecting an analytical method and sampling locations?

When selecting a method to monitor for cyanotoxins, states and authorized tribes should consider cost and practicality of various monitoring methods, and reliability and comparability of results, among other factors. More than 100 microcystin congeners are known to exist, although the majority of toxicological data on the effects of microcystins are available for microcystin-LR (a frequently monitored congener). Therefore, EPA established its recommended criterion for microcystins based on microcystin-LR and used it as a surrogate for other microcystin congeners. Two congeners of cylindrospermopsin have been identified. Analytical methods must be sufficiently sensitive to detect the cyanotoxins in question at concentrations below the criteria magnitude. Analytical results may vary depending on the analytical methods used.

Methods vary widely in sensitivity, rapidity, cost, and ease of use. As described below, there are both rapid screening tests and laboratory methods used to detect and identify cyanobacterial cells; determine the presence, absence, or count of individual congeners; or measure the concentration of total cyanotoxins in a water sample. In waterbodies that have been prioritized for monitoring, it may be cost effective to use field test kits at several locations throughout the waterbody prior to selecting samples for lab-based analysis (see below for description of field test kits). Some analytical methods measure the sum of all congeners while other methods measure specific congeners of interest. Each method has specific requirements for sample preparation/processing and analytical standards. These should also be considered when planning monitoring programs. EPA does not recommend any single method to monitor

for cyanotoxins and refers readers to the National Environmental Methods Index (https://www.nemi.gov/home/) for information on analytical methods.

Methods for quantifying cyanotoxins (total or individual congener concentrations) include:

- Lab-based enzyme–linked immunosorbent assay (ELISA) methods, to measure cyanotoxins in question. The ELISA method is typically run with only a microcystin-LR standard for comparison but can quantify a broad range of microcystin congeners (especially if using an ADDA-based antibody) and other compounds that are similar to microcystin congeners (e.g., nodularin).
- Field test kits (e.g., Abraxis test strip, Envirologix QualiTube). These field-based methods do not require laboratory instrumentation and can produce semi-quantitative results within about an hour; however, their relatively high limit of quantification (approximately 1-10 µg/L) may better suit these methods for screening purposes.
- Protein phosphatase inhibition assay (PPIA). This method has varying degrees of specificity depending on its substrate composition and may react to compounds in the sample, other than microcystins.
- High performance liquid chromatography (HPLC) combined with ultraviolet/photodiode array detectors (UV/PDA). This method requires known toxin standards to be run alongside the water sample(s) to quantify the toxin concentration(s) and its results are limited to only those congeners for which standards are available and analyzed. LC-UV methods are based on a non-selective detector and co-eluting interferents can prevent accurate identification of components and quantitation. It is less sensitive than mass spectrometry methods (see below).
- Liquid chromatography/tandem mass spectrometry (LC/MS/MS).[14] Like HPLC-PDA, this method typically requires known toxin standards to be run alongside the water sample(s) to quantify the toxin concentration(s) and its results are limited to only those congeners for which standards are available and analyzed. It is, however, the most precise method for quantitation of analytes (such as specific microcystin variants) if standards are available. May require use of solid phase extraction for analytes with weak product ion abundance (microcystins). The LC-MS/MS MMPB (2-methyl-3(methoxy)-4-phenylbutyic acid) method, however, analyzes the chemically cleaved ADDA group common to all microcystin variants and provides an alternative LC-based approach for analyzing total microcystins. The MMPB method may also detect microcystins break-down products and could potentially overestimate total microcystins concentrations in some settings.

EPA recognizes that several states or authorized tribes may monitor for cyanobacterial cell densities in addition to, or in lieu of, monitoring for cyanotoxins. Two methods for quantifying cyanobacterial cells include microscopy and quantitative polymerase chain reaction (qPCR) and microarrays/DNA chips.

Like other aspects of monitoring programs, decisions on sample location are guided by the management questions being addressed. For example, when monitoring recreational waters for public health protection, it's reasonable to target sample collection toward capturing the highest potential exposure risks; therefore, EPA recommends that states and authorized tribes collect single grab samples from designated swimming areas, near the shoreline, or composite samples from points within the splash zones where children play. Monitoring for other objectives may result in sampling at other areas of a waterbody (e.g., if sampling for source water protection, rather than the protection of recreational

[14] For more information on single laboratory validated methods for detecting cyanotoxins in freshwaters by LC/MS/MS, see (1) https://www.epa.gov/water-research/single-laboratory-validated-method-determination-cylindrospermopsin-and-anatoxin to detect cylindrospermopsin and anatoxin-a, and (2) https://www.epa.gov/water-research/single-laboratory-validated-method-determination-microcystins-and-nodularin-ambient to detect microsystins and nodularin (combined intracellular and extracellular).

uses).

EPA has produced technical materials to aid in the development of cyanobacteria and cyanotoxin monitoring programs, including information on available testing methods and sampling logistics. See materials at:

- For recommendations related to establishing a sampling and monitoring program, see: https://www.epa.gov/cyanohabs/recommendations-cyanobacteria-and-cyanotoxin-monitoring-recreational-waters;
- For a summary of methods used to detect cyanobacteria and cyanotoxins in water, see: Determination of Cyanotoxins in Drinking and Ambient Freshwaters; and,
- For a set of frequently asked questions related to laboratory analysis for microcystins in drinking water, see: Frequently Asked Questions: Laboratory Analysis for Microcystins in Drinking Water.

10. What data and information should states and authorized tribes assemble and evaluate to complete CWA water quality assessments with EPA's recommended cyanotoxin criteria?

States and authorized tribes are required under 40 CFR 130.7(b)(5) to assemble and evaluate all existing and readily available water quality-related data and information when determining which waterbodies belong on the state's or authorized tribe's CWA section 303(d) list of impaired and threatened waters. For states and authorized tribes that have adopted the recommended recreational criteria for total microcystins and cylindrospermopsin and are interested in assessing against the criteria, "readily available data and information" would include observed concentration levels for microcystins and cylindrospermopsin collected by the state, authorized tribe, or other stakeholders (e.g., citizen science water monitoring groups).

In addition to observed concentration levels of microcystins and cylindrospermopsin, EPA considers advisories to be existing and readily available data and information. Hence, for states and authorized tribes that implement the recommended swimming advisory levels but do not adopt the recommended criteria, advisories can be used to support water quality assessments using other applicable WQS (e.g., designated uses and narrative criteria). EPA recommends that states and authorized tribes include within their assessment methodologies information about how the state or tribe will use advisories to evaluate attainment of narrative water quality criteria and associated designated uses.

11. Should states and authorized tribes update their assessment methodology to include the evaluation of cyanotoxin data and information? If so, what should be considered in these updates?

States and authorized tribes should consider updating their water quality assessment methodology to address any water quality standard adopted by the state or authorized tribe. An assessment methodology constitutes the decision process that a state or authorized tribe employs to determine the water quality attainment status of waters in the state or on tribal lands. Under 40 CFR 130.7(b)(6), states and authorized tribes are required to provide documentation to EPA to support their determination to include or not include waters on its impaired and threatened waters list. States and authorized tribes are required to include a description of the methodology used to develop the list; a description of the data and information used to identify waters for the list, including a description of the data and information used by the jurisdiction; and, a rational for any decisions to not use existing and readily available data and information to develop the list.

EPA encourages states and authorized tribes to make the assessment methodology available to the public for review and comment prior to, or along with, solicitations for data and information. Such engagement helps facilitate stakeholder input to the state's or authorized tribe's assessment of water quality status, including recreational use assessments. If states and authorized tribes choose to adopt the recommended cyanotoxin recreational criteria, EPA recommends that they update their assessment methodologies to account for any criteria-specific considerations. For states and authorized tribes that use the swimming advisory recommendation, EPA also encourages them to describe in their assessment methodology how, if at all, the advisory information will be used to support water quality assessments using other applicable WQS (e.g., designated uses and narrative criteria).

12. What happens if a state or authorized tribe does not have sufficient data to make an assessment determination?

EPA recommends that states and authorized tribes submit an integrated water quality report that uses a five-category approach for classifying the WQS attainment status for each waterbody in their jurisdiction. Consistent with this approach, EPA recommends that waterbody segments be placed in waterbody assessment Category 3 of the Integrated Report when there is insufficient available data and/or information to make a use-support determination. Information on the appropriate use of Category 3 can be found in EPA's 2009 memo to assist states and authorized tribes in the preparation of the 2010 Integrated Water Quality Reports.[15]

13. What factors should be considered in defining waterbody segmentation (e.g., if dividing a waterbody into smaller assessment units for advisories and 303(d) listings)?

States and authorized tribes have flexibility to define the segmentation for waterbodies within their jurisdiction. Information on segmenting waters can be found in EPA's 2005 memo to assist states and authorized tribes in the preparation of the 2006 Integrated Water Quality Reports.[16]

14. How should states and authorized tribes approach waterbody assessments for a waterbody that is already on the CWA section 303(d) list?

Consistent with any applicable water quality standard, states and authorized tribes make future assessment decisions based on an evaluation of existing and readily available water quality-related data and information against the water quality standard and accompanying assessment method. The assessment decision informs whether a waterbody should be identified as impaired or threatened on the jurisdiction's Section 303(d) list.

States and authorized tribes can decide not to include a waterbody/pollutant combination that was previously identified as impaired or threatened on a state's or tribe's 303(d) list (also known as, "delist") for several reasons, including: (a) the water quality standard is now being met, (b) there were flaws in the

[15] USEPA (U.S. Environmental Protection Agency). 2009. Information Concerning 2010 Clean Water Act Sections 303(d), 305(b), and 314 Integrated Reporting and Listing Decisions. U.S. Environmental Protection Agency, Office of Water, Washington, DC. https://www.epa.gov/sites/production/files/2015-10/documents/2009_05_06_tmdl_guidance_final52009.pdf.

[16] USEPA (U.S. Environmental Protection Agency). 2005. Guidance for 2006 Assessment, Listing and Reporting Requirements Pursuant to Sections 303(d), 305(b) and 314 of the Clean Water Act. U.S. Environmental Protection Agency, Office of Water, Washington, DC. https://www.epa.gov/sites/production/files/2015-10/documents/2006irg-report.pdf

original listing, (c) a TMDL was developed and approved by EPA, or (d) other point sources or nonpoint source controls are expected to meet WQS as described in 40 CFR 130.7(a)(b)(1).

Implementation Questions about Water Quality Management Plans (e.g., TMDLs)

15. How should states and authorized tribes complete TMDLs for waterbodies that are listed under CWA section 303(d) as impaired or threatened due to cyanotoxins?

States and authorized tribes can develop TMDLs for the pollutant(s) that promote development of cyanobacterial blooms and elevated levels of microcystins and cylindrospermopsin above EPA's recommended criteria. As previously discussed in this document, nitrogen and phosphorus loadings can provide optimal conditions for the formation of cyanobacterial blooms and cyanotoxin production. Although accurately deriving the quantity of nutrient pollution reduction necessary to meet WQS for cyanotoxins (or cyanobacteria) is a complex process, states and authorized tribes can use tools EPA is currently developing to better understand the relationship between the causal variables—pollutant loads of nitrogen and phosphorus—and the response variables of the cyanotoxin microcystins.[17] States and authorized tribes may also want to conduct site-specific studies to help refine these relationships and should consider the state of the science as they prioritize and schedule TMDLs to address cyanotoxins.

[17] Additional research is currently being conducted by EPA and others to improve the understanding between cyanotoxins and nutrient pollution. For example, see:

1. Lester L. Yuan and Amina I. Pollard, Managing microcystin: identifying national-scale thresholds for total nitrogen and chlorophyll-*a*, *Freshwater Biology*, 59, 9, (1970-1981), (2014).
2. Lester L. Yuan and Amina I. Pollard, Deriving nutrient targets to prevent excessive cyanobacterial densities in U.S. lakes and reservoirs, *Freshwater Biology*, 60, 9, (1901-1916), (2015).
3. Lester L. Yuan and Amina I. Pollard, Using National-Scale Data to Develop Nutrient–Microcystin Relationships That Guide Management Decisions, *Environmental Science & Technology*, 51, 12, (6972), (2017).

National Recommended Water Quality Criteria - Human Health Criteria Table

As prepared by New Mexico Environment Department Surface Water Quality Bureau
For Informational Purposes Only
February 2021

NOTE: (P)= Priority Pollutant

Human Health Criteria Calculation Matrix
Human Health Criteria and Methods for Toxics
Organoleptic Effects Criteria Table
Aquatic Life Criteria Table

60	*Carcinogens (304(a) criteria was adjusted manually one decimal place to right)
94	had updates in 2015
23	304(a) Criteria is equavlent to WQS - No changes needed (11 have no numeric criteria in 304(a) or WQS)
60	304(a) Criteria is MORE Stringent, need to consider adoption
25	304(a) Criteria is LESS Stringent
14	Have a 304(a) numeric criteria but not found in Table J (or elsewhere)
122	total number of pollutants on 304(a) HH-OO

Human health ambient water quality criteria represent specific levels of chemicals or conditions in a water body that are not expected to cause adverse effects to human health. EPA provides recommendations for "water + organism" and "organism only" human health criteria for states and authorized tribes to consider when adopting criteria into their water quality standards. These human health criteria are developed by EPA under Section 304(a) of the Clean Water Act.

Select pollutant name for current criteria document.

Already converted to 1 extra incident in 100,000

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
Acenaphthene (P)	304(a) is MORE stringent	83329	90	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		990		
Acrolein (P)	304(a) is LESS stringent	107028	400	2015			9		
Acrylonitrile (P)	304(a) is LESS stringent	107131	70	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	7	C	This is the same but the cancer risk for NM is 10-5 so this should be 70 ug/L
Aldrin (P)	304(a) is MORE stringent	309002	0.0000077	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.0005	C,P	
alpha-Hexachlorocyclohexane (HCH) (P)	304(a) is MORE stringent	319846	0.0039	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.049	C	
alpha-Endosulfan (P)	304(a) is MORE stringent	959988	30	2015			89		
Anthracene (P)	304(a) is MORE stringent	120127	400	2015			40000		
Antimony (P)	Same-No Change Needed	7440360	640	1980	This criterion was revised to reflect EPA's q1* or RfD as contained in the Integrated Risk Information System (IRIS) as of May 17, 2002. The fish tissue bioconcentration factor (BCF) is from the 1980 Ambient Water Quality Criteria document. EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		640	P	
Arsenic (P)	304(a) is MORE stringent	7440382	1.4	1992	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This recommended water quality criterion for arsenic refers to the inorganic form only.	Cancer Endpoint	9	C,P	
Asbestos (P)	Same-No Change Needed	1332214	—	1991	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		—	-	

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Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
Barium	Same-No Change Needed	7440393	—	1986	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.This human health criterion is the same as originally published in the Quality Criteria for Water, 1976 ("Red Book") which predates the 1980 methodology and did not utilize the fish ingestion BCF approach. This same criterion value is published in the Quality Criteria for Water, 1986 ("Gold Book").		—	-	
Benzene (P)	304(a) is MORE stringent	71432	102	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	510	C	The HH-OO is based on a range while the drinking water MCL is lower (5ug/L). Should use the drinking water MCL
Benzidine (P)	304(a) is LESS stringent	92875	0.11	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.002	C	
Benzo(a)anthracene (P)	304(a) is MORE stringent	56553	0.013	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.18	C	
Benzo(a)pyrene (P)	304(a) is MORE stringent	50328	0.0013	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	0.18	C,P	
Benzo(b)fluoranthene (P)	304(a) is MORE stringent	205992	0.013	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.18	C	
Benzo(k)fluoranthene (P)	304(a) is MORE stringent	207089	0.13	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.18	C	
Beryllium (P)	Same-No Change Needed	7440417	—	—	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		—	-	
beta-Hexachlorocyclohexane (HCH) (P)	304(a) is MORE stringent	319857	0.14	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.17	C	
beta-Endosulfan (P)	304(a) is MORE stringent	3.3E+07	40	2015			89		
Bis(2-Chloro-1-methylethyl) Ether (P)	304(a) is MORE stringent	108601	4000	2015			65000		
Bis(2-Chloroethyl) Ether (P)	304(a) is LESS stringent	111444	22	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	5.3	C	
Bis(2-Ethylhexyl) Phthalate (P)	304(a) is MORE stringent	117817	3.7	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	22	C	

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
Bis(Chloromethyl) Ether	Not In 900	542881	0.17	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	Not in 900		
Bromoform (P)	304(a) is MORE stringent	75252	1200	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	1400	C	
Butylbenzyl Phthalate (P)	304(a) is MORE stringent	85687	1	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	1900		
Cadmium (P)	Same-No Change Needed	7440439	—	—	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		—	-	
Carbon Tetrachloride (P)	304(a) is LESS stringent	56235	50	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	16	C	
Chlordane (P)	304(a) is MORE stringent	57749	0.0032	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.0081	C,P	
Chlorobenzene (P)	304(a) is MORE stringent	108907	800	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects. EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		1600	-	
Chlorodibromomethane (P)	304(a) is LESS stringent	124481	210	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	130	C	
Chloroform (P)	304(a) is MORE stringent	67663	2,000	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		4700	C	Endpoint from EPA does not indicate it was based off of carcinogenic enpoint
Chlorophenoxy Herbicide (2,4-D)	Not In 900	94757	12,000	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		Not in 900	-	
Chlorophenoxy Herbicide (2,4,5-TP) [Silvex]	Not In 900	93721	400	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		Not in 900	-	
Chromium (III) (P)	Same-No Change Needed	1.6E+07	—	—	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		—	-	
Chromium (VI) (P)	Same-No Change Needed	1.9E+07	—	—	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		—	-	

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
Chrysene (P)	304(a) is LESS stringent	218019	1.3	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	0.18	C	
Copper (P)	Same-No Change Needed	7440508	—	1992	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). This chemical has a criterion for organoleptic (taste and odor) effects. In some cases, the organoleptic criterion may be more stringent. EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	—	-	
Cyanide (P)	304(a) is LESS stringent	57125	400	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		140	-	
Dibenzo(a,h)anthracene (P)	304(a) is MORE stringent	53703	0.0013	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.18	C	
Dichlorobromomethane (P)	304(a) is LESS stringent	75274	270	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	170	C	
Dieldrin (P)	304(a) is MORE stringent	60571	0.000012	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.00054	C,P	
Diethyl Phthalate (P)	304(a) is MORE stringent	84662	600	2015			44000	-	
Dimethyl Phthalate (P)	304(a) is MORE stringent	131113	2,000	2015			1100000	-	
Di-n-Butyl Phthalate (P)	304(a) is MORE stringent	84742	30	2015			4500	-	
Dinitrophenols	Not In 900	2.6E+07	1,000	2015			Not in 900	-	There is a 2,4-dintrophenol CAS# 51-28-5 with HH-OO of 5300
Endosulfan Sulfate (P)	304(a) is MORE stringent	1031078	40	2015			89	-	Listed as endosulfan sulfate
Endrin (P)	304(a) is MORE stringent	72208	0.03	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		0.06	-	
Endrin Aldehyde (P)	304(a) is LESS stringent	7421934	1	2015			0.3	-	
Ethylbenzene (P)	304(a) is MORE stringent	100414	130	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		2100	-	
Fluoranthene (P)	304(a) is MORE stringent	206440	20	2015			140	-	
Fluorene (P)	304(a) is MORE stringent	86737	70	2015			5300	-	
gamma-Hexachlorocyclohexane (HCH) [Lindane] (P)	304(a) is LESS stringent	58899	4.4	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		1.8	-	

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
Heptachlor (P)	304(a) is MORE stringent	76448	0.000059	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	0.00079	C	
Heptachlor Epoxide (P)	304(a) is MORE stringent	1024573	0.00032	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	0.00039	C	
Hexachlorobenzene (P)	304(a) is MORE stringent	118741	0.00079	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	0.0029	C,P	
Hexachlorobutadiene (P)	304(a) is MORE stringent	87683	0.1	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	180	C	
Hexachlorocyclohexane (HCH) -Technical	Not In 900	608731	0.1	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	Not in 900	-	
Hexachlorocyclopentadiene (P)	304(a) is MORE stringent	77474	4	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects. EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		1100	-	
Hexachloroethane (P)	304(a) is MORE stringent	67721	1	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	33	C	
Indeno(1,2,3-cd)pyrene (P)	304(a) is MORE stringent	193395	0.013	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.18	C	
Isophorone (P)	304(a) is LESS stringent	78591	18,000	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	9600	C	
Manganese	304(a) is MORE stringent	7439965	100	1993	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects. The Human Health for the consumption of Water + Organism criterion for manganese is not based on toxic effects, but rather is intended to minimize objectionable qualities such as laundry stains and objectionable tastes in beverages.		-	-	
Methylmercury (P)	Same-No Change Needed	2.3E+07	0.3 mg/kg	2001	This fish tissue residue criterion for methylmercury is based on a total fish consumption rate of 0.0175 kg/day.		0.3 mg/kg	P	
Methoxychlor	Not In 900	72435	0.02	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		Not in 900	-	
Methyl Bromide (P)	304(a) is LESS stringent	74839	10000	2015			1500	-	

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
Methylene Chloride (P)	304(a) is LESS stringent	75092	10,000	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	5900	C	
Nickel (P)	Same-No Change Needed	7440020	4,600	1998	This criterion was revised to reflect EPA's q1* or RfD as contained in the Integrated Risk Information System (IRIS) as of May 17, 2002. The fish tissue bioconcentration factor (BCF) is from the 1980 Ambient Water Quality Criteria document.		4600	P	
Nitrates	Same-No Change Needed	1.5E+07	—	1986	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		—	-	
Nitrobenzene (P)	304(a) is MORE stringent	98953	600	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		690	-	
Nitrosamines	Not In 900	—	12.4	1980	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	Not in 900		
Nitrosodibutylamine	Not In 900	924163	2.2	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	Not in 900		
Nitrosodiethylamine	Not In 900	55185	12.4	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	Not in 900		
Nitrosopyrrolidine	Not In 900	930552	340	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	Not in 900		
N-Nitrosodimethylamine (P)	Same-No Change Needed	62759	30	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	30	C	
N-Nitrosodi-n-Propylamine (P)	Same-No Change Needed	621647	5.1	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	5.1	C	
N-Nitrosodiphenylamine (P)	Same-No Change Needed	86306	60	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	60	C	
Pathogen and Pathogen Indicators	Same-No Change Needed	—	—	2012	See EPA's 2012 Recreational Water Quality Criteria. For Shellfish see Quality Criteria for Water 1986 ("Gold Book")		—	—	
Pentachlorobenzene	Not In 900	608935	0.1	2015			Not in 900	-	
Pentachlorophenol (P)	304(a) is MORE stringent	87865	0.4	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects. EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	30	C	
pH	Same-No Change Needed	—	—	1986			—	-	Listed as criteria for specific ALUs

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
Phenol (P)	304(a) is MORE stringent	108952	300,000	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		860000	-	
Polychlorinated Biphenyls (PCBs) (P)	Same-No Change Needed		0.00064	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). This criterion applies to total PCBs (e.g., the sum of all congener or all isomer or homolog or Aroclor analyses). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	0.00064	C,P	
Pyrene (P)	304(a) is MORE stringent	129000	30	2015			4000	-	
Selenium (P)	Same-No Change Needed	7782492	4200	2002	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		4200	P	
Solids Dissolved and Salinity	Same-No Change Needed	—	—	1986			—	-	
Tetrachloroethylene (P)	304(a) is LESS stringent	127184	290	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). **EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.**	Cancer Endpoint	33	C,P	
Thallium (P)	Same-No Change Needed	7440280	0.47	2003			0.47	P	
Toluene (P)	304(a) is MORE stringent	108883	520	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		15000	-	
Toxaphene (P)	304(a) is LESS stringent	8001352	0.0071	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	0.0028	C	
Trichloroethylene (P)	304(a) is MORE stringent	79016	70	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	300	C	
Vinyl Chloride (P)	304(a) is MORE stringent	75014	16	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	24	C	
Zinc (P)	Same-No Change Needed	7440666	26,000	2002	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		26000	P	
1,1,1-Trichloroethane (P)	304(a) is MORE stringent	71556	200,000	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		-	-	
1,1,2,2-Tetrachloroethane (P)	304(a) is MORE stringent	79345	30	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	40	C	

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
1,1,2-Trichloroethane (P)	304(a) is MORE stringent	79005	89	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	160	C	
1,1-Dichloroethylene (P)	304(a) is LESS stringent	75354	20,000	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		7100	C	Endpoint from EPA does not indicate it was based off of carcinogenic enpoint
1,2,4,5-Tetrachlorobenzene	Not In 900	95943	0.03	2015			Not in 900	-	
1,2,4-Trichlorobenzene (P)	304(a) is MORE stringent	120821	0.76	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations. This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	70	-	
1,2-Dichlorobenzene (P)	304(a) is LESS stringent	95501	3,000	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		1300	-	
1,2-Dichloroethane (P)	304(a) is LESS stringent	107062	6500	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	370	C	
1,2-Dichloropropane (P)	304(a) is LESS stringent	78875	310	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	150	C	
1,2-Diphenylhydrazine (P)	Same-No Change Needed	122667	2	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	2	C	
Trans-1,2-Dichloroethylene (P)	304(a) is MORE stringent	156605	4,000	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		10000	-	
1,3-Dichlorobenzene (P)	304(a) is MORE stringent	541731	10	2015			960	-	
1,3-Dichloropropene (P)	304(a) is MORE stringent	542756	120	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	210	C	
1,4-Dichlorobenzene (P)	304(a) is LESS stringent	106467	900	2015	EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.		190	-	
2,3,7,8-TCDD (Dioxin) (P)	Same-No Change Needed	1746016	5.10E-08	2002	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). EPA has issued a Maximum Contaminant Level (MCL) for this chemical which may be more stringent. See EPA's National Primary Drinking Water Regulations.	Cancer Endpoint	5.10E-08	C,P	Listed only as Dioxin with no CAS
2,4,5-Trichlorophenol	Not In 900	95954	600	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		Not in 900	-	

Pollutant	HH-OO Variance From 304(a)	CAS Number	304(a) Criteria Human Health for the consumption of Organism Only (µg/L)	Publication Year	Notes	*Criteria Enpoint (C or P)	NM WQ Criteria for HH-OO	Type	Notes
2,4,6-Trichlorophenol (P)	304(a) is LESS stringent	88062	28	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right). The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.	Cancer Endpoint	24	C	
2,4-Dichlorophenol (P)	304(a) is MORE stringent	120832	60	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		290	-	
2,4-Dimethylphenol (P)	304(a) is LESS stringent	105679	3,000	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		850	-	
2,4-Dinitrophenol (P)	304(a) is MORE stringent	51285	300	2015			5300	-	
2,4-Dinitrotoluene (P)	304(a) is MORE stringent	121142	17	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	34	C	
2-Chloronaphthalene (P)	304(a) is MORE stringent	91587	1000	2015			1600	-	
2-Chlorophenol (P)	304(a) is LESS stringent	95578	800	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		150	-	
2-Methyl-4,6-Dinitrophenol (P)	304(a) is MORE stringent	534521	30	2015			280	-	
3,3'-Dichlorobenzidine (P)	304(a) is LESS stringent	91941	1.5	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.28	C	
3-Methyl-4-Chlorophenol (P)	Not In 900	59507	2,000	2015	The criterion for organoleptic (taste and odor) effects may be more stringent. See National Recommended Water Quality Criteria - Organoleptic Effects.		Not in 900	-	
p,p'-Dichlorodiphenyldichloroethane (DDD) (P)	304(a) is MORE stringent	72548	0.0012	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.0022	C,P	Listed under general DDT and derivatives
p,p'-Dichlorodiphenyldichloroethylene (DDE) (P)	304(a) is MORE stringent	72559	0.00018	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.0022	C,P	Listed under general DDT and derivatives
p,p'-Dichlorodiphenyltrichloroethane (DDT) (P)	304(a) is MORE stringent	50293	0.0003	2015	This criterion is based on carcinogenicity of 10-6 risk. Alternate risk levels may be obtained by moving the decimal point (e.g., for a risk level of 10-5, move the decimal point in the recommended criterion one place to the right).	Cancer Endpoint	0.0022	C,P	Listed under general DDT and derivatives

United States Environmental Protection Agency

Office of Water
Office of Science and Technology

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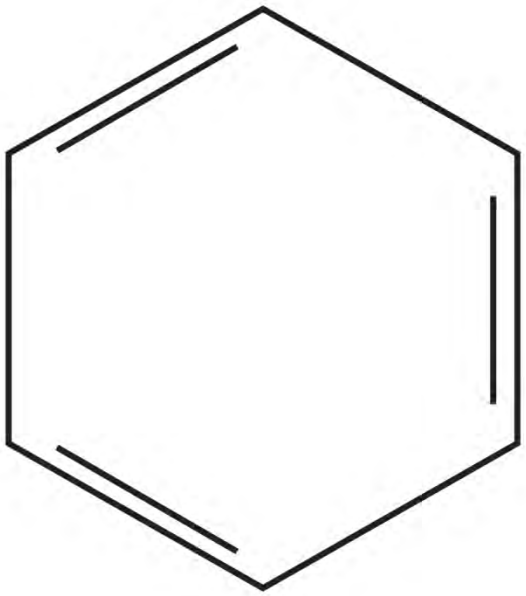
June 2015

Update of Human Health Ambient Water Quality Criteria:

Benzene
71-43-2



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Office of Science and Technology
Office of Water
U.S. Environmental Protection Agency
Washington, DC 20460

Table of Contents

1 Introduction: Background and Scope of Update

EPA's recommended ambient water quality criteria (AWQC) for human health are scientifically derived numeric values that EPA has determined will adequately protect human health from the adverse effects of pollutants in ambient water.

Section 304(a)(1) of the Clean Water Act (CWA) requires EPA to develop and publish, and from time to time revise, recommended criteria for the protection of water quality that accurately reflect the latest scientific knowledge. Water quality criteria developed under section 304(a) are based solely on data and scientific judgments on the relationship between pollutant concentrations and environmental and human health effects. Section 304(a) criteria do not reflect consideration of economic impacts or the technological feasibility of meeting pollutant concentrations in ambient water.

EPA's recommended section 304(a) criteria provide technical information for states and authorized tribes[a] to consider and use in adopting water quality standards that ultimately provide the basis for assessing water body health and controlling discharges of pollutants into waters of the United States. Under the CWA and its implementing regulations, states and authorized tribes are required to adopt water quality criteria to protect the designated uses of waters (e.g., public water supply, aquatic life, recreational use, industrial use). EPA's recommended water quality criteria do not substitute for the CWA or regulations, nor are they regulations themselves. Thus, EPA's recommended criteria do not impose legally binding requirements. States and authorized tribes may adopt, where appropriate, other scientifically defensible water quality criteria that differ from these recommendations.

The water quality criteria that are the subject of this document are national AWQC recommendations for human health issued under CWA section 304(a). Unless expressly indicated otherwise, all references to "criteria," "water quality criteria," "ambient water quality criteria recommendations," or similar variants thereof are references to national AWQC recommendations for human health.

In this 2015 update, EPA has revised the human health criteria for benzene to reflect the latest scientific information, including updated exposure factors (body weight [BW], drinking water intake [DI] rate, and fish consumption rate [FCR]), bioaccumulation factors (BAFs), and human health toxicity values (reference dose [RfD] multiplied by relative source contribution [RSC] or 10^{-6} divided by cancer slope factor [CSF]). The criteria continue to be based on EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health*, which is referred to as the "2000 Methodology" in this document (USEPA 2000a). EPA accepted written scientific views from the public on the draft updated human health criteria for this chemical (and 93 others) from May through August 2014.

[a] Throughout this document, the term *states* means the 50 states, the District of Columbia, the Commonwealth of Puerto Rico, the Virgin Islands, Guam, American Samoa, and the Commonwealth of the Northern Mariana Islands. The term *authorized tribe* or *tribe* means an Indian tribe authorized for treatment in a manner similar to a state under CWA section 518 for the purposes of section 303(c) water quality standards.

It is important for states and authorized tribes to consider any new or updated section 304(a) recommended criteria as part of their triennial review process to ensure that state or tribal water quality standards reflect current science and protect applicable designated uses. These final 2015 updated section 304(a) human health criteria recommendations supersede EPA's previous recommendations.

2 Problem Formulation

Problem formulation provides a strategic framework for water quality criteria development by focusing on the most relevant endpoints and increasing the transparency of the effects assessment. The structure of this criteria document is intended to be consistent with general concepts of effects assessments as described in EPA's *Framework for Human Health Risk Assessment to Inform Decision Making* (USEPA 2014a).

In developing AWQC, EPA currently follows the assessment method outlined in its 2000 Methodology (USEPA 2000a). The 2000 Methodology describes different approaches for addressing water and non-water exposure pathways to derive human health AWQC depending on the toxicological endpoint of concern, the toxicological effect (noncarcinogenic or carcinogenic), and whether toxicity is considered a linear or threshold effect. Water sources of exposure include both consuming drinking water and eating fish or shellfish from inland and nearshore waters that have been exposed to pollutants in the water body. For pollutants that exhibit a threshold of exposure before deleterious effects occur, as is the case for noncarcinogens and nonlinear carcinogens, EPA applies an RSC to account for other potential human exposures to the pollutant (USEPA 2000a). Other sources of exposure might include, but are not limited to, exposure to a particular pollutant from ocean fish or shellfish consumption (which is not included in the FCR), non-fish food consumption (e.g., consumption of fruits, vegetables, grains, meats, or poultry), dermal exposure, and inhalation exposure.

For substances for which the toxicity endpoint is carcinogenicity based on a linear low-dose extrapolation, only the exposures from drinking water and fish ingestion are reflected in human health AWQC; that is, non-water sources are not explicitly included and no RSC is applied (USEPA 2000a). In these situations, AWQC are derived with respect to the *incremental* lifetime cancer risk posed by the presence of a substance in water, rather than an individual's total risk from all sources of exposure. The resulting criterion represents the water concentration that is expected to increase an individual's lifetime risk of cancer from exposure to the particular pollutant by no more than one chance in one million for the general population. EPA calculates AWQC at a 10^{-6} (one in one million) cancer risk level for the general population (USEPA 2000a). The 2000 Methodology recommends that states set human health criteria cancer risk levels for the target general population at either 10^{-5} or 10^{-6} and also notes that states and authorized tribes can choose a more stringent risk level, such as 10^{-7}.

For substances that are carcinogenic, EPA takes an integrated approach and considers both cancer and noncancer effects when deriving AWQC (USEPA 2000a; USEPA 2000b). Where sufficient data are available, EPA derives AWQC using both carcinogenic and noncarcinogenic toxicity endpoints and recommends the lower value for the AWQC. The AWQC might not utilize

the value obtained from the cancer analysis if it is less protective than that derived from the noncancer endpoint.

3 Criteria Formulas: Analysis Plan

Human health AWQC for toxic pollutants are necessary to protect any designated uses related to ingestion of water and ingestion of aquatic organisms. These uses can include, but are not limited to, recreation in and on the water, consumption of fish or shellfish (including consumption associated with fishing or shellfish harvesting), and protection of drinking water supplies.

The derivation of human health AWQC requires information about both the toxicological endpoints of concern for water pollutants and the pathways of human exposure to those pollutants. EPA considers the following two primary pathways of human exposure to pollutants present in a particular water body when deriving human health 304(a) AWQC: (1) direct ingestion of drinking water obtained from the water body and (2) consumption of fish or shellfish obtained from the water body.

The equations for deriving human health AWQC for noncarcinogenic effects and carcinogenic effects are presented as Eqs. 1 and 2. EPA derives recommended human health AWQC based on the consumption of both water and aquatic organisms (Eq. 1) and based on the consumption of aquatic organisms alone (Eq. 2). The use of one criterion over the other depends on the designated use of a particular water body or water bodies (i.e., drinking water source and/or fishable waters). EPA recommends applying organism-only AWQC (Eq. 2) to a water body where the designated use includes supporting fishable uses under section 101(a) of the CWA but the water body is not a drinking water supply source (e.g., non-potable estuarine waters that support fish or shellfish for human consumption) (USEPA 2000a).

EPA recommends including the drinking water exposure pathway for ambient surface waters where drinking water is a designated use for the following reasons: (1) drinking water is a designated use for surface waters under the CWA, and therefore criteria are needed to ensure that this designated use can be protected and maintained; (2) although they are rare, some public water supplies provide drinking water from surface water sources without treatment; (3) even among the majority of water supplies that do treat surface waters, existing treatments might not be effective for reducing levels of particular contaminants; and (4) in consideration of the Agency's goals of pollution prevention, ambient waters should not be contaminated to a level where the burden of achieving health objectives is shifted away from those responsible for pollutant discharges and placed on downstream users that must bear the costs of upgraded or supplemental water treatment (USEPA 2000a).

The equations for deriving the criteria values are as follows (USEPA 2000a):

For consumption of water and organisms:

$$AWQC\ (\mu g/L) = \frac{\text{toxicity value (mg/kg-d)} \times BW\ (kg) \times 1{,}000\ (\mu g/mg)^{b}}{DI\ (L/d) + \sum_{i=2}^{4} (FCR_i\ (kg/d) \times BAF_i\ (L/kg))} \quad \text{(Eq. 1)}$$

For consumption of organisms only:

$$AWQC\ (\mu g/L) = \frac{\text{toxicity value (mg/kg-d)} \times BW\ (kg) \times 1{,}000\ (\mu g/mg)^{c}}{\sum_{i=2}^{4} (FCR_i\ (kg/d) \times BAF_i\ (L/kg))} \quad \text{(Eq. 2)}$$

Where:

AWQC	=	ambient water quality criteria
toxicity value	=	RfD x RSC (mg/kg-d) for noncarcinogenic effects or 10^{-6}/CSF (kg-d/mg) for carcinogenic effects[d]
RSC	=	relative source contribution (applicable to only noncarcinogenic and nonlinear low-dose extrapolation for carcinogenic effects)
BW	=	body weight
DI	=	drinking water intake
$\sum_{i=2}^{4}$	=	summation of values for aquatic trophic levels (TLs), where the letter *i* stands for the TLs to be considered, starting with TL2 and proceeding to TL4
FCR_i	=	fish consumption rate for aquatic TLs 2, 3, and 4
BAF_i	=	bioaccumulation factor for aquatic TLs 2, 3, and 4

EPA rounds AWQC to the number of significant figures in the least precise parameter as described in the 2000 Methodology (USEPA 2000a, section 2.7.3).

4 Exposure Factors

4.1 Body Weight

EPA updated the default BW assumption to 80.0 kg based on National Health and Nutrition Examination Survey (NHANES) data from 1999 to 2006 as reported in Table 8.1 of EPA's *Exposure Factors Handbook* (USEPA 2011). The updated BW represents the mean weight for adults ages 21 and older. EPA's previously recommended BW assumption for adults was 70 kg, which was based on the mean BW of adults from the NHANES III database (1988–1994) and a 1989 study conducted by the National Cancer Institute (USEPA 2000a).

[b] 1,000 µg/mg is used to convert the units of mass from milligrams to micrograms.
[c] 1,000 µg/mg is used to convert the units of mass from milligrams to micrograms.
[d] 10^{-6} or 1 in 1,000,000 risk level for the general population.

4.2 Drinking Water Intake

EPA updated the default DI to 2.4 L/d, rounded from 2.414 L/d, based on NHANES data from 2003 to 2006 as reported in EPA's *Exposure Factors Handbook* (USEPA 2011, Table 3-23). This rate represents the per capita estimate of combined direct and indirect community water[e] ingestion at the 90th percentile for adults ages 21 and older. EPA selected the per capita rate for the updated DI because it represents the average daily dose estimates; that is, it includes both people who drank water during the survey period and those who did not, which is appropriate for a national-scale assessment such as CWA section 304(a) national human health criteria development (USEPA 2011, section 3.2.1).

EPA's updated DI of 2.4 L/d is consistent with the 2000 Methodology. In that document, EPA recommended a default DI of 2 L/d, which represented the per capita community water ingestion rate at the 86th percentile for adults surveyed in the U.S. Department of Agriculture's 1994–1996 Continuing Survey of Food Intake by Individuals (CSFII) analysis (USEPA 2000a, section 4.3.2.1).

4.3 Fish Consumption Rate

The updated FCR for the general adult population is 22.0 g/d, or 0.0220 kg/d (USEPA 2014b, Table 9a). This FCR represents the 90th percentile per capita consumption rate of fish from inland and nearshore waters for U.S. adults ages 21 years and older based on NHANES data from 2003–2010. The 95 percent confidence interval (CI) of the 90th percentile per capita FCR is 19.1 g/d and 25.4 g/d. This updated FCR replaces EPA's previously recommended default FCR of 17.5 g/d, which represented an estimate of the 90th percentile per capita consumption rate of fish from inland and nearshore waters for U.S. adults ages 21 years and older. That default FCR was based on USDA's CSFII 1994–1996 data (USEPA 2002a).

As recommended in the 2000 Methodology, EPA updated the AWQC to reflect trophic level- (TL-) specific FCRs to better represent human dietary consumption of fish. An organism's trophic position in the aquatic food web can have an important effect on the magnitude of bioaccumulation of certain chemicals. The TL-specific FCRs are numbered 2, 3, and 4, and they account for different categories of fish and shellfish species based on their position in the aquatic food web: TL2 accounts for benthic filter feeders; TL3 accounts for forage fish; and TL4 accounts for predatory fish (USEPA 2000a).

EPA used the following TL-specific FCRs to derive the updated AWQC: TL2 = 7.6 g/d (0.0076 kg/d) (95 percent CI [6.4, 9.1] g/d); TL3 = 8.6 g/d (0.0086 kg/d) (95 percent CI [7.2, 10.2] g/d); and TL4 = 5.1 g/d (0.0051 kg/d) (95 percent CI [4.0, 6.4] g/d). Each TL-specific FCR represents the 90th percentile per capita consumption rate of fish and shellfish from inland and nearshore waters from that particular TL for U.S. adults ages 21 years and older (USEPA 2014b,

[e] *Community water* includes direct and indirect use of tap water for household uses and excludes bottled water and other sources (USEPA 2011, section 3.3.1.2). *Direct ingestion* is defined as direct consumption of water as a beverage, while *indirect ingestion* includes water added during food preparation (e.g., cooking, rehydration of beverages) but not water intrinsic to purchased foods (USEPA 2011, section 3.1).

Tables 16a, 17a, and 18a). The sum of these three TL-specific FCRs is 21.3 g/d, which is within the 95 percent CI of the overall FCR of 22.0 g/d. EPA recommends using the TL-specific FCRs when deriving AWQC; however, the overall FCR rate (22.0 g/d) may be used if a simplified approach is preferred.

4.4 Bioaccumulation Factor

4.4.1 Approach

Several attributes of the bioaccumulation process are important to understand when deriving national BAFs for use in developing national recommended section 304(a) AWQC. First, the term *bioaccumulation* refers to the uptake and retention of a chemical by an aquatic organism from all surrounding media, such as water, food, and sediment. The term *bioconcentration* refers to the uptake and retention of a chemical by an aquatic organism from water only. For some chemicals (particularly those that are highly persistent and hydrophobic), the magnitude of bioaccumulation by aquatic organisms can be substantially greater than the magnitude of bioconcentration. Thus, an assessment of bioconcentration alone might underestimate the extent of accumulation in aquatic biota for those chemicals. Accordingly, the EPA guidelines presented in the 2000 Methodology emphasize using, when possible, measured or estimated BAFs, which account for chemical accumulation in aquatic organisms from all potential exposure routes (USEPA 2000a).

EPA estimated BAFs for this updated AWQC using EPA's 2000 Methodology (USEPA 2000a) and its *Technical Support Document, Volume 2: Development of National Bioaccumulation Factors* (Technical Support Document, Volume 2) (USEPA 2003a). Specifically, these documents provide a framework for identifying alternative procedures to derive national TL-specific BAFs for a chemical based on the chemical's properties (e.g., ionization and hydrophobicity), metabolism, and biomagnification potential (USEPA 2000a; USEPA 2003a).

EPA's approach for developing national BAFs represents the long-term average bioaccumulation potential of a pollutant in aquatic organisms that are commonly consumed by humans across the United States. National BAFs are not intended to reflect fluctuations in bioaccumulation over short periods (e.g., a few days) because human health AWQC are generally designed to protect humans from long-term (lifetime) exposures to waterborne chemicals (USEPA 2003a).

EPA followed the approach described in Figure 3-1 of the Technical Support Document, Volume 2 (USEPA 2003a). EPA used peer-reviewed, publicly available information to classify each chemical using this framework to derive the most appropriate BAFs according to EPA's 2000 Methodology (USEPA 2000a). The framework provides six alternatives, or procedures, resulting in up to four possible methods for each chemical, based on the chemical's properties. These four methods follow:

- **BAF Method.** This method uses measured BAFs derived from data obtained from field studies. Field-measured BAFs were normalized by adjusting for the water-dissolved portions of the chemical and the lipid fraction of fish tissue for each species, as well as the fraction of the total concentration of chemical in water that is freely dissolved. EPA averaged multiple field BAFs using a geometric mean of the normalized BAFs by species and TL; then EPA further averaged the BAFs across species to compute TL baseline BAFs. The national-level BAF adjusts the TL baseline BAFs by national default values for lipid content, dissolved and particulate organic carbon content, and the n-octanol-water partition coefficient (K_{ow}). EPA chose the recommended 50th percentile dissolved and particulate organic carbon content for the national-level default values, as described in section 6.3 of the Technical Support Document, Volume 2 (USEPA 2003a).
- **BSAF Method.** This method uses biota-sediment accumulation factors (BSAFs) to estimate BAFs. EPA did not use measured BSAFs to calculate national BAFs because the two major compilations of these data—EPA's Biota-Sediment Accumulation Factor Data Set, Version 1.0 (USEPA 2015a), and the U.S. Army Corps of Engineers' BSAF database (USACE 2015)—have not been peer-reviewed.
- **BCF Method.** This method uses BAFs estimated from laboratory-measured bioconcentration factors (BCFs) with or without adjustment by a food chain multiplier. Similar to field BAFs, laboratory-measured BCFs are normalized with the lipid fraction and the fraction of the total concentration of chemical in water that is freely dissolved, then multiplied by the food chain multiplier where applicable. Multiple values are averaged using a geometric mean across species and then across TL to compute baseline BAFs. The national-level BAF adjusts the TL baseline BAFs by national default values for lipid content, dissolved and particulate organic carbon content, and the K_{ow}. EPA chose the recommended 50th percentile dissolved and particulate organic carbon content for the national-level default values, as described in section 6.3 of the Technical Support Document, Volume 2 (USEPA 2003a).
- **K_{ow} Method.** This method predicts BAFs based on a chemical's K_{ow}, with or without adjustment using a food chain multiplier, as described in section 5.4 of the Technical Support Document, Volume 2 (USEPA 2003a).

Following the decision framework presented in Figure 3-1 of the Technical Support Document, Volume 2 (USEPA 2003a), EPA selected one of the six procedures to develop a national-level BAF for this chemical. For a given procedure, EPA selected the method that provided BAF estimates for all three TLs (TL2–TL4) in the following priority:

1. BAF estimates using the BAF method (i.e., based on field-measured BAFs) if possible.
2. BAF estimates using the BCF method if (a) the BAF method did not produce estimates for all three TLs and (b) the BCF method produced national-level BAF estimates for all three TLs.
3. BAF estimates using the K_{ow} method if (a) Procedure 1 or 3 was applicable (see Figure 3-1 of the Technical Support Document, Volume 2 [USEPA 2003a]) and (b) the BAF and BCF methods did not produce BAF estimates for all three TLs.

In cases where the procedure called for the BAF method but there were fewer than three TL estimates and the K_{ow} method did not apply (i.e., Procedures 2, 4, 5, and 6), EPA used the BAF method estimate for the reported TLs by averaging the estimates using a geometric mean when there were two BAFs and using the single estimate when only one was available. EPA did not mix values from the BAF and BCF methods. If the BAF method did not have sufficient reliable data for any TLs, EPA used the BCF method estimates in the same manner. If none of the four methods provided sufficient data, or if none were appropriate for the procedure, EPA used the BCF from the previously recommended 2002/2003 criteria (USEPA 2002b; USEPA 2003b).

EPA primarily used field-measured BAFs and laboratory-measured BCFs available from peer-reviewed, publicly available databases (Arnot and Gobas 2006; Environment Canada 2006) to develop national BAFs. If field-measured BAFs and laboratory-measured BCFs were not available from those sources, EPA selected K_{ow} values from peer-reviewed sources (i.e., Agency for Toxic Substances and Disease Registry [ATSDR] preferentially, followed by U.S. Department of Health and Human Services' Hazardous Substances Data Bank) for use in calculating national BAFs using the K_{ow} method described in EPA's Technical Support Document, Volume 2 (USEPA 2003a). For those chemicals for which the K_{ow} method was not applicable based on the Technical Support Document, Volume 2 (USEPA 2003a), EPA performed open literature searches of peer-reviewed journal articles to find field-measured BAFs or laboratory-measured BCFs.

4.4.2 Chemical-specific BAFs

EPA selected national BAF values of 3.6 L/kg (TL2), 4.5 L/kg (TL3), and 5.0 L/kg (TL4) for benzene. EPA followed the framework for selection of methods for deriving national BAFs in Figure 3-1 of the Technical Support Document, Volume 2 (USEPA 2003a) to select a procedure for estimating national BAFs for benzene. Based on the characteristics of this chemical, EPA selected Procedure 3 for deriving a national BAF value. Benzene has the following characteristics:

- Nonionic organic chemical (USDHHS 2014)
- Low hydrophobicity (log $K_{ow} < 4$); log $K_{ow} = 2.13$ (ATSDR 2007)
- Low/unknown metabolism

EPA was not able to locate peer-reviewed, field-measured BAFs or lab-measured BCFs for TLs 2, 3, and 4. Therefore, EPA used the K_{ow} method to derive the national BAF values for this chemical:

TL2 = 3.6 L/kg
TL3 = 4.5 L/kg
TL4 = 5.0 L/kg

5 Hazard Identification and Dose Response

5.1 Approach

EPA considered all available toxicity values for both noncarcinogenic and carcinogenic toxicological effects to develop this updated AWQC for benzene. As described in the 2000 Methodology (USEPA 2000a), where data are available EPA derives AWQC for both noncarcinogenic and carcinogenic effects and recommends the more protective value for the AWQC. (See section 7, Criteria Derivation: Analysis.)

For noncarcinogenic toxicological effects, EPA uses a chronic-duration oral RfD to derive human health AWQC. An RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure of the human population to a substance that is likely to be without an appreciable risk of deleterious effects during a lifetime. An RfD is typically derived from a laboratory animal dosing study in which a no-observed-adverse-effect level (NOAEL), lowest-observed-adverse-effect level (LOAEL), or benchmark dose can be obtained. Uncertainty factors are applied to reflect the limitations of the data (USEPA 2000a).

For carcinogenic toxicological effects, EPA uses an oral CSF to derive human health AWQC. The oral CSF is an upper bound, approximating a 95 percent confidence limit, on the increased cancer risk from a lifetime oral exposure to a stressor.

For this update, EPA conducted a systematic search of eight peer-reviewed, publicly available sources to obtain the toxicity value (RfD or CSF) for use in developing AWQC. EPA's primary source of toxicity values for developing human health AWQC is its Integrated Risk Information System (IRIS) program (USEPA 2015b). EPA also systematically searched for toxicological assessments from the following EPA program offices, other national and international programs, and state programs:

- EPA, Office of Pesticide Programs (USEPA 2015c)
- EPA, Office of Pollution Prevention and Toxics (USEPA 2015d)
- EPA, Office of Water (USEPA 2015e)
- EPA, Office of Solid Waste and Emergency Response (USEPA 2015f)
- U.S. Department of Health and Human Services, ATSDR (ATSDR 2015)
- Health Canada (HC 2015)
- California Environmental Protection Agency, Office of Environmental Health Hazard Assessment (CalEPA 2014)

After identifying and documenting all available toxicity values, EPA followed a systematic process to select the toxicity values used to derive the AWQC for noncarcinogenic and carcinogenic effects. EPA selected IRIS toxicity values to derive the updated AWQC if *any* of the following conditions were met:

1. EPA's IRIS toxicological assessment was the only available source of a toxicity value.
2. EPA's IRIS toxicological assessment was the most current source of a toxicity value.

3. EPA's IRIS program was reassessing the chemical in question and had published the draft Toxicological Review for public review and comment, discussion at a public meeting, and subsequent expert peer review.[f]
4. The toxicity value from a more current toxicological assessment from a source other than EPA IRIS was based on the same principal study and was numerically the same as an older EPA IRIS toxicity value.
5. A more current toxicological assessment from a source other than EPA IRIS was available, but it did *not* include the relevant toxicity value (chronic-duration oral RfD or CSF).
6. A more current toxicological assessment from a source other than EPA IRIS was available, but it did *not* introduce new science (e.g., the toxicity value was not based on a newer principal study) or use a more current modeling approach compared to an older EPA IRIS toxicological assessment.

EPA selected the toxicity value from a peer-reviewed, publicly available source other than EPA IRIS to derive the updated AWQC if *any* of the following conditions were met:

1. The chemical is currently used as a pesticide, and EPA Office of Pesticide Programs had a toxicity value that was used in pesticide registration decision-making.
2. A toxicological assessment from a source other than EPA IRIS was the only available source of a toxicity value.
3. A more current toxicological assessment from a source other than EPA IRIS introduced new science (e.g., the toxicity value was based on a newer principal study) or used a more current modeling approach compared to an older EPA IRIS toxicological assessment.

5.2 Chemical-specific Toxicity Value

5.2.1 Reference Dose

In place of an RfD, EPA selected a chronic oral minimal risk level (MRL) of 5×10^{-4} mg/kg-d (0.0005 mg/kg-d) for benzene based on a 2007 ATSDR assessment (ATSDR 2007). A chronic oral MRL is an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects for a chronic duration (365 days and longer).

ATSDR identified the inhalation occupational exposure study by Lan et al. (2004a; 2004b) as the critical study and decreased B cell counts in benzene-exposed workers served as the effect for the determination of the point of departure for the derivation for the chronic-duration oral MRL for benzene. ATSDR cited toxicokinetic data in humans and animals exposed to low levels of benzene that demonstrate absorption of approximately 50 percent of an inhaled dose and essentially 100 percent of an oral dose as rationale for the use of a route-to-route extrapolation. The lower-bound confidence limit on the benchmark dose ($BMDL_{0.25sdADJ}$) was

[f] Equivalent to Step 4 in the July 2013 EPA Process for Developing IRIS Health Assessments. Available online at http://www.epa.gov/iris/process.htm.

0.014 mg/kg-d. In deriving the MRL, ATSDR applied a composite uncertainty factor of 30 to account for route-to-route extrapolation (3) and intraspecies variation (10) (ATSDR 2007).

EPA identified two other RfD sources through the systematic search described in section 5: a 2002 EPA IRIS assessment (USEPA 2002c) and a 2001 California EPA assessment (CalEPA 2001). Based on the selection process described in section 5, the 2007 ATSDR RfD is preferred for use in AWQC development at this time. The ATSDR assessment is the most current source and relied on a newer study (Lan et al. 2004a; Lan et al. 2004b) compared to the 2002 IRIS assessment (Rothman et al. 1996). Additionally, ATSDR used a method of route-to-route extrapolation that is consistent with the IRIS assessment (USEPA 2002c).

5.2.2 Cancer Slope Factor

Under the 1996 EPA *Proposed Guidelines for Carcinogen Risk Assessment* (USEPA 1996), benzene is classified as Group A, "known human carcinogen" (USEPA 2000c).

EPA selected a CSF range of 1.5×10^{-2} per mg/kg-d (0.015 per mg/kg-d) to 5.5×10^{-2} per mg/kg-day (0.055 per mg/kg-day) for benzene based on a 2000 EPA IRIS assessment (USEPA 2000c). EPA's IRIS program derived the CSF using a principal studies by Rinsky et al. (1981; 1987), Paustenbach et al. (1993), Crump (1994), and USEPA (1998; 1999) based on the development of leukemia in humans with occupational inhalation exposure to benzene (USEPA 2000c).

EPA identified one other CSF source through the systematic search described in section 5: a 2001 California EPA assessment (CalEPA 2001). Based on the selection process described in section 5, the 2000 EPA IRIS CSF is preferred for use in AWQC development at this time. The CalEPA CSF is based on studies that IRIS considered in their assessment but did not use quantitatively (Paxton et al. 1994; Hayes et al. 1997).

6 Relative Source Contribution

6.1 Approach

The RSC component of the AWQC calculation allows a percentage of the RfD's exposure to be attributed to the consumption of ambient water and fish and shellfish from inland and nearshore waters when there are other potential exposure sources. The RSC describes the portion of the RfD available for AWQC-related sources (USEPA 2000a); the remainder of the RfD is allocated to other sources of the pollutant. The rationale for this approach is that for pollutants exhibiting threshold effects, the objective of the AWQC is to ensure that an individual's total exposure from all sources does not exceed that threshold level. Exposures outside the RSC include, but are not limited to, exposure to a particular pollutant from ocean fish and shellfish consumption (which is not included in the FCR), non-fish food consumption (e.g., fruits, vegetables, grains, meats, poultry), dermal exposure, and respiratory exposure.

EPA derived an RSC for each chemical included in this 2015 update by using the Exposure Decision Tree approach described in the 2000 Methodology (USEPA 2000a). To use that approach, EPA compiled information for each chemical on its uses, chemical and physical properties, occurrences in other potential sources (e.g., air, food), and releases to the environment, as well as regulatory restrictions on other sources that are specific to the chemical (e.g., air quality standards, food tolerance levels). The ATSDR "Toxicological Profiles" (ATSDR 2015) were the primary sources for this information. EPA used the Hazardous Substance Data Bank (HSDB) (USDHHS 2015) from the National Library of Medicine's Toxicology Data Network (TOXNET) as the primary source for chemicals without ATSDR Toxicological Profiles. Both sources are peer-reviewed compilations of chemical information.

EPA used additional references, including the following, to obtain specific types of information and to supplement the information from ATSDR and the HSDB:

- EPA's Six-Year Reviews (drinking water data) (USEPA 2009a; USEPA 2009b).
- Food and Drug Administration (FDA) Total Diet Study (USFDA 2015).
- FDA Everything Added to Food in the United States (USFDA 2013).
- EPA National Lake Fish Tissue Study (USEPA 2009c).
- EPA Toxic Release Inventory (USEPA 2015g).
- International Bottled Water Association Standards of Quality (IBWA 2012).
- National Oceanic and Atmospheric Administration (NOAA) Mussel Watch (NOAA 2014).
- Additional sources as needed.

To determine the RSC to be used in the AWQC calculation, EPA then used the information compiled for each chemical to address the questions posed in the Exposure Decision Tree. Some of the important items evaluated in the Exposure Decision Tree follow:

- The adequacy of the data available for each relevant exposure source and pathway.
- The availability of sufficient information to characterize the likelihood of exposure to relevant sources.
- Whether there are significant known or potential uses/sources other than the source of concern (i.e., ambient water and fish/seafood from those waters).
- Whether information on each source is available to make a characterization of exposure.

In cases where there is a lack of environmental or exposure data, or both, the Exposure Decision Tree approach results in a recommended RSC of 20 percent. This 20 percent value for the RSC may be replaced where sufficient data are available to develop a scientifically defensible alternative value. When appropriate, if scientific data demonstrating that sources and routes of exposure other than water and fish from inland and nearshore waters are not anticipated for the pollutant in question, the RSC may be raised to 80 percent based on the available data (USEPA 2000a).

6.2 Chemical-specific RSC

EPA derived recommended AWQC for this chemical using both noncarcinogenic and carcinogenic toxicity endpoints (RfD and CSF). For comparative purposes only, a default RSC of 20 percent was applied for AWQC derivation for noncarcinogenic effects (USEPA 2000a).

7 Criteria Derivation: Analysis

Table 1 summarizes the model inputs used to derive the 2015 updated human health AWQC that are protective of exposure to benzene from consuming drinking water and eating fish and shellfish (organisms) from inland and nearshore waters. The criteria calculations are presented below. These updated criteria recommendations are based on the 2000 Methodology (USEPA 2000a) and the updated exposure assumptions described above. (See section 4, Exposure Factors; section 5, Hazard Identification and Dose Response; and section 6, Relative Source Contribution.)

Table 1. Summary of Input Parameters for 2015 Human Health AWQC for Benzene

Input Parameter		Value
RfD		0.0005 mg/kg-d
CSF		0.015 per mg/kg-d to 0.055 per mg/kg-d
RSC		0.20
BW		80.0 kg
DI		2.4 L/d
FCR	TL2	0.0076 kg/d
	TL3	0.0086 kg/d
	TL4	0.0051 kg/d
BAF	TL2	3.6 L/kg
	TL3	4.5 L/kg
	TL4	5.0 L/kg

7.1 AWQC for Noncarcinogenic Toxicological Effects

For consumption of water and organisms:

$$\text{AWQC (µg/L)} = \frac{\text{toxicity value (RfD [mg/kg-d]} \times \text{RSC)} \times \text{BW (kg)} \times 1{,}000\ (\mu g/mg)}{\text{DI (L/d)} + \sum_{i=2}^{4} (\text{FCR}_i\ (\text{kg/d}) \times \text{BAF}_i\ (\text{L/kg}))}$$

$$= \frac{0.0005\ \text{mg/kg-d} \times 0.20 \times 80.0\ \text{kg} \times 1{,}000\ \mu g/mg}{2.4\ \text{L/d} + ((0.0076\ \text{kg/d} \times 3.6\ \text{L/kg}) + (0.0086\ \text{kg/d} \times 4.5\ \text{L/kg}) + (0.0051\ \text{kg/d} \times 5.0\ \text{L/kg}))}$$

$$= 3.21\ \mu g/L$$

$$= 3\ \mu g/L\ \text{(rounded)}$$

For consumption of organisms only:

$$\text{AWQC } (\mu g/L) = \frac{\text{toxicity value (RfD [mg/kg-d] } \times \text{ RSC)} \times \text{BW (kg)} \times 1{,}000\ (\mu g/mg)}{\sum_{i=2}^{4} (FCR_i\ (kg/d) \times BAF_i\ (L/kg))}$$

$$= \frac{0.0005\ mg/kg\text{-}d \times 0.20 \times 80.0\ kg \times 1{,}000\ \mu g/mg}{(0.0076\ kg/d \times 3.6\ L/kg) + (0.0086\ kg/d \times 4.5\ L/kg) + (0.0051\ kg/d \times 5.0\ L/kg)}$$

$$= 87.4\ \mu g/L$$

$$= 90\ \mu g/L \text{ (rounded)}$$

7.2 AWQC for Carcinogenic Toxicological Effects

For consumption of water and organisms:

$$\text{AWQC } (\mu g/L) = \frac{\text{toxicity value } (10^{-6} / \text{CSF}) \text{ [mg/kg-d]} \times \text{BW (kg)} \times 1{,}000\ (\mu g/mg)}{\text{DI (L/d)} + \sum_{i=2}^{4} (FCR_i\ (kg/d) \times BAF_i\ (L/kg))}$$

- Lower CSF

$$= \frac{(10^{-6} / 0.015)\ mg/kg\text{-}d \times 80.0\ kg \times 1{,}000\ \mu g/mg}{2.4\ L/d + ((0.0076\ kg/d \times 3.6\ L/kg) + (0.0086\ kg/d \times 4.5\ L/kg) + (0.0051\ kg/d \times 5.0\ L/kg))}$$

$$= 2.141\ \mu g/L$$

$$= 2.1\ \mu g/L \text{ (rounded)}$$

- Upper CSF

$$= \frac{(10^{-6} / 0.055)\ mg/kg\text{-}d \times 80.0\ kg \times 1{,}000\ \mu g/mg}{2.4\ L/d + ((0.0076\ kg/d \times 3.6\ L/kg) + (0.0086\ kg/d \times 4.5\ L/kg) + (0.0051\ kg/d \times 5.0\ L/kg))}$$

$$= 0.5838\ \mu g/L$$

$$= 0.58\ \mu g/L \text{ (rounded)}$$

For consumption of organisms only:

$$AWQC\ (\mu g/L) = \frac{\text{toxicity value } (10^{-6} / CSF)\ [mg/kg\text{-}d] \times BW\ (kg) \times 1{,}000\ (\mu g/mg)}{\sum_{i=2}^{4} (FCR_i\ (kg/d) \times BAF_i\ (L/kg))}$$

- Lower CSF

$$= \frac{(10^{-6} / 0.015)\ mg/kg\text{-}d \times 80.0\ kg \times 1{,}000\ \mu g/mg}{(0.0076\ kg/d \times 3.6\ L/kg) + (0.0086\ kg/d \times 4.5\ L/kg) + (0.0051\ kg/d \times 5.0\ L/kg)}$$

$$= 58.25\ \mu g/L$$

$$= 58\ \mu g/L\ (rounded)$$

- Upper CSF

$$= \frac{(10^{-6} / 0.055)\ mg/kg\text{-}d \times 80.0\ kg \times 1{,}000\ \mu g/mg}{(0.0076\ kg/d \times 3.6\ L/kg) + (0.0086\ kg/d \times 4.5\ L/kg) + (0.0051\ kg/d \times 5.0\ L/kg)}$$

$$= 15.89\ \mu g/L$$

$$= 16\ \mu g/L\ (rounded)$$

7.3 AWQC Summary

EPA derived the AWQC for benzene using both noncarcinogenic and carcinogenic toxicity endpoints. The updated human health AWQC for noncarcinogenic effects for benzene are **3 µg/L** for consumption of water and organisms and **90 µg/L** for consumption of organisms only. The updated human health AWQC for carcinogenic effects (at a 10^{-6} cancer risk level) for benzene are **0.58–2.1 µg/L** for consumption of water and organisms and **16–58 µg/L** for consumption of organisms only. EPA recommends the lower AWQC, based on the carcinogenic effects of benzene, as the updated human health AWQC (Table 2). These updated criteria replace EPA's previously published values (USEPA 2002b).

Table 2. Summary of EPA's Previously Recommended (2002) and Updated (2015) Human Health AWQC for Benzene

	2002 Human Health AWQC	2015 Human Health AWQC
Water and Organism	0.61 - 2.2 µg/L	**0.58 - 2.1 µg/L**
Organism Only	14 - 51 µg/L	**16 - 58 µg/L**

These AWQC are intended to be protective of the general adult population from an increased cancer risk due to exposure to benzene at a 10^{-6}, or one in one million, risk level. The 10^{-6} risk level associated with the AWQC represents the concentration that would be expected to increase an individual's lifetime cancer risk from exposure to the particular pollutant by no more than one chance in one million, regardless of the additional lifetime cancer risk due to exposure, if any, to that particular substance from other sources.

8 Criteria Characterization

The updated 2015 human health AWQC for benzene take into account current data on health effects and exposure input parameters, consistent with the 2000 Methodology (USEPA 2000a). The following paragraphs describe the individual influence of each of the revised inputs and exposure assumptions on the overall change in value.

Body Weight

EPA's updated AWQC assume a higher BW compared to the previously recommended 2002 criteria, reflecting a recent rise in average adult BW among the U.S. population. The updated BW assumption of 80.0 kg, based on recent survey data from the 1999−2006 NHANES data, is 10 kg greater than the previous assumption of 70 kg. Assuming all other input parameters remain constant, a higher average BW in the AWQC calculations (Eqs. 1 and 2 above) results in higher AWQC. That is, as BW increases, the level of a contaminant in water at or below which negative health effects are not anticipated from a lifetime of exposure also increases.

Drinking Water Intake

The updated DI assumption is 2.4 L/d, which is higher than the previously recommended rate of 2 L/d. Assuming all other input parameters remain constant, a higher DI assumption in the AWQC calculations (Eqs. 1 and 2 above) results in lower AWQC. That is, as DI increases, and thus overall exposure increases, the level of a contaminant in water at or below which negative health effects are not anticipated from a lifetime of exposure decreases.

Fish Consumption Rate

The updated FCR for fish and shellfish from inland and nearshore waters is 22.0 g/d; the TL-specific FCRs are 7.6 g/d, 8.6 g/d, and 5.1 g/d for TLs 2, 3, and 4, respectively. The previously recommended FCR was 17.5 g/d. Assuming all other input parameters remain constant, a higher FCR assumption in the AWQC calculations (Eqs. 1 and 2 above) results in lower AWQC. That is, as fish consumption increases, and thus overall exposure increases, the level of a contaminant in water at or below which negative health effects are not anticipated from a lifetime of exposure decreases.

Bioaccumulation Factor

The national lower (TL2), mid (TL3), and upper (TL4) TL BAFs used in the updated AWQC (Eqs. 1 and 2 above) are 3.6, 4.5, and 5.0 L/kg wet-weight, respectively. These BAFs were derived using EPA's 2000 Methodology (USEPA 2000a) and its Technical Support Document, Volume 2 (USEPA 2003a). These national TL BAFs replace EPA's previously recommended BCF of 5.2 L/kg.

As an additional line of evidence, EPA used model-estimated BAFs from the Estimation Program Interface (EPI) Suite (USEPA 2012) to support field-measured or predicted BAFs developed using the four methods described above. The BCFBAF program within EPI Suite estimates fish BAFs by using K_{ow} and biotransformation data from a model designed by Arnot and Gobas (2003). The model includes mechanistic processes for bioaccumulation, such as chemical uptake from the

water at the gill surface and from the diet, chemical elimination at the gill surface, fecal egestion, growth dilution, and metabolic biotransformation. Other processes included in the calculations are bioavailability in the water column (only the freely dissolved fraction can bioconcentrate) and absorption efficiencies at the gill and in the gastrointestinal tract. The model requires the K_{ow} of the chemical and the normalized whole-body metabolic biotransformation rate constant as input parameters to predict BAF values. The EPI Suite model estimates are as follows:

TL2 = 8.9 L/kg
TL3 = 10.01 L/kg
TL4 = 14.79 L/kg

Assuming all other input parameters remain constant, lower BAFs or BCFs result in higher AWQC. That is, as bioaccumulation or bioconcentration of a contaminant in fish and shellfish decreases, the level of a contaminant in water at or below which negative health effects are not anticipated from a lifetime of exposure increases.

The utilization of a national-level BAF rather than a BCF better represents the amount of a contaminant accumulating in an organism because it accounts not only for the organism's exposure to the pollutant in the water column, but also from the food chain and surrounding environment as well as biotransformation of the pollutant in the organism due to metabolic processes. The utilization of the three TLs of fish and shellfish consumed, as opposed to representing all TLs of fish and shellfish consumed by a single value, allows for better exposure representation.

Reference Dose

In place of an RfD, EPA selected a chronic oral MRL of 0.0005 mg/kg-d for benzene based on a 2007 ATSDR assessment (ATSDR 2007). EPA used the MRL of 0.0005 mg/kg-d to derive AWQC for noncarcinogenic effects. EPA did not derive AWQC for noncarcinogenic effects of benzene in its previous criteria update (USEPA 2002d).

Cancer Slope Factor

EPA retained a CSF range of 0.015 per mg/kg-d to 0.055 per mg/kg-day for benzene based on a 2000 EPA IRIS assessment (USEPA 2000c; USEPA 2002d). EPA used this CSF range to derive AWQC for carcinogenic effects. Assuming all other input parameters remain constant, no change in the values used for the CSF in the AWQC calculations (Eqs. 1 and 2) results in no change in AWQC.

Relative Source Contribution

An RSC of 20 percent was used for comparative purposes to calculate AWQC for noncarcinogenic effects. Previously, the recommended AWQC were derived for carcinogenic effects only, and therefore, an RSC was not included.

9 Chemical Names and Synonyms

- Benzene (CAS Number 71-43-2)
- Benzol
- Coal naphtha
- Cyclohexatriene
- Phene
- Phenyl hydride
- Polystream
- Pyrobenzol

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United States Environmental Protection Agency

National Recommended Water Quality Criteria - Aquatic Life Criteria Table

Related Information

- Aquatic Life Criteria and Methods for Toxics
- Human Health Criteria Table
- Organoleptic Effects Criteria Table

This table contains the most up to date criteria for aquatic life ambient water quality criteria. Aquatic life criteria for toxic chemicals are the highest concentration of specific pollutants or parameters in water that are not expected to pose a significant risk to the majority of species in a given environment or a narrative description of the desired conditions of a water body being "free from" certain negative conditions. The table below lists EPA's recommended aquatic life criteria. State and tribal governments may use these criteria or use them as guidance in developing their own.

Select pollutant name for current criteria document.

On this page:

National Recommended Aquatic Life Criteria table

Pollutant (P = Priority Pollutant)	CAS Number	Freshwater CMC[1] (acute) (µg/L)	Freshwater CCC[2] (chronic) (µg/L)	Publication Year	Notes
Acrolein (P)	107028	3ug/L	3ug/L	2009	
Aesthetic Qualities	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Aldrin (P)	309002	3.0	—	1980	These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines.
Alkalinity	—	—	20000	1986	The CCC of 20mg/L is a minimum value except where alkalinity is naturally lower, in which case the criterion cannot be lower than 25% of the natural level.

alpha-Endosulfan (P)	959988	0.22	0.056	1980	These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines. This value was derived from data for endosulfan and is most appropriately applied to the sum of alpha-endosulfan and beta-endosulfan.
Aluminum pH 5.0 - 10.5	7429905	--	--	2018	The criteria is based on the water chemistry data (for pH, hardness and DOC) entered into the criteria calculator for a given location.
Ammonia	7664417	—	—	2013 (Freshwater), 1989 (Saltwater)	Freshwater criteria are ph, temperature and life-stage dependent. Saltwater criteria are pH and temperature dependent.
Arsenic	7440382	340	150	1995	This recommended water quality criterion was derived from data for arsenic (III), but is applied here to total arsenic. Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the

					water column. Refer to Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria.
Atrazine	1912249				
Bacteria	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
beta-Endosulfan (P)	33213659	0.22	0.056	1980	These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines. This value was derived from data for endosulfan and is most appropriately applied to the sum of alpha-endosulfan and beta-endosulfan.
Boron	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Cadmium (P)	7440439	1.8	0.72	2016	Freshwater acute and chronic criteria are hardness-dependent and were normalized to a

					hardness of 100 mg/L as CaCO3 to allow the presentation of representative criteria values. . Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column. Refer to Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria.
Carbaryl	63252	2.1	2.1	2012	
Chlordane (P)	57749	2.4	0.0043	1980	These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines.
Chloride	16887006	860000	230000	1988	
Chlorine	7782505	19	11	1986	
Chlorpyrifos	2921882	0.083	0.041	1986	

Chromium (III) (P)	16065831	570	74	1995	Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column. Refer to Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria. The freshwater criterion for this metal is expressed as a function of hardness (mg/L). The value given here corresponds to a hardness of 100 mg/L.
Chromium (VI) (P)	18540299	16	11	1995	Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column. Refer to Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria.
Color	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Copper (P)	7440508	—	—	2007	Freshwater criteria calculated using the Biotic Ligand Model. Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column. Refer to Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria.

Cyanide (P)	57125	22	5.2	1985	These recommended water quality criteria are expressed as µg free cyanide (CN/L).
Demeton	8065483	—	0.1	1985	
Diazinon	333415	0.17ug/L	0.17ug/L	2005	
Dieldrin (P)	60571	0.24	0.056	1995	The freshwater CCC criterion and both Saltwater criteria are based are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines.
Endrin (P)	72208	0.086	0.036	1995	The derivation of the CCC for this pollutant did not consider exposure through the diet, which is probably important for aquatic life occupying upper trophic levels.
gamma-BHC (Lindane) (P)	58899	0.95	—	1995	The Saltwater CCC criterion is based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to

					obtain a value that is more comparable to a CMC derived using the 1985 Guidelines.
Gases, Total Dissolved	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Guthion	86500	—	0.01	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Hardness	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Heptachlor (P)	76448	0.52	0.0038	1980	These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines.
Heptachlor Epoxide (P)	1024573	0.52	0.0038	1981	These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a

					value that is more comparable to a CMC derived using the 1985 Guidelines. This value was derived from data for heptachlor and there was insufficient data to determine relative toxicities of heptachlor and heptachlor epoxide.
Iron	7439896	—	1000	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Lead (P)	7439921	65	2.5	1984	Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column. Refer to Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria. The freshwater criterion for this metal is expressed as a function of hardness (mg/L). The value given here corresponds to a hardness of 100 mg/L.
Malathion	121755	—	0.1	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Mercury (P)	7439976 22967926	1.4	0.77	1995	Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column. Refer to Office of Water Policy

					and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria.
Methoxychlor	72435	—	0.03	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Methyl Tertiary-Butyl Ether (MTBE)					
Mirex	2385855	—	0.001	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Nickel (P)	7440020	470	52	1995	Freshwater and saltwater criteria for metals are expressed in terms of the dissolved metal in the water column. Refer to Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria. The freshwater criterion for this metal is expressed as a function of hardness (mg/L). The value given here corresponds to a hardness of 100 mg/L.
Nonylphenol	84852153	28 ug/L	6.6 ug/L	2005	
Nutrients	—	—	—	—	Refer to EPA's Ecoregional criteria for Total Phosphorus, Total Nitrogen, Chlorophyll *a* and

					Water Clarity (Secchi depth for lakes; turbidity for streams and rivers) (and Level III Ecoregional criteria)
Oil and Grease	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Oxygen, Dissolved Freshwater Oxygen, Dissolved Saltwater	7782447	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for freshwater. For saltwater, Refer to Aquatic Life Criteria for Dissolved Oxygen (Saltwater) Cape Cod to Cape Hatteras.
Parathion	56382	0.065	0.013	1995	
Pentachlorophenol (P)	87865	19	15	1995	Freshwater aquatic life values for pentachlorophenol are expressed as a function of pH and values displayed in table correspond to a pH of 7.8.
pH	—	—	6.5 – 9	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement. For open ocean waters where the depth is substantially greater than the euphotic zone, the pH should not be changed more than 0.2 units from the naturally occurring variation or any case outside the range of 6.5 to 8.5. For shallow, highly productive coastal and estuarine areas where naturally occurring pH variations

					approach the lethal limits of some species, changes in pH should be avoided but in any case should not exceed the limits established for fresh water, i.e., 6.5-9.0.
Phosphorus Elemental	7723140	—	—	1986	
Polychlorinated Biphenyls (PCBs) (P)	—	—	0.014	—	This criterion applies to total PCBs, (e.g., the sum of all congener or all isomer or homolog or Aroclor analyses.)
Selenium (P)	7782492	—	---	2016 Freshwater 1999 Saltwater	Refer to Aquatic Life Ambient Water Quality Criterion for Selenium -Freshwater 2016 for narrative statement.
Silver (P)	7440224	3.2	—	1980	
Solids Suspended and Turbidity	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.
Sulfide-Hydrogen Sulfide	7783064	—	2.0	1986	
Tainting Substances	—	—	—	1986	Refer to Quality Criteria for Water, 1986 ("Gold Book") for narrative statement.

Temperature	—	—	—	1986	Criteria is species dependent. Refer to Quality Criteria for Water, 1986 ("Gold Book").
Toxaphene (P)	8001352	0.73	0.0002	1986	
Tributyltin (TBT)	—	0.46	0.072	2004	
Zinc (P)	7440666	120	120	1995	
4,4'-DDT (P)	50293	1.1	0.001	1980	

Appendix A

Conversion Factors for Dissolved Metals

Metal	Freshwater CMC	Freshwater CCC	Saltwater CMC	Saltwater CCC
Arsenic	1.000	1.000	1.000	1.000
Cadmium	1.136672-[(ln hardness)(0.041838)]	1.101672-[(ln hardness)(0.041838)]	0.994	0.994

Conversion Factors for Dissolved Metals				
Metal	**Freshwater CMC**	**Freshwater CCC**	**Saltwater CMC**	**Saltwater CCC**
Chromium III	0.316	0.860	—	—
Chromium VI	0.982	0.962	0.993	0.993
Copper	0.960	0.960	0.83	0.83
Lead	1.46203-[(ln hardness)(0.145712)]	1.46203-[(ln hardness)(0.145712)]	0.951	0.951
Mercury	0.85	0.85	0.85	0.85
Nickel	0.998	0.997	0.990	0.990
Selenium	—	—	0.998	0.998
Silver	0.85	—	0.85	—

Conversion Factors for Dissolved Metals

Metal	Freshwater CMC	Freshwater CCC	Saltwater CMC	Saltwater CCC
Zinc	0.978	0.986	0.946	0.946

Appendix B

Parameters for Calculating Freshwater Dissolved Metals Criteria That Are Hardness-Dependent

Chemical	mA	bA	mC	bC	Freshwater Conversion Factors (CF)	
					CMC	CCC
Cadmium	0.9789	-3.866	0.7977	-3.909	1.136672-[(*ln*hardness)(0.041838)]	1.101672-[(*ln*hardness)(0.041838)]
Chromium III	0.8190	3.7256	0.8190	0.6848	0.316	0.860

Parameters for Calculating Freshwater Dissolved Metals Criteria That Are Hardness-Dependent

Chemical	mA	bA	mC	bC	Freshwater Conversion Factors (CF)	
					CMC	CCC
Lead	1.273	-1.460	1.273	-4.705	1.46203-[(*ln*hardness)(0.145712)]	1.46203-[(*ln*hardness)(0.145712)]
Nickel	0.8460	2.255	0.8460	0.0584	0.998	0.997
Silver	1.72	-6.59	—	—	0.85	—
Zinc	0.8473	0.884	0.8473	0.884	0.978	0.986

Hardness-dependant metals' criteria may be calculated from the following:
CMC (dissolved) = exp{mA [*ln*(hardness)]+ bA} (CF)
CCC (dissolved) = exp{mC [*ln*(hardness)]+ bC} (CF)

1/ CMC: Criterion Maximum Concentration
2/ CCC: Criterion Continuous Concentration

National Recommended Water Quality Criteria- Aquatic Life Criteria Table

As prepared by New Mexico Environment Department Surface Water Quality Bureau

For Informational Purposes Only

February 2021

Acute		Chronic	
45	304(a) Criteria is equavlent to WQS - No changes needed	38	304(a) Criteria is equavlent to WQS - No changes needed
6	304(a) Criteria is MORE Stringent, need to consider adoption	14	304(a) Criteria is MORE Stringent, need to consider adoption
2	304(a) Criteria is LESS Stringent	1	304(a) Criteria is LESS Stringent
0	Have a 304(a) numeric criteria but not found in Table J (or elsewhere)	0	Have a 304(a) numeric criteria but not found in Table J (or elsewhere)
8	Have a Hardness Based Criteria under 900	8	Have a Hardness Based Criteria under 900

		Acute			Chronic			
Pollutant (P = Priority Pollutant)	**CAS Number**	**Freshwater CMC1 (acute) (ug/L)**	**NM WQS Criteria Acute**	**Update Needed**	**Freshwater CCC2 (chronic) (µg/L)**	**NM WQS Criteria Chronic**	**Update Needed**	**Publication Year**
Aldrin (P)	309002	3	3	Equivalent	—	—	Equivalent	1980
alpha-Endosulfan (P)	959988	0.22	0.22	Equivalent	0.056	0.056	Equivalent	1980
beta-Endosulfan (P)	33213659	0.22	0.22	Equivalent	0.056	0.056	Equivalent	1980
Chlordane (P)	57749	2.4	2.4	Equivalent	0.0043	0.0043	Equivalent	1980
Heptachlor (P)	76448	0.52	0.52	Equivalent	0.0038	0.0038	Equivalent	1980
Lead (P)	7439921	65	hardness based	900 Hardness based	2.5	hardness based	900 Hardness based	1980
Silver (P)	7440224	3.2	hardness based	900 Hardness based	—	hardness based	900 Hardness based	1980
4,4'-DDT (P)	50293	1.1	1.1	Equivalent	0.001	0.001	Equivalent	1980
Heptachlor Epoxide (P)	1024573	0.52	0.52	Equivalent	0.0038	0.0038	Equivalent	1981
Cyanide (P)	57125	22	22	Equivalent	5.2	5.2	Equivalent	1985
Demeton	8065483	—	—	Equivalent	0.1	—	304a more stringent	1985
Aesthetic Qualities	—	—	—	Equivalent	—	—	Equivalent	1986
Alkalinity	—	—	—	Equivalent	20000	—	304a more stringent	1986
Bacteria	—	—	—	Equivalent	—	—	Equivalent	1986
Boron	—	—	—	Equivalent	—	—	Equivalent	1986
Chlorine	7782505	19	19	Equivalent	11	11	Equivalent	1986
Chlorpyrifos	2921882	0.083	—	304a more stringent	0.041	—	304a more stringent	1986
Color	—	—	—	Equivalent	—	—	Equivalent	1986
Gases, Total Dissolved	—	—	—	Equivalent	—	—	Equivalent	1986
Guthion	86500	—	—	Equivalent	0.01	—	304a more stringent	1986
Hardness	—	—	—	Equivalent	—	—	Equivalent	1986
Iron	7439896	—	—	Equivalent	1000	—	304a more stringent	1986
Malathion	121755	—	—	Equivalent	0.1	—	304a more stringent	1986

Pollutant (P = Priority Pollutant)	CAS Number	Freshwater CMC1 (acute) (ug/L)	NM WQS Criteria Acute	Update Needed	Freshwater CCC2 (chronic) (µg/L)	NM WQS Criteria Chronic	Update Needed	Publication Year
Methoxychlor	72435	—	—	Equivalent	0.03	—	304a more stringent	1986
Mirex	2385855	—	—	Equivalent	0.001	—	304a more stringent	1986
Oil and Grease	—	—	—	Equivalent	—	—	Equivalent	1986
Oxygen, Dissolved Freshwater	7782447	—	—	Equivalent	—	—	Equivalent	1986
pH	—	—	—	Equivalent	6.5 – 9	6.5 – 9	Equivalent	1986
Phosphorus Elemental	7723140	—	—	Equivalent	—	—	Equivalent	1986
Solids Suspended and Turbidity	—	—	—	Equivalent	—	—	Equivalent	1986
Sulfide-Hydrogen Sulfide	7783064	—	—	Equivalent	2	—	304a more stringent	1986
Tainting Substances	—	—	—	Equivalent	—	—	Equivalent	1986
Temperature	—	—	—	Equivalent	—	—	Equivalent	1986
Toxaphene (P)	8001352	0.73	0.73	Equivalent	0.0002	0.0002	Equivalent	1986
Chloride	16887006	860000	—	304a more stringent	230000	—	304a more stringent	1988
Arsenic	7440382	340	340	Equivalent	150	150	Equivalent	1995
Chromium (III) (P)	16065831	570	hardness based	900 Hardness based	74	hardness based	900 Hardness based	1995
Chromium (VI) (P)	18540299	16	16	Equivalent	11	11	Equivalent	1995
Dieldrin (P)	60571	0.24	0.24	Equivalent	0.056	0.056	Equivalent	1995
Endrin (P)	72208	0.086	0.086	Equivalent	0.036	0.036	Equivalent	1995
gamma-BHC (Lindane) (P)	58899	0.95	0.95	Equivalent	—	—	Equivalent	1995
Mercury (P)	7439976	1.4	1.4	Equivalent	0.77	0.77	Equivalent	1995
Nickel (P)	7440020	470	hardness based	900 Hardness based	52	hardness based	900 Hardness based	1995
Parathion	56382	0.065	—	304a more stringent	0.013	—	304a more stringent	1995
Pentachlorophenol (P)	87865	19	19	Equivalent	15	15	Equivalent	1995
Zinc (P)	7440666	120	hardness based	900 Hardness based	120	hardness based	900 Hardness based	1995
Tributyltin (TBT)	—	0.46	—	304a more stringent	0.072	—	304a more stringent	2004
Diazinon	333415	0.17	0.17	Equivalent	0.17	0.17	Equivalent	2005
Nonylphenol	84852153	28	28	Equivalent	6.6	6.6	Equivalent	2005
Copper (P)	7440508	—	hardness based	900 Hardness based	—	hardness based	900 Hardness based	2007
Acrolein (P)	107028	3	—	304a more stringent	3	—	304a more stringent	2009
Carbaryl	63252	2.1	—	304a more stringent	2.1	—	304a more stringent	2012
Cadmium (P)	7440439	1.8	hardness based	900 Hardness based	0.72	hardness based	900 Hardness based	2016
Aluminum pH 5.0 - 10.5	7429905	—	hardness based	900 Hardness based	—	hardness based	900 Hardness based	2018
Nutrients	—	—	—	Equivalent	—	—	Equivalent	—

Pollutant (P = Priority Pollutant)	CAS Number	Freshwater CMC1 (acute) (ug/L)	NM WQS Criteria Acute	Update Needed	Freshwater CCC2 (chronic) (µg/L)	NM WQS Criteria Chronic	Update Needed	Publication Year
Polychlorinated Biphenyls (PCBs) (P)	—	—	2	304a less stringent	0.014	0.014	Equivalent	—
Ammonia	7664417	—	—	Equivalent	—	—	Equivalent	2013 (Freshwater), 1989 (Saltwater)
Selenium (P)	7782492	—	20	304a less stringent	—	5	304a less stringent	2016 Freshwater, 1999 Saltwater
Atrazine	1912249	—	—	Equivalent			Equivalent	
Methylmercury	22967926	—	—	Equivalent			Equivalent	
Methyl Tertiary-Butyl Ether (MTBE)		—	—	Equivalent			Equivalent	

United States
Environmental Protection
Agency

Office of Water
4304T

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December 2018

FINAL
AQUATIC LIFE AMBIENT WATER QUALITY CRITERIA FOR ALUMINUM
2018

EPA-822-R-18-001

FINAL

AQUATIC LIFE

AMBIENT WATER QUALITY CRITERIA FOR

ALUMINUM - 2018

(CAS Registry Number 7429-90-05)

December 2018

U.S. ENVIRONMENTAL PROTECTION AGENCY
OFFICE OF WATER
OFFICE OF SCIENCE AND TECHNOLOGY
HEALTH AND ECOLOGICAL CRITERIA DIVISION
WASHINGTON, D.C.

NOTICES

This document provides information to states and tribes authorized to establish water quality standards under the Clean Water Act (CWA), to protect aquatic life from toxic effects of aluminum. Under the CWA, states and tribes are to establish water quality criteria to protect designated uses. State and tribal decision makers retain the discretion to adopt approaches that are scientifically defensible that differ from these criteria to reflect site-specific conditions. While this document contains the Environmental Protection Agency's (EPA) scientific recommendations regarding ambient concentrations of aluminum that protect aquatic life, the Aluminum Criteria Document does not substitute for the CWA or the EPA's regulations; nor is it a regulation itself. Thus, the document does not impose legally binding requirements on the EPA, states, tribes, or the regulated community, and might not apply to a particular situation based upon the circumstances. The EPA may update this document in the future. This document has been approved for publication by the Office of Science and Technology, Office of Water, U.S. Environmental Protection Agency.

Mention of trade names or commercial products does not constitute endorsement or recommendation for use. This document can be downloaded from: https://www.epa.gov/wqc/aquatic-life-criteria-and-methods-toxics.

FOREWORD

The Clean Water Act (CWA) Section 304(a)(l) (P.L. 95-217) directs the Administrator of the Environmental Protection Agency (EPA) to publish water quality criteria that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that might be expected from the presence of pollutants in any body of water, including groundwater. This document is a final ambient water quality criteria (AWQC) document for the protection of aquatic life based upon consideration of all available information relating to effects of aluminum on aquatic organisms.

The term Water Quality Criteria is used in two sections of the CWA, Section 304(a)(l) and Section 303(c)(2). The term has different meanings in each section. In Section 304, the term represents a non-regulatory, scientific assessment of ecological and human health effects. Criteria presented in this document are such a scientific assessment of ecological effects. In section 303, if water quality criteria associated with specific surface water uses are adopted by a state or the EPA as water quality standards, they become the CWA water quality standards applicable in ambient waters within that state or authorized tribe. Water quality criteria adopted in state water quality standards could have the same numerical values as recommended criteria developed under section 304. However, in some situations states might want to adjust water quality criteria developed under section 304 to reflect local water chemistry or ecological conditions. Alternatively, states and authorized tribes may develop numeric criteria based on other scientifically defensible methods, but the criteria must be protective of designated uses. It is not until their adoption as part of state water quality standards, and subsequent approval by the EPA under section 303(c), that criteria become CWA applicable water quality standards. Guidelines to assist the states and authorized tribes in modifying the criteria presented in this document are contained in the Water Quality Standards Handbook (U.S. EPA 2014).

This document presents recommendations only. It does not establish or affect legal rights or obligations. It does not establish a binding requirement and cannot be finally determinative of the issues addressed. The EPA will make decisions in any particular situation by applying the CWA and the EPA regulations on the basis of specific facts presented and scientific information then available.

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ACRONYMS

ACR	Acute to Chronic Ratio
AIC	Akaike Information Criterion
AVS	Acid Volatile Sulfide
AWQC	Ambient Water Quality Criteria
BAF	Bioaccumulation Factor
BCF	Bioconcentration Factor
BIC	Bayesian Information Criterion
CCC	Criterion Continuous Concentration
CMC	Criterion Maximum Concentration
CV	Chronic Value (expressed in this document as an EC_{20})
CWA	Clean Water Act
DOC	Dissolved Organic Carbon
ECOTOX	Ecotoxicology Database
EC_x	Effect Concentration at X Percent Effect Level
ELS	Early-Life Stage
EPA	Environmental Protection Agency
EU	European Union
FACR	Final Acute-to-Chronic Ratio
FAV	Final Acute Value
FCV	Final Chronic Value
FDA	US Food and Drug Administration
GMAV	Genus Mean Acute Value
GMCV	Genus Mean Chronic Value
IC_x	Inhibitory Concentration at X Percent Level
LC_x	Lethal Concentration at X Percent Survival Level
LOEC	Lowest Observed Effect Concentration
MATC	Maximum Acceptable Toxicant Concentration (expressed mathematically as the geometric mean of the NOEC and LOEC)
MDR	Minimum Data Requirement
MLR	Multiple Linear Regression
NAWQA	USGS National Water Quality Assessment Program
NOAA	National Oceanic and Atmospheric Administration
NOEC	No Observed Effect Concentration
NPDES	National Pollutant Discharge Elimination System
QA/QC	Quality Assurance and Quality Control
SMAV	Species Mean Acute Value
SMCV	Species Mean Chronic Value
TMDL	Total Maximum Daily Load
TRAP	Toxicity Relationship Analysis Program
US	United States
USGS	United States Geological Survey
WQC	Water Quality Criteria
WQS	Water Quality Standards

EXECUTIVE SUMMARY

The U.S. Environmental Protection Agency (EPA) is updating its aquatic life ambient water quality criteria (AWQC) recommendation for aluminum, in accordance with the provisions of section 304(a) directing the EPA to revise AWQC from time to time to reflect the latest scientific knowledge. The recommended aluminum aquatic life AWQC were developed using peer reviewed methods and data that are acceptable for the derivation of criteria, as described in the EPA's 1985 "*Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*" (Stephan et al. 1985, referred to herein as "1985 Guidelines"). The previous aquatic life AWQC for aluminum were developed in 1988 (EPA 440/5-86-008). These 2018 final recommended aquatic life AWQC for aluminum supersedes the 1988 recommended criteria.

The 2017 draft aquatic life AWQC for aluminum were posted to the Federal Register (Docket ID: EPA-HQ-OW-2017-0260) in late July 2017 for public comment. The public comment period was open for 90 days and closed in late October 2017. Public comments received were incorporated and addressed in these final AWQC, where applicable. The EPA responses to all of the public comments can be found on the website for the aluminum criteria (https://www.epa.gov/wqc/aquatic-life-criteria-aluminum).

Literature searches for laboratory tests published from 1988 to 2017 identified new studies describing the toxicity of aluminum to aquatic life. The EPA supplemented these studies with additional data made available by researchers in late-2017 and 2018. The EPA conducted a full evaluation of available data to determine test acceptability for criteria development. Appendix A of "*Quality Criteria for Water 1986*" (U.S. EPA 1986) provides an in-depth discussion of the minimum requirements for data quality needed to develop AWQC for aquatic life.

This update to the recommended aluminum aquatic life AWQC establishes freshwater criteria magnitude values resulting from the interactions of aluminum and three water chemistry parameters: pH, total hardness, and dissolved organic carbon (DOC). It also expands the toxicity database to include those studies conducted in waters with pH values below 6.5. There were insufficient data to establish an estuarine/marine aluminum criteria.

Multiple linear regression (MLR) models were developed to characterize the bioavailability of aluminum in aquatic systems, based on the effects of pH, total hardness and

DOC on aluminum toxicity (DeForest et al. 2018a,b). These authors used a dataset comprised of 22 chronic tests with the fathead minnow (*Pimephales promelas*), and 23 chronic tests with an invertebrate (*Ceriodaphnia dubia*) to evaluate the ability of MLR models to predict chronic toxicity of aluminum as a function of pH, total hardness and DOC water chemistry conditions. These three parameters are considered to be the most influential for aluminum bioavailability and can be used to explain the range of differences in the observed toxicity values. These datasets were supplemented in 2018 with an additional nine *C. dubia* toxicity tests and nine *P. promelas* toxicity tests to expand the range of water chemistry conditions for model development (OSU 2018a,b,d). All of the toxicity test data used in the model were subjected to independent external expert peer review.

Two models, one for invertebrates and one for vertebrates, were used to normalize freshwater aluminum toxicity values. These separate models correspond to effects on invertebrates and vertebrates due to differing effects of pH, total hardness and DOC on aluminum bioavailability and toxicity, and therefore enable the criteria magnitudes to be calculated as a function of the unique chemistry conditions at a given site. The EPA conducted both independent external expert peer review and internal reviews of these models, published by DeForest et al. (2018a,b), to verify the results. The updated aluminum criteria were derived using these MLR models to normalize the freshwater acute and chronic toxicity data. The MLR equations applied to the acute toxicity data were those developed using chronic tests, with the expectation that the effect of water chemistry on bioavailability remains consistent across exposure duration.

Freshwater Criteria Update

The 1988 aluminum freshwater criteria (U.S. EPA 1988) are expressed as total recoverable aluminum. Acid soluble aluminum was considered but not used because the methods were not developed. These updated 2018 criteria are also based on total recoverable aluminum concentrations.

The 1988 criteria did not consider the variable effects of water chemistry on aluminum toxicity, but simply specified that the recommended criteria only applied to a pH range of 6.5 to 9.0. The 2018 final aluminum recommended AWQC take into account the effects of pH, total hardness and DOC on aluminum toxicity.

The 1988 freshwater acute criterion was based on data from eight species of invertebrates and seven species of fish for a total of 15 species grouped into 14 genera. This 2018 freshwater acute criterion update is based on data from 13 species of invertebrates, eight species of fish, and one species of frog for a total of 22 species grouped into 20 genera.

The freshwater acute criterion represents the concentration of aluminum at which approximately 95% of genera in a freshwater aquatic ecosystem should be protected if the one-hour average (duration) concentration of total aluminum is not exceeded more than once in three years (frequency). The magnitude of the criterion depends on the water chemistry conditions in the waterbody, using the MLR models to normalize the freshwater acute toxicity data. As a result, the acute criterion will vary with water chemistry conditions. Example acute criteria values for various water chemistry conditions are presented in **Appendix K** (*Recommended Criteria for Various Water Chemistry Conditions*) and can also be calculated with the Aluminum Criteria Calculator V.2.0[1].

The 1988 aluminum freshwater chronic dataset included two species of invertebrates and one fish species grouped into three genera. This 2018 criteria update includes new chronic data for an additional nine species, and consists of eight invertebrate and four fish species grouped into 12 genera. With the addition of one study from **Appendix H** (*Other Data on Effects of Aluminum to Freshwater Aquatic Organisms*), the Minimum Data Requirements (MDRs) for direct calculation (using a sensitivity distribution, as described in the 1985 Guidelines) of the Final Chronic Value (FCV) were fulfilled. This method does not require the use of an acute to chronic ratio (ACR).

Like the acute criterion, the freshwater chronic criterion is also dependent on the water chemistry of the waterbody. Therefore, it is also a function of the MLR models used to normalize the chronic toxicity data. Example chronic criteria (CCC) for various water chemistry conditions are presented in **Appendix K** (*Recommended Criteria for Various Water Chemistry Conditions*) and can also be calculated with the Aluminum Criteria Calculator V.2.0.

The empirical toxicity test data used to develop the MLR models were developed under a range of water chemistry conditions (for more detail, see **Section 4** of this document). The MLRs were then used to normalize all of the toxicity data used in the criteria calculations. MLR models

[1] https://www.epa.gov/wqc/aquatic-life-criteria-aluminum

are useful for characterizing trends in data, but should be used with caution when extrapolating beyond the range of data used for model development.

The bounds for pH of the models ranged from 6.0-8.7. The EPA criteria calculator is designed to allow the user to extrapolate beyond the pH values used to generate the MLR models. The criteria calculator can be used to address all waters within a pH range of 5.0 to 10.5. This is reflected in the criteria lookup tables in **Appendix K**. The EPA took this approach so that the recommended criteria can be calculated for, and will be protective of, a broader range of natural waters found in the U.S. Extrapolated criteria values outside of the empirical pH data tend to be more conservative (i.e., lower values) and will be more protective of the aquatic environment in situations where pH plays a critical role in aluminum toxicity. Criteria values generated outside of the range of the pH conditions of the toxicity tests underlying the MLR models are more uncertain than values within the pH conditions of the MLR toxicity tests, and thus should be considered carefully and used with caution.

The bounds for total hardness of the models ranged from 9.8 to 428 mg/L. Since a decrease in total hardness tends to increase aluminum toxicity, the EPA concludes that it is reasonable to extrapolate below the lower bound of the empirical hardness data of 9.8 mg/L to enable generation of more stringent criteria at low hardnesses. This is consistent with existing EPA approaches to address low end hardness values (U.S. EPA 2002). Therefore, hardness input values in the criteria calculator can be entered that are less than 9.8 mg/L down to a limit of 0.01 mg/L. However, hardness input values into the criteria calculator will be bounded at the approximate upper limit of the empirical MLR models' underlying hardness data, at a maximum of 430 mg/L total hardness (as $CaCO_3$). The user can input hardness values greater than 430 mg/L for total hardness into the criteria calculator, but the criteria magnitude will reach its maximum value at 430 mg/L total hardness (as $CaCO_3$), and criteria magnitudes will not increase or decrease by increasing the hardness above 430 mg/L total hardness (as $CaCO_3$). This is also consistent with existing EPA guidance on high end hardness caps (U.S. EPA 2002). This recommendation is reflected in the criteria lookup tables provided in **Appendix K**. The EPA took this approach to ensure that the recommended criteria are protective of a broader range of natural waters found in the U.S. Criteria values generated beyond the lower bound of the hardness conditions of the toxicity tests underlying the MLR models are more uncertain than values within the hardness bounds of the MLR toxicity test data.

The bounds for DOC of the models ranged from 0.08 to 12.3 mg/L. Since most natural waters contain some DOC, the lower bound of the empirical toxicity test data (0.08 mg/L) is the lowest value that can be entered into the criteria calculator; thus no extrapolation below the lowest empirical DOC of 0.08 mg/L is provided. Similar to hardness, the criteria values generated will be bounded at the upper limit of the empirical MLR models' underlying DOC data, at a maximum 12.0 mg/L DOC in the criteria calculator. The user can input DOC values greater than 12.0 mg/L into the calculator, but the criteria magnitude will reach its maximum value at 12.0 mg/L DOC, and criteria magnitudes will not increase or decrease by increasing the DOC above 12.0 mg/L. This limitation on the maximum DOC value is also reflected in the criteria lookup tables provided in **Appendix K**. This is consistent with the existing approach for hardness (U.S. EPA 2002) to provide for protection of aquatic organisms through the use of protective, conservative values under water chemistry conditions beyond the upper limits of the empirical toxicity test data.

In addition to **Appendix K** look-up tables, the EPA created a user-friendly **Aluminum Criteria Calculator V.2.0** (Aluminum Criteria Calculator V.2.0.xlsm) that allows users to enter site-specific values for pH, total hardness and DOC to calculate the appropriate recommended freshwater acute and chronic criteria magnitudes for site-specific parameters and will generate criteria magnitude values based on the bounds described above.

2018 Recommended Aluminum Aquatic Life AWQC and the 1988 Criteria[a]

Version	**Freshwater Acute** (1-hour, total aluminum)	**Freshwater Chronic** (4-day, total aluminum)
2018 AWQC (vary as a function of a site's pH, DOC and total hardness)	1-4,800 µg/L[b]	0.63-3,200 µg/L[b]
1988 AWQC (pH 6.5 – 9.0, across all total hardness and DOC ranges)	750 µg/L	87 µg/L

[a] Values are recommended not to be exceeded more than once every three years on average.
[b] Criteria values will be different under differing water chemistry conditions as identified in this document, as described in Appendix K and applied in the Aluminum Criteria Calculator.

Estuarine/Marine Criteria Update

As with the 1988 AWQC for aluminum, there are still insufficient data on estuarine and marine species to fulfill the MDRs as specified in the 1985 Guidelines. As a result, the EPA cannot recommend criteria for estuarine/marine waters at this time. The 1985 Guidelines require

that data from a minimum of eight families are needed to calculate an estuarine/marine Final Acute Value (FAV). New acute toxicity data for five families representing five species of estuarine/marine organisms are available for aluminum; no data were previously available. The most sensitive species was the polychaete worm (*Ctenodrilus serratus*) with a Species Mean Acute Value (SMAV) of 97.15 µg/L total aluminum, and the most tolerant species was a copepod (*Nitokra spinipes*) with a SMAV of 10,000 µg/L. No acceptable acute tests on estuarine/marine fish species were available. There are no estuarine/marine chronic toxicity data for fish or other genera that meet the test acceptability and quality assurance and quality control (QA/QC) principles as outlined in the 1985 Guidelines. Thus acute and chronic aluminum toxicity data for estuarine and marine species remain a data gap.

1 INTRODUCTION AND BACKGROUND

The United States Environmental Protection Agency (EPA) establishes national recommended Ambient Water Quality Criteria (AWQC) as authorized under section 304(a)(1) of the Clean Water Act (CWA). Section 304(a)(1) aquatic life criteria serve as recommendations to states and authorized tribes by defining ambient water concentrations that will protect against unacceptable adverse ecological effects to aquatic life resulting from exposure to pollutants found in water, consistent with the 1985 Guidelines. Section 304(a) recommended aquatic life criteria are developed to provide for the protection and propagation of fish and shellfish. Once the EPA publishes final section 304(a) recommended water quality criteria, states and authorized tribes may adopt these criteria into their water quality standards to protect designated uses of water bodies. States and authorized tribes may adopt water quality criteria that reflect adjustments to the EPA's recommended section 304(a) criteria to reflect local environmental conditions and human exposure patterns. Alternatively, states and authorized tribes may derive numeric criteria based on other scientifically defensible methods that protect the designated use. After adoption, states and authorized tribes submit new and revised water quality standards (WQS) to the EPA for review and approval or disapproval under CWA section 303(c). When approved by the EPA, the state or authorized tribe's WQS become the applicable WQS for CWA purposes. Such purposes include identification of impaired waters and establishment of Total Maximum Daily Loads (TMDLs) under CWA section 303(d) and derivation of water quality-based effluent limitations in permits issued under the CWA Section 402 National Pollutant Discharge Elimination System (NPDES) permit program.

As required by the CWA, the EPA periodically reviews and revises section 304(a) AWQC to ensure the criteria accurately reflect the latest scientific knowledge. The EPA previously published AWQC recommendations for aluminum in 1988 (EPA-440/5-86-008[2]), and is updating these criteria through its authority under CWA section 304(a). Water quality criteria are developed following the guidance outlined in the EPA's "*Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*" (Stephan et al. 1985) (herein referred to as the "1985 Guidelines"). This document describes

[2] https://www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table

scientifically defensible water quality criteria values for aluminum pursuant to CWA section 304(a), derived utilizing best available data in a manner consistent with the 1985 Guidelines.

2 PROBLEM FORMULATION

Problem formulation provides a strategic framework to develop water quality criteria by providing an overview of a chemical's sources and occurrence, fate and transport in the environment, and toxicological characteristics and factors affecting toxicity. A problem formulation uses this information to develop a conceptual model and identify the most relevant chemical properties and endpoints for evaluation. The structure of this effects assessment for aluminum is consistent with the EPA's *Guidelines for Ecological Risk Assessment* (U.S. EPA 1998a). This ecological effects assessment describes scientifically defensible water quality criteria values for aluminum under CWA section 304(a)(1).

2.1 Overview of Aluminum Sources and Occurrence

This section provides an overview of available reliable information from the peer-reviewed literature that characterizes sources and occurrence of aluminum in the environment. Aluminum is the third most abundant element and the most common metal in the Earth's crust, comprising about eight percent of the lithosphere (CRC 2000). It is typically found in complexation with oxygen (as oxides) and silica (as silicates), but rarely in the elemental state (Greenwood and Earnshaw 1997). Aluminum is found in most rocks, particularly igneous rocks, containing aluminosilicate minerals (Staley and Haupin 1992), and associated with clays and soil/sediments. Different water column forms include monomeric, polymeric, particulate (suspended) and colloidal forms of aluminum. Ions such as chloride, fluoride, nitrate, phosphate and sulfate form soluble complexes with aluminum, as do fulvic and humic acids (U.S. EPA 1988).

Aluminum enters the aquatic environment from both natural and anthropogenic sources, with natural sources typically dominating occurrence (Lantzy and MacKenzie 1979). This is due to the abundance of aluminum in rocks and minerals released by weathering (Lee and Von Lehmden 1973; Sorenson et al. 1974). Other natural aluminum sources include volcanic activity and acidic spring waters (USGS 1993; Varrica et al. 2000).

Anthropogenic releases are primarily associated with industrial processes and include air emissions, wastewater effluent and solid waste (ATSDR 2008). Anthropogenic sources include

fossil fuel combustion, aluminum production (mining and smelting) and aluminum present in fertilizers used in agriculture (Lantzy and MacKenzie 1979; Lee and Von Lehmden 1973; Ondov et al. 1982; Que Hee et al. 1982). Alum (potassium aluminum sulfate), used as a coagulant to clarify drinking water and wastewater, can also be a source of aluminum if this water is discharged to aquatic systems (Gidde et al. 2012).

A common source of aluminum in freshwater systems is from the mobilization of aluminum from rocks and soils by acid precipitation, heavy rains, or snow melt (Bjerknes et al. 2003). For estuaries and oceans, the primary source of aluminum is from riverine discharges, with the majority of the introduced aluminum sorbed to the surface of clay particles in estuarine sediments (Hydes and Liss 1977). However, aluminum that is either bound to clays or complexed to dissolved organic carbon can be converted to the reactive species upon mixing with high pH and high salinity ocean waters (Bjerknes et al. 2003; Rosseland et al. 1998; Teien et al. 2006a). The mechanism of this conversion is not well understood.

Aluminum is still actively mined in the U.S. from bauxite, the primary aluminum ore (mainly in Arkansas), with approximately 2 million metric tons produced in 2014. This raw domestic feedstock, plus imported bauxite and recycled aluminum, are currently processed at nine U.S. smelters into refined products (Bray 2015; USGS 2013). Because of aluminum's properties (light weight, resistance to corrosion, electrical conductivity, and durability), it has many diverse uses including: the transportation industry (automobiles, airplanes, trucks, railcars, marine vessels, etc.); packaging (cans, foil, etc.); construction (windows, doors, siding, etc.); consumer durables (appliances, cooking utensils, etc.); electrical transmission lines; and machinery (USGS 2013). Aluminum is also used in wastewater treatment to reduce effluent phosphorus levels (Tchobanoglous et al. 2003) and in the pharmaceutical industry in antacids and as a food additive (Government of Canada 1998).

The Water Quality Data Portal (https://www.waterqualitydata.us/) is an extensive database of environmental measurements available to identify concentrations of chemical contaminants, including aluminum, in surface waters such as rivers and streams. The results are reported in filtered and unfiltered categories. The terms filtered, dissolved, unfiltered, and total and their relationships, as defined by the U.S. Geological Survey (USGS), are presented below. "Dissolved" refers to constituents that exist in chemical solution in a water sample. "Filtered" pertains to constituents in a water sample passed through a filter membrane of specified pore

diameter, most commonly 0.45 micrometer or less for inorganic analytes. Therefore, for interpretation, the filtered samples (prior to acidification) will be assumed to be dissolved aluminum. “Total” pertains to the constituents in an unfiltered, representative water-suspended-sediment sample. This term is used only when the analytical procedure includes an acid digestion procedure that ensures measurement of at least 95 percent of the constituent present in both the dissolved and suspended phases of the sample. Therefore, for interpretation, the unfiltered samples are assumed to be total recoverable aluminum.

Aluminum data for freshwater systems were obtained from the Water Quality Data Portal (accessed 2/16/17) for data representing years 1991 to 2017. A total of 7,483 surface water samples were collected (4,991 filtered samples and 2,492 unfiltered samples) in that timeframe and analyzed for dissolved and total aluminum, respectively. The range of concentrations reported for dissolved aluminum was 0.8 µg/L to a maximum concentration reported of 20,600 µg/L. The range of total aluminum concentrations across all sites was a minimum of 0.9 µg/L, with a maximum reported total concentration of 210,000 µg/L. Groundwater concentrations of dissolved aluminum (filtered using a 0.45 micrometer filter) from the USGS National Water Quality Assessment Program (NAWQA) database collected during 1992-2003 are presented in **Figure 1**, and had a 90th percentile concentration of dissolved aluminum concentrations of 11 µg/L.

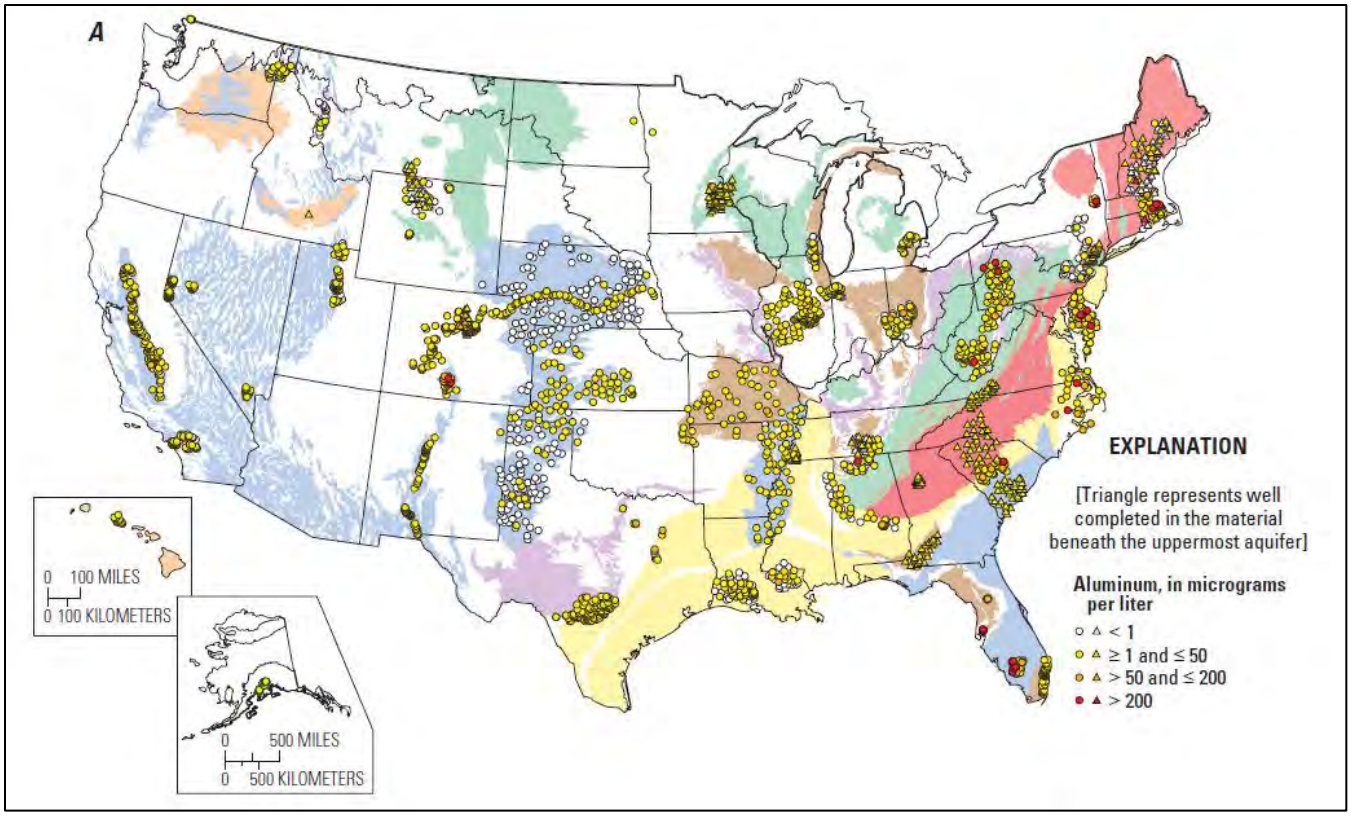


Figure 1. Geographic Distribution of Dissolved Aluminum Concentrations in Groundwater Collected from Wells as Part of the National Water-Quality Assessment Program, 1992–2003.
(Ayotte et al. 2011, used with permission.)

Aluminum concentrations in marine and estuarine waters are generally lower than levels found in freshwater systems, especially compared to acid-impacted areas (Gensemer and Playle 1999). Data for dissolved aluminum in coastal and marine waters were compiled from the scientific literature by Angel et al. (2016) and indicate that concentrations range from 0.5 to 2 µg/L in coastal waters, and from 0.008 to 0.68 µg/L in the open ocean. Other researchers have also reported that values are generally ≤1 µg/L in ocean waters (Brown et al. 2010; Hydes and Liss 1977; Tria et al. 2007). At the typical ocean pH of 8.0-8.3, aluminum forms complexes with hydroxide ion, primarily as $Al(OH)_4$, which precipitates out of solution. This largely explains the low concentrations in marine waters.

Much of the early to mid-1970s metals data in samples from natural waters are considered erroneously high due to contamination from sampling methods or containers. These flaws were corrected with the implementation of clean sampling techniques and guidance provided by U.S. EPA's Method 1669: Sampling Ambient Water for Trace Metals at EPA Water

Quality Criteria Levels (U.S. EPA. 2004). This method was designed to support water quality monitoring programs authorized under the Clean Water Act, specifically created for measuring toxic metals at the low part-per-trillion to low part-per-billion range (U.S. EPA 1996).

Average concentrations of total aluminum in the atmosphere were observed to range from 0.005 to 0.18 μg/m^3 (Hoffman et al. 1969; Potzl 1970; Sorenson et al. 1974). These concentrations are dependent on the location, weather conditions and industrial activity in the area with most of the airborne aluminum present in the form of small suspended particles of soil (dust) (ATSDR 2008). It should be noted that aluminum concentrations in air samples are often dependent upon the aluminum levels of the entrained soil particles, especially if measured as total aluminum. Goncharuk et al. (2012) sampled sea aerosols from the lower portion of the troposphere in the Black Sea (2002-2008), the Caspian Sea (2002-2006), the Baltic Sea (2001-2008), the White, Barents and Kara Seas (2005-2007) and high-altitude arctic regions in the Arctic and South Atlantic Oceans. Air samples were collected by aerosol filters for 3 to 5 hours during headwind conditions in the direction of atmospheric phenomenon. Most reported atmospheric total aluminum concentrations were less than 1 μg/m^3. The authors noted that the lowest concentrations were found at the high-altitude northern arctic regions, with increasing levels observed for the Western Arctic seas, and the highest concentrations reported for the most southerly located Black and Caspian Seas. They suggested that this northern to southern increasing concentration trend could be due to differential anthropogenic loading to the respective water bodies, and also with the increasing emissions of domestic and industrial wastes, wastewater, and emergency discharges of toxicants. Urban and industrial areas can have higher atmospheric total aluminum concentrations with levels reported from 0.4 to 8.0 μg/m^3 (Cooper et al. 1979; Dzubay 1980; Kowalczyk et al. 1982; Lewis and Macias 1980; Moyers et al. 1977; Ondov et al. 1982; Pillay and Thomas 1971; Sorenson et al. 1974; Stevens et al. 1978).

Total aluminum concentrations in North Atlantic precipitation collected in 1988 ranged from 6.1 to 827 μg/L (Lim and Jickells 1990). This is similar to a recent study that collected rainfall from two Mexico locations: a rural forested region 80 km south and downwind of Mexico City and Mexico City itself (Garcia et al. 2009). Average total aluminum precipitation concentrations reported in the rural area (107.2 μg/L, range of 28.8-222.7 μg/L) were higher than observed in the urban area (83.9 μg/L, range 35.8-125.4 μg/L). Samples of wet deposition collected in semi-rural Dexter, Michigan, had an average total aluminum concentration of 57

µg/L (Landis and Keeler 1997). Much lower levels of total aluminum were found in rainfall samples collected in Japan during 2000 and 2002 where average concentrations ranged from 2.71 to 6.06 µg/L (Takeda et al. 2000; Vuai and Tokuyama 2011). Atmospheric precipitation (i.e., rain and snow) samples collected in the U.S. have contained up to 1,200 µg/L total aluminum (Dantzman and Breland 1970; DOI 1971; Fisher et al. 1968; USGS 1964). No available information was found reporting concentrations of aluminum in fog.

Due to the abundance of aluminum in the earth's crust, soil concentrations can range widely from approximately 700 mg/kg to over 100,000 mg/kg (Shacklette and Boerngen 1984; Sorenson et al. 1974), averaging 71,000 mg/kg (Frink 1996). These concentrations are generally dependent on local geology and associated vegetation types and can vary within the same area, often strongly correlated with its clay content (Ma et al. 1997). Total aluminum concentrations in 1,903 soil samples collected from the continental U.S., Hawaii, Virgin Islands, Guam and Puerto Rico ranged from 500 to 142,000 mg/kg (Burt et al. 2003). In streambed sediment samples collected from locations in the conterminous U.S. from 1992 to 1996, aluminum concentrations ranged from 1.4 to 14% (by weight) (Rice 1999). Marsh/estuarine sediment samples collected from nine sampling sites within or along Georgia's Cockspur Island and McQueen's Island at Fort Pulaski's National Monument, a salt marsh ecosystem, had aluminum concentrations ranging from 17 to 820 mg/kg dry weight (Kumar et al. 2008).

Aluminum may form a precipitate when aluminum-rich water meets less acidic water. This precipitate mix, referred to as a floc, may include other co-precipitated ions, as well as nutrients, suspended materials and microorganisms. Removal of phosphorus from water has been observed in laboratory studies (Auvraya et al. 2006; Gilmore 2009; Matheson 1975; Minzoni 1984; Peterson et al. 1974; Westholm 2006) and in lake field studies (Knapp and Soltero 1983; Pilgrim and Brezonik 2005; Reitzel et al. 2005). Turbidity due to clay has been removed from pond waters using aluminum sulfate (Boyd 1979). Unz and Davis (1975) hypothesized that aluminum floc might coalesce bacteria and concentrate organic matter in effluents, thus assisting the biological sorption of nutrients. Aluminum sulfate (or alum) has been used to flocculate algae from water (McGarry 1970; Minzoni 1984; Zarini et al. 1983).

2.2 Environmental Fate and Transport of Aluminum in the Aquatic Environment

Aluminum (CAS Number 7429-90-05) is a silver white, malleable, and ductile metal that is odorless, and has a molecular weight of 26.98 g/mole (HSDB 2008). It has a density of 2.70

g/cm^3, a melting point of 660°C, a boiling point of 2,327°C, a vapor pressure of 1 mm Hg at 1,284°C, and is insoluble in water (CRC 2000; HSDB 2008). The n-octanol/water partitioning coefficient (K_{ow}), organic-carbon normalized partition coefficient (K_{oc}), and Henry's law constant for aluminum are unknown.

The chemistry of aluminum in surface water is complex because of the following properties: 1) it is amphoteric, meaning it is more soluble in acidic solutions and in basic solutions than in circumneutral solutions; 2) specific ions such as chloride, fluoride, nitrate, phosphate and sulfate form soluble complexes with aluminum; 3) it can form strong complexes with fulvic and humic acids; 4) hydroxide ions can connect aluminum ions to form soluble and insoluble polymers (e.g. gibbsite, corundum); and 5) under at least some conditions, solutions of aluminum in water approach chemical equilibrium rather slowly, with monomeric species of aluminum transforming into insoluble polymers which precipitate out of solution over time (Angel et al. 2016; Campbell et al. 1983; Hem 1968a,b; Hem and Roberson 1967; Hsu 1968; Roberson and Hem 1969; Smith and Hem 1972).

Aluminum exists as inorganic, monomeric species (Al^{3+}, $Al(OH)^{2+}$, $Al(OH)_2^+$, $Al(OH)_3$, and $Al(OH)_4^-$), as amorphous $Al(OH)_3$ leading to gibbsite formation and precipitation, and as polynuclear species such as the tridecameric Al_{13} polynuclear species (Gensemer and Playle 1999). The chemistry of aluminum in aquatic environments is complex, and several comprehensive reviews on its biological effects have been published (e.g., Driscoll and Schecher 1988; Gensemer and Playle 1999; Gostomski 1990; Havas 1986a,b; Havas and Jaworski 1986; Howells et al. 1990; Lewis 1989; Lydersen and Lofgren 2002; Rosseland et al. 1990; Scheuhammer 1991; Sigel and Sigel 1988; Sparling and Lowe 1996a; Sposito 1989, 1996; Wilson 2012; Yokel and Golub 1997). Effects on the aquatic community and considerations for criteria development are addressed below.

Aluminum from both natural and anthropogenic sources is transported by several means. Natural aluminum transport mechanisms include rock and mineral weathering, volcanic activity and acidic spring waters (USGS 1993; Varrica et al. 2000). Anthropogenic releases include air emissions, effluent dischargers and solid waste leaching. Aluminum is transported through the atmosphere as windblown particulate matter and is deposited onto land and water by wet and dry deposition. Atmospheric loading rates of aluminum to Lake Michigan have been estimated at 5

million kg/year (Eisenreich 1980), and at 0.1 g/m^2-year on Massachusetts Bay (Golomb et al. 1997).

Factors such as pH, temperature, and presence of complexing ions influence the fate and transport of aluminum in the environment. Of primary importance to understanding aluminum fate and behavior are its interactions with pH (see **Figure 2**). At neutral pH, aluminum is nearly insoluble, but its solubility increases exponentially as the pH reaches either acidic (pH<6) or basic (pH>8) conditions (Gensemer and Playle 1999). At pH values between 6.5 and 9.0 in fresh water, aluminum occurs predominantly in solution as monomeric, dimeric, and polymeric hydroxides and as complexes with fulvic and humic acids, chloride, phosphate, sulfate, and less common anions. The K_{sp} (solubility product) of aluminum hydroxide (gibbsite) ranges from 1.06×10^{-33} (Gayer et al. 1958) to 3.7×10^{-15} at 25°C (CRC 2000). Thus, aluminum hydroxide is insoluble compared to the more soluble salts used to determine aluminum toxic effect levels to aquatic species (aluminum chloride $K_{sp} = 2.04 \times 10^{4}$, aluminum nitrate $K_{sp} = 2.16 \times 10^{3}$, and aluminum sulfate $K_{sp} = 6.92 \times 10^{1}$) (CRC 2000).

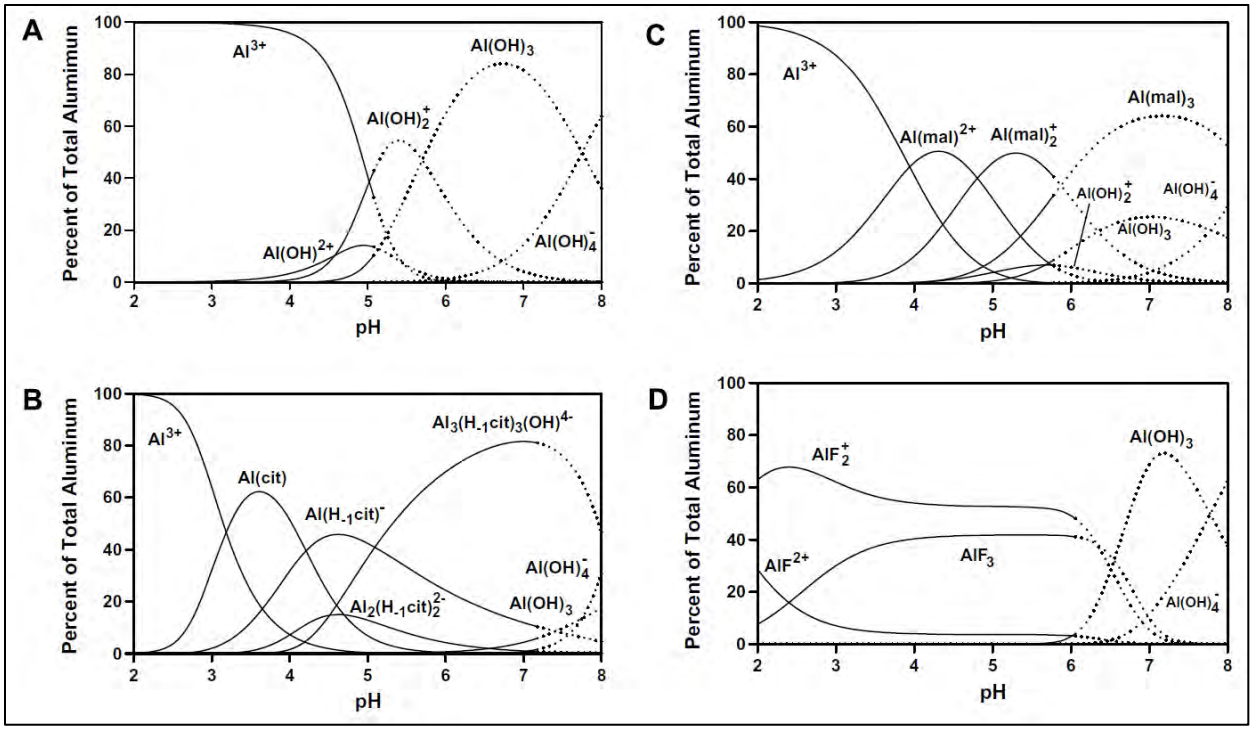


Figure 2. Results of Al Speciation Calculations at a Total of 65 µM Al in the Absence of Ligands (panel A) and in the Presence of Citrate (65 µM) (panel B), Maltolate (195 µM) (panel C), and Fluoride (260 µM) (panel D) in the pH Range 2 to 8.
The dotted lines indicate solutions that would be supersaturated with respect to freshly prepared $Al(OH)_3$. (Zhou et al. 2008, Figure 1, used with permission.)

Aluminum solubility increases at lower temperatures and in the presence of complexing ligands (both inorganic and organic) (ATSDR 2008; Lydersen, 1990; Wilson 2012). These two characteristics are significant because episodic acidic pulses in streams, for example during winter snowmelt, maximize the solubility of aluminum if pH drops to 5.5 or lower (Schofield 1977; Wilson 2012), and therefore may mobilize aluminum.

In the early 1980s the impacts of acid rain and aluminum toxicity were observed in aquatic and terrestrial environments in specific regions of the U.S., most notably in the northeastern part of the country where aquatic systems had limited buffering capacity to prevent pH changes. Researchers observed that aluminum can be a major factor responsible for the demise of biotic communities since the toxicant becomes more soluble and potentially more toxic to aquatic biota at acidic pH (Gensemer and Playle 1999).

2.3 Mode of Action and Toxicity

Aluminum has no biologically important functions or beneficial properties to aquatic life, and is therefore considered a non-essential metal (Eichenberger 1986; Exley 2003; Tchounwou

et al. 2012; Williams 1999; Wood 1984, 1985). It has been identified as the cause of harmful effects on fish and wildlife, but is not a known teratogen, carcinogen or mutagen (Leonard and Gerber 1988). The specific mechanisms of aluminum toxicity to aquatic organisms have been investigated extensively for fish and to a lesser extent for aquatic invertebrates.

For invertebrates, it is postulated that aluminum disrupts concentrations of specific ions, primarily resulting in a loss of sodium (Hornstrom et al. 1984). Elevated levels of aluminum affect ion regulation and the respiratory efficiency of sensitive species (Sparling and Lowe 1996a). Havas (1985) found that aluminum interfered with salt regulation in *Daphnia magna*, which caused a reduction in whole body sodium and chloride concentrations, resulting in death. In addition, aluminum has been shown to increase respiration, and thereby energy demands among mayfly species (Herrmann and Andersson 1986).

For fish, the gill is the primary site of aluminum toxic action, resulting in ionoregulatory, osmoregulatory and respiratory dysfunction. The gill is the primary site of aluminum toxicity under either acidic or alkaline conditions (Wilson 2012). Under acidic conditions, aluminum disrupts the barrier properties of the gill epithelium by binding with functional groups at both the apical gill surface and intracellularly within the lamellar epithelial cells (Exley et al. 1991). At reduced pH (<6.5), aluminum will accumulate on the gill surface resulting in physical damage to the epithelial cells that subsequently causes a loss of plasma ions (Na^+, Cl^-), reduced ion uptake and gas exchange. At alkaline pH (>8), the negatively charged aluminate anion dominates which also disrupts gill function, but to a lesser degree due to the lack of binding of the aluminate anion to the negatively charged gill surface. The subsequent necrosis of the epithelial cells causes a loss of plasma ions (Na^+, Cl^-), reduced osmolality and gas exchange, and if severe enough, the death of the fish (Dietrich 1988; Dietrich and Schlatter 1989a,b; Leivestad et al. 1980; Mallatt 1985; Muniz and Leivestad 1980a,b; Rosseland and Skogheim 1984, 1987). Mitigation of these toxic effects was observed with moderate concentrations of calcium (Brown 1981b), high concentrations of humic acids (Baker and Schofield 1982; Driscoll et al. 1980), and high concentrations of silica (Birchall et al. 1989). Fish in low pH waters with high aluminum concentrations will accumulate aluminum on the gill surface (Rosseland et al. 1990). Bjerknes et al. (2003) observed elevated aluminum concentrations in the gills of dead and "sluggish" Atlantic salmon (*Salmo salar*) associated with ruptured atria, which the authors suggested may have resulted from hypercapnia (abnormally elevated carbon dioxide levels in the blood) caused

by circulatory distress from the clogging of gills with aluminum. The specific mechanisms of aluminum toxicity at alkaline pH are not well understood.

In laboratory toxicity tests, organisms are exposed to a mixture of dissolved and particulate aluminum depending on how long the acidic aluminum stock solution has been allowed to equilibrate prior to dosing the organisms (Angel et al. 2016). Over time (minutes) as the aluminum from the stock solution equilibrates with the test water and the pH increases, the monomeric species of aluminum transform to the newly-formed insoluble polymeric hydroxide species, which are more toxic (Cardwell et al. 2018). Thus, soon after test initiation, there is a transformation period of rapid speciation changes from short-lived transient amorphous and colloidal forms of aluminum (from minutes to a few hours) to more stable crystalline forms that can take days to form (Gensemer et al. 2018). Aged stock solutions (aluminum solutions that have been given sufficient time (i.e., hours to days) to form more stable forms of aluminum) have been shown to be less toxic than those that are not aged (Exley et al. 1996; Witters et al. 1996). Unfortunately, many studies included for criteria derivation did not describe stock solution age prior to test initiation, and this variable therefore cannot be factored into the toxicity assessment.

Several investigators have found different trends in the toxicity of aluminum under different pH conditions, and toxicity of aluminum appears to be lowest at neutral pH (approximately 7), with toxicity tending to increase with either increasing or decreasing pH (above and below neutral pH). Freeman and Everhart (1971) found that the lethal time to 50% of the rainbow trout decreased (i.e., was more toxic) as the pH increased from 6.8 to 8.99 when rainbow trout were exposed in flow-through tests to the same nominal (unmeasured) aluminum concentration. They concluded that soluble aluminum was the toxic form. Hunter et al. (1980) observed the same relationship of increasing toxicity with rainbow trout over a pH range of 7.0 to 9.0 in chronic static renewal toxicity studies (also nominal aluminum exposures). Call (1984) conducted measured static acute toxicity studies with fathead minnows at pH of 7.61 and 8.05 and showed a slight increase in toxicity at increased pH. However, in another measured static acute toxicity study with a different species, rainbow trout, Call (1984) found a decrease in toxicity as pH increased for the studies conducted at pH 7.31 and 8.17. Thus, generally, most studies show that aluminum toxicity increases as pH increases in the range of approximately 7.0 to 9.0.

Regarding toxicity at low pH, Freeman and Everhart (1971) also observed the greater toxicity at acidic pH 6.52 in static renewal tests with rainbow trout. In a measured static acute toxicity study with rainbow trout by Call (1984), tests were conducted with pH measurements of 6.59, 7.31 and 8.17. The greatest toxicity was observed at the acidic pH of 6.59. The tests conducted by Freeman and Everhart (1971) and Hunter et al. (1980) were static renewal or flow-through and showed the lowest acute values. The flow-through and renewal tests are considered to be a more reliable way to conduct toxicity tests for aluminum because the dosed chemical is more likely to remain in solution at the desired concentration, and less likely to drop below nominal levels due to precipitation and/or adherence to test vessel surfaces. In addition, because the polymerization of aluminum hydroxide is a relatively slow process, the chemical form of aluminum might have differed from test to test due to the amount of time the aluminum was in stock and test solutions.

The influence of pH on aluminum speciation and associated toxicity to aquatic organisms is readily apparent and highlights the importance of pH control during toxicity tests. Depending on the pH at test initiation, the greatest potential for pH drift would be static exposures, followed by static-renewal and finally flow-through studies. All of the studies evaluated for criteria derivation reported pH, and most included the standard deviation of the measurements, thus providing a rough estimate of pH drift during the exposure. Only selected studies, however, described pH drift for individual tests (e.g., ENSR 1992c,d; European Aluminum Association 2009).

Driscoll et al. (1980) tested postlarvae of brook trout and white suckers under slightly acidic conditions and concluded that only inorganic forms of aluminum were toxic to fish. Hunter et al. (1980) reported that the toxicity of test solutions was directly related to the concentration of dissolved aluminum that passed through a 0.45 μm membrane filter.

In dilute aluminum solutions, formation of particles and the large insoluble polynuclear complexes known as floc is primarily a function of the concentration of organic acids and the hydroxide ion. Time for particle formation varies from less than one minute to several days depending upon the source of aluminum (i.e., aluminum chloride, aluminum nitrate), the pH and the presence of electrolytes and organic acids (Snodgrass et al. 1984). When particles form an aggregate large enough to become visible, the floc is white in color, and tends to settle. Mats of aluminum floc have been reported blanketing a stream bed (Hunter et al. 1980). Laboratory

studies conducted at alkaline pH levels have reported floc in the exposure chambers (Brooke 1985; Call 1984; Lamb and Bailey 1981; Zarini et al. 1983). The floc did not appear to affect most aquatic species. However, the swimming ability of *Daphnia magna* was impeded by "fibers" of flocculated aluminum trailing from the carapaces. Additionally, the mobility and feeding of midges also was affected, ultimately resulting in death (Lamb and Bailey 1981). Bottom-dwelling organisms may be impacted more by aluminum floc in the field than in the laboratory due to the greater floc layer thickness observed in the field relative to laboratory exposures (U.S. EPA 1988), but this will also depend on the water velocity and mixing in both the field and the laboratory.

Aquatic plant toxicity to aluminum can be dependent on the speciation of aluminum which is controlled by pH. In a study of cell growth rate of the green alga, *Chlorella pyrenoidosa*, to aluminum, Helliwell et al. (1983) found that decreased cell growth occurred in the pH range of 5.8 to 6.2. This is near the pH of minimum solubility of aluminum and maximum concentration of $Al(OH)_2^+$. They found that the toxicity of aluminum decreased as pH increased from 6.2 to 7 or as pH decreased from 5.8 to 4.7, and they hypothesized that the monovalent hydroxide is the most toxic form. Seip et al. (1984) stated that "the simple hydroxides ($Al(OH)^{+2}$ and $Al(OH)_2^+$) are regarded as the most dangerous forms, while organically bound aluminum and polymeric forms are less toxic or essentially harmless." However, one study found algae productivity and biomass were seldom affected if the pH is above 3.0 (Sparling and Lowe 1996a). Aluminum and acid toxicity tend to be additive to some algae when the pH is less than 4.5. Because aluminum binds with inorganic phosphorus, it may reduce the availability of this nutrient thereby reducing productivity (Sparling and Lowe 1996a). As shown in **Appendix E** and **Appendix H**, the effects of aluminum on algae productivity and biomass are dependent on the pH, total hardness and DOC of the exposure solutions.

2.3.1 Water Quality Parameters Affecting Toxicity

Bioavailability of aluminum is affected by water chemistry parameters such as pH, total hardness and DOC, and to a lesser extent fluoride. The pH of waters affects aluminum speciation and solubility. Aluminum can sorb to dissolved organic carbon (DOC), such as humic and fulvic acids, and form organic aluminum complexes. An increase in DOC in waters reduces the bioavailability of aluminum to aquatic organisms as a result of this binding (Wilson 2012). Hardness also has an effect on the toxicity of aluminum, as the cation Al^{+3} competes with other

cations present in water such as calcium (Ca^{+2}) for uptake (Gensemer and Playle 1999). The observed effect of total hardness may be due to one or more of a number of usually interrelated ions, such as hydroxide, carbonate, calcium, and magnesium. Acute tests were conducted at four different levels of water total hardness with *Ceriodaphnia dubia* (ENSR 1992d), demonstrating that daphnids were more than 138 times more sensitive to aluminum in soft water than in hard water (**Appendix A** *Acceptable Acute Toxicity Data of Aluminum to Freshwater Aquatic Animals*). Data in **Appendix A** also indicate that aluminum was more toxic to *Daphnia magna*, brook trout, and fathead minnows in soft water than in hard water. In contrast, no apparent total hardness-toxicity relationship was observed for rainbow trout exposed to three different total hardness levels at a controlled pH of 8.3 (Gundersen et al. 1994). This is consistent with data recently published by DeForest et al. (2018a) and Gensemer et al. (2018) demonstrating that there is a reduced effect of total hardness at elevated pH levels.

Development of the Biotic Ligand Model (BLM - formerly the "gill model") and multi-parameter linear regression models in recent years were intended to better account for the water chemistry parameters that most strongly affect the bioavailability, and hence toxicity, of metals to aquatic life. The BLM, a mechanistic model that uses a series of submodels to quantify the capacity of metals to accumulate or bind to active sites on the gills of aquatic organisms, estimates the bioavailable portion of dissolved metals in the water column based on site-specific water quality parameters such as pH, hardness, and DOC (McGeer et al. 2000; Meyer et al. 1999; Pagenkopf 1983; Paquin et al. 1999; U.S. EPA 1999a, 2000). Multiple linear regression (MLR) models are statistical in nature and can also take into account pH, total hardness and DOC. While MLR models are less complex than BLM models, they also estimate the bioavailability of aluminum to aquatic species. The EPA evaluated the use of empirical, non-mechanistic MLR models for aluminum (DeForest et al. 2018a) as a bioavailability-based approach for deriving water quality criteria as well as a BLM model for aluminum (Santore et al. 2018). Note that the aluminum BLM developed by Santore et al. (2018) differs from earlier BLMs for other metals, because the aluminum BLM accounts for the dissolved and precipitated fraction of aluminum. Previous BLMs for other metals only account for the dissolved fraction of the metal.

The EPA decided to use an empirical MLR approach in this aluminum criteria update rather than a BLM model due to: 1) the relative simplicity and transparency of the model, 2) the relative similarity to the available BLM model outputs, and 3) the decreased number of input

data on water chemistry needed to derive criteria at different sites. An external peer review of an approach using a pH and total hardness equation-based criteria, an MLR approach, and a BLM approach for aluminum criteria development was conducted in 2015 and peer-reviewers' comments were considered in the selection of the MLR-based criteria approach. The EPA independently examined and verified the quality and fit of the DeForest et al. (2018a,b) MLR models before applying them in this criteria document.

2.4 Conceptual Model

Conceptual models consist of a written description and diagram (U.S. EPA 1998a) that illustrate the relationships between human activities, stressors, and ecological effects on assessment endpoints. The conceptual model links exposure characteristics with the ecological endpoints important for management goals.

2.4.1 Conceptual Diagram

Aluminum can originate from both natural and anthropogenic sources (Lantzy and MacKenzie 1979). The environmental fate properties of aluminum indicate that weathering/erosion, volcanic activity, runoff/leaching, groundwater recharge, spray drift from aluminum-containing pesticides, and atmospheric deposition represent potential transport mechanisms of aluminum to surface water habitats for aquatic organisms (ATSDR 2008). These transport mechanisms are depicted in the conceptual model below for natural (i.e., weathering and erosion, volcanic activity) and anthropogenic sources of aluminum to the environment (i.e., wastewater treatment, resource extraction, smelting/manufacturing operations, agricultural uses and fossil fuel combustion) (**Figure 3**). The model also depicts exposure pathways for biological receptors of concern (e.g., aquatic animals) and the potential attribute changes (i.e., effects such as reduced survival, growth and reproduction) in the receptors due to aluminum exposure. A solid line indicates a major pathway and a dashed line indicates a minor pathway. Aquatic assessments address exposure primarily through anthropogenic releases, runoff and atmospheric deposition.

The conceptual model provides a broad overview of how aquatic organisms can potentially be exposed to aluminum. Derivation of criteria focuses on effects on survival, growth and reproduction of aquatic organisms. However, the pathways, receptors, and attribute changes depicted in **Figure 3** may be helpful for states and authorized tribes as they adopt criteria into standards and need to evaluate potential exposure pathways affecting designated uses.

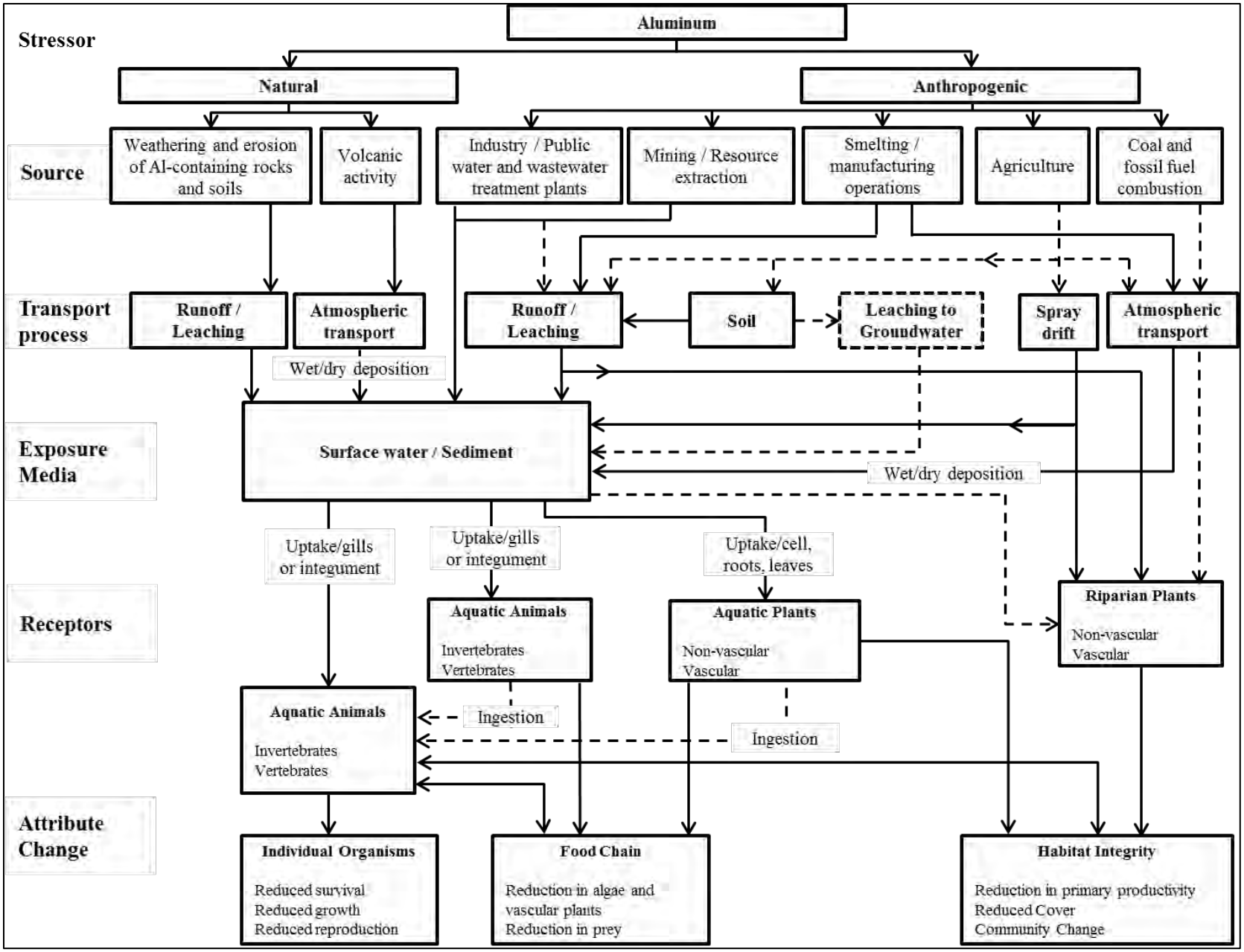


Figure 3. Conceptual Model for Aluminum Effects on Aquatic Organisms.
(Dotted lines indicate exposure pathways that have a lower likelihood of contributing to ecological effects).

2.5 Assessment Endpoints

Assessment endpoints are defined as the explicit expressions of the environmental values to be protected and are comprised of both the ecological entity (e.g., a species, community, or other entity) and the attributes or characteristics of the entity to be protected (U.S. EPA 1998a). Assessment endpoints may be identified at any level of organization (e.g., individual, population, community). In the context of the CWA, aquatic life criteria for toxic substances are typically determined based on the results of toxicity tests with aquatic organisms, for which adverse effects on growth, reproduction, or survival are measured. This information is aggregated into a genus sensitivity analysis that characterizes an impact to the aquatic community. Criteria are designed to be protective of the vast majority of aquatic animal taxa in an aquatic community (i.e., approximately the 95^{th} percentile of genera based on tested aquatic animals representing the aquatic community per the 1985 Guidelines recommendations (Stephan et al 1985). Assessment endpoints consistent with the criteria developed in this document are summarized in **Table 1**.

The concept of using laboratory toxicity tests to protect North American bodies of water and resident aquatic species is based on the theory that effects occurring to a species in controlled laboratory tests will generally occur to the same species in comparable field situations. Since aquatic ecosystems are complex and diversified, the 1985 Guidelines require acceptable data be available for at least eight genera with a specified taxonomic diversity (the standard eight-family minimum data requirement, or MDR). The intent of the eight-family MDR is to serve as a typical surrogate sample community representative of the larger and generally much more diverse natural aquatic community, not necessarily the most sensitive species in a given environment. For many aquatic life criteria, enough data are available to describe a sensitivity distribution to represent the distribution of sensitivities in natural ecosystems. In addition, since aquatic ecosystems can tolerate some stress and occasional adverse effects, protection of all species at all times and places is not deemed necessary. The intent is to protect approximately 95 percent of a group of diverse taxa, with special consideration given to any commercially and recreationally important species (Stephan et al 1985). Thus, if properly derived and used, the combination of a freshwater or estuarine/marine acute and chronic aquatic life criteria should provide an appropriate degree of protection of aquatic organisms and their uses from acute and

chronic toxicity to animals, toxicity to plants, and bioaccumulation by aquatic organisms (Stephan et al. 1985).

Table 1. Summary of Assessment Endpoints and Measures of Effect Used in Criteria Derivation.

Assessment Endpoints for the Aquatic Community	Measures of Effect
Survival, growth, and reproduction of freshwater fish, other freshwater vertebrates, and invertebrates	For acute effects: LC_{50}, EC_{50} For chronic effects: EC_{20}, MATC (only used when an EC_{20} could not be calculated for the genus), EC_{10} (for bioaccumulative compounds)
Survival, growth, and reproduction of estuarine/marine fish and invertebrates	For acute effects: LC_{50}, EC_{50} For chronic effects: EC_{20}, MATC (only used when an EC_{20} could not be calculated for the genus), EC_{10} (for bioaccumulative compounds)
Maintenance and growth of aquatic plants from standing crop or biomass (freshwater and estuarine/marine)	LOEC, EC_{20}, EC_{50}, IC_{50}, reduced growth rate, cell viability, calculated MATC

MATC = Maximum acceptable toxicant concentration (geometric mean of NOEC and LOEC)
NOEC = No observed effect concentration
LOEC = Lowest observed effect concentration
LC_{50} = Lethal concentration to 50% of the test population
$EC_{50}/EC_{20}/EC_{10}$ = Effect concentration to 50%/20%/10% of the test population
IC_{50} = Concentration of aluminum at which growth is inhibited 50% compared to control organism growth

2.6 Measurement Endpoints

Measurement endpoints (**Table 1**) are the measures of ecological effect used to characterize or quantify changes in the attributes of an assessment endpoint or changes in a surrogate entity or attribute, in this case a response to chemical exposure (U.S. EPA 1998a). Toxicity data are used as measures of direct and indirect effects on representative biological receptors. The selected measures of effects for the development of aquatic life criteria encompass changes in the growth, reproduction, and survival of aquatic organisms (Stephan et al. 1985).

The toxicity data used for the development of aquatic life criteria depend on the availability of applicable toxicity test outcomes, the acceptability of test methodologies, and an in-depth evaluation of the acceptability of each specific test, as performed by the EPA. Measurement endpoints for the development of aquatic life criteria are derived using acute and chronic toxicity studies for representative test species, which are then quantitatively and qualitatively analyzed, as described in the Analysis Plan below. Measurement endpoints

considered for each assessment endpoint in this criteria document are summarized in **Table 1**. The following sections discuss toxicity data requirements for the fulfillment of these measurement endpoints.

2.6.1 Overview of Toxicity Data Requirements

The EPA has specific data requirements to assess the potential effects of a stressor on an aquatic ecosystem and develop CWA section 304(a) aquatic life criteria as described in the 1985 Guidelines (Stephan et al 1985). Acute toxicity test data (short term effects on survival) for species from a minimum of eight diverse taxonomic groups are required for the development of acute criteria to ensure the protection of various components of an aquatic ecosystem.

- Acute toxicity test data for species from a minimum of eight diverse taxonomic groups. The diversity of tested species is intended to ensure protection of various components of an aquatic ecosystem.
    - The acute freshwater requirement is fulfilled with the following eight minimum data requirements:
        - the family Salmonidae in the class Osteichthyes
        - a second family in the class Osteichthyes, preferably a commercially or recreationally important warmwater species (e.g., bluegill, channel catfish, etc.)
        - a third family in the phylum Chordata (may be in the class Osteichthyes or may be an amphibian, etc.)
        - a planktonic crustacean (e.g., cladoceran, copepod, etc.)
        - a benthic crustacean (e.g., ostracod, isopod, amphipod, crayfish, etc.)
        - an insect (e.g., mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge, etc.)
        - a family in a phylum other than Arthropoda or Chordata (e.g., Rotifera, Annelida, Mollusca, etc.)
        - a family in any order of insect or any phylum not already represented
    - The acute estuarine/marine requirement is fulfilled with the following eight minimum data requirements:
        - two families in the phylum Chordata
        - a family in a phylum other than Arthropoda or Chordata
        - either the Mysidae or Penaeidae family
        - three other families not in the phylum Chordata (may include Mysidae or Penaeidae, whichever was not used above)
        - one from any other family
- Chronic toxicity test data (longer-term survival, growth, or reproduction) are required for a minimum of three taxa, with at least one chronic test being from an acutely-sensitive species.

- Acute-chronic ratios (ACRs) can be calculated with data from species of aquatic animals from at least three different families if the following data requirements are met:
    - at least one is a fish
    - at least one is an invertebrate
    - for freshwater chronic criterion: at least one is an acutely sensitive freshwater species (the other two may be estuarine/marine species) or for estuarine/marine chronic criterion: at least one is an acutely sensitive estuarine/marine species (the other two may be freshwater species).

The 1985 Guidelines also require at least one acceptable test with a freshwater alga or vascular plant. If plants are among the aquatic organisms most sensitive to the chemical, results of a plant in another phylum should also be available. Data on toxicity to aquatic plants are examined to determine whether plants are likely to be unacceptably affected by concentrations below those expected to cause unacceptable effects on aquatic animals. As discussed in **Section 3.4** and **Section 5.2**, based on available data the relative sensitivity of fresh and estuarine/marine algae and plants to aluminum (**Appendix E** *Acceptable Toxicity Data of Aluminum to Freshwater Aquatic Plants* and **Appendix F** *Acceptable Toxicity Data of Aluminum to Estuarine/Marine Aquatic Plants*) is less than vertebrates and invertebrates, so plant criteria were not developed. This trend was apparent for all conditions, as vertebrate and invertebrate generated criteria values were always less than alga EC_{20}s (DeForest et al. 2018a), except at unrealistically high pH and very high total hardness.

2.6.2 Measures of Effect

The assessment endpoints for aquatic life criteria are based on survival, growth and reproduction of the assessed taxa per the 1985 Guidelines (Stephan et al 1985). The measures of effect are provided by the acute and chronic toxicity data. These toxicity endpoints (expressed as genus mean values) are used in the sensitivity distribution of the aquatic community at the genus level to derive the aquatic life criteria. Endpoints used in this assessment are listed in **Table 1**. Studies that had unacceptable control survival were not used (i.e., studies where acute and chronic control mortality was >10% and >20%, respectively), regardless of test conditions.

Measure of Aluminum Exposure Concentration

Only data from toxicity tests conducted using chloride, nitrate and sulfate salts (either anhydrous or hydrated) are used in this effects assessment. This is consistent with the EPA's

previous 1988 aluminum aquatic life AWQC document. This document addresses the toxicity of total aluminum to freshwater organisms in the pH range of 5.0 to 10.5. The 1988 AWQC addressed waters with a pH between 6.5 and 9.0 (U.S. EPA 1988) to be consistent with the recommended aquatic life pH criteria (U.S. EPA 1986). The pH range for freshwater was expanded, in part, because of the complex chemistry of aluminum in surface waters, the available toxicity data demonstrated an increased sensitivity of freshwater aquatic species in low pH (i.e., pH<6.5), and the expanded range represents a fuller range of pH conditions in natural waters. Tests conducted in pH water less than 5 were deemed too low to be used quantitatively due to a mixture effect from the combined stress of both low pH and aluminum on the test organisms, and the inability to discern a particular effect level to either low pH or elevated aluminum concentration.

Aluminum chemistry in surface waters is extremely complex, and so measurement uncertainty can be high if only one form of aluminum is taken into account. A thorough understanding of aluminum toxicity is complicated by the need to distinguish between aqueous and particulate aluminum, and between inorganic and organic forms of aluminum (Driscoll and Postek 1996; Gensemer and Playle 1999). Laboratory dilution waters do not contain suspended solids, clays or particulate matter where aluminum may be bound (unless specifically investigated). Therefore, a distinction needs to be made in how the EPA interprets the measurements of aluminum in water, so that extrapolating laboratory data to natural waters is better understood. There is also a complication as the available measurement methods (i.e., total, total recoverable, acid soluble, pH 4 extractable and dissolved) present different challenges when applied to natural and laboratory waters. In application to natural waters, total, total recoverable, and acid soluble methods may be confounded by measuring aluminum in aluminum silicate (i.e., clay).

Laboratory Exposures

The 1988 AWQC considered using dissolved aluminum concentrations to set aquatic life criteria, however not enough data were available to allow derivation of a criterion based on dissolved aluminum. The EPA also noted at the time that organisms would be exposed to both dissolved and undissolved aluminum from laboratory exposures. The lack of data prevented any definitive analysis.

Data are now available to compare toxicity of aluminum using total aluminum (unfiltered test samples that were acidified) and dissolved aluminum (operationally defined as filtered with typically a 0.45 µm filter before acidification). The total aluminum concentrations in laboratory test solutions will contain dissolved monomeric and precipitated forms (e.g., aluminum hydroxides) of aluminum. Dissolved concentrations will not contain these precipitated forms.

In tests with brook trout at low pH and total hardness, toxic effects increased with increasing concentrations of total aluminum even though the corresponding concentration of dissolved aluminum was relatively constant (Cleveland et al. 1989). This phenomenon was also observed in several chronic studies with widely varying test concentrations and conditions (renewal and flow-through exposures) at pH 6 conducted by the Oregon State University (e.g., 2012a,e), where toxic effects increased with increasing total aluminum concentrations, while measured concentrations of dissolved and monomeric aluminum changed very little with increasing total aluminum concentrations.

In filtration studies at pH 8 with the fathead minnow, both acute and chronic toxicity tests indicated no toxicity when the test water was 0.2 µm filtered prior to exposure (Gensemer et al. 2018). Toxicity was only observed when the test solutions were unfiltered. Furthermore, dose-response relationships were only observed using total aluminum; relationships were not observed using measurements of dissolved or monomeric forms (Gensemer et al. 2018). This same effect was observed in 7-day exposures at pH 7 and 8 with the daphnid (*Ceriodaphnia dubia*) where filtered test solutions were less toxic than unfiltered solutions (Gensemer et al. 2018).

Therefore, because measurements of dissolved aluminum do not reflect the full spectrum of forms of aluminum that results in toxicity, all laboratory exposure data used for criteria derivation will be based on measurements of total aluminum. Measurements with methods using lesser degrees of acidification (that is, acid soluble and pH 4 extractable) are generally not available. If aluminum criteria are based on dissolved concentrations, toxicity will be underestimated, because aluminum hydroxide precipitates that contribute to toxicity would not be measured (GEI Consultants, Inc. 2010; U.S. EPA 1988). All concentrations from toxicity tests are expressed as total aluminum in this document (unless otherwise specified).

Natural Waters

Researchers rely on operationally defined procedures to evaluate the concentration and forms of aluminum in natural waters, and the accuracy of these methods is difficult to evaluate,

resulting in uncertainty regarding the actual amount of aluminum present in various forms (Driscoll and Postek 1996). Total aluminum concentrations in natural waters are determined using a wide variety of digestion procedures at varied extraction times, resulting in a range of operational methods and uncertainty in measured values (Driscoll and Postek 1996). Furthermore, particulate material comprises a continual size distribution making measurement of dissolved concentrations dependent on the filter-pore size used (Driscoll and Postek 1996).

A major complication for extrapolating total aluminum concentrations measured in laboratory waters to natural waters is the test method used. The 1988 AWQC for aluminum were based on acid-soluble concentrations (operationally defined as the aluminum that passes through a 0.45 µm filter after the sample has been acidified with nitric acid to a pH between 1.5 and 2.0). In the early 1990s, the EPA converted most metals criteria (excluding aluminum) to the dissolved measurement. With the acid-soluble method seldom used and insufficiently different from total, (U.S. EPA 1999c) the EPA expressed the aluminum criterion as total recoverable aluminum, with a caution that a Water-Effect Ratio would often be needed. The EPA uses the terms "total" and "total recoverable" synonymously for effluent guidelines and permitting under NPDES programs (U.S. EPA 1988b). The current EPA Test Method for measuring total recoverable aluminum in ambient water and wastewater uses inductively coupled plasma-atomic emission spectrometry and inductively-coupled plasma-mass spectrometry (U.S. EPA 1994a,b). The methods recommend that the sample first be solubilized by gentle refluxing with nitric and hydrochloric acids (i.e., digestion to pH<2) when an aqueous sample contains undissolved material. After cooling, the sample is made up to volume, then mixed and either centrifuged or allowed to settle overnight prior to analysis. This process dissolves the monomeric and polymeric forms of aluminum, in addition to colloidal, particulate and clay-bound aluminum. Applying the aluminum criteria to total recoverable aluminum is considered conservative because it includes monomeric (both organic and inorganic) forms, polymeric and colloidal forms, as well as particulate forms and aluminum sorbed to clays (Wilson 2012). However, under natural conditions not all of these forms would be biologically available to aquatic species (e.g., clay-bound aluminum).

EPA Methods 200.7 and 200.8 are the only currently approved methods for measuring aluminum in natural waters and wastewater for NPDES permits (U.S. EPA 1994a,b). Research on new analytical methods is ongoing to address concerns with including aluminum bound to

particulate matter (i.e., clay) in the total recoverable aluminum concentrations (OSU 2018c). One approach would not acidify the sample to pH less than 2 but rather to pH 4 (pH 4 extracted method) to better capture the bioavailable fraction of aluminum (CIMM 2016, OSU 2018c). In the pH 4 extraction method, sodium acetate buffer is added to the sample to reach the desired pH, followed by sample agitation for a specified period of time, and finally 0.45 µm sample filtration. The sample is then acidified with nitric acid before inductively coupled plasma-optical emission spectrometry analysis.

To further explore this issue, researchers conducted an aluminum analysis of 12 natural freshwater sources throughout the United States with various concentrations of total suspended solids using four different aluminum methods (i.e., total, acid-soluble, pH 4 extracted and dissolved) (OSU 2018c). The total method (consistent with EPA methods 200.7 and 200.8) acidified the sample to pH 2 before analysis; the acid soluble method acidified the sample to pH<2, held the sample for 16 hours and then filtered the sample with a 0.45 µm filter; the pH extraction method acidified the sample to pH 4.0-4.2, held the sample for three hours, and then filtered the sample with a 0.45 µm filter; and lastly, the dissolved method filtered the sample before acidification. As expected, the total method typically had elevated measured aluminum concentrations compared to the levels quantified by the three other test methodologies. This trend was most evident with natural waters that had high total suspended solids. The validation of the pH 4 extraction method is still on-going, with the expectation that this approach will better estimate the bioavailable fraction of aluminum in natural waters.

Acute Measures of Effect

The acute measures of effect on aquatic organisms are the LC_{50}, EC_{50}, and IC_{50}. LC stands for "Lethal Concentration," and a LC_{50} is the concentration of a chemical that is estimated to kill 50 percent of the test organisms. EC stands for "Effect Concentration," and the EC_{50} is the concentration of a chemical that is estimated to produce a specific effect in 50 percent of the test organisms. IC stands for "Inhibitory Concentration," and the IC_{50} is the concentration of a chemical that is estimated to inhibit some biological process (e.g., growth) in 50 percent of the test organisms. Acute data that were determined to have acceptable quality and to be useable in the derivation of water quality criteria as described in the 1985 Guidelines for the derivation of a freshwater and estuarine/marine criteria are presented in **Appendix A** (*Acceptable Acute Toxicity*

Data of Aluminum to Freshwater Aquatic Animals) and **Appendix B** (*Acceptable Acute Toxicity Data of Aluminum to Estuarine/Marine Aquatic Animals*), respectively.

Chronic Measures of Effect

The endpoint for chronic exposure for aluminum is the EC_{20}, which represents a 20 percent effect/inhibition concentration. This is in contrast to a concentration that causes a low level of reduction in response, such as an EC_5, which is rarely statistically significantly different from the control treatment. A major reduction, such as 50 percent, is not consistent with the intent of establishing chronic criteria to protect populations from long-term effects. The EPA selected an EC_{20} to estimate a low level of effect for aluminum that would typically be statistically different from control effects, but not severe enough to cause chronic effects at the population level (see U.S. EPA 1999b). Reported NOECs (No Observed Effect Concentrations) and LOECs (Lowest Observed Effect Concentrations) were only used for the derivation of a chronic criterion when an EC_{20} could not be calculated for the genus. A NOEC is the highest test concentration at which none of the observed effects are statistically different from the control. A LOEC is the lowest test concentration at which the observed effects are statistically different from the control. When LOECs and NOECs are used, a Maximum Acceptable Toxicant Concentration (MATC) is calculated, which is the geometric mean of the NOEC and LOEC.

Regression analysis was used to characterize a concentration-effect relationship and to estimate concentrations at which chronic effects are expected to occur. For the calculation of the chronic criterion, point estimates (e.g., EC_{20}s) were selected for use as the measure of effect rather than MATCs, as MATCs are highly dependent on the concentrations tested (as are the NOECs and LOECs from which they are derived). Point estimates also provide additional information that is difficult to determine with an MATC, such as a measure of magnitude of effect across a range of tested concentrations. Author reported EC_{20}s were used when provided, otherwise point estimates were calculated from raw toxicity data using the EPA's Toxicity Relationship Analysis Program (TRAP). Chronic toxicity data that met the test acceptability and quality assurance and quality control (QA/QC) criteria in the 1985 Guidelines for the derivation of freshwater and estuarine/marine criteria are presented in **Appendix C** (*Acceptable Chronic Toxicity Data of Aluminum to Freshwater Aquatic Animals*) and **Appendix D** (*Acceptable Chronic Toxicity Data of Aluminum to Estuarine/Marine Aquatic Animals*), respectively.

2.7 Analysis Plan

During CWA section 304(a) criteria development, the EPA reviews and considers all relevant toxicity test data. Information available for all relevant species and genera are reviewed to identify whether: 1) data from acceptable tests meet data quality standards; and 2) the acceptable data meet the minimum data requirements (MDRs) as outlined in the 1985 Guidelines (Stephan et al. 1985; U.S. EPA 1986). The taxa represented by the different MDR groups represent taxa with different ecological, trophic, taxonomic and functional characteristics in aquatic ecosystems, and are intended to be a representative subset of the diversity within a typical aquatic community. In most cases, data on freshwater and estuarine/marine species are grouped separately to develop separate freshwater and estuarine/marine criteria. Thus, where data allow, four criteria are developed (acute freshwater, acute estuarine/marine, chronic freshwater, and chronic estuarine/marine). If plants are more sensitive than vertebrates and invertebrates, plant criteria are developed.

Table 2 provides a summary of the toxicity data used to fulfill the MDRs for calculation of acute and chronic criteria for both freshwater and estuarine/marine organisms. For aluminum, there are acceptable toxicity data for derivation of a freshwater acute criterion with all of the freshwater MDRs being met. The acceptable acute toxicity data encompass four phyla, 14 families, 20 genera and 22 species (**Table 2**). Acceptable estuarine/marine acute toxicity data are only available for three phyla, five families, five genera and five species. Consequently, only five of the eight MDRs are met for the estuarine/marine acute criterion; and no acceptable acute test data on fish species were available. Therefore, the EPA cannot develop an acute estuarine/marine criterion at this time. The chronic toxicity data for direct calculation of the FCV for the freshwater criterion consisted of seven of the eight freshwater MDRs (the missing MDR was the "other chordate"). However, the 1985 Guidelines still allow derivation of a chronic criterion (see **Section 2.6.1**). Because derivation of a chronic freshwater criterion is important for environmental protection, the EPA examined qualitative data for the Chordate MDR from **Appendix H** (*Other Data on Effects of Aluminum to Freshwater Aquatic Organisms*) and selected an amphibian test to fulfill that MDR. The species did not rank in the lowest four normalized Genus Mean Chronic Values (GMCVs) (the numeric-criteria-driving portion of the sensitivity distribution), and thus its use to fulfill the missing MDR is considered justified (U.S. EPA 2008). There are not enough chronic toxicity data for direct calculation of the FCV for the

estuarine/marine criteria (no acceptable estuarine/marine chronic studies), thus the EPA did not derive chronic estuarine/marine criterion. Aluminum toxicity data on estuarine/marine species remain a data gap; additional acute and chronic toxicity testing on estuarine/marine taxa would be needed in order to derive estuarine/marine criteria for aluminum.

Table 2. Summary of Acceptable Toxicity Data Used to Fulfill the Minimum Data Requirements in the 1985 Guidelines for Aluminum.

Family Minimum Data Requirement (Freshwater)	Acute (Phylum / Family / Genus)	Chronic (Phylum / Family / Genus)
Family Salmonidae in the class Osteichthyes	Chordata / Salmonidae / Oncorhynchus	Chordata / Salmonidae / Salvelinus
Second family in the class Osteichthyes	Chordata / Centrarchidae / Lepomis	Chordata / Cyprinidae / Pimephales
Third family in the phylum Chordata	Chordata / Cyprinidae / Pimephales	Chordata / Ranidae / Rana*
Planktonic Crustacean	Arthropoda / Daphniidae / Ceriodaphnia	Arthropoda / Daphniidae / Ceriodaphnia
Benthic Crustacean	Arthropoda / Crangonyctidae / Crangonyx	Arthropoda / Hyalellidae / Hyalella
Insect	Arthropoda/ Chironomidae/ Chironomus	Arthropoda / Chironomidae / Chironomus
Family in a phylum other than Arthropoda or Chordata	Mollusca / Physidae / Physa	Mollusca / Lymnaeidae / Lymnaea
Family in any order of insect or any phylum not already represented	Annelida / Naididae / Nais	Annelida / Aeolosomatidae / Aeolosoma
Family Minimum Data Requirement (Estuarine/Marine)	**Acute (Phylum / Family / Genus)**	**Chronic (Phylum / Family / Genus)**
Family in the phylum Chordata	No acceptable data	No acceptable data
Family in the phylum Chordata	No acceptable data	No acceptable data
Either the Mysidae or Penaeidae family	No acceptable data	No acceptable data
Family in a phylum other than Arthropoda or Chordata	Mollusca / Ostreidae / Crassostrea	No acceptable data
Family in a phylum other than Chordata	Annelida / Nereididae / Neanthes	No acceptable data
Family in a phylum other than Chordata	Annelida / Capitellidae / Capitella	No acceptable data
Family in a phylum other than Chordata	Annelida / Ctenodrilidae / Ctenodrilus	No acceptable data
Any other family	Arthropoda / Ameiridae / Nitokra	No acceptable data

* Data used qualitatively, see Section 3.2.1.

	Freshwater Acute			Freshwater Chronic			Estuarine/Marine Acute			Estuarine/Marine Chronic		
Phylum	**Families**	**GMAVs**	**SMAVs**	**Families**	**GMCVs**	**SMCVs**	**Families**	**GMAVs**	**SMAVs**	**Families**	**GMCVs**	**SMCVs**
Annelida	1	1	1	1	1	1	3	3	3	-	-	-
Arthropoda	5	7	9	3	4	4	1	1	1	-	-	-
Chordata	5	9	9	2	4	4	-	-	-	-	-	-
Mollusca	3	3	3	2	2	2	1	1	1	-	-	-
Rotifera	-	-	-	1	1	1	-	-	-	-	-	-
Total	**14**	**20**	**22**	**9**	**12**	**12**	**5**	**5**	**5**	**0**	**0**	**0**

2.7.1 pH, Total Hardness and DOC Normalization

Although many factors might affect the results of toxicity tests of aluminum to aquatic organisms (Sprague 1985), water quality criteria can quantitatively take into account only factors for which enough data are available to show that the factor similarly affects the results of tests with a variety of species. A variety of approaches were evaluated for the development of the freshwater aluminum criteria due to aluminum's unique chemistry and geochemical effects on its bioavailability. These included empirical models that directly relate water chemistry conditions to metal bioavailability and include single parameter regression models (e.g., hardness adjustment equations) and a variety of MLRs. The mechanistic models evaluated included an aluminum BLM model and a simplified aluminum BLM model. For further discussion, see **Section 5.3.5**.

A recent publication by Gensemer et al. (2018) summarized short-term aluminum chronic toxicity data across a range of pH, total hardness, and DOC values. Three-day toxicity tests measuring growth with the green alga (*Pseudokirchneriella subcapitata*), 7-day reproduction tests with the cladoceran (*Ceriodaphnia dubia*), and 7-day mean biomass tests with the fathead minnow (*Pimephales promelas*) were compiled to evaluate how the effect of pH, total hardness, and DOC alters aluminum bioavailability. The *P. subcapitata* data consisted of 27 tests with dilution water parameters that ranged from 6.14-8.0 for pH, 22-121 mg/L total hardness and 0.3-4.0 mg/L DOC (DeForest et al. 2018a). The *C. dubia* data consisted of 23 tests with test parameters that ranged from 6.3-8.1 for pH, 9.8-123 mg/L total hardness and 0.1-4 mg/L DOC (DeForest et al. 2018a). The fathead minnow data consisted of 22 tests with test parameters that ranged from 6.0-8.0 for pH, 10.2-127 mg/L total hardness and 0.08-5.0 mg/L DOC (DeForest et al. 2018a). DeForest et al. (2018a) used these data to evaluate the ability of MLR models to predict chronic toxicity of aluminum as a function of multiple combinations of pH, total hardness, and DOC conditions. These three parameters are thought to be the most influential for aluminum bioavailability and can be used to explain the scale of differences in the observed toxicity values (Cardwell et al. 2018; Gensemer et al. 2018). As a result of the public comments on the draft of this document released into the Federal Register, data on an additional nine *C. dubia* and nine *P. promelas toxicity* tests were obtained in order to expand the ranges of water chemistry conditions for model development. The new toxicity data expanded the DOC range up to 12.3 mg/L for *C. dubia* and 11.6 mg/L for *P. promelas* and the hardness range up to 428 mg/L

and 422 mg/L, respectively. These new data were subjected to an independent, external expert peer review, and an EPA quality review, prior to their use in the aluminum criteria. The external expert peer review comments on these new data obtained by the EPA in 2018 and the EPA's response to the external expert peer reviews can be found on the EPA website for the aluminum criteria (https://www.epa.gov/wqc/aquatic-life-criteria-aluminum).

The approach described by DeForest et al. (2018a,b) incorporated pH, total hardness, and DOC into MLR models to determine if the estimation of aluminum bioavailability to animals in freshwater aquatic systems could be applicable in the development of aluminum water quality criteria. The approach resulted in the creation of multiple MLR models that could be used for the development of aluminum water quality criteria following European Union (EU) (ECB 2003) and the EPA methodologies (Stephan et al. 1985). Only the MLR model development for the fathead minnow and *C. dubia* using EC_{20} effects concentrations is described below. Note that while a 7-day survival and growth test for *P. promelas* is not defined as an early-life stage (ELS) test per the 1985 Guidelines, testing demonstrated that it produced sensitivity values for total aluminum comparable to those generated via an acceptable ELS test (DeForest et al. 2018a, Table S1), and therefore, is considered appropriate to use for MLR model development.

MLR models for each species were developed using a multi-step process and the general approach is briefly described below. For more detailed information, figures, tables, and statistical results, please see DeForest et al. (2018a,b) and Brix et al. (2017). The authors first examined if any of the relationships between the dependent variable (total aluminum effect concentrations) and the three main effect terms (pH, total hardness and DOC; all independent variables) were non-linear. Effect concentrations (EC_{20}s) for each species were plotted against each independent variable using data where the other two parameters were held constant. Overall, EC_{20}s increased with each independent variable. However, there was some evidence of a unimodal relationship with pH, with increased EC_{20}s around pH 7 and decreasing EC_{20}s at low and high pH, as well as potential differences regarding the effects of total hardness at low and high pH (DeForest et al. 2018a). To account for these potential nonlinearities, the three potential two-way interactions (i.e., pH:hardness, DOC:hardness and pH:hardness) for each of the three main effect terms were added. Finally, a squared pH term was included in the initial models to account for the potential unimodal relationship between pH and aluminum bioavailability (DeForest et al. 2018a).

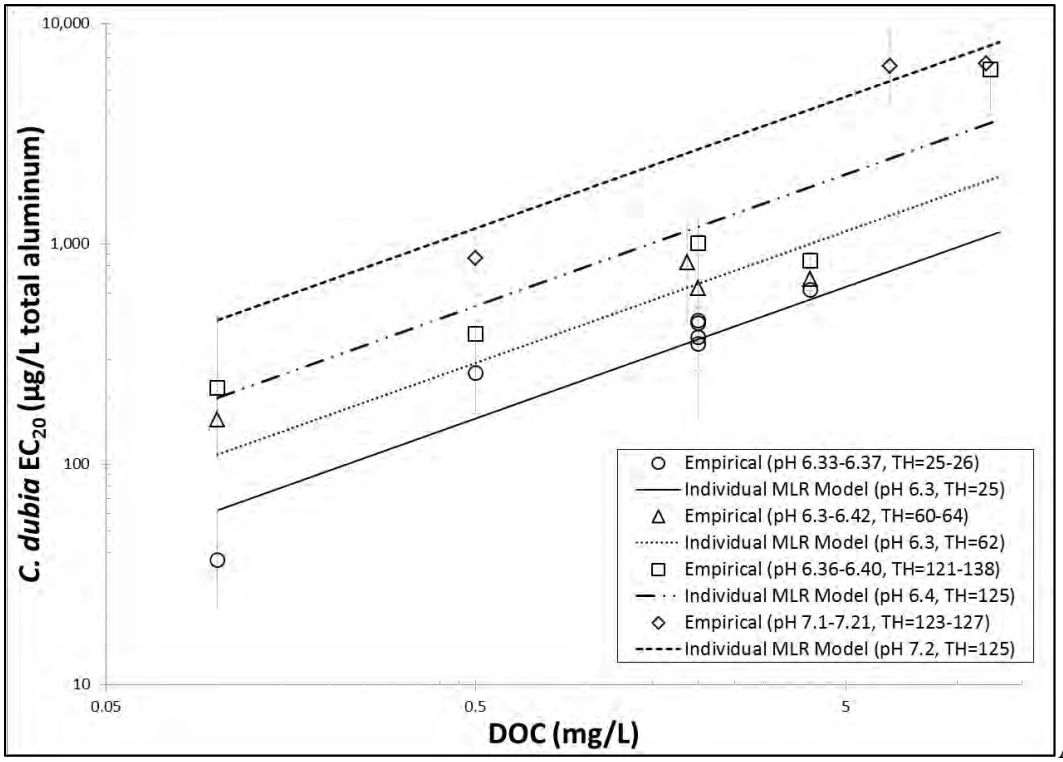
Beginning with a seven-parameter model consisting of the three main effect terms (pH, total hardness and DOC), the three two-way interactions for the main effects, and a squared pH term, a final model was developed for each species using a step-wise procedure. In this procedure, the original model was compared to a series of simpler models by removing one or more of the four "higher-level" terms (i.e., the three interaction terms and the squared pH term), until the most parsimonious model was developed. Each potential model was evaluated using Akaike Information Criterion (AIC) and Bayesian Information Criterion (BIC). The overall goodness of fit of a model increases with each additional model term. AIC and BIC penalize a model's goodness-of-fit by a factor related to the number of parameters in the model (DeForest et al. 2018a). AIC and BIC are minimized for the model that best balances overall goodness-of-fit and model complexity, as too many terms in the model may over extrapolate from the dataset making it less useful, whereas too few terms reduces its precision.

DeForest et al. (2018b) re-evaluated the original published models supplemented with the new data and developed a pooled MLR model based on the combined *C. dubia* and *P. promelas* datasets. A pooled model approach is described in Brix et al. (2017) for copper. In a pooled MLR model approach, species-specific intercepts are used to account for the differences in species sensitivity. The same procedures were used to develop a pooled model as was done for the individual species MLR models.

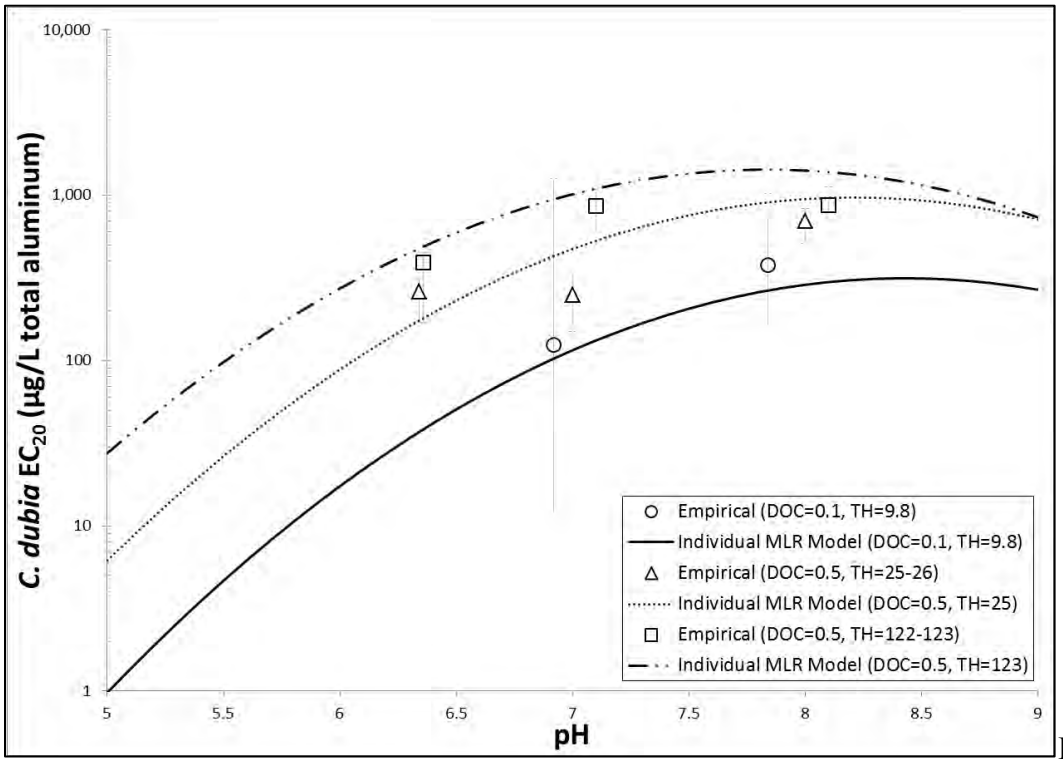
For *C. dubia*, the final individual MLR model, based on AIC and BIC, included both the pH:hardness interaction and the squared pH term (DeForest et al. 2018b). The negative pH^2 term accounts for the fact that aluminum bioavailability decreases from pH 6 to pH 7 and then increases from pH 7 to pH 8, which is expected given the unique solubility chemistry of aluminum (DeForest et al. 2018a). The negative pH:hardness term is reflective of the decreasing effects of total hardness mitigating toxicity as pH increases (DeForest et al. 2018a). The adjusted R^2 for the final model was 0.880, compared to an R^2 of 0.67 for the model consisting of the three main independent variables [pH, ln(total hardness), and ln(DOC)]. In the final MLR model, predicted EC_{20}s were within a factor of two of observed values used to create the model for 97% of the tests (DeForest et al. 2018b). The comparison of MLR predicted versus observed *C. dubia* values where one water chemistry parameter was varied is seen in **Figure 4** and **Figure 5**. No clear pattern was observed in the residuals over a wide range of water chemistry conditions or

relative to single independent variables (Figure S3-Figure S6, DeForest et al. 2018a). The final individual MLR model for *C. dubia* is:

$$C.dubia\ EC_{20} = e^{\left[-32.523+[0.597\times\ln(DOC)]+[2.089\times\ln(hard)]+(8.802\times pH)-(0.491\times pH^2)-[0.230\times pH:\ln(hard)]\right]}$$

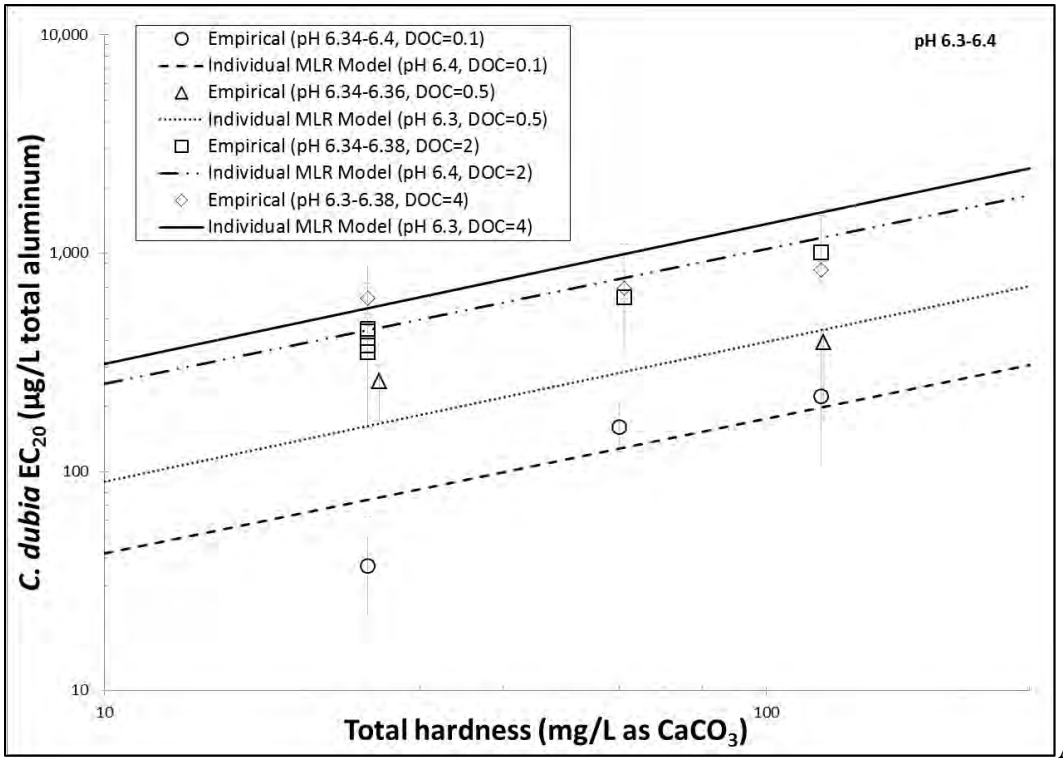


A

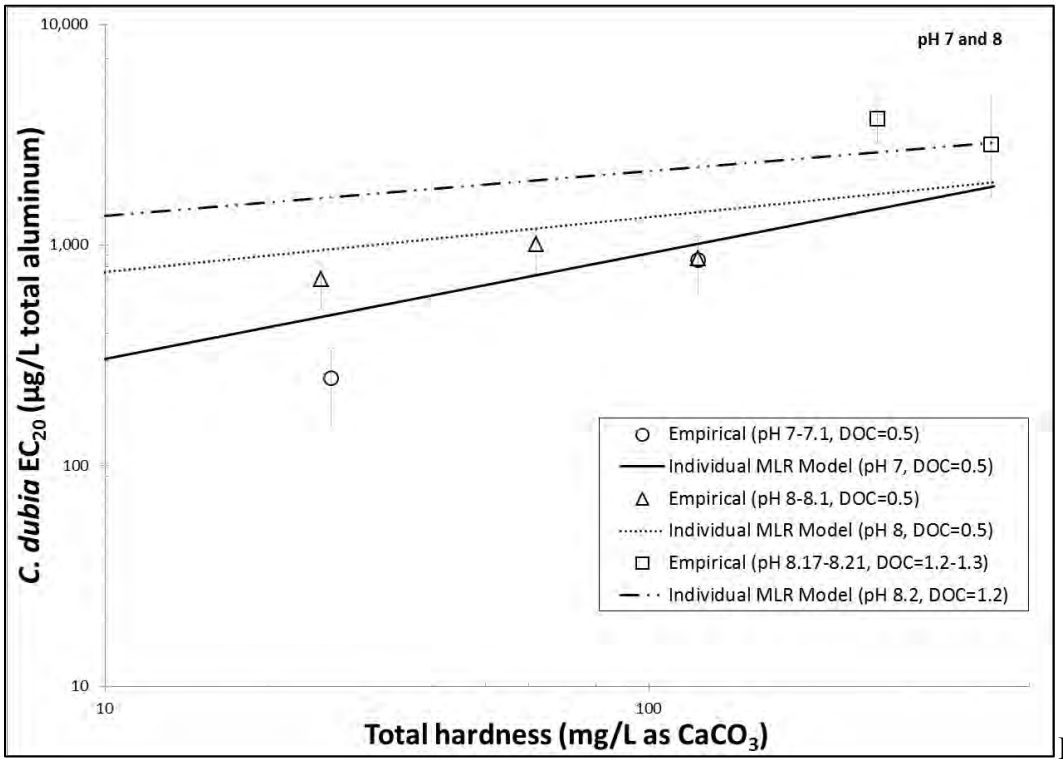


B

Figure 4. Observed and Individual MLR-Predicted Aluminum EC_{20}s (±95% CLs) for *C. dubia* where DOC or pH was Varied.
(Panel A: DOC is varied; Panel B: pH is varied; Adapted from Figure 2, from DeForest et al. 2018a, used with permission).



A

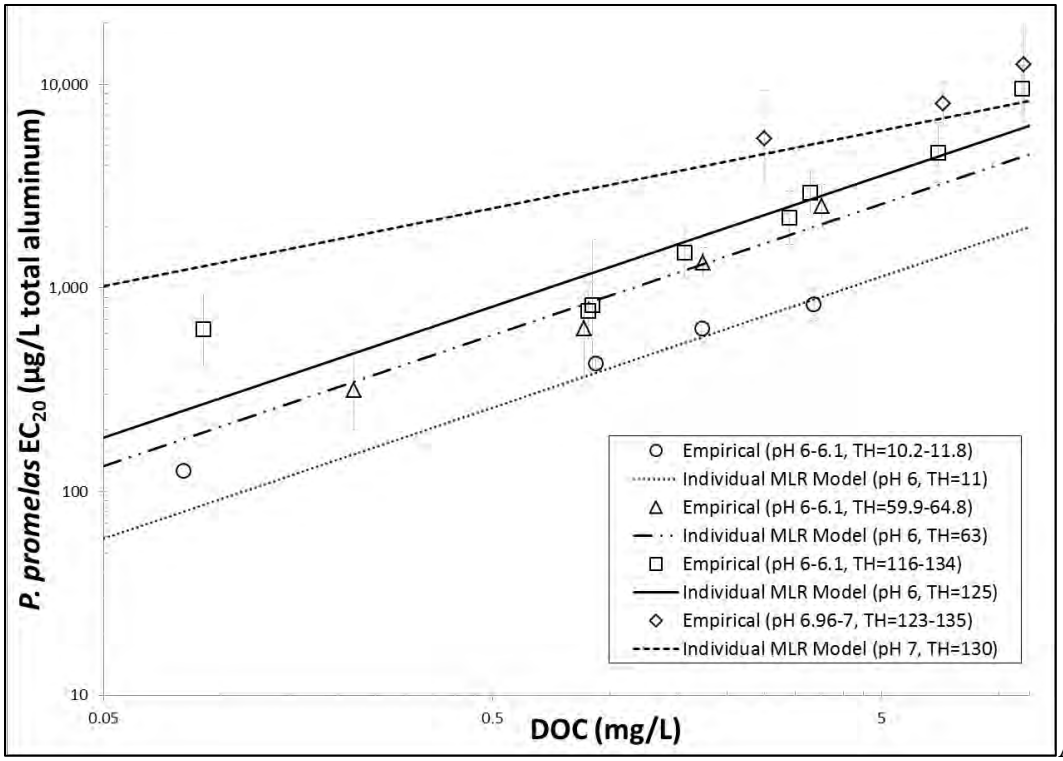


B

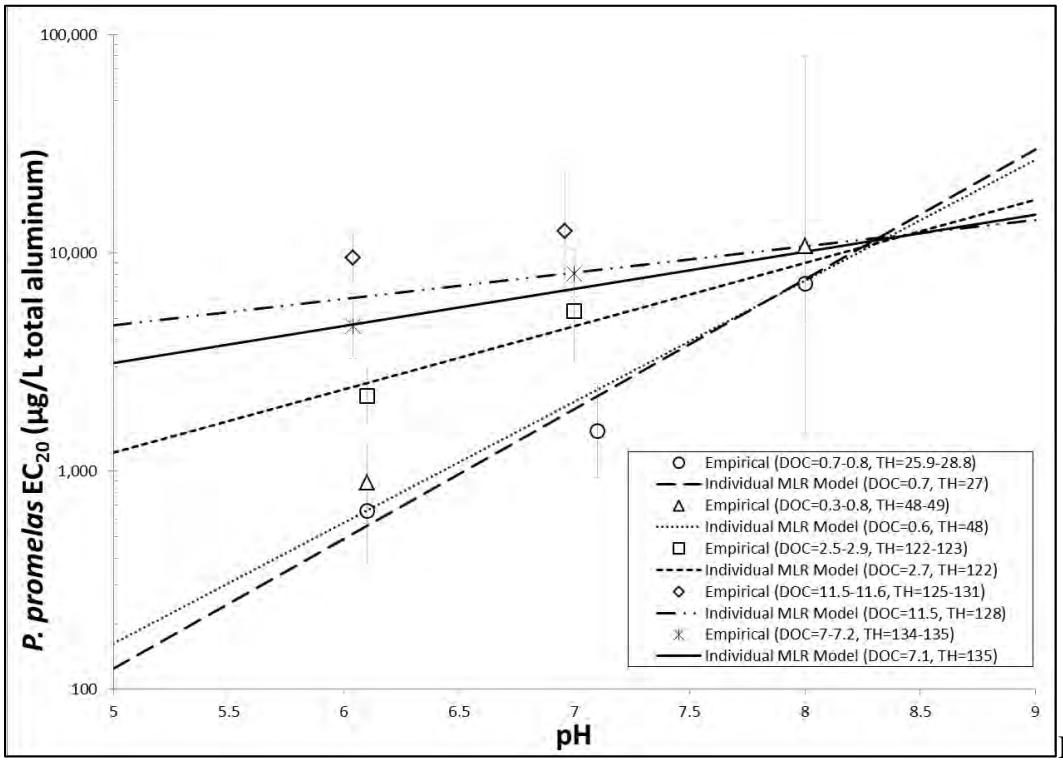
Figure 5. Observed and Individual MLR-Predicted Aluminum EC_{20}s (±95% CLs) for *C. dubia* where Total Hardness was Varied.
(Panel A: pH 6.3-6.4, Panel B: pH 7 and 8; Adapted from Figure 2, from DeForest et al. 2018a, used with permission).

For *P. promelas*, the final individual model, based on AIC and BIC, included the pH:hardness and pH:DOC interaction terms (DeForest et al. 2018b). The pH:hardness interaction term was retained because of the unique chemistry of aluminum where total hardness has less of a mitigating effect on bioavailability at higher pH levels (DeForest et al. 2018a; Gensemer et al. 2018). The adjusted R^2 for the final model was 0.923, compared to an R^2 of 0.85 for the model consisting of the three main independent variables [ln(DOC), pH, and ln(hardness)]. In the final MLR model, predicted EC_{20}s were within a factor of two of observed values used to create the model for 97% of the tests (DeForest et al. 2018b). The comparison of MLR predicted versus observed *P. promelas* values where one water chemistry parameter was varied is provided in **Figure 6** and **Figure 7**. Again, no clear pattern was observed in the residuals over a wide range of water chemistry conditions or relative to single independent variables (Figure S3-Figure S6, DeForest et al. 2018a). The final individual MLR model for *P. promelas* is:

$$P.promelas\ EC_{20} = e^{[-7.371+[2.209\times\ln(DOC)]+[1.862\times\ln(hard)]+(2.041\times pH)-[0.232\times pH:\ln(hard)]-[0.261\times pH:\ln(DOC)]]}$$



A



B

Figure 6. Observed and Individual MLR-Predicted Aluminum EC_{20}s (±95% CLs) for *P. promelas* where DOC or pH was Varied.
(Panel A: DOC, Panel B: pH; Adapted from Figure 3, from DeForest et al. 2018a, used with permission).

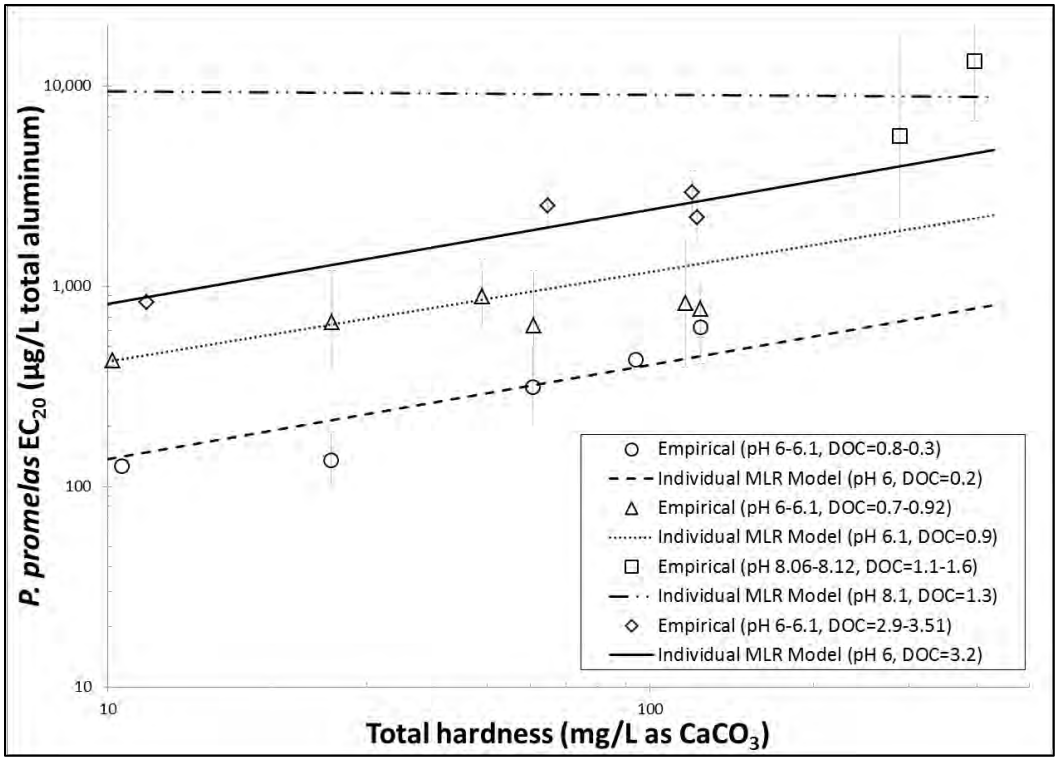


Figure 7. Observed and Individual MLR-Predicted Aluminum EC_{20}s (±95% CLs) for *P. promelas* where Total Hardness was Varied.
(Adapted from Figure 3, from DeForest et al. 2018a, used with permission).

The pooled MLR model performed similarly as the individual (fish and invertebrate) MLR models (DeForest et al. 2018b). The adjusted R^2 value, based on the BIC, was 0.882 and includes the pH:hardness interaction term. The pooled MLR model had a similar to identical level of accuracy as the individual MLR models with 97% of *C. dubia* and 94% of *P. promelas* predicted EC_{20}s within a factor of two of observed values (DeForest et al. 2018b). However, a comparison of the residuals between the observed and predicted values for the two models (individual vs. pooled MLR) showed that the individual models' residuals had smaller standard deviations. Additionally, the pooled model had some patterns in the residuals of the predictions relative to the independent variables (e.g., pH). There were no patterns in the residuals for either the *C. dubia* or *P. promelas* individual MLR models. The EPA elected to use the individual fish and invertebrate models in the final recommended aluminum aquatic life AWQC, instead of a pooled model for the above reasons. This modeling approach is also consistent with the approach in the draft 2017 aluminum criteria document. Additional analysis comparing the performance to the two model approaches (individual vs. pooled MLR) is presented in **Appendix L** (*EPA's MLR Model Comparison of DeForest et al. (2018b) Pooled and Individual-Species Model Options*).

The models developed followed the trends seen in the empirical data, 1) at pH 6 predicted effects concentrations increased with both total hardness and DOC concentrations, 2) at pH 7 predicted effect concentrations increased with DOC concentrations, but not total hardness, and 3) at pH 8 predicted effect concentrations increased with DOC concentrations, but predicted effect concentrations decreased with increased total hardness concentrations (DeForest et al. 2018a). The individual species models developed by DeForest et al. (2018b) were used to normalize the freshwater acute and chronic data in **Appendix A** and **Appendix C**, respectively. Invertebrate data were normalized using the individual MLR model for *C. dubia*, and vertebrate data were normalized using the individual MLR model for *P. promelas*. Invertebrate and vertebrate freshwater aluminum toxicity data were normalized with the following equations:

$$Invertebrate\ Normalized\ EC_{20}/LC_{50}$$

$$= e^{\left[\begin{array}{c}(\ln EC_{20,test}/LC_{50,test})-[0.597\times(\ln DOC_{test}-\ln DOC_{target})]-[8.802\times(pH_{test}-pH_{target})]-[2.089\times(\ln hard_{test}-\ln hard_{target})]\\ +[0.491\times(pH_{test}^2-pH_{target}^2)]+\left[0.230\times[(pH_{test}\times\ln hard_{test})-(pH_{target}\times\ln hard_{target})]\right]\end{array}\right]}$$

$$Vertebrate\ Normalized\ EC_{20}/LC_{50}$$

$$= e^{\left[\begin{array}{c}\left(\ln\frac{EC_{20,test}}{LC_{50,test}}\right)-[2.209\times(\ln DOC_{test}-\ln DOC_{target})]-[2.041\times(pH_{test}-pH_{target})]-[1.862\times(\ln hard_{test}-\ln hard_{target})]\\ +\left[0.261\times[(pH_{test}\times\ln DOC_{test})-(pH_{target}\times\ln DOC_{target})]\right]\\ +\left[0.232\times[(pH_{test}\times\ln hard_{test})-(pH_{target}\times\ln hard_{target})]\right]\end{array}\right]}$$

where:

$EC_{20,test}$	=	reported chronic total aluminum effect concentration in µg/L
$LC_{50,test}$	=	reported acute total aluminum effect concentration in µg/L
DOC_{test}	=	reported test DOC concentration in mg/L
pH_{test}	=	reported test pH
$hard_{test}$	=	reported test total hardness concentration in mg/L as $CaCO_3$
DOC_{target}	=	DOC value to normalize to in mg/L
pH_{target}	=	pH value to normalize to
$hard_{target}$	=	total hardness value to normalize to in mg/L as $CaCO_3$

Throughout this document, unless otherwise stated, effect concentrations were normalized to pH 7, total hardness of 100 mg/L and DOC of 1 mg/L. This example scenario is illustrative only and

is not meant to represent water quality characteristics typical of U.S. natural waters. Normalized values will be different under differing water chemistry conditions as identified in this document.

2.7.2 Acute Criterion

Acute criteria are derived from the sensitivity distribution of compiled genus mean acute values (GMAVs), calculated from species mean acute values (SMAVs) of acceptable data. SMAVs are calculated using the geometric mean for all acceptable toxicity tests within a given species (e.g., all tests for *Daphnia magna*). If only one test is available, the SMAV is that test value by default. As stated in the 1985 Guidelines, flow-through measured test data are normally given preference over other test exposure types (i.e., renewal, static, unmeasured) for a species, when available. When relationships are apparent between life-stage and sensitivity, only values for the most sensitive life-stage are considered. GMAVs are then calculated using the geometric means of all SMAVs within a given genus (e.g., all SMAVs for genus *Daphnia* - *Daphnia pulex*, *Daphnia magna*). If only one SMAV is available for a genus, then the GMAV is represented by that value. GMAVs are then rank-ordered by sensitivity from most sensitive to least sensitive.

Acute criteria are based on the Final Acute Value (FAV). The FAV is determined by regression analysis based on the four most sensitive genera (reflected as GMAVs) in the data set to interpolate or extrapolate (as appropriate) to the 5^{th} percentile of the sensitivity distribution represented by the tested genera. The intent of the eight MDRs is to serve as a representative sample of the aquatic community. These MDRs represent different ecological, trophic, taxonomic and functional differences observed in the natural aquatic ecosystem. Use of a sensitivity distribution where the criteria values are based on the four most sensitive taxa in a triangular distribution represents a censored statistical approach that improves estimation of the lower tail (where most sensitive taxa are) when the shape of the whole distribution is uncertain, while accounting for the total number of genera within the whole distribution.

The acute criterion, defined as the Criterion Maximum Concentration (CMC), is the FAV divided by two, which is intended to provide an acute criterion protective of nearly all individuals in such a genus. The use of the factor of two to reduce the FAV to the criterion magnitude is based on analysis of 219 acute toxicity tests on a range of chemicals, as described in the *Federal Register* on May 18, 1978 (43 FR 21506-18). For each of these tests, mortality data were used to determine the highest test concentration that did not cause mortality greater than that observed in the control for that particular test (which would be between 0 and 10% for

an acceptable acute test). Thus, dividing the LC_{50}-based FAV by two decreases potential acute effects to a level comparable to control mortality levels. Therefore, the acute criterion is expected to protect 95% of species in a representative aquatic community from acute effects.

2.7.3 Chronic Criterion

The chronic criterion, defined as the Criterion Continuous Concentration (CCC), may be determined by one of two methods. If all eight MDRs are met with acceptable chronic test data, then the chronic criterion is derived using the same method used for the acute criterion, employing chronic values (e.g., EC_{20}) estimated from acceptable toxicity tests. In cases where fewer chronic data are available (i.e., must have at least three chronic tests from taxa that also have appropriate acute toxicity data), the chronic criterion can be derived by determining an appropriate acute-chronic ratio (ACR).

The criteria presented are the EPA's estimate of maximum concentrations of aluminum to protect most aquatic organisms from any unacceptable short- or long-term effects. Results of such intermediate calculations such as Species Mean Acute Values (**Appendix A** and **Appendix B**) and chronic values (**Appendix C** and **Appendix D**) are specified to four significant figures to prevent round-off error in subsequent calculations; the number of places beyond the decimal point does not reflect the precision of the value. The acute and chronic criteria are rounded to two significant figures.

3 EFFECTS ANALYSES

Data for aluminum were obtained from studies published in the open literature and identified in a literature search using the ECOTOXicology database (ECOTOX) as meeting data quality standards. ECOTOX is a source of high quality toxicity data for aquatic life, terrestrial plants, and wildlife. The database was created and is maintained by the EPA, Office of Research and Development, and the National Health and Environmental Effects Research Laboratory's Mid-Continent Ecology Division. The latest comprehensive literature search for this document via ECOTOX was conducted in 2017 and supplemented by additional data researchers made available to the EPA in 2018.

A further evaluation of the quality of the available data was performed by the EPA to determine test acceptability for criteria development. Appendix A of *Quality Criteria for Water*

1986 (U.S. EPA 1986) provides an in-depth discussion of the minimum data requirements and data quality requirements for aquatic life criteria development.

3.1 Acute Toxicity to Aquatic Animals

All available reliable data relating to the acute effects of total aluminum on aquatic animals were considered in deriving the aluminum criteria. Data suitable (in terms of test acceptability and quality in a manner consistent with the 1985 Guidelines) for the derivation of a freshwater and an estuarine/marine FAV are presented in **Appendix A** (*Acceptable Acute Toxicity Data of Aluminum to Freshwater Aquatic Animals*) and **Appendix B** (*Acceptable Acute Toxicity Data of Aluminum to Estuarine/Marine Aquatic Animals*), respectively. Most fish and invertebrate data are LC_{50} measures from acute toxicity tests that were 96 hours in duration, except the tests for cladocerans, midges, mysids and certain embryos and larvae of specific estuarine/marine groups, which were 48 hours in duration and typically EC_{50} endpoints (per the 1985 Guidelines).

3.1.1 Freshwater

Twenty-two freshwater species encompassing 20 genera are represented in the dataset of acceptable data for acute toxicity to aluminum. The water quality conditions for these 118 toxicity tests ranged from 5.0-8.3 for pH, 2-220 mg/L as $CaCO_3$ for total hardness, and 0.48-4.0 mg/L for DOC. Since these three parameters affect the bioavailability, and hence toxicity of aluminum, all of the acceptable acute toxicity data presented in **Appendix A** were normalized to standardized water quality conditions using the MLR equations described in the Analysis Plan (**Section 2.7.1**). However, the dilution water DOC concentration was not reported for a number of acute studies presented in **Appendix A**. In this situation, where only the DOC was lacking, default values were used for several different dilution waters using a methodology documented in the 2007 freshwater copper AWQC document (see Appendix C, U.S. EPA 2007b). Specifically, the default DOC value for: 1) laboratory prepared reconstituted water is 0.5 mg/L, 2) Lake Superior water is 1.1 mg/L, 3) city tap and well water is 1.6 mg/L, and 4) Liberty Lake, Washington water is 2.8 mg/L. These values were determined from empirical data obtained for each source water.

Once normalized, the toxicity data were compiled (i.e., based on the geometric mean for each species and genus) and ranked by GMAV into a sensitivity distribution. Normalizing the toxicity data to the same pH, total hardness and DOC levels allows comparisons to be made

because the MLR derived equations address the differences seen in the magnitude of effects when comparing across conditions. However, because the 118 toxicity tests were each conducted at different water quality conditions, the MLR derived equations may have either a minor or major effect on the magnitude of the observed reported effects depending on the set of conditions to which the tests are normalized. Thus, the relative sensitivity rankings can change depending on what pH, hardness and DOC concentrations are selected for normalization (see **Appendix K** for examples).

All values reported in this section are normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$, and DOC of 1.0 mg/L (see **Section 2.7.1** for more information). Several species tested were not exposed to aluminum concentrations high enough or low enough to allow calculation of an LC_{50} (i.e., the LC_{50} is a "greater than" or "less than" value). The decision rule for using these non-definitive LC_{50}s to calculate SMAVs is consistent with methods used previously in criteria development. The freshwater ammonia AWQC document explains how chronic values (e.g., EC_{20}s) can be evaluated for potential use in deriving SMCVs (U.S. EPA 2013). The methodology is based on the finding that "greater than" values for concentrations of low magnitude, and "less than" values for concentrations of high magnitude do not generally add significant information to the toxicity analysis. The decision rule was applied as follows: "greater than" (>) low chronic values and "less than" (<) high chronic values were not used in the calculation of the SMCV; but "less than" (<) low chronic values and a "greater than" (>) high chronic values were included in the SMCV (U.S. EPA 2013). This approach was also followed for acute SMAV calculations.

While non-definitive SMAVs were ranked in **Table 3** according to the highest concentration used in the test, the value does not necessarily imply a accurate ranking of sensitivities. Again, in this section and below, the relative rankings are presented for comparative purposes and only apply when the set of chemistry conditions are pH 7, total hardness of 100 mg/L and DOC of 1.0 mg/L. SMAVs ranged from 1,836 µg/L for the cladoceran, *Daphnia pulex*, to 119,427 µg/L for the snail, *Melanoides tuberculata*. There is no apparent trend between freshwater taxon and acute sensitivity to aluminum (**Table 3**). The smallmouth bass, *Micropterus dolomieu*, represents the second most sensitive genus; cladocerans represent the first and fourth most sensitive genera; fish genera rank second, third, sixth and seventh in the sensitivity distribution; and an ostracod (*Stenocypris*) ranks fifth.

Other fish species were less sensitive with SMAVs of 18,913 µg/L for the brook trout, *Salvelinus fontinalis*, greater than 22,095 µg/L for the fathead minnow, *Pimephales promelas*, greater than 31,087 µg/L for the green sunfish, *Lepomis cyanellus*, and greater than 21,779 µg/L for the Rio Grande silvery minnow, *Hybognathus amarus.* The midge (*Chironomus plumosus*, SMAV = 25,216 µg/L), the aquatic air-breathing snail (*Physa sp.*, SMAV = 41,858 µg/L), and the freshwater juvenile mussel (*Lampsilis siliquoidea*, SMAV = >29,492 µg/L) were comparatively insensitive to aluminum.

Summary of Studies Used in Acute Freshwater Determination

The taxa used in calculating the acute criterion (the lowest four ranked GMAVs) depends on the set of water quality conditions for which the criterion is being derived. Based on the analysis in **Appendix K** (*Recommended Criteria for Various Water Chemistry Conditions*), a combination of several genera will rank in the lowest four. Those acute studies used to calculate the GMAVs are summarized below. The normalized values mentioned below are for pH of 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1.0 mg/L.

Invertebrates

Cladoceran, *Daphnia*

The pH/total hardness/DOC-normalized GMAV of 2,325 µg/L aluminum for *Daphnia* is based on the SMAVs for two cladoceran species, *Daphnia magna* and *D. pulex*. The *D. magna* normalized SMAV (2,944 µg/L) is based on the geometric mean of five 48-hr EC_{50}s (ranged from 713.2 to 15,625 µg/L aluminum) as reported by Biesinger and Christensen (1972), European Aluminum Association (2009), Kimball (1978) and Shephard (1983). All tests were static that exposed <24-hr old neonates, and only the Kimball (1978) test measured aluminum concentrations and did not use nominal concentrations. The *D. pulex* normalized SMAV (1,836 µg/L) is based on only one static-renewal unmeasured toxicity test conducted by Griffitt et al. (2008).

Cladoceran, *Ceriodaphnia*

Two species of *Ceriodaphnia*, *C. dubia* and *C. reticulata*, are used to derive the pH/total hardness/DOC-normalized GMAV of 7,771 µg/L aluminum. The *C. dubia* SMAV of 5,863 µg/L aluminum is calculated from 52 normalized EC_{50} values that ranged from 322.4 to greater than 88,933 µg/L aluminum (ENSR 1992d; European Aluminum Association 2009, 2010; Fort and Stover 1995; Gensemer et al. 2018; Griffitt et al. 2008; McCauley et al. 1986; Soucek et al.

2001). The tests were a mix of static or renewal exposures with either measured or unmeasured aluminum concentrations. The *C. reticulata* normalized SMAV of 10,299 µg/L aluminum is based on the two flow-through measured test results reported by Shephard (1983).

Ostracod, *Stenocypris major*

Shuhaimi-Othman et al. (2011a, 2013) reported a 96-hr LC_{50} of 3,102 µg/L aluminum for the ostracod, *S. major*, which equates to a pH/total hardness/DOC-normalized LC_{50}/SMAV/GMAV of 8,000 µg/L total aluminum. The adult organisms were exposed to static-renewal conditions and the test solutions were measured.

Worm, *Nais elinguis*

Shuhaimi-Othman et al. (2012a, 2013) reported a 96-hr LC_{50} of 3,874 µg/L aluminum for the worm, *Nais elinguis* which equates to a pH/total hardness/DOC-normalized LC_{50}/SMAV/GMAV of 9,224 µg/L total aluminum. Adult worms were exposed to aluminum sulfate under static-renewal conditions and the test solutions were measured.

Vertebrates

Rainbow trout, *Oncorhynchus mykiss*

Eight acute toxicity tests for the rainbow trout (*O. mykiss*) were used to calculate the pH/total hardness/DOC-normalized SMAV of 3,312 µg/L aluminum reported by Gundersen et al. (1994). The eight flow-through measured normalized LC_{50}s ranged from 1,680 to 7,216 µg/L aluminum.

Atlantic salmon, *Salmo salar*

Two acceptable acute values reported by Hamilton and Haines (1995) were used to calculate the SMAV/GMAV for the Atlantic salmon, *S. salar*. The sac fry were exposed in static, unmeasured chambers at a total hardness of 6.8 mg/L (as $CaCO_3$) and two different pH levels. The 96-hr LC_{50} values were 584 and 599 µg/L total aluminum conducted at pH levels of 5.5 and 6.5, respectively. The corresponding pH/total hardness/DOC-normalized values are 20,749 and 3,599 and the resulting normalized SMAV/GMAV for the species is 8,642 µg/L total aluminum.

Smallmouth bass, *Micropterus dolomieu*

Three acceptable acute values from one study (reported in both Kane 1984; Kane and Rabeni 1987) are available for the smallmouth bass, *M. dolomieu*. The 48-hr post hatch larva were exposed in static, measured concentration chambers at a total hardness of ~12 mg/L (as $CaCO_3$) and three different pH levels. The LC_{50} values were 130, greater than 978.4 and greater

than 216.8 µg/L total aluminum conducted at pH levels of 5.05, 6.25 and 7.5, respectively. The corresponding pH/total hardness/DOC-normalized values are 2,442, greater than 3,655 and greater than 153.4 µg/L. The SMAV/GMAV of 2,988 µg/L for the species/genus is based on the geometric mean of the normalized LC_{50} of 2,442 and greater than 3,655 µg/L total aluminum since the other value (greater than 153.4) is unbounded (i.e., greater than value), and is considered a "greater than" (>) low acute value.

GMAVs for 20 freshwater genera are provided in **Table 3**, and the four most sensitive genera were within a factor of 3.3 of each other. The freshwater FAV (the 5th percentile of the genus sensitivity distribution, intended to protect 95 percent of the genera) for aluminum normalized to a pH 7, total hardness of 100 mg/L and DOC of 1.0 mg/L is 1,961 µg/L, calculated using the procedures described in the 1985 Guidelines. The FAV is an estimate of the concentration of aluminum corresponding to a cumulative probability of 0.05 in the acute toxicity values for the genera with which acceptable acute tests have been conducted (**Table 4**). The FAV is lower than all of the GMAVs for the tested species. The FAV is then divided by two for reasons described above (see **Section 2.7.2**). Based on the above, the FAV/2, which is the freshwater continuous maximum concentration (CMC), for aluminum normalized to a pH 7, total hardness of 100 mg/L and DOC of 1.0 mg/L is 980 µg/L total aluminum (rounded to two significant figures) and is expected to be protective of 95% of freshwater genera potentially exposed to aluminum under short-term conditions (**Figure 8**). However, the freshwater acute toxicity data are normalized using MLR equations that predict the bioavailability and hence toxicity of aluminum under different water chemistry conditions. Thus, the value of the criterion for a given site will depend on the specific pH, total hardness, and DOC concentrations at the site (see **Appendix K** *Recommended Criteria for Various Water Chemistry Conditions* for additional criteria values and four most sensitive genera for each set of conditions).

Table 3. Ranked Freshwater Genus Mean Acute Values at pH 7, Total Hardness of 100 mg/L, and DOC of 1.0 mg/L.

(Note: Values will be different under differing water chemistry conditions as identified in this document).

Rank[a]	GMAV (µg/L total Al)	Genus	Species	SMAV[b] (µg/L total Al)
20	119,427	Melanoides	Snail, *Melanoides tuberculata*	119,427
19	>70,647	Paratanytarsus	Midge, *Paratanytarsus dissimilis*	>70,647
18	41,858	Physa	Snail, *Physa sp.*	41,858
17	>31,087	Lepomis	Green sunfish, *Lepomis cyanellus*	>31,087
16	>29,492	Lampsilis	Fatmucket, *Lampsilis siliquoidea*	>29,492
15	>27,766	Hyalella	Amphipod, *Hyalella azteca*	>27,766
14	25,216	Chironomus	Midge, *Chironomus plumosus*	25,216
13	>22,095	Pimephales	Fathead minnow, *Pimephales promelas*	>22,095
12	>21,779	Hybognathus	Rio Grande silvery minnow, *Hybognathus amarus*	>21,779
11	18,913	Salvelinus	Brook trout, *Salvelinus fontinalis*	18,913
10	>18,563	Hyla	Green tree frog, *Hyla cinerea*	>18,563
9	12,901	Crangonyx	Amphipod, *Crangonyx pseudogracilis*	12,901
8	9,224	Nais	Worm, *Nais elinguis*	9,224
7	9,061	Poecilia	Guppy, *Poecilia reticulata*	9,061
6	8,642	Salmo	Atlantic salmon, *Salmo salar*	8,642
5	8,000	Stenocypris	Ostracod, *Stenocypris major*	8,000
4	7,771	Ceriodaphnia	Cladoceran, *Ceriodaphnia dubia*	5,863
			Cladoceran, *Ceriodaphnia reticulata*	10,299
3	3,312	Oncorhynchus	Rainbow trout, *Oncorhynchus mykiss*	3,312
2	2,988	Micropterus	Smallmouth bass, *Micropterus dolomieu*	2,988
1	2,325	Daphnia	Cladoceran, *Daphnia magna*	2,944
			Cladoceran, *Daphnia pulex*	1,836

[a] Ranked from the most resistant to the most sensitive based on Genus Mean Acute Value.

[b] From Appendix A: Acceptable Acute Toxicity Data of Aluminum to Freshwater Aquatic Animals (all values normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$, and DOC of 1.0 mg/L).

Table 4. Freshwater Final Acute Value and Criterion Maximum Concentration (normalized to pH 7, total hardness of 100 mg/L and DOC of 1.0 mg/L).

(See Appendix K for acute criterion under different water chemistry conditions).

Calculated Freshwater FAV based on 4 lowest values: Total Number of GMAVs in Data Set = 20

Rank	Genus	GMAV (µg/L)	lnGMAV	$(lnGMAV)^2$	P=R/(n+1)	SQRT(P)
4	Ceriodaphnia	7,771	8.96	80.25	0.190	0.436
3	Oncorhynchus	3,312	8.11	65.70	0.143	0.378
2	Micropterus	2,988	8.00	64.04	0.095	0.309
1	Daphnia	2,325	7.75	60.08	0.048	0.218
		Σ (Sum):	**32.82**	**270.1**	**0.476**	**1.34**

S^2 = 31.13 **S = slope**

L = 6.334 **L = X-axis intercept**

A = 7.581 **A = lnFAV**

P = cumulative probability

FAV = 1,961 µg/L total aluminum

CMC (acute criterion) = 980 µg/L total aluminum (rounded to two significant figures)

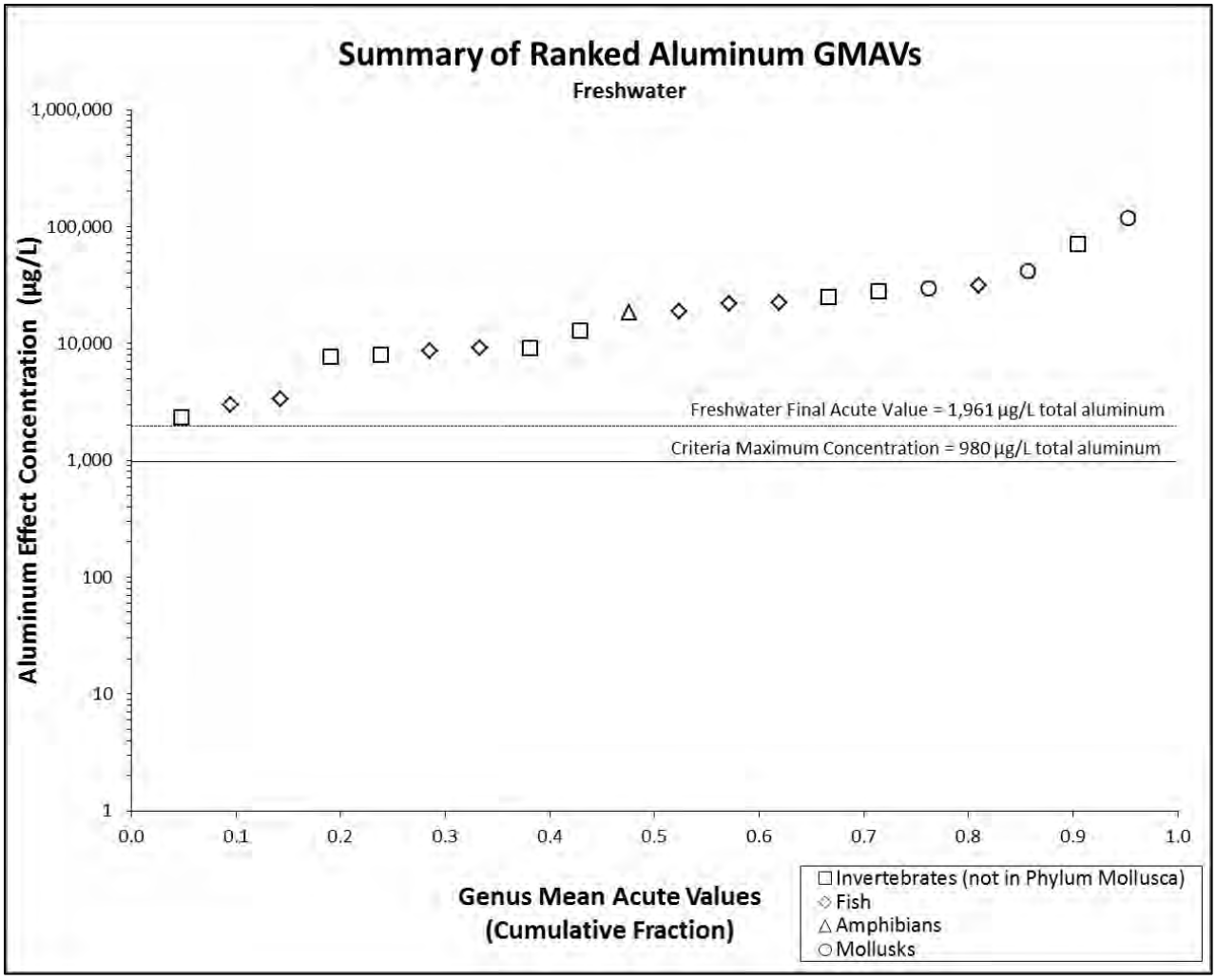


Figure 8. Ranked Summary of Total Aluminum Genus Mean Acute Values (GMAVs) - Freshwater at pH 7, Total Hardness of 100 mg/L, and DOC of 1.0 mg/L.

3.1.2 Estuarine/Marine

The 1985 Guidelines require that data from a minimum of eight families are needed to calculate an estuarine/marine FAV. Notably, no acceptable test data on fish species were available (**Figure 9**). Since data are available for only five families, an estuarine/marine FAV (and consequently the EPA cannot derive an estuarine/marine acute criterion).

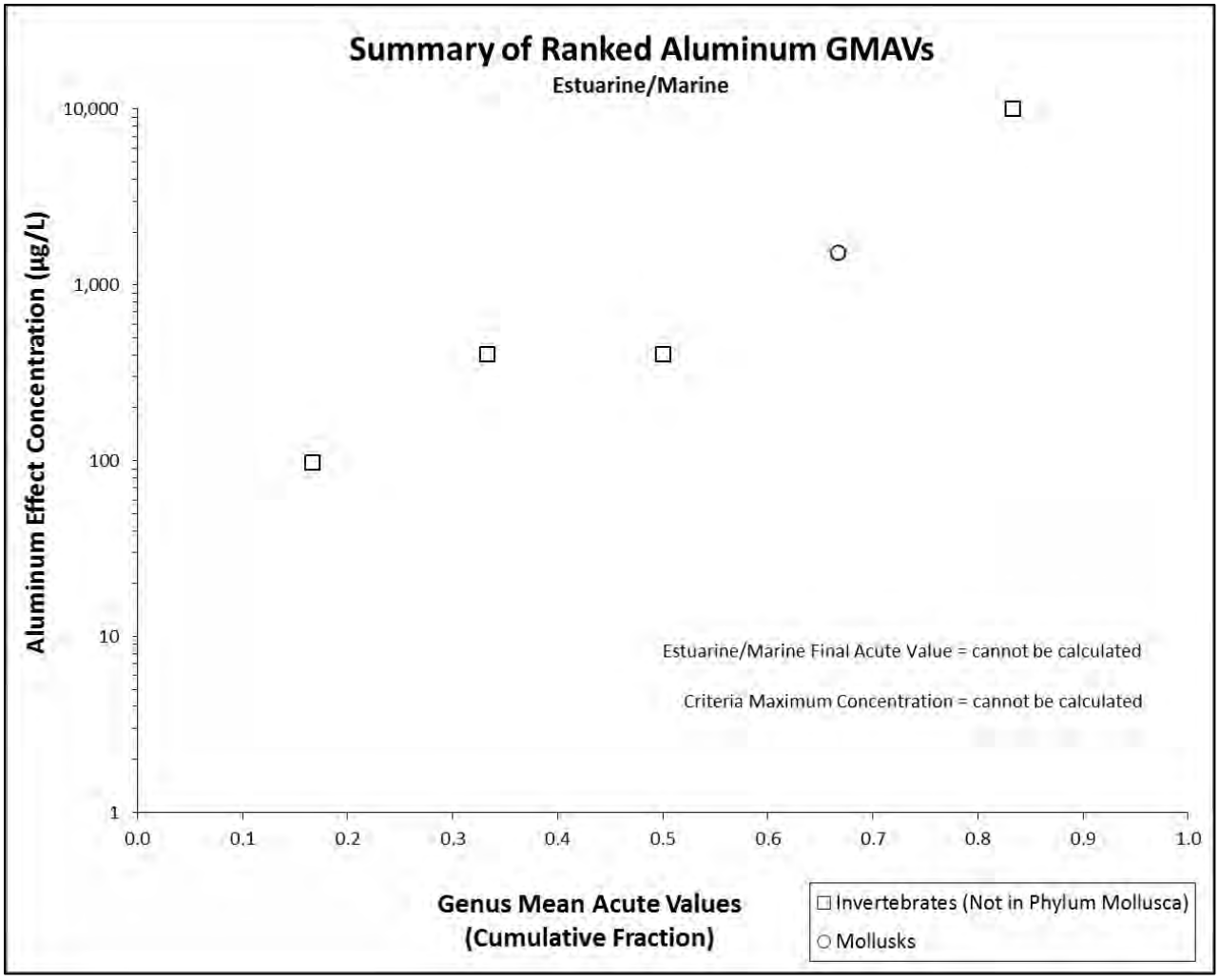


Figure 9. Ranked Summary of Total Aluminum Genus Mean Acute Values (GMAVs) - Estuarine/Marine.

3.2 Chronic Toxicity to Aquatic Animals

3.2.1 Freshwater

Freshwater chronic toxicity data that meet the test acceptability and quality assurance/control criteria (in a manner consistent with the 1985 Guidelines) are presented in **Appendix C** (*Acceptable Chronic Toxicity Data of Aluminum to Freshwater Aquatic Animals*). All tests were conducted with measured concentrations of total aluminum and measurement endpoints are EC_{20}s for all but one test where an EC_{20} could not be calculated. Details on chronic tests are described below. As with the freshwater acute SMAVs/GMAVs, the relative SMCV/GMCV rankings will change depending on the specific pH, total hardness and DOC values selected for data normalization. And as also described for the acute studies, the same

DOC default values were used for select chronic tests where the DOC concentration was lacking for specific dilution waters as provided by U.S. EPA (2007b). In addition, the DOC value reported by Cleveland et al. (1989) was applied to the studies by McKee et al. (1989), Palawski et al. (1989) and Buckler et al. (1995). All four studies used the same dilution water preparation, a mixture of well water and reverse osmosis-treated well water to obtain a low hardness (~13 mg/L as $CaCO_3$), and all four studies reported using the same dilution water preparation from Cleveland et al. (1986).

Aluminum chronic toxicity data are available for twelve species of freshwater organisms: two mollusks (a freshwater mussel and a snail), five other invertebrate species (a rotifer, two cladocerans, a midge, an oligochaete and an amphipod) and four fish species (fathead minnow, zebrafish, Atlantic salmon and brook trout). The water quality conditions for these 59 toxicity tests ranged from 5.1-8.7 for pH, 11.8-428 mg/L as $CaCO_3$ for total hardness, and 0.33-12.3 mg/L for DOC. All chronic values were normalized using the same MLR derived equations as the acute data (see **Section 2.7.1**). If aluminum reduced survival and growth, the product of these variables (biomass) was analyzed (when possible), rather than analyzing them separately (U.S. EPA 2013).

In this section and below, the relative rankings only apply when the set of chemistry conditions are pH 7, total hardness of 100 mg/L and DOC of 1.0 mg/L. Ranked GMCVs are provided in **Table 5**. The fish genus *Salmo*, represented by Atlantic salmon, was the most sensitive genus, and the least sensitive genus was represented by an oligochaete. There is no apparent trend between freshwater taxon and chronic sensitivity to aluminum.

Invertebrates

The chronic toxicity of aluminum to the freshwater unionid mussel, *Lampsilis siliquoidea*, was evaluated by Wang et al. (2016, 2018). Six-week old juvenile mussels were exposed under flow-through measured conditions for 28 days to five aluminum nitrate concentrations and dilution water control composed of a well water/deionized water mix adjusted to a nominal pH of 6.0 and total hardness of 100 mg/L as $CaCO_3$. The calculated biomass EC_{20} of 169 µg/L was reported in the study, with a corresponding normalized EC_{20} of 1,026 µg/L (normalized to pH 7, total hardness = 100 mg/L as $CaCO_3$ and DOC = 1.0 mg/L).

Several chronic aluminum studies were conducted in separate laboratories with the cladoceran, *Ceriodaphnia dubia* (CECM 2014; ENSR 1992b; European Al Association 2010;

Gensemer et al. 2018; McCauley et al. 1986; OSU 2018a). Aluminum chloride was evaluated by McCauley et al. (1986) at the University of Wisconsin-Superior using life cycle studies (*C. dubia* neonates, ≤16-hr old) in Lake Superior water (both raw and treated dechlorinated city water) to determine ACRs at near neutral pH. Five test concentrations plus a dilution water control were renewed three times over seven days, and the number of young per surviving adult was found to be significantly inhibited at 2,600 and 2,400 µg total aluminum/L in each respective dilution water. The EC_{20} and MATC were estimated to be 1,780 and <1,100 µg/L, respectively, or 2,031 and <925.5 µg/L after normalization. Poor dose response in the treated dechlorinated city water exposure prevented calculation of an EC_{20}.

Three-brood, 6-day static-renewal toxicity tests were conducted with aluminum chloride at four hardness levels using <24-hr old *C. dubia* neonates (ENSR 1992b). Reconstituted dilution water was prepared at nominal 25, 50, 100 and 200 mg/L total hardness as $CaCO_3$ and pH of 7.65, 7.7, 8.2 and 8.45, respectively. The mean number of young produced per female was the most sensitive endpoint with normalized (to pH 7, total hardness = 100 mg/L as $CaCO_3$ and DOC = 1.0 mg/L) EC_{20}s of 2,602, 1,077, 708.8 and 746.8 µg/L, respectively (**Appendix C**).

The Center for the Ecotoxicology and Chemistry of Metals (CECM 2014) and the European Al Association (2010) also evaluated the effect of aluminum on the survival and reproduction of *C. dubia* at different pH and total hardness levels. Less than 24-hr old neonates were exposed to aluminum nitrate for seven days using reconstituted laboratory water established at different nominal total hardness (25, 60 or 120 mg/L as $CaCO_3$), DOC (0.5, 2 or 4 mg/L) and pH (6.3, 7.0 or 8.0) levels. Test solutions were renewed daily and the pH was maintained with synthetic buffers (as summarized in Gensemer et al. 2018). Reproduction was the most sensitive endpoint, with EC_{20}s ranging from 36.6 to 1,011.6 µg/L aluminum, and corresponding normalized (to pH 7, total hardness = 100 mg/L as $CaCO_3$ and DOC = 1.0 mg/L) EC_{20}s ranging from 291.7 to 2,072 µg/L (**Appendix C**). A similar experiment was run with another cladoceran, *Daphnia magna*, except water chemistry parameters were not varied (European Al Association 2010; Gensemer et al. 2018). Less than 24-hr old neonates were exposed to aluminum nitrate for 21 days at a total hardness of 140 mg/L as $CaCO_3$, pH 6.3 and DOC of 2 mg/L. Again, reproduction (young per female) was the most sensitive endpoint with a reported EC_{20} of 791.0 µg/L total aluminum. The normalized SMCV/GMCV for the species is 985.3 µg/L.

Oregon State University researchers conducted nine additional aluminum toxicity studies with *Ceriodaphnia dubia* in 2018. The results of these tests allowed the EPA to expand on the bounds of the MLR model. Less than 24-hr old neonates were exposed to one of five aluminum nitrate concentrations for seven days using reconstituted laboratory water established at different nominal total hardness (60-400 mg/L as $CaCO_3$), DOC (1.0-14.0 mg/L) and pH (6.3-8.8) levels (OSU 2018a). Reproduction was the most sensitive endpoint with effect concentrations ranging from 828.6 to 6,612 µg/L total aluminum (1,170 to 2,308 µg/L when normalized using the MLR equation).

Two acceptable *Hyalella azteca* chronic studies are available for aluminum based on recently recommended culture and control conditions (Mount and Hockett 2015; U.S. EPA 2012). Researchers at Oregon State University exposed 7-9 day old juvenile amphipods to five aluminum nitrate concentrations diluted with a well water/reverse osmosis water mix for 42 days under flow-through conditions and a nominal pH of 6 (Cardwell et al. 2018; OSU 2012h). A small amount of artificially-formulated sediment was provided as substrate during the test. Biomass was the most sensitive endpoint, with a 28-day EC_{20} of 199.3 µg/L and a normalized EC_{20} of 665.9 µg/L aluminum (the 28-day results were used since the 79 percent control survival after 42 days was slightly below the 80 percent minimum requirement).

Wang et al. (2016, 2018) also conducted a *H. azteca* chronic test where 7-day old juvenile amphipods were exposed under flow-through measured conditions for 28 days to five aluminum nitrate concentrations and dilution water control composed of a well water/deionized water mix adjusted to a nominal pH of 6.0 and total hardness of 100 mg/L as $CaCO_3$. Silica sand was provided as a substrate. The calculated biomass EC_{20} was 425 µg/L, with a corresponding normalized EC_{20} of 2,890 µg/L (normalized to pH 7, total hardness = 100 mg/L as $CaCO_3$ and DOC = 1.0 mg/L).

Oregon State University (2012f) conducted a 28-day life cycle test with the midge, *Chironomus riparius*, in a mixture of well water and reverse osmosis water (pH range of 6.3-6.9). The authors reported an EC_{20} for the number of eggs per case to be 3,387 µg/L, or 8,181 µg total aluminum/L when normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1.0 mg/L. Palawski et al. (1989) also exposed *C. riparius*, but for 30 days at two pH levels (5.6 and 5.0). Larval midge (<24-hr) were exposed to five aluminum sulfate concentrations with a control under flow-through conditions. Adult midge emergence was significantly inhibited at

61.4 and 235.2 µg/L aluminum, at pH 5.6 and 5.0, with calculated EC_{20}s of 29.55 and 84.42 µg/L and normalized EC_{20}s of 1,075 and 15,069 µg/L, respectively. The resultant normalized SMCV of 5,099 µg/L is calculated from all three test results.

Oregon State University also conducted chronic studies for three invertebrate species: an oligochaete, *Aeolosoma sp.*; a rotifer, *Brachionus calyciflorus*; and the great pond snail, *Lymnaea stagnalis* (Cardwell et al. 2018; OSU 2012b,c,e). All tests were conducted with aluminum nitrate, and at a nominal pH of 6.0. The normalized EC_{20}s from the aforementioned studies are 20,514 (oligochaete 17-day population count), 1,845 (48-hr rotifer population count) and 5,945 (pond snail 30-day biomass) µg/L, respectively (**Appendix C**). The researchers also conducted a series of validation studies in 2018 with the rotifer and great pond snail at nominal pH 6.3, with various hardness and DOC levels (OSU 2018e,f). The normalized EC_{20}s ranged from 2,132 to 6,653 µg/L for *Brachionus calyciflorus* and 1,812 to 3,902 µg/L for *Lymnaea stagnalis*.

Vertebrates

Kimball (1978) conducted an early life stage test using fathead minnow (*Pimephales. promelas*) fertilized eggs (16 to 40-hr old) in flowing hard well water. Six treatments of aluminum sulfate plus control replicated four times were used to expose fish for 28 days post-hatch, and aluminum concentrations were measured three times per week during the study. Biomass was more sensitive to the aluminum exposures than percent hatchability, growth and survival, with a resulting EC_{20} of 6,194 µg/L, or 2,690 µg/L when normalized.

The chronic toxicity of aluminum to fathead minnows and zebrafish (*Danio rerio*) was also evaluated by OSU (2012g, 2013) and summarized in Cardwell et al. (2018). Fish were exposed under flow-through conditions in the same dilution water and pH as described above for the amphipod and midge tests (OSU 2012f,h). Less than 24-hr old fertilized fathead minnow eggs and less than 36-hr post fertilization zebrafish were exposed to aluminum nitrate for 33 days. Fathead minnow fry survival was the most sensitive endpoint with a calculated EC_{20} of 428.6 µg/L, and normalized EC_{20} of 2,154 µg/L. Zebrafish biomass was the most sensitive endpoint with a calculated EC_{20} of 234.4 µg/L (1,342 µg/L when normalized).

An early life cycle test was also conducted with brook trout (*Salvelinus fontinalis*). Brook trout eyed eggs were exposed to four aluminum sulfate concentrations at pH 5.7 and 6.5 for 60 days (Cleveland et al. 1989). Both exposures were conducted using flow-through conditions and soft water (total hardness = 12.5 mg/L as $CaCO_3$). The survival and hatching of eyed eggs and

the survival, growth, behavioral and biochemical responses of the resultant larvae and juveniles were measured during the exposure. The incomplete hatch endpoint reported in the study was not used after further analysis and communication with the authors because the incomplete hatch endpoint may or may not be a transient effect. The incompletely hatched larvae (based on chorion attachment) were removed daily from the study and not fully evaluated further for survivability. In addition, exposure to acidic waters increased the percentage of incomplete hatched larvae (Cleveland et al. 1986; Ingersoll et al. 1990c), and therefore it is difficult to distinguish between the effects of pH versus aluminum. Therefore, the lack of information and uncertainty with the endpoint led to the decision to not use the data from the study to develop the criteria document. The biomass EC_{20} for the test conducted at pH 5.7 was 143.5 µg/L, and at pH 6.5 the biomass EC_{20} was 164.4 µg/L. The normalized EC_{20}s at pH 5.7 and 6.5 were 1,076 µg/L and 378.7 µg/L, respectively.

Atlantic salmon eyed eggs were exposed to flow-through conditions for 60 days at pH 5.7 and a total hardness of 12.7 mg/L as $CaCO_3$ in reconstituted water (McKee et al. 1989). Salmon weight and survival NOEC and LOEC were 71 and 124 µg aluminum/L, respectively. The calculated biomass EC_{20} for the study was 61.56 µg/L (**Appendix C**). Buckler et al. (1995) also reported a chronic *Salmo salar* study initiated with eyed eggs in reconstituted water (total hardness of 12.7 mg/L as $CaCO_3$) that continued for 60 days post-hatch under flow-through exposure conditions. Time to hatch was not significantly affected at pH 5.7 and 264 µg/L, the highest test concentration evaluated. Survival at 60 days post hatch was reduced at 124 µg/L, with an estimated EC_{20} of 154.2 µg/L (normalized EC_{20} = 1,088 µg/L).

When calculating the Atlantic salmon EC_{20}s for the two studies (Buckler et al. 1995 and McKee et al. 1989), it was observed that the studies listed the same test concentrations and similar dose response for the same test measurements, but reported different endpoints between the two studies. It appears that the Buckler et al. (1995) study was a republication of the previous study performed by McKee et al. (1989), and therefore, only the most sensitive EC_{20} was used in the calculation of the SMCV. The most sensitive EC_{20} of 61.56 µg/L (or 434.4 µg/L when normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1.0 mg/L), was based on a 60-day reduction in fish biomass.

Only seven of the eight MDRs are met for direct calculation of the FCV, with the third family in the phylum Chordata missing. Because derivation of a chronic freshwater criterion is

important for environmental protection, the EPA examined qualitative data in **Appendix H** (*Other Data on Effects of Aluminum to Freshwater Aquatic Organisms*) to determine if any "Other Data" can be used to fulfill the missing MDR group, and selected an amphibian test to fulfill that MDR.

The MDR for the third family in the phylum Chordata was fulfilled using results of an abbreviated life cycle test initiated with wood frog (*Rana sylvatica*) larvae (Gosner stage 25) and continued through metamorphosis (Peles 2013). The NOEC for survival and growth normalized to a pH 7, total hardness of 100 mg/L and DOC of 1.0 mg/L was 10,684 µg/L, with a chronic value of greater than 10,684 µg/L. The study was not included in **Appendix C** because the test pH (4.68-4.70) was lower than 5. If not for the marginally lower pH (Peles 2013), this study would have been an acceptable chronic test for criterion derivation. The addition of this other chronic test does not directly affect the calculation of the FCV as the species does not rank in the lowest four GMCVs (the numeric-criteria-driving portion of the sensitivity distribution). The species was the most sensitive value from the qualitative data that could be used to fulfill the MDR and the test had a minor deviation in pH. After adding this additional study, the chronic dataset consists of 13 freshwater species representing 13 freshwater genera (**Table 5**).

The four most sensitive GMCVs are from the core quantitative chronic dataset and represent taxa which have been determined to be the most sensitive to aluminum. Based on these rankings, the resultant chronic criterion is 380 µg/L total aluminum at pH 7, total hardness of 100 mg/L (as $CaCO_3$) and DOC of 1.0 mg/L (**Table 6**). The chronic toxicity data are normalized using the MLR equations described in the Analysis Plan that account for the effects of pH, total hardness, and DOC on bioavailability and hence toxicity of aluminum. Thus, the value of the criterion for a given site will depend on the specific pH, total hardness, and DOC concentrations at the site (see **Appendix K** *Recommended Criteria for Various Water Chemistry Conditions* for additional criteria values and four most sensitive genera for each set of conditions). The EPA is confident that the criteria values generated using the MLR models are protective of approximately 95% of freshwater genera in an ecosystem that are potentially exposed to aluminum under long-term conditions (**Figure 10**).

Table 5. Ranked Genus Mean Chronic Values at pH 7, Total Hardness of 100 mg/L, and DOC of 1.0 mg/L.

(Note: Values will be different under differing water chemistry conditions as identified in this document).

Rank[a]	GMCV (µg/L total Al)	Genus	Species	SMCV[b] (µg/L total Al)
13	20,514	Aeolosoma	Oligochaete, *Aeolosoma sp.*	20,514
12	>10,684	Rana	Wood frog,[c] *Rana sylvatica*	>10,684
11	5,099	Chironomus	Midge, *Chironomus riparius*	5,099
10	3,539	Brachionus	Rotifer, *Brachionus calyciflorus*	3,539
9	3,119	Lymnaea	Great pond snail, *Lymnaea stagnalis*	3,119
8	2,407	Pimephales	Fathead minnow, *Pimephales promelas*	2,407
7	1,387	Hyalella	Amphipod, *Hyalella azteca*	1,387
6	1,342	Danio	Zebrafish, *Danio rerio*	1,342
5	1,181	Ceriodaphnia	Cladoceran, *Ceriodaphnia dubia*	1,181
4	1,026	Lampsilis	Fatmucket, *Lampsilis siliquoidea*	1,026
3	985.3	Daphnia	Cladoceran, *Daphnia magna*	985.3
2	638.2	Salvelinus	Brook trout, *Salvelinus fontinalis*	638.2
1	434.4	Salmo	Atlantic salmon, *Salmo salar*	434.4

[a] Ranked from the most resistant to the most sensitive based on Genus Mean Chronic Value.

[b] From Appendix C: Acceptable Chronic Toxicity Data of Aluminum to Freshwater Aquatic Animals (all values normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$, and DOC of 1.0 mg/L).

[c] Fulfills MDR for third family in phylum Chordata, used only qualitatively.

Table 6. Freshwater Final Chronic Value and Criterion Continuous Concentration (normalized to pH 7, total hardness of 100 mg/L and DOC of 1.0 mg/L).

(See Appendix K for chronic criterion under different water chemistry conditions).

Calculated Freshwater FCV based on 4 lowest values: Total Number of GMCVs in Data Set = 13

Rank	Genus	GMCV (µg/L)	lnGMCV	$(lnGMCV)^2$	P=R/(n+1)	SQRT(P)
4	Lampsilis	1,026	6.93	48.07	0.286	0.535
3	Daphnia	985.3	6.89	47.51	0.214	0.463
2	Salvelinus	638.2	6.46	41.71	0.143	0.378
1	Salmo	434.4	6.07	36.89	0.071	0.267
		Σ (Sum):	**26.36**	**174.2**	**0.714**	**1.64**

S^2 =	12.423	**S = slope**
L =	5.142	**L = X-axis intercept**
A =	5.930	**A = lnFCV**
		P = cumulative probability

FCV = 376.3 µg/L total aluminum

CCC (chronic criterion) = 380 µg/L total aluminum (rounded to two significant figures)

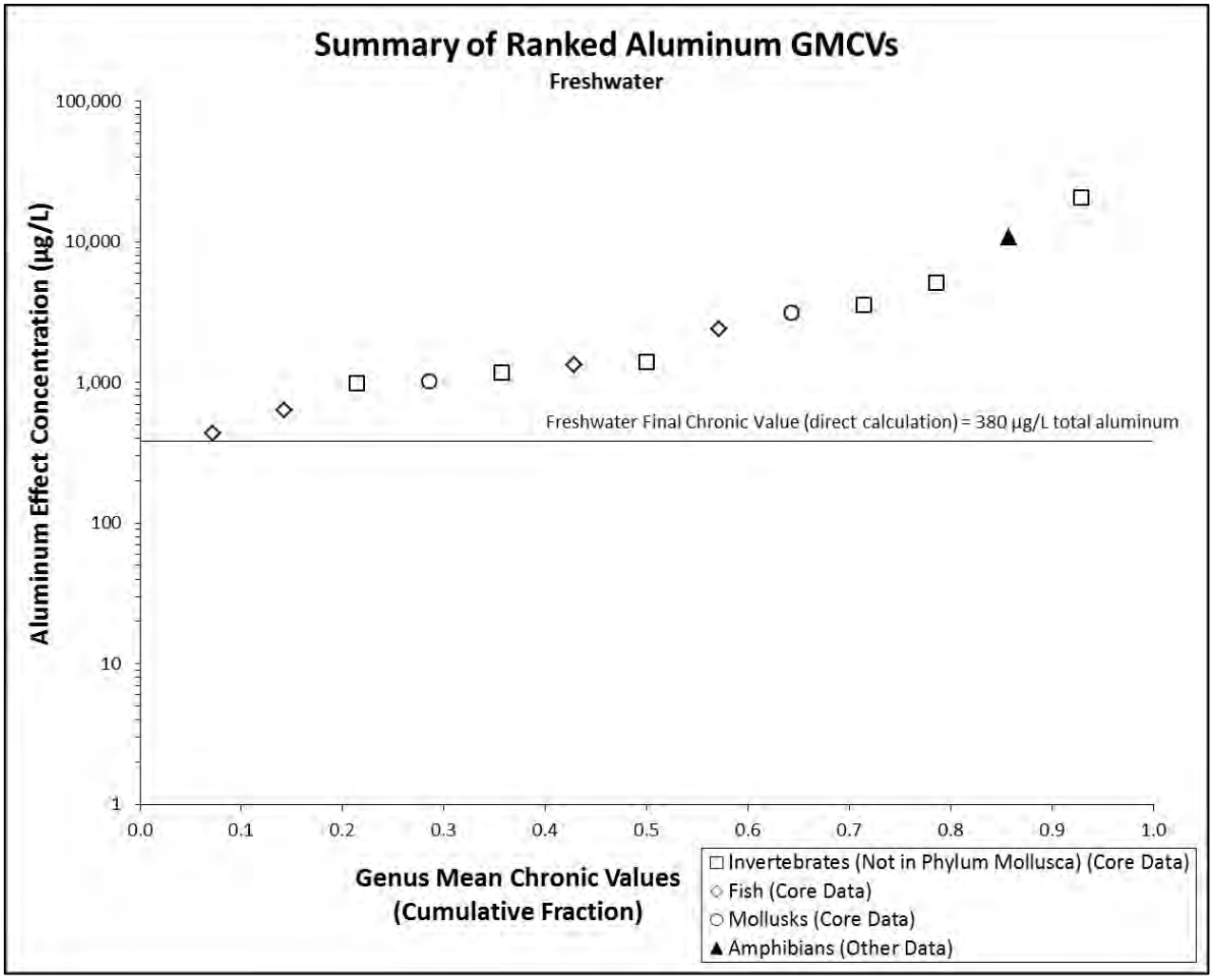


Figure 10. Ranked Summary of Total Aluminum Genus Mean Chronic Values (GMCVs) – Freshwater Supplemented with Other Data to Fulfill Missing MDRs at pH 7, Total Hardness of 100 mg/L, and DOC of 1.0 mg/L.

3.2.2 Estuarine/Marine

There are no estuarine/marine chronic toxicity data that meet the test acceptability and quality assurance/control criteria in a manner consistent with the 1985 Guidelines in **Appendix D** (*Acceptable Chronic Toxicity Data of Aluminum to Estuarine/Marine Aquatic Animals*).

3.3 Bioaccumulation

Aluminum bioaccumulates in aquatic organisms, although increased accumulation through trophic levels in aquatic food chains (i.e., biomagnification) is not usually observed (Suedel et al. 1994, U.S. EPA 2007a). Total uptake generally depends on the environmental aluminum concentration, exposure route and the duration of exposure (McGeer et al. 2003).

Desouky et al. (2002) reported that the bioavailability of aluminum to a grazing invertebrate is influenced by both oligomeric silica and humic acid, and that aluminum bound to humic acid may still be bioavailable via grazing. Bioconcentration Factors (BCFs) and bioaccumulation factors (BAFs) typically vary with the bioavailable concentration of metals in water, with higher BCFs occurring at lower metal concentrations (McGeer et al. 2003). In marine sediments, metal bioavailability is altered by increased acid volatile sulfide (AVS) content (Casas and Crecelius 1994), and ligand concentration (Skrabal et al. 2000). Bioaccumulation and toxicity via the diet are considered unlikely relative to direct waterborne aluminum toxicity (Handy 1993; Poston 1991). This conclusion is also supported by the lack of any biomagnification within freshwater invertebrates that are likely to be prey of fish in acidic, aluminum-rich rivers (Herrmann and Frick 1995; Otto and Svensson 1983; Wren and Stephenson 1991). The opposite phenomena, trophic dilution up the food chain, has been suggested (King et al. 1992). A more detailed discussion of bioaccumulation factors is provided in the Effects Characterization section (**Section 5.1.6**).

No U.S. Food and Drug Administration (FDA) action level or other maximum acceptable concentration in tissue, as defined in the 1985 Guidelines, is available for aluminum. Therefore, a Final Residue Value cannot be calculated for fish tissue.

3.4 Toxicity to Aquatic Plants

No aluminum toxicity tests with important aquatic plant species in which the concentrations of test material were measured and the endpoint was biologically important are available in the literature. Therefore, the EPA could not determine a Final Plant Value. However, analysis of plant data provides evidence that criteria magnitudes that are protective of aquatic animals will also be protective of aquatic plants. Effects on aquatic plants are discussed qualitatively in the Effects Characterization section (**Section 5.2**).

4 SUMMARY OF NATIONAL CRITERIA

4.1 Freshwater

The 2018 final aluminum criteria are derived using multiple linear regression (MLR) models that incorporate pH, total hardness, and DOC as input parameters to normalize the acute and chronic toxicity data to a set of predetermined water quality conditions. The MLR equations account for the effects of pH, total hardness and DOC on the bioavailability, and hence toxicity

of aluminum. The numeric magnitude of the criteria (acute or chronic criterion) for a given set of conditions, therefore, will depend on the specific pH, total hardness and DOC concentrations used for normalization. The relative GMAVs/GMCVs rankings and subsequent four most sensitive genera used to calculate the criteria will depend on the data normalization conditions selected. The acute and chronic criteria for a given set of input conditions (pH, total hardness and DOC) are numeric magnitude values that are protective for that set of input conditions. The recommended criteria for aluminum can be calculated in two different ways: 1) use the lookup tables provided (see **Appendix K** *Recommended Criteria for Various Water Chemistry Conditions*) to find the numeric aluminum acute and chronic criteria corresponding to the pH, total hardness and DOC conditions of interest, or 2) use the Aluminum Criteria Calculator V.2.0 (*Aluminum Criteria Calculator V.2.0.xlsm*) to enter the pH, total hardness and DOC conditions of interest.

For the purposes of illustration, the following criteria magnitude values are provided at pH 7, total hardness 100 mg/L and DOC of 1.0 mg/L. The resulting numeric values represent the concentrations at which freshwater aquatic organisms would have an appropriate level of protection if the one-hour average concentration of total aluminum does not exceed (in µg/L):

Criterion Maximum Concentration (CMC) =
980 µg/L total aluminum at a pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1.0 mg/L;

and if the four-day average concentration of total aluminum does not exceed (in µg/L):

Criterion Continuous Concentration (CCC) =
380 µg/L total aluminum at pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1.0 mg/L.

The criteria value for the specific water chemistry conditions of interest are recommended not to be exceeded more than once every three years on average.

The above illustrative criteria values would vary under other water chemistry conditions for the three water quality parameters (pH, total hardness and DOC) that affect the expression of aluminum toxicity (see **Appendix K** *Recommended Criteria for Various Water Chemistry Conditions*). **Table 7** provides a detailed break-down of the freshwater acute (CMC) and chronic

(CCC) criteria across different pH and total hardness levels when the DOC = 1.0 mg/L. **Appendix K** provides additional criteria values across pH and total hardness levels when DOC = 0.1, 0.5, 2.5, 5, 10 and 12 mg/L, and provides the four most sensitive genera for both the acute and chronic criteria. The empirical toxicity test data that the EPA used to develop the MLR models were developed under a specific range of water chemistry conditions as described below.

The pH of toxicity test waters ranged from 6.0-8.7. Specifically, *Ceriodaphnia dubia* toxicity test data ranged 6.3-8.7 for pH (only one *C. dubia* toxicity test was conducted at pH 8.7; the majority of tests were conducted at pH less than 8.3); *Pimephales promelas* toxicity test data ranged 6.0-8.12 for pH. The EPA has determined that for pH it is reasonable to allow the user to extrapolate beyond these values for criteria derivations. The criteria calculator can be used to address all waters within a pH range of 5.0 to 10.5. Thus, criteria values for pH input values beyond the range of the underlying empirical pH data used for model development (pH 6.0 to 8.2) can be generated using the criteria calculator. This is also reflected in the criteria lookup tables in **Appendix K**. The EPA took this approach for pH so that the recommended criteria can be provided for, and thus are protective of, a broader range of U.S. natural waters. Extrapolated criteria values outside of the empirical pH data tend to be more protective of the aquatic environment (i.e., lower criteria values) in situations where pH plays a critical role in aluminum toxicity. However, criteria values generated outside of the range of the pH conditions of the toxicity tests underlying the MLR models are more uncertain than values within the pH conditions of the MLR toxicity tests, and thus should be considered carefully and used with caution. Although the EPA has provided model predictions of criteria values outside the empirical range for pH, these values may warrant further exploration and consideration for site-specific criteria. Additional information regarding the uncertainty associated with the MLR models is provided in **Section 5.3.6** and **Appendix L**.

The total hardness of toxicity test waters ranged from 9.8 to 428 mg/L. More specifically, total hardness (as $CaCO_3$) ranged from 9.8-428 mg/L for *Ceriodaphnia dubia* toxicity tests and from 10.2-422 mg/L for *Pimephales promelas* toxicity tests. Since a decrease in total hardness tends to increase aluminum toxicity, the EPA has determined it is reasonable to extrapolate on the lower bound of the hardness data to enable generation of more stringent criteria at low hardnesses beyond the limit of the empirical data. Thus, hardness input values in the criteria calculator can be entered that are less than 9.8 mg/L down to a limit of 0.01 mg/L. This is

consistent with existing EPA approaches to low end hardness (U.S. EPA 2002). However, criteria values are bounded at the approximate upper limit of the empirical MLR models' underlying hardness data, at a maximum of 430 mg/L total hardness (as $CaCO_3$). The user can input hardness values into the criteria calculator that are greater than 430 mg/L for total hardness, but the criteria magnitude will reach its maximum value at 430 mg/L total hardness (as $CaCO_3$), and criteria magnitudes will not increase or decrease by increasing the hardness above 430 mg/L total hardness (as $CaCO_3$). This is also consistent with existing EPA guidance on high end hardness "caps" (U.S. EPA 2002). These total hardness bound approaches are also reflected in the criteria lookup tables in **Appendix K**. The EPA took this approach so that the recommended criteria can be provided for, and will be protective of, a broader range of natural waters found in the U.S. Criteria values generated beyond the lower bound of the hardness conditions of the toxicity tests underlying the MLR models are more uncertain than values within the hardness bounds of the MLR toxicity test data.

The DOC of toxicity test waters ranged from 0.08 to 12.3 mg/L. More specifically DOC ranged from 0.1-12.3 mg/L for *Ceriodaphnia dubia* toxicity tests and 0.08-11.6 mg/L for *Pimephales promelas* toxicity tests. Since most natural waters contain some DOC, the lower bound of the empirical toxicity test data (0.08 mg/L) is the lowest value that can be entered into the criteria calculator; thus no extrapolation below the lowest empirical DOC of 0.08 mg/L is provided. The criteria values generated with the criteria calculator are bounded at the upper limit of the empirical MLR models' underlying DOC data, at a maximum 12.0 mg/L DOC. The user can input DOC values greater than 12.0 mg/L into the calculator, but the criteria magnitude will reach its maximum value at 12.0 mg/L DOC, and criteria magnitudes will not increase or decrease by increasing the DOC above 12.0 mg/L. This is also reflected in the criteria lookup tables in **Appendix K**. This is consistent with the existing approach for hardness (U.S. EPA 2002) to provide for protection of aquatic organisms through the use of protective, conservative values when water chemistry conditions are beyond the upper limits of the empirical toxicity test data.

The EPA created the Aluminum Criteria Calculator V.2.0 (*Aluminum Criteria Calculator V.2.0.xlsm*) that allows users to enter the pH, total hardness and DOC based on water sampling and automatically calculates freshwater criteria for these site-specific parameters based on the bounds described above. Existing data on these water chemistry parameters may be helpful in

determining criteria calculator input values. The criteria calculator gives a warning when any of the water quality parameters entered are "outside MLR model inputs," to alert end users. As noted above, total hardness and DOC concentrations entered into the calculator that are greater than the bounds recommended will automatically default to a maximum limit; pH values that are outside the bounds recommended (i.e., pH<6, pH>8.2) can be used, but should be considered carefully and used with caution. As displayed in **Table 7** and **Appendix K**, total hardness and DOC are bounded at a maximum of 430 mg/L as $CaCO_3$ and 12.0 mg/L, respectively. **Table 7** shows example freshwater acute (CMC) and chronic (CCC) criteria at DOC of 1.0 mg/L and various water total hardness levels and pH, with additional tables for other DOC values are provided in **Appendix K**.

Table 7. Freshwater Acute and Chronic Criteria at Example Conditions of DOC of 1.0 mg/L and Various Water Total Hardness Levels and pH.

(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Total Hardness	Acute Criteria (µg/L total aluminum)												
	pH												
	5.0	5.5	6.0	6.5	7.0	7.5	8.0	8.2	8.5	9.0	9.5	10.0	10.5
10	*4.0*	*19*	70	190	430	810	1,200	1,200	*1,300*	*1,100*	*720*	*370*	*150*
25	*9.5*	*40*	130	310	620	1,100	1,400	1,500	*1,400*	*1,100*	*660*	*310*	*110*
50	*18*	*72*	210	430	790	1,300	1,700	1,700	*1,600*	*1,100*	*610*	*270*	*90*
75	*27*	*100*	260	520	900	1,400	1,800	1,800	*1,700*	*1,100*	*590*	*240*	*79*
100	*35*	*130*	320	590	980	1,500	1,900	1,900	*1,700*	*1,100*	*570*	*230*	*72*
150	*51*	*170*	400	700	1,100	1,600	2,100	2,100	*1,800*	*1,100*	*550*	*210*	*63*
200	*67*	*220*	470	790	1,200	1,700	2,200	2,200	*1,900*	*1,100*	*540*	*200*	*57*
250	*82*	*260*	540	870	1,300	1,800	2,200	2,200	*1,900*	*1,100*	*530*	*190*	*53*
300	*98*	*300*	600	950	1,400	1,900	2,300	2,300	*2,000*	*1,100*	*520*	*180*	*50*
350	*110*	*340*	650	1,000	1,500	1,900	2,300	2,300	*2,000*	*1,200*	*510*	*180*	*48*
400	*130*	*380*	700	1,100	1,600	2,000	2,400	2,400	*2,100*	*1,200*	*500*	*170*	*46*
430	*140*	*400*	730	1,100	1,600	2,000	2,400	2,400	*2,100*	*1,200*	*500*	*170*	*45*

Total Hardness	Chronic Criteria (µg/L total aluminum)												
	pH												
	5.0	5.5	6.0	6.5	7.0	7.5	8.0	8.2	8.5	9.0	9.5	10.0	10.5
10	*2.5*	*12*	47	110	240	500	730	770	*790*	*670*	*450*	*230*	*95*
25	*5.9*	*25*	81	160	300	580	970	930	*890*	*680*	*410*	*190*	*71*
50	*11*	*46*	110	200	340	620	1,100	1,100	*980*	*690*	*380*	*170*	*56*
75	*17*	*66*	140	220	360	640	1,100	1,200	*1,000*	*700*	*370*	*150*	*49*
100	*22*	*85*	160	240	380	650	1,100	1,300	*1,100*	*700*	*360*	*140*	*45*
150	*32*	*120*	190	260	400	660	1,100	1,300	*1,100*	*710*	*350*	*130*	*39*
200	*42*	*140*	210	290	420	670	1,100	1,300	*1,200*	*710*	*340*	*120*	*36*
250	*51*	*160*	230	300	430	670	1,100	1,300	*1,300*	*720*	*330*	*120*	*33*
300	*61*	*180*	250	320	440	680	1,100	1,300	*1,300*	*720*	*320*	*110*	*31*
350	*71*	*200*	260	330	450	680	1,100	1,300	*1,400*	*720*	*320*	*110*	*30*
400	*80*	*220*	280	340	470	680	1,100	1,300	*1,400*	*720*	*310*	*110*	*29*
430	*86*	*230*	290	350	470	680	1,100	1,300	*1,400*	*720*	*310*	*110*	*28*

4.2 Estuarine/Marine

Insufficient data are available to fulfill the MDRs for estuarine/marine criteria development, therefore no criteria are recommended at this time.

5 EFFECTS CHARACTERIZATION

This section characterizes the potential effects of aluminum on aquatic life based on available test data and describes additional lines of evidence not used directly in the criteria calculations, but which support the 2018 criteria values. This section also provides a summary of the uncertainties and assumptions associated with the criteria derivation and explanations for the decisions the EPA made regarding data acceptability and usage in the effects assessment. Finally, this section describes substantive differences between the 1988 aluminum AWQC and the 2018 update.

5.1 Effects on Aquatic Animals

5.1.1 Freshwater Acute Toxicity

The EPA identifies several acute studies that did not meet data quality screening guidelines for inclusion in criterion calculations (**Appendix H** *Other Data on Effects of Aluminum to Freshwater Aquatic Organisms*), but showed similar ranges of toxicity and are presented here to provide additional supporting evidence of the observed toxicity of aluminum to aquatic organisms.

Among Mollusca studies where the pH was greater than or equal to 5, Harry and Aldrich (1963) observed adverse 24-hr effects to the snail *Taphius glabratus* exposed to aluminum at concentrations between 1,000-5,000 µg/L in distilled water (**Appendix J** *List of Aluminum Studies Not Used in Document Along with Reasons*). In contrast, the 24-hr LC_{50} of 130,500 µg/L (65,415 µg/L when normalized to conditions in **Appendix A**) for the zebra mussel *Dreissena polymorpha* (Mackie and Kilgour 1995) was insensitive, similar to the mollusk *Physa sp.* (**Appendix A**). In a series of 96-hr tests conducted at low pH and total hardness (15.3 mg/L as $CaCO_3$) levels, Mackie (1989) found that *Pisidium casertanum* and *Pisidium compressum* did not reach 50% mortality at 1,000 µg/L when pH was 3.5, and 400 µg/L when pH was 4.0 and 4.5; the highest concentrations tested. When these concentrations are normalized to the conditions in **Appendix A**, LC_{50}s for the species would be greater than 412,645 to greater than 72,075,634 µg/L.

Among cladocerans, Call et al. (1984) observed an unidentified *Ceriodaphnia* species to be similarly acutely sensitive to identified *Ceriodaphnia* species in acceptable tests, with pH/total hardness/DOC-normalized 48-hr LC_{50} values of 2,277 µg/L and 3,083 µg/L. Also

similar to results observed among acceptable tests and supporting studies, *Daphnia sp.* was more acutely sensitive than *Ceriodaphnia sp.* For example, Havas and Likens (1985b) observed reduced survival in *Daphnia catawba* for a test with a non-standard test duration (72 hours) at a pH/total hardness/DOC-normalized concentration of 4,341 µg/L; Khangarot and Ray (1989) observed a normalized 48-hr LC_{50} of 23,665 µg/L for *Daphnia magna* exposed to an unacceptable form of aluminum (aluminum ammonium sulfate); and Havas (1985) observed a normalized 48-hr LOEC based on survival of 1,343 µg/L in *Daphnia magna* using lake water as dilution water.

Although no data from benthic crustaceans were used to calculate the freshwater acute criterion (not one of the four most sensitive genera), evidence suggests they are somewhat acutely sensitive to aluminum. The isopod *Asellus aquaticus* was found to be somewhat sensitive to aluminum, with a pH/total hardness/DOC-normalized 72-hr LC_{50} of 12,284 µg/L that was not included because of the test duration (Martin and Holdich 1986). The isopod values would fall 8th out of 20 in relative acute sensitivity to aluminum, despite the decreased length of the acute test over standard acute invertebrate test durations. Both Borgmann et al. (2005) and Mackie (1989) conducted acute toxicity tests with the amphipod *Hyalella azteca*. Seven-day LC_{50}s from the two Borgmann et al. (2005) studies comparing soft reconstituted water and dechlorinated tap water were 212.7 and greater than 2,978 µg/L, respectively (values normalized to **Appendix A** conditions), but these data were not included because of both test length and unacceptable control mortality. Three (pH/total hardness/DOC) adjusted unbounded *H. azteca* LC_{50} values reported by Mackie (1989) ranged from greater than 4,455 to greater than 178,365 µg/L, the highest concentrations tested. The lowest of these values would rank this taxon 4th in the acute genus sensitivity. These data were included in **Appendix H** because of uncertainty regarding whether bromide and chloride concentrations in dilution water met the recently established testing requirements for *H. azteca* (Mount and Hockett 2015; U.S. EPA 2012). The author was not able to provide details regarding bromide and chloride water concentrations, but noted that there was 100% survival in the experiment, suggesting that conditions were met (Gerry Mackie, personal communication, March 2013). In addition, no substrate was provided for the test organisms. Although some substrate is recommended for water only tests with *H. azteca*, the absence of substrate does not invalidate a test result (Mount and Hockett 2015; U.S. EPA 2012). Because the value is unbounded (i.e., a greater than value), the study most likely overestimates

the toxicity of aluminum to this species, since the test failed to reach 50% mortality at the highest concentrations tested.

Studies by Vuori (1996) (caddisfly), Mackie (1989) (damselfly) and Rockwood et al. (1990) (dragonfly) suggest some insects may be acutely sensitive to aluminum, but these tests were either conducted at pH<5 (Mackie 1989), or used an atypical endpoint for acute exposures (Rockwood et al. 1990; Vuori 1996). However, when the concentrations are normalized to the conditions in **Appendix A**, LC_{50}s for the damselfly would be greater than 412,645 to greater than 72,075,634 µg/L. (Note: Rockwood et al. and Vuori did not report test hardness so values could not be normalized).

Consistent with data used to calculate the freshwater acute criterion, vertebrates were no more or less sensitive overall to aluminum than invertebrates. Also consistent with vertebrate data from **Appendix H**, acute toxicity data for fish, while variable, provide additional evidence that freshwater fish are acutely sensitive to aluminum. DeLonay et al. (1993) observed reduced 7-day survival of *Oncorhynchus aguabonita* alevin and swim-up larvae exposed to 18,359 µg/L aluminum (pH/total hardness/DOC-normalized). Cutthroat trout (*O. clarkii*) alevin and swim-up larvae also exposed at pH 5 for seven days exhibited reduced survival at 482.0 µg/L (60% reduction) and 340.8 µg/L (~50% reduction) (pH/total hardness/DOC-normalized), respectively (Woodward et al. 1989). Both studies were excluded from acute criteria calculations because of the atypical acute test duration.

In two studies examining the effects of aluminum on rainbow trout survival, pH/total hardness/DOC-normalized *O. mykiss* LC_{50}s after 6 and 7-12 days, respectively, were 2,837 and 460.0 µg/L (Birge et al. 2000; Orr et al. 1986). In two tests with embryo/larva rainbow trout at pH 6.5 and 7.2, Holtze (1983) observed no reduction in survival after an 8-day exposure to 2,544 and 1,023 µg/L aluminum, respectively, when normalized. While these studies demonstrated the sensitivity of rainbow trout survival to aluminum, they were excluded from acute criteria calculations because of atypical acute test durations. In contrast, Hunter et al. (1980) observed 40% mortality at pH/total hardness/DOC-normalized concentration of 18,009 µg/L for rainbow trout, suggesting that rainbow trout could possibly be more tolerant to aluminum than reported by the previous studies. However, this study had only one treatment concentration, did not provide information regarding replicates or the number of fish per replicate, and the fish were fed

during the study, precluding it from consideration as a reliable toxicity prediction and for criteria derivation.

Unlike the observed results of the acceptable acute studies, other data for the Family Salmonidae appears to be acutely insensitive to aluminum. In a series of eight 4- and 5-day tests with juvenile Atlantic salmon (*Salmo salar)* conducted at pH 4.42-5.26, Roy and Campbell (1995, 1997) observed pH/total hardness/DOC-normalized LC_{50}s ranging from 2,170-47,329 µg/L. Similarly, Wilkinson et al. (1990) observed a 7-day LC_{50} at pH 4.5 of 88 µg/L (or 13,060 µg/L when normalized to **Appendix A** conditions) for juvenile Atlantic salmon. These studies were not included in the acute criteria calculations because of either a non-standard duration, exposure at pH<5, or both.

Among warm water fishes, goldfish embryos (*Carassius auratus*) were highly sensitive to aluminum, with a 7- to 12-day pH/total hardness/DOC-normalized LC_{50} of 271.1 µg/L (Birge et al. 2000). While this value is below the acute criterion at the same normalized conditions (980 µg/L), the study provided little exposure details and exceeded the duration for an acceptable acute exposure toxicity test, therefore, it is likely overestimating the acute toxicity of aluminum to the species. Fathead minnow (*Pimephales promelas*) sensitivity, however, was variable across studies. In two tests that were excluded because test fish were fed, pH/total hardness/DOC-normalized 96-hr and 8-day LC_{50}s were 19,324 and 12,702 µg/L, respectively (Kimball 1978). In a 96-hour test that was excluded because measured total dissolved aluminum concentrations were greater than reported nominal total aluminum concentrations for all but the highest two treatment concentrations, suggesting total aluminum exposures were greater than reported, the pH/total hardness/DOC-normalized 96-hour LC_{50} was greater than 572.8 µg/L (Palmer et al. 1989). In contrast, Buhl (2002) observed a pH/total hardness/DOC-normalized 96-hr EC_{50} for death and immobility of greater than 21,779 µg/L for this species. Birge et al. (1978) and Birge et al. (2000) found largemouth bass (*Micropterus salmoides*) to be sensitive to aluminum, with 8-day and 7- to 12-day pH/total hardness/DOC-normalized LC_{50}s of 124.6 and 156.1 µg/L, respectively. In contrast, Sanborn (1945) observed no mortality in juvenile *M. salmoides* after a 7-day exposure to a pH/total hardness/DOC-normalized concentration of 45,181 µg/L.

Amphibians appear to be less acutely sensitive to aluminum than fish based on the very limited data available, but their sensitivity is highly variable and appears to depend upon life stage, with embryos being more sensitive than tadpoles. In a series of tests with leopard frogs

(*Rana pipiens*) of different tadpole life stages conducted at low (4.2-4.8) pH and low (2.0 mg/L) total hardness, Freda and McDonald (1990) observed pH/total hardness/DOC-normalized 4 to 5-day LC_{50}s ranging from greater than 57,814 to greater than 490,582 µg/L. In two separate studies conducted at pH 4.5 and low total hardness, the pH/total hardness/DOC-normalized 96-hr LC_{50} for American toad (*Bufo americanus*) tadpoles was 358,450 µg/L (Freda et al. 1990); and the pH/total hardness/DOC-normalized 96-hr LC_{50} for the green tree frog (*Hyla cinerea*) was 200,373 µg/L (Jung and Jagoe 1995). In contrast, when *R. pipiens* embryos were exposed to aluminum for 10-11 days at a higher pH range (7.0-7.8), Birge et al. (2000) observed a normalized LC_{50} of 73.94 µg/L. Birge et al. (2000) also found embryonic spring peepers (*Pseudacris crucifer*) and embryonic Fowler's toads (*Bufo fowleri*) to be highly sensitive to aluminum, with a 7-day normalized LC_{50} of 73.94 and 230.0 µg/L, respectively. These values exceed the typical duration for an acute exposure for the species and therefore overestimate the toxicity of aluminum when comparing them to the acute criterion. However, aluminum sensitivity among amphibian embryos was not always greater than tadpole life stages, as the pH/total hardness/DOC-normalized 96-hr LC_{50} for *R. pipiens* embryos at pH 4.8 was 74,782 µg/L (Freda et al. 1990), similar to the LC_{50}s of *R. pipiens* tadpoles (Freda and McDonald 1990).

5.1.2 Freshwater Chronic Toxicity

Several chronic studies were identified as not meeting quality screening guidelines for inclusion in criterion calculations (**Appendix H** *Other Data on Effects of Aluminum to Freshwater Aquatic Organisms*), but showed similar ranges of toxicity and are presented here to provide additional supporting evidence of the potential toxicity of aluminum to aquatic organisms.

In two unmeasured lifecycle (3-brood) tests, IC_{25}s based on reproduction for *Ceriodaphnia dubia* were 566 and 641 µg/L (pH not reported so values could not be normalized), were within the range of observed acceptable chronic values for this species (Zuiderveen and Birge 1997). In three unmeasured 21-day *Daphnia magna* tests, LC_{50} and reproductive EC_{16} and EC_{50} pH/total hardness/DOC-normalized endpoints were 1,162, 265.6 and 564.3 µg/L, respectively (Biesinger and Christensen 1972). These values are within the range of acceptable chronic data reported for the cladoceran *C. dubia* (**Appendix C**).

Among fish species, the pH/total hardness/DOC-normalized 28-day EC_{50} (death and deformity) for *O. mykiss* of 457.4 µg/L (Birge 1978; Birge et al. 1978) was similar to chronic

values for acceptable tests with other cold water test species. In addition, the 16-day normalized LC_{50}s for rainbow trout at two different test total hardness levels (20.3 and 103 mg/L as $CaCO_3$) observed by Gundersen et al. (1994) were 485.2 and 1,084 µg/L, respectively. However, the 16-day exposures were about one-fourth the duration of an acceptable ELS test for a salmonid (ASTM 2013). In a 28-day test of *S. fontinalis* conducted at pH 4.4, the pH/total hardness/DOC-normalized MATC for survival was 2,523 µg/L (Ingersoll et al. 1990a). Even though the duration of this test was insufficient and the pH was below 5, it provides additional evidence of the sensitivity of brook trout, a commercially and recreationally important species. Several short-term (7-day) chronic tests conducted by Oregon State University (OSU 2012a) with the fathead minnow at pH 6 and across a range of total hardness and DOC concentrations revealed that both an increase in total hardness and DOC reduced the toxicity of aluminum (non-normalized EC_{20}s ranged from 127.2 to 2,938 µg/L or 1,718 to 7,220 µg/L when normalized to the test conditions in **Appendix C**).

5.1.3 Freshwater Field Studies

Field studies have been conducted to measure effects of aluminum additions to control phosphorus concentrations in lakes, to validate parallel laboratory exposures, and to investigate the effects of acid deposition in aquatic systems. Aluminum sulfate was continuously added for 35 days to the Cuyahoga River 500 meters upstream of Lake Rockwell to control phosphorus concentrations in the reservoir. Artificial colonization substrata were placed at five locations along the treatment reach five weeks before the release, sampled on the day of the release, redeployed after collecting invertebrates immediately before the release, and then sampled weekly throughout the 35-day aluminum addition. After one week of treatment, invertebrate densities declined throughout the study reach, and were completely absent from a site 60 meters downstream of the release point. Once treatment was stopped, invertebrate densities recovered and replaced after approximately three weeks by rapidly colonizing oligochaete taxa (Barbiero et al. 1988).

In Little Rock Lake, WI, sulfuric acid was added to half of the lake between 1984-1990, resulting in a decrease in pH from 6.05 to 4.75 and an increase in aqueous aluminum from 7 to 42 µg/L. The other half of the lake served as a control, where aluminum increased from 7 to 14 µg/L and pH decreased from 6.04 to 5.99 during the same time period (Eaton et al. 1992). In parallel laboratory experiments in 1988, eggs of several fish species were exposed to aluminum

concentrations ranging from 8.1-86.9 µg/L and pH values ranging from 4.5-5.5 until seven days' post hatch. In both the acidified portion of the lake and in laboratory exposures at comparable aluminum and pH levels, mortality was higher than in controls (Eaton et al. 1992). However, mortality of control fish in both the *in-situ* and laboratory exposures exceeded the minimum 80 percent survival acceptable guideline for tests of this duration.

Additional field studies have evaluated the effects of aluminum and acidification on different trophic level communities. Havens and Decosta (1987) acidified the circumneutral Lake O'Woods (WV) to pH 4.8 and compared phytoplankton and zooplankton assemblages with and without the addition of 300 µg/L aluminum. They observed similar species in all conditions, but the aluminum dosed water exhibited a decrease in chlorophyll *a* concentrations and a drop in zooplankton abundances over the 49-day observation period, while the acidified condition without aluminum addition only exhibited a drop in chlorophyll *a*. The algal biomass decrease was attributed to the initial co-precipitation of phosphorus and/or algal cells with the aluminum hydroxide at circumneutral pH. Bukaveckas (1989) reported similar declines in algal biomass when acidic, aluminum-rich waters are neutralized with lime. In contrast, aluminum addition produced a more pronounced difference in algal community structure and succession when Havens and Heath (1990) gradually acidified (pH 4.5) and dosed East Twin Lake (OH) with 200 µg/L aluminum.

Increased drift of invertebrates (Ephemeroptera, Diptera and Orthocladiinae chironomids) in an acidified (pH~5) stream dosed with 280 µg/L aluminum was observed relative to a non-dosed stream at the same ~5 pH level (Hall et al. 1987). Ormerod et al. (1987), however, found little added effect of 350 µg/L aluminum on stream invertebrates compared with the effects of acidification alone (pH~4.3). In contrast, brown trout and Atlantic salmon showed significantly increased mortalities in the acidified aluminum condition (50 to 87%) relative to the acid-only treatment (7 to 10%). Baldigo and Murdoch (1997) deployed caged brook trout in selected New York Catskill Mountain streams where the pH, aluminum concentration and other stream conditions fluctuated naturally over time. They noted that fish mortality correlated best with high inorganic aluminum concentrations and low water pH (4.4-5.2), with 20 percent mortality observed for brook trout exposed to greater than or equal to 225 µg/L inorganic monomeric aluminum for two days. They also observed, based on regression analysis, that a vast majority (74-99%) of the variability in mortality could be explained by either the mean or median

inorganic monomeric aluminum concentration, and that the mortality was highly related to inorganic monomeric aluminum, pH, dissolved organic carbon, calcium and chloride concentrations. Bulger et al. (1993) also reported that water pH and monomeric inorganic aluminum concentrations best predicted brown trout populations of 584 Norwegian lakes. Lakes with 133 µg/L aluminum and a pH of 4.8 were devoid of brown trout (39% of the 584 lakes), whereas lakes with 11 µg/L aluminum and a pH of 6.0 had healthy brown trout populations.

5.1.4 Estuarine/Marine Acute Toxicity

SMAVs for five genera representing five species of estuarine/marine organisms were calculated for aluminum (**Table 8**). SMAVs and GMAVs were equal since there is only one species present per genus. The most sensitive genus was the polychaete worm (*Ctenodrilus serratus*), with a SMAV of 97.15 µg/L, followed by two other polychaete worms (*Capitella capitata* and *Neanthes arenaceodentata*) with SMAVs of 404.8 and greater than 404.8 µg/L, respectively. The most tolerant genus was a copepod (*Nitokra spinipes*) with a SMAV of 10,000 µg/L (**Figure 9**). However, the freshwater acute criterion (980 µg/L total aluminum) is much higher than the most sensitive acute estuarine/marine species LC_{50} (97.15 µg/L total aluminum). Thus, at least some invertebrate estuarine/marine species would not be protected if the freshwater acute aluminum criterion was applied in those systems.

Table 8. Ranked Estuarine/Marine Genus Mean Acute Values.

Rank[a]	GMAV (µg/L total Al)	Species	SMAV (µg/L total Al)[b]
5	10,000	Copepod, *Nitokra spinipes*	10,000
4	>1,518	American oyster, *Crassostrea virginica*	>1,518
3	>404.8	Polychaete worm, *Neanthes arenaceodentata*	>404.8
2	404.8	Polychaete worm, *Capitella capitata*	404.8
1	97.15	Polychaete worm, *Ctenodrilus serratus*	97.15

[a] Ranked from the most resistant to the most sensitive based on Genus Mean Acute Value.
[b] From Appendix B: Acceptable Acute Toxicity Data of Aluminum to Estuarine/Marine Aquatic Animals.

In contrast to freshwater, only a few acute studies were identified as not meeting screening guidelines for inclusion in criterion calculations (**Appendix I** *Other Data on Effects of Aluminum to Estuarine/Marine Aquatic Organisms*), but showed similar ranges of toxicity. As

with other non-conforming studies previously described, the results are presented here to provide additional supporting evidence of the potential toxicity of aluminum to estuarine/marine organisms. In one of these studies, a cohort of sea urchin embryos (*Paracentrotus lividus*) exposed to 539.6 µg/L aluminum for 72-hr exhibited increased developmental defects by 69.7% (Caplat et al. 2010). Although this study was not considered acceptable because the control group exhibited 19.3% defects indicative of some health deficiency, the effect level was comparable to the acute effect levels observed in **Appendix B**. In 24-hr exposures to aluminum added as potassium aluminum sulfate, LC_{50}s for the crab species *Eupagurus bernhardus* and *Carcinus maenas*, the snail *Littorina littorea*, and the mussel *Mytilus edulis* were extremely high, ranging from a low of 250,000 µg/L for *E. bernhardus* to greater than 6,400,000 µg/L for the two mollusk species (Robinson and Perkins 1977). Although these studies were unacceptable because of the atypical acute test duration, they suggest that some saltwater taxa are highly tolerant to acute aluminum exposure.

5.1.5 Estuarine/Marine Chronic Toxicity

There are no acceptable saltwater chronic data available for aluminum (**Appendix D**). However, the EPA identified several chronic studies that did not meet screening guidelines for inclusion in criterion calculations, but provided supporting evidence of potential chronic toxicity of aluminum to aquatic organisms in estuarine/marine environments (**Appendix I** *Other Data on Effects of Aluminum to Estuarine/Marine Aquatic Organisms*). Petrich and Reish (1979) observed a 21-day MATC for reproduction in the polychaete *C. serratus* of 28.28 µg/L. Consistent with acceptable acute test results for this species, this chronic test suggests that polychaetes may be chronically sensitive to aluminum. This study was excluded because of the test duration. In a "semi-chronic" 12-day study of the effects of aluminum on daggerblade grass shrimp (*Palaemonetes pugio*) embryos, the LC_{50} was 1,079 µg/L (Rayburn and Aladdin 2003). This study was not included because it was longer than an acceptable 48-hr acute test, and it was not a full life cycle test.

5.1.6 Bioaccumulation

Three acceptable studies examined the effects of waterborne aluminum bioaccumulation in aquatic organisms (**Appendix G** *Acceptable Bioaccumulation Data of Aluminum by Aquatic Organisms*). Cleveland et al. (1991a) exposed 30-day old brook trout to 200 µg/L of aluminum in test waters at three pH levels (5.3, 6.1, and 7.2) for 56 days. After 56 days, trout were

transferred to water of the same pH with no aluminum amendments and held for 28 days. Fish were sampled for whole body aluminum on days 3, 7, 14, 28 and 56 of the exposure; and on days 3, 7, 14 and 28 of the depuration period. The estimated time to achieve steady state whole body aluminum concentrations was 1.5 days at pH 5.3, 4.2 days at pH 6.1, and 1.7 days at pH 7.2. Bioconcentration factors (BCF) were inversely related to pH: 142 at pH 5.3, 104 at pH 6.1, and 14.2 at pH 7.2. Mortality was also highest at pH 5.3 and lowest at pH 7.2. In a separate study, Buckler et al. (1995) continuously exposed Atlantic salmon beginning as eyed eggs to four aluminum treatment levels (33, 71, 124, 264 µg/L) at pH 5.5 for 60 days after the median hatch date. Fish were sampled for whole body aluminum after 15, 30, 45, and 60 days post median hatch. After 60 days, average mortality was 15% in the 124 µg/L treatment and 63% in the 264 µg/L treatment. The mortality NOEC and LOEC were 71 and 124 µg/L, respectively. BCFs were directly related to exposure concentration, and were 76, 154, and 190 at treatment levels 33, 71, and 124 µg/L, respectively. A BCF could not be calculated for the 264 µg/L treatment level because there were insufficient surviving fish to analyze. Snails, *Lymnaea stagnalis*, held in neutral pH for 30 days and 242 µg/L total aluminum reached steady with a reported BCF of 4.26 in the digestive gland (Dobranskyte et al. 2004).

As reported in the literature, aquatic organisms can accumulate metals from both aqueous and dietary exposure routes. The relative importance of each, however, is dependent upon the chemical. Aluminum adsorbs rapidly to gill surface from the surrounding water, but cellular uptake from the water is slow, with gradual accumulation by the internal organs over time (Dussault et al. 2001). Bioaccumulation and toxicity via the diet are considered highly unlikely based on studies by Handy (1993) and Poston (1991), and also supported by the lack of any biomagnification within freshwater invertebrates that are likely to be prey of fish in acidic, aluminum-rich rivers (Herrmann and Frick 1995; Otto and Svensson 1983; Wren and Stephenson 1991). The opposite phenomena, trophic dilution up the food chain, has been suggested based on the lowest aluminum accumulation exhibited by fish predators (perch) and highest by the phytoplankton that their zooplankton prey were consuming (King et al. 1992). Thus, the low aluminum BCFs reported in the literature are supported by the slow waterborne uptake and the lack of dietary accumulation.

5.2 Effects on Aquatic Plants

Aquatic plant data are not used to derive the criteria for aluminum. However, a summary of available data is presented below. For freshwater algae, aluminum effect concentrations ranged from 50 µg/L to 6,477 µg/L, with most effect levels below 1,000 µg/L (**Appendix E** *Acceptable Toxicity Data of Aluminum to Freshwater Aquatic Plants*). Studies for freshwater macrophytes are limited, but available data suggest freshwater macrophytes are more tolerant to aluminum than freshwater algae. The effect concentration for Eurasian watermilfoil is 2,500 µg/L based on root weight (Stanley 1974), which is near the upper range of freshwater algae sensitivities. Several 3-day tests with the green alga *Pseudokirchneriella subcapitata* at pH 6, 7 and 8 across a range of total hardness and DOC concentrations revealed that both an increase in pH, total hardness and DOC reduced the toxicity of aluminum (European Aluminum Association 2009). DeForest et al. (2018a) used these 27 toxicity tests (as summarized in Gensemer et al. 2018) to develop a MLR model to explain the effects of water chemistry on algal toxicity. The MLR model developed was:

$$P.\,subcapitata\ EC_{20} = e^{\left[-61.952+[1.678\times\ln(DOC)]+[4.007\times\ln(hard)]+(17.019\times pH)-(1.020\times pH^2)-[0.204\times pH{:}\ln(DOC)]-[0.556\times pH{:}\ln(hard)]\right]}$$

The MLR model for *P. subcapitata* was within a factor of two for 100% of the predicted versus observed values (DeForest et al. 2018a). Most of the acceptable toxicity data for freshwater aquatic plants (**Appendix E**) did not report all three water quality parameters (i.e., pH, total hardness and DOC) preventing the use of applying the alga based MLR equation to the data. The EPA contacted authors and in limited cases, the authors were able to provide rough estimates of some of the missing information. Normalized lowest observed effect concentrations (LOECs) for the twenty-one day tests as reported by Pilsbury and Kingston (1990) were 3,482 µg/L, while normalized 4-day EC_{50}s for *P. subcapitata* were 620 and 1,067 µg/L (Call et al. 1984). These values are above the chronic criterion at the same test conditions, suggesting that the criteria developed using aquatic animals will also be protective of aquatic plants. This was also observed when normalizing the 3-day *P. subcapitata* test in **Appendix H** (*Other Data on Effects of Aluminum to Freshwater Aquatic Organisms*) with normalized effect concentrations ranging from 161 to 5,113 µg/L. The geometric mean of these values was 1,653 µg/L (Note:

these tests were excluded from the acceptable table due to the insufficient test duration, less than 4 days).

In contrast to other freshwater plants, duckweed is highly tolerant to aluminum, with an effect concentration based on reduced growth of greater than 45,700 µg/L (Call et al. 1984). For the one acceptable study of a saltwater plant (Seagrass, *Halophila stipulacea*), less than 50% mortality of teeth cells was observed at 26.98 µg/L, and more than 50% mortality of teeth cells observed at 269.8 µg/L (Malea and Haritonidis 1996). In a shorter duration study, the saltwater algal species, *Dunaliella tertiolecta*, also exhibited sensitivity to aluminum, but the effect concentration was higher at 18,160 µg/L (**Appendix I** *Other Data on Effects of Aluminum to Estuarine/Marine Aquatic Organisms*). Although aquatic plant data are not normalized using the alga based MLR equation, the effect levels observed are similar to the available animal data, and the recommended criteria should therefore be protective for algae and aquatic plants.

5.3 Identification of Data Gaps and Uncertainties for Aquatic Organisms

Data gaps and uncertainty were identified for the aluminum criteria. A number of uncertainties are associated with calculation of the freshwater Final Acute Value (FAV) as recommended by the 1985 Guidelines, and include use of limited data for a species or genus, acceptability of widely variable data for a genus, application of adjustment factors, extrapolation of laboratory data to field situations, and data normalization with a MLR model.

5.3.1 Acute Criteria

There are a number of cases in the acute database where only one acute test is used to determine the Species Mean Acute Value (SMAV) and subsequently the Genus Mean Acute Value (GMAV) is based on the one acute test. In this situation, there is a level of uncertainty associated with the GMAV based on the one test result since it does not incorporate the range of values that would be available if multiple studies were available. Such a GMAV is still valid, however, in spite of the absence of these additional data because it represents the best available data and to exclude this data would create an unnecessary data gap. Additionally, many of the acute studies did not report a definitive LC_{50} (i.e., yielded greater than values) because the highest concentration used did not cause more than 50% mortality. This adds more uncertainty since the true LC_{50} is unknown.

The acute criterion is set as equal to half of the FAV to represent a low level of effect for the fifth percentile genus, rather than a 50% effect. This adjustment factor was derived from an

analysis of 219 acute toxicity tests with a variety of chemicals (see 43 FR 21506-21518 for a complete description) where mortality data were used to determine the highest tested concentration that did not cause mortality greater than that observed in the control (or between 0 and 10%). Application of this adjustment factor is justified because that concentration represents minimal acute toxicity to the species.

5.3.2 Chronic Criteria

The freshwater FCV calculation is also influenced by the limited availability of data and the use of qualitative data to fulfill the one remaining family (Chordata) MDR. The aluminum freshwater chronic database is comprised of 12 species and subsequently 12 genera that provide seven of the eight MDR families as recommended in the 1985 Guidelines. In order to satisfy the eight-family requirement, the dataset included a wood frog (*Rana sylvatica*) chronic study that was relegated to **Appendix H** due to minor methodology issues (pH<5). While this study does not quantitatively affect the criterion value, it was used to fulfill the MDRs per the 1985 Guidelines, thereby allowing direct calculation of the FCV (see **Section 2.7.3**). Additional testing of other species and families in the Phylum Chordata would reduce the uncertainty in the FCV.

5.3.3 Laboratory to Field Exposures

Application of water-only laboratory toxicity tests to develop water quality criteria to protect aquatic species is a basic premise of the 1985 Guidelines, supported by the requirements of a diverse assemblage of eight families and the intended protection goal of 95 percent of all genera. Confirmation has been reported by a number of researchers (Clements and Kiffney 1996; Clements et al. 2002; Mebane 2006; Norberg-King and Mount 1986), thereby indicating that on the whole, extrapolation from the laboratory to the field is a scientifically valid and protective approach for aquatic life criteria development.

The unique chemistry of aluminum (speciation changes and the transient precipitates formed during toxicity testing) and difference between geological aluminum materials suspended in natural water are additional areas of uncertainty (Angel et al. 2016; Cardwell et al. 2018; Gensemer et al. 2018). The use of total aluminum concentrations is justified for laboratory toxicity test data (see **Section 2.6.2**); where the total aluminum concentration is in either a dissolved or precipitated form (Santore et al. 2018). However, natural water samples may also contain other species of aluminum that are not biologically available (i.e., suspended particles, clays and aluminosilicate minerals) (Santore et al. 2018; Wilson 2012). This creates uncertainty

because the total recoverable aluminum concentrations measured in natural waters may overestimate the potential risks of toxicity to aquatic organisms.

EPA Methods 200.7 and 200.8 are the only currently approved methods for measuring aluminum in natural waters and wastes for NPDES permits (U.S. EPA 1994a,b). Research on new analytical methods is ongoing to address concerns with including aluminum bound to particulate matter (i.e., clay) in the total recoverable aluminum concentrations (OSU 2018c). One approach would not acidify the sample to pH less than 2 but rather to pH 4 (pH 4 extracted method) to better capture the bioavailable fraction of aluminum (CIMM 2016, OSU 2018c). Thus, this draft pH 4 extracted method under development is expected to reduce the uncertainty regarding bioavailable aluminum measurements in the aquatic environment.

5.3.4 Lack of Toxicity Data for Estuarine/Marine Species and Plants

Since limited acceptable acute and chronic data are available for estuarine/marine species, the EPA could not derive estuarine/marine acute and chronic aluminum criteria at this time. In addition, very few acceptable aquatic vascular plant studies are available.

5.3.5 Bioavailability Models

Aluminum toxicity is strongly affected by water chemistry, through its effects on bioavailability. The understanding of the interactions between aluminum species, water characteristics, and aquatic toxicity data has led to the development of several bioavailability models. There are currently two different approaches that take into account aluminum bioavailability in relation to aquatic toxicity that are considered applicable to the development of aquatic life criteria: empirical models that relate toxicity to water chemistry; and Biotic Ligand Models that encompass both abiotic and biotic mechanistic factors determining toxicity.

Initially in considering the array of approaches for criteria development, the EPA considered using an empirical total hardness adjustment equation for criteria development. However, studies that tested aluminum at pH 6 for a variety of organisms (OSU 2012a, 2012b, 2012c, 2012d, 2012e, 2012f, 2012g, 2012h, 2013) indicated additional water chemistry parameters affected bioavailability, and hence aquatic effects of aluminum. In addition, new data are available that supported the development of MLR models that incorporate pH and total hardness. Also, a mechanistic BLM model for aluminum was recently developed (Santore et al. 2018). Finally, an approach described in DeForest et al. (2018a,b) incorporated pH, total hardness and DOC into empirical MLR models to determine if the estimation of aluminum

bioavailability to animals in freshwater aquatic systems could be applicable in the development of aluminum water quality criteria. The approach resulted in the creation of multiple MLR models that could be used for the development of aluminum water quality criteria methodologies. Both MLR models and the BLM model include the same toxicity test data, with the BLM including additional data on the accumulation of aluminum on the gills of Atlantic salmon (Santore et al 2018). The MLR approach empirically curve-fits log-log pH, total hardness and DOC relationships (with interaction terms) to the empirical data. The BLM uses a mechanistic model based on an underlying theory of how water chemistry input parameters affect aluminum toxicity, although it still has empirically derived factors.

An external peer review of the different aluminum aquatic life criteria approaches was conducted in November 2016 to provide a comparison of the several available approaches to generating aluminum criteria that reflect water quality condition impacts on toxicity. Approaches compared included a 10-parameter BLM, a simplified-BLM approach (e.g., pH, total hardness, dissolved organic carbon, temperature), and MLR models to facilitate evaluation of the most appropriate approaches to consider for aluminum toxicity modeling. The EPA conducted three additional external peer reviews in 2018 regarding the new toxicity data and re-fitted MLR models on: 1) the new invertebrate toxicity tests on *C. dubia* (OSU 2018a); 2) the new fish toxicity tests on *P. promelas* (OSU 2012b); and 3) the new individual and pooled MLRs developed by Deforest et al. (2018b). Based on external peer review comments, ease of use, and transparency, the EPA applied the DeForest et al. (2018b) individual species MLR model to normalize the freshwater acute and chronic data (**Appendix A** and **Appendix C**) and derived the aluminum criteria using the criteria development approaches described in the 1985 Guidelines. The EPA independently examined and verified the quality and fit of the DeForest et al. (2018a,b) MLR models before applying them in this final criteria document.

5.3.6 pH, Total Hardness and DOC MLR Models

There are additional uncertainties, beyond those described above, associated with the normalization of aluminum toxicity data using the MLR models developed by DeForest et al. (2018b). The models were developed with chronic toxicity data from two animal species, one invertebrate (*C. dubia*; a sensitive species) and one fish (fathead minnow; a moderately sensitive species). Incorporating additional species in the model development would improve the representativeness of all species and further validate the MLR model use across species. Though

the pH, total hardness, and DOC do explain the majority of differences seen in the toxicity data between the two species, there are two MLR models developed (invertebrate *C. dubia* model and vertebrate *P. promelas* model), which better delineate the differences in their uptake of aluminum. Because the arthropod phylum is highly diverse, there is some uncertainty in the application of the *C. dubia* model across other invertebrate taxa. However, among fish (and amphibians), the MLR approach that uses a model optimized solely for those taxa is the best model to use as opposed to a BLM which uses one model to normalize the data for multiple taxa for criteria calculations. Thus, the MLR-based criteria derivation specific to the most sensitive taxa may address additional uncertainty because some of the model differences may be a function of the species physiology in addition to bioavailability, and hence the MLR approach may better capture taxa physiologic differences in sensitivity across different water chemistry conditions. The models are, however, applied across gross taxonomy (vertebrate vs. invertebrate), creating some additional uncertainty. Finally, only chronic data were used in model development, and application to acute toxicity data assumes that the same relationships are present. All of these uncertainties associated with the model are areas where additional research would be helpful.

The models were developed using data that encompass a pH range of 6.0-8.7, DOC range of 0.08-12.3 mg/L and total hardness range of 9.8-428 mg/L (as $CaCO_3$). The authors (DeForest et al. 2018a) noted that the empirical data evaluated support a reduced total hardness effect at higher pH levels (i.e., 8-9), but limited data are available. Additional chronic aluminum toxicity testing at higher pH levels would be useful for further validating the MLR models (i.e., there is a limited amount of data at pH>8). When any of the water quality parameters selected is outside model inputs, the Aluminum Criteria Calculator V.2.0 (*Aluminum Criteria Calculator V.2.0.xlsm*) flags these values and defaults to the maximum bounds for DOC and total hardness. Values generated outside the recommended water quality parameter for pH (6.0-8.2) should be treated with caution because extrapolating beyond the conditions used for model development is highly uncertain. Of particular concern is the quadratic term (pH^2) in the *C. dubia* MLR model which can compound issues with extrapolating. Additional toxicity tests conducted and pH<6.0 and pH> 8.5 would further define behavior of this model.

5.4 Protection of Endangered Species

Although the dataset for aluminum is not extensive, it does include some data representing species that are listed as threatened or endangered by the U.S. Fish and Wildlife Service and/or National Oceanic and Atmospheric Administration (NOAA) Fisheries. Summaries are provided here describing the available aluminum toxicity data for listed species indicating that the 2018 aluminum criteria are expected to be protective of these listed species, based on available scientific data.

5.4.1 Key Acute Toxicity Data for Listed Fish Species

Tests relating to effects of aluminum on several threatened and endangered freshwater fish species are available (certain populations threatened, and others endangered): rainbow trout, *Oncorhynchus mykiss* with a normalized SMAV of 3,312 µg/L (Call et al. 1984; Gundersen et al. 1994; Holtze 1983); Rio Grande silvery minnow, *Hybognathus amarus* with a normalized SMAV of greater than 21,779 µg/L (Buhl 2002); and Atlantic salmon, *Salmo salar* with a SMAV of 8,642 µg/L (Hamilton and Haines 1995). For this comparison, all SMAVs are normalized to a pH 7, a total hardness of 100 mg/L as $CaCO_3$ and a DOC of 1.0 mg/L. All of the normalized SMAVs are above the recommended acute criterion (CMC) of 980 µg/L at the same pH, total hardness and DOC levels. There are no acceptable acute toxicity data for endangered or threatened estuarine/marine aquatic fish species.

5.4.2 Key Chronic Toxicity Data for Listed Fish Species

While there are no acceptable chronic toxicity data for estuarine/marine endangered and/or threatened fish species, there is one acceptable early life-stage test conducted with the endangered freshwater fish, Atlantic salmon, *Salmo salar*. The test, conducted at a pH of 5.7, yielded a pH/total hardness/DOC-normalized species mean chronic value (SMCV) of 434.4 µg/L (McKee et al. 1989). This value is greater than the recommended chronic criterion of 380 µg/L at the same total hardness, DOC and pH.

5.4.3 Concerns about Federally Listed Endangered Mussels

Some researchers have expressed concerns that mussels may be more sensitive to the effects of aluminum than other organisms. A study by Kadar et al. (2001) indicated that adult *Anodonta cygnea* mussels may be sensitive to aluminum at concentrations above 250 µg/L, with reductions in mean duration of shell opening of 50% at 500 µg/L aluminum in the water column

(at circumneutral pH) when compared to paired controls. This suggests that chronic elevated aluminum concentrations could lead to feeding for shorter durations with potential implications for survival and growth, and possibly even reproduction. Pynnonen (1990) conducted toxicity tests with two freshwater mussels in the Unionidae family (*Anodonta anatina* and *Unio pictorum*). In both species, pH had a significant effect on accumulation of aluminum in the gills. The *Anodonta* mussel species in the two studies described above are not native to the United States and are included in **Appendix J** (*List of Aluminum Studies Not Used in Document Along with Reasons*). While the *Anodonta* mussel species in these two studies are not native, there are species of the *Anodonta* genus present in the United States. Simon (2005) provides an additional line of evidence that indicates mussels may be more sensitive to the effects of aluminum than other organisms. In a 21-day chronic aluminum toxicity test conducted at circumneutral pH with juvenile unionid freshwater mussel *Villosa iris*, growth was significantly reduced at aluminum levels above 337 µg/L.

New data are available for this update on aluminum toxicity to the fatmucket mussel (*Lampsilis siliquoidea*), another freshwater mussel in the family Unionidae. While the 96-hr LC_{50} juvenile test failed to elicit an acute 50% response at the highest concentration tested (6,302 µg/L total aluminum, or 29,492 µg/L when normalized), the 28-day biomass-normalized SMCV ranked as the fourth most sensitive genus in the dataset. The SMCV is greater than the most sensitive species, Atlantic salmon, and the freshwater criterion value. Thus, the chronic criterion is expected to be protective of this and related species. The fatmucket mussel tested is not a threatened and/or endangered species, but the genus *Lampsilis* contains several listed species with a wide distribution across the United States, and is also member of the family Unionidae.

Freshwater mussels in the family Unionidae are known to be sensitive to a number of chemicals, including metals and organic compounds (Wang et al 2018; U.S. EPA 2013). The EPA's 2013 Aquatic Life Ambient Water Quality Criteria for Ammonia in Freshwater indicates many states in the continental U.S. have freshwater unionid mussel fauna in at least some of their waters (Abell et al. 2000; Williams and Neves 1995; Williams et al. 1993). Roughly one-quarter of the approximately 300 freshwater unionid mussel taxa in the United States are Federally-listed as endangered or threatened species. Additional testing on endangered mussel species, or closely related surrogates, would be useful to further examine the potential risk of aluminum exposures to endangered freshwater mussels.

5.5 Comparison of 1988 and 2018 Criteria Values

The 1988 aluminum freshwater acute criterion was based on data from eight species of invertebrates and seven species of fish grouped into 14 genera. This 2018 update now includes 13 species of invertebrates, eight species of fish, and one frog species for a total of 22 species grouped into 20 genera. The data in the previous AWQC were not normalized to any water chemistry conditions making it difficult to compare the magnitude of the two criteria.

The 1988 aluminum freshwater chronic criterion was set at 87 µg/L across a pH range 6.5 to 9.0, and across all total hardness and DOC ranges, based on a dataset that included two species of invertebrates and one fish species. This 2018 criteria update includes new data for an additional nine species, and consists of eight invertebrates and four fish species grouped into 12 genera and is a function of pH, total hardness and DOC. Addition of the frog (*Rana sylvatica*) data from **Appendix H** satisfied the MDR for the one missing family (Chordata), thereby allowing for direct calculation of the FCV.

Like the previous AWQC for aluminum, there are still insufficient data to fulfill the estuarine/marine MDRs as per the 1985 Guidelines, therefore the EPA did not derive estuarine/marine criteria at this time. New toxicity data for five genera representing five species of estuarine/marine organisms are presented in this update; no data were available in 1988.

Table 9. Comparison of the 2018 Recommended Aluminum Aquatic Life AWQC and the 1988 Criteria.

Version	Freshwater Acute[a] (1-hour, total aluminum)	Freshwater Chronic[a] (4-day, total aluminum)
2018 AWQC (vary as a function of a site's pH, DOC and total hardness)	1-4,800 µg/L	0.63-3,200 µg/L
1988 AWQC (pH 6.5 – 9.0, across all total hardness and DOC ranges)	750 µg/L	87 µg/L

[a] Values are recommended not to be exceeded more than once every three years on average.
Note: 2018 Criteria values will be different under differing water chemistry conditions as identified in this document, and can be calculated using the Aluminum Criteria Calculator V.2.0 (*Aluminum Criteria Calculator V.2.0.xlsm*) or found in the tables in Appendix K. See Appendix K for specific comparisons of 1988 and 2018 criteria values across water chemistry parameter ranges.

6 UNUSED DATA

For this 2018 criteria update document, the EPA considered and evaluated all available data that could be used to derive the new acute and chronic criteria for aluminum in fresh and

estuarine/marine waters. A substantial amount of those data were associated with studies that did not meet the basic QA/QC requirements in a manner consistent with the 1985 Guidelines (see Stephan et al. 1985) and reflecting best professional judgments of toxicological effects. A list of all other studies considered, but removed from consideration for use in deriving the criteria, is provided in **Appendix J** (*List of Aluminum Studies Not Used in Document Along with Reasons*) with rationale indicating the reason(s) for exclusion. Note that unused studies from previous AWQC documents were not re-evaluated.

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Appendix A ACCEPTABLE ACUTE TOXICITY DATA OF ALUMINUM TO FRESHWATER AQUATIC ANIMALS

Appendix A. Acceptable Acute Toxicity Data of Aluminum to Freshwater Aquatic Animals
(Bold values are used in SMAV calculation).
(Species are organized phylogenetically).

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Worm (adult, 1.0 cm), *Nais elinguis*	R, M, T	Aluminum sulfate	17.89 (±1.74)	6.51 (±0.01)	3.2	3,874	**9,224**	9,224	Shuhaimi-Othman et al. 2012a, 2013
Snail (adult), *Physa sp.*	S, M, T	Aluminum chloride	47.4 (±4.51)	6.59	1.1[d]	>23,400	**>52,593**	-	Call 1984; Call et al. 1984
Snail (adult), *Physa sp.*	S, M, T	Aluminum chloride	47.4 (±4.51)	7.55	1.1[d]	30,600	**27,057**	-	Call 1984; Call et al. 1984
Snail (adult), *Physa sp.*	S, M, T	Aluminum chloride	47.4 (±4.51)	8.17	1.1[d]	>24,700	>19,341[c]	-	Call 1984; Call et al. 1984
Snail (adult), *Physa sp.*	S, M, T	Aluminum chloride	47.4 (±4.51)	7.46	1.1[d] (aged solution)	55,500	**51,539**	41,858	Call 1984; Call et al. 1984
Snail (adult, 1.5-2.0 cm, 22.5 mg), *Melanoides tuberculata*	R, M, T	Aluminum sulfate	18.72 (±1.72)	6.68 (±0.22)	3.2	68,230	**119,427**	119,427	Shuhaimi-Othman et al. 2012b, 2013
Fatmucket (juvenile, 6 d), *Lampsilis siliquoidea*	R, M, T	Aluminum chloride	107 (±6.3)	8.19 (±0.22)	0.5	>54,300	>57,735[f]	-	Ivey et al. 2014
Fatmucket (juvenile, 7-8 d, 0.38 mm), *Lampsilis siliquoidea*	F, M, T	Aluminum nitrate	106 (104-108)	6.12 (6.10-6.13)	0.48	>6,302	**>29,492**	>29,492	Wang et al. 2016, 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, A	Aluminum chloride	50.0	7.42 (±0.02)	1.1[d]	1,900	**1,771**	-	McCauley et al. 1986
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, A	Aluminum chloride	50.5	7.86 (±0.04)	1.1[d]	1,500	**1,170**	-	McCauley et al. 1986
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, A	Aluminum chloride	50.0	8.13 (±0.03)	1.1[d]	2,560	**1,974**	-	McCauley et al. 1986

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Cladoceran (<24 hr), *Ceriodaphnia dubia*	R, M, T	Aluminum chloride	25 (24-26)	7.5 (7.0-8.0)	0.5[d]	720	**1,321**	-	ENSR 1992d
Cladoceran (<24 hr), *Ceriodaphnia dubia*	R, M, T	Aluminum chloride	49 (46-52)	7.65 (7.3-8.0)	0.5[d]	1,880	**2,516**	-	ENSR 1992d
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum chloride	95 (94-96)	7.9 (7.7-8.1)	0.5[d]	2,450	**2,559**	-	ENSR 1992d
Cladoceran (<24 hr), *Ceriodaphnia dubia*	R, M, T	Aluminum chloride	193 (192-194)	8.05 (7.8-8.3)	0.5[d]	>99,600	**>88,933**	-	ENSR 1992d
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, NR	Aluminum sulfate	90 (80-100)	7.15 (7.0-7.3)	0.5[d]	3,727	**5,243**	-	Fort and Stover 1995
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, NR	Aluminum sulfate	90 (80-100)	7.15 (7.0-7.3)	0.5[d]	5,673	**7,981**	-	Fort and Stover 1995
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, NR	Aluminum sulfate	89	8.2	0.5[d]	2,880	**3,189**	-	Soucek et al. 2001
Cladoceran (<24 hr), *Ceriodaphnia dubia*	R, U, T	Aluminum chloride	142 (±2)	8.2 (±1)	1.6[d]	153,440	**77,169**		Griffitt et al. 2008
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.01 (5.99-6.03)	0.5[d]	71.12	**2,009**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.05 (6.02-6.07)	2	686.5	**7,721**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.09 (6.03-6.15)	4	1,558.1	**10,568**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.01 (5.95-6.06)	0.5[d] (solution aged 3 hrs)	68.1	**1,924**	-	European Al Association 2009; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.03 (5.95-6.10)	0.5[d] (solution aged 27 hrs)	163.0	**4,394**	-	European Al Association 2009; Gensemer et al. 2018

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	5.97 (5.92-6.01)	0.5[d] (solution aged 51 hrs)	178.5	**5,546**	-	European Al Association 2009; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	5.92 (5.87-5.96)	0.5[d] (solution aged 99 hrs)	141.0	**4,945**	-	European Al Association 2009; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.99 (6.96-7.01)	0.5[d]	>1,300	**>5,842**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	7.85 (7.77-7.93)	0.5[d]	>5,000	**>9,735**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.80 (6.55-7.04)	2	>10,000	**>26,061**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	7.82 (7.49-8.14)	2	>15,000	**>12,984**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	6.77 (6.51-7.03)	4	>10,000	**>18,075**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	7.66 (7.39-7.93)	4	>15,000	**>9,538**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	7.91 (7.82-7.99)	0.5[d] (solution aged 3 hrs)	>2,000	**>3,793**	-	European Al Association 2009; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	10.6	7.89 (7.83-7.95)	0.5[d] (solution aged 27 hrs)	>2,000	**>3,812**	-	European Al Association 2009; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	6.04 (6.02-6.05)	0.5[d]	110.8	**867.5**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	5.98 (5.90-6.05)	2	1,137.1	**4,376**	-	European Al Association 2009

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	5.73 (5.39-6.06)	4	8,046.7	**34,704**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	6.71 (6.44-6.98)	0.5[d]	>10,000	**>26,800**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	7.83 (7.74-7.92)	0.5[d]	>5,000	**>5,975**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	6.79 (6.55-7.03)	2	>10,000	**>10,615**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	7.67 (7.41-7.92)	2	>15,000	**>8,154**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	6.68 (6.35-7.01)	4	>15,000	**>12,073**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	60	7.62 (7.35-7.89)	4	>15,000	**>5,487**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	6.06 (5.97-6.14)	2	3,386.8	**6,889**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	5.60 (5.22-5.97)	4	10,484.2	**34,985**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	6.93 (6.84-7.02)	0.5[d]	>5,000	**>7,361**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	7.88 (7.80-7.95)	0.5[d]	>5,000	**>4,896**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	6.76 (6.43-7.09)	2	>15,000	**>11,400**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	7.71 (7.46-7.95)	2	>15,000	**>6,471**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	6.60 (6.21-6.98)	4	>15,000	**>9,047**	-	European Al Association 2009
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, U, T	Aluminum nitrate	120	7.60 (7.32-7.87)	4	>15,000	**>4,366**	-	European Al Association 2009

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	6.03 (6.02-6.03)	0.5[d] (stock solution not buffered)	119.71	**3,227**	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	6.03 (6.02-6.03)	0.5[d] (stock solution buffered)	274.78	**7,407**	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	6.03 (6.02-6.03)	0.5[d] (test solution MES buffered)	119.98	**3,234**	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	6.07 (6.06-6.07)	0.5[d] (0.0 µM PO_4 in test solution)	92.495	**2,273**	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	6.09 (6.08-6.09)	0.5[d] (12.0 µM PO_4 in test solution)	313.37	7,355[g]	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	6.10 (6.09-6.11)	0.5[d] (60.0 µM PO_4 in test solution)	332.35	7,625[g]	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	7.08 (7.06-7.09)	0.5[d] (test solution HCl buffered)	>886.4	**>3,528**	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	7.79 (7.70-7.88)	0.5[d] (test solution HEPES buffered)	>4,278.3	**>8,625**	-	European Al Association 2010

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Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	10.6	7.53 (7.45-7.61)	0.5[d] (test solution NaOH adjusted)	132.04	**322.4**	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	60.0	6.01 (5.99-6.03)	0.5[d] (stock solution not buffered)	463.26	**3,845**	-	European Al Association 2010
Cladoceran (<24 hr), *Ceriodaphnia dubia*	S, M, T	Aluminum nitrate	60.0	5.99 (5.98-5.99)	0.5[d] (stock solution buffered)	>859.0	**>7,415**	5,863	European Al Association 2010
Cladoceran (0-24 hr), *Ceriodaphnia reticulata*	S, U, T	Aluminum chloride	45.1	7.25 (6.8-7.7)	1.1[d]	2,800	3,070[f]	-	Shephard 1983
Cladoceran (0-24 hr), *Ceriodaphnia reticulata*	F, M, T	Aluminum chloride	45.1	6.0	1.1[d]	304	**1,967**	-	Shephard 1983
Cladoceran (0-24 hr), *Ceriodaphnia reticulata*	F, M, T	Aluminum chloride	4.0	5.5	1.1[d]	362	**53,910**	10,299	Shephard 1983
Cladoceran (0-24 hr), *Daphnia magna*	S, U, NR	Aluminum chloride	48.5 (44-53)	7.8 (7.4-8.2)	1.1[d]	3,900	**3,117**	-	Biesinger and Christensen 1972
Cladoceran (0-24 hr), *Daphnia magna*	S, M, T	Aluminum sulfate	220	7.60 (7.05-8.15)	1.6[d]	38,200	**15,625**	-	Kimball 1978
Cladoceran (0-24 hr), *Daphnia magna*	S, U, T	Aluminum chloride	45.1	7.25 (6.8-7.7)	1.1[d]	2,800	**3,070**	-	Shephard 1983
Cladoceran (<24 hr), *Daphnia magna*	S, U, T	Aluminum nitrate	168	5.99 (5.98-5.99)	0.5[d]	>500	**>2,075**	-	European Al Association 2009
Cladoceran (<24 hr), *Daphnia magna*	S, U, T	Aluminum nitrate	168	6.98 (6.97-6.98)	0.5[d]	>500	>598.9[c]	-	European Al Association 2009
Cladoceran (<24 hr), *Daphnia magna*	S, U, T	Aluminum nitrate	168	7.93 (7.92-7.94)	0.5[d]	>500	>449.2[c]	-	European Al Association 2009

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Cladoceran (<24 hr), *Daphnia magna*	S, U, T	Aluminum nitrate	168	7.92 (7.90-7.93)	0.5[d]	795.0	**713.2**	-	European Al Association 2009
Cladoceran (<24 hr), *Daphnia magna*	S, U, T	Aluminum nitrate	168	7.95 (7.92-7.97)	2	>1,200	>472.9[c]	-	European Al Association 2009
Cladoceran (<24 hr), *Daphnia magna*	S, U, T	Aluminum nitrate	168	7.93 (7.92-7.94)	3	>1,200	>369.9[c]	2,944	European Al Association 2009
Cladoceran (adult), *Daphnia pulex*	R, U, T	Aluminum chloride	142 (±2)	8.2 (±1)	1.6[d]	3,650	**1,836**	1,836	Griffitt et al. 2008
Ostracod (adult, 1.5 mm, 0.3 mg), *Stenocypris major*	R, M, T	Aluminum sulfate	15.63 (±2.74)	6.51 (±0.01)	3.2	3,102	**8,000**	8,000	Shuhaimi-Othman et al. 2011a, 2013
Amphipod (4 mm), *Crangonyx pseudogracilis*	R, U, T	Aluminum sulfate	50 (45-55)	6.75 (6.7-6.8)	1.6[d]	9,190	**12,901**	12,901	Martin and Holdich 1986
Amphipod (juvenile, 7 d, 1.32 mm), *Hyalella azteca*	F, M, T	Aluminum nitrate	105 (103-108)	6.13 (6.09-6.16)	0.48	>5,997	**>27,766**	>27,766	Wang et al. 2016, 2018
Midge (3rd-4th instar larvae), *Chironomus plumosus*	S, U, T	Aluminum chloride	80	7.0 (±0.5)	1.6[d]	30,000	**25,216**	25,216	Fargasova 2001, 2003
Midge (2nd-3rd instar larvae), *Paratanytarsus dissimilis*	S, U, T	Aluminum sulfate	17.43	7.28 (6.85-7.71)	2.8[d]	>77,700	**>70,647**	>70,647	Lamb and Bailey 1981, 1983
Rainbow trout (alevin), *Oncorhynchus mykiss*	S, U, T	Aluminum sulfate	14.3	5.5	0.4	160	10,037[f]	-	Holtze 1983

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Rainbow trout (alevin), *Oncorhynchus mykiss*	S, U, T	Aluminum sulfate	14.3	5.5	0.4	310	8,467[f]	-	Holtze 1983
Rainbow trout (fingerling), *Oncorhynchus mykiss*	S, M, T	Aluminum chloride	47.4 (±4.51)	6.59 (±0.15)	1.1[d]	7,400	13,495[f]	-	Call et al. 1984
Rainbow trout (fingerling), *Oncorhynchus mykiss*	S, M, T	Aluminum chloride	47.4 (±4.51)	7.31 (±0.89)	1.1[d]	14,600	11,879[f]	-	Call et al. 1984
Rainbow trout (fingerling), *Oncorhynchus mykiss*	S, M, T	Aluminum chloride	47.4 (±4.51)	8.17 (±0.42)	1.1[d]	>24,700	>7,664[f]	-	Call et al. 1984
Rainbow trout (fingerling), *Oncorhynchus mykiss*	S, M, T	Aluminum chloride	47.4 (±4.51)	7.46 (±0.14)	1.1[d] (18 d aged solution)	8,600	5,915[f]	-	Call et al. 1984
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	26.35 (25.3-27.4)	7.61 (7.58-7.64)	0.5[d]	>9,840	**>7,216**	-	Gundersen et al. 1994
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	45.5 (44.6-46.4)	7.59 (7.55-7.62)	0.5[d]	>8,070	**>5,766**	-	Gundersen et al. 1994
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	88.05 (86.6-89.5)	7.60 (7.58-7.62)	0.5[d]	>8,160	**>5,390**	-	Gundersen et al. 1994
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	127.6 (124.8-130.4)	7.61 (7.58-7.64)	0.5[d]	>8,200	**>5,164**	-	Gundersen et al. 1994
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	23.25 (21.9-24.6)	8.28 (7.97-8.58)	0.5[d]	6,170	**1,685**	-	Gundersen et al. 1994
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	35.4 (33.1-37.7)	8.30 (8.02-8.58)	0.5[d]	6,170	**1,680**	-	Gundersen et al. 1994
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	83.6 (83.0-84.2)	8.31 (8.06-8.56)	0.5[d]	7,670	**2,180**	-	Gundersen et al. 1994

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	F, M, T	Aluminum chloride	128.5 (112.5-144.5)	8.31 (8.06-8.56)	0.5[d]	6,930	**2,026**	3,312	Gundersen et al. 1994
Atlantic salmon (sac fry, ≈0.2 g), *Salmo salar*	S, U, T	Aluminum chloride	6.8 (6.6-7.0)	5.5	0.5[d]	584	**20,749**	-	Hamilton and Haines 1995
Atlantic salmon (sac fry, ≈0.2 g), *Salmo salar*	S, U, T	Aluminum chloride	6.8 (6.6-7.0)	6.5	0.5[d]	599	**3,599**	8,642	Hamilton and Haines 1995
Brook trout (14 mo., 210 mm, 130 g), *Salvelinus fontinalis*	F, M, T	Aluminum sulfate	-	6.5	-	3,600	NA[e]	-	Decker and Menendez 1974
Brook trout (14 mo., 210 mm, 130 g), *Salvelinus fontinalis*	F, M, T	Aluminum sulfate	-	6.0	-	4,400	NA[e]	-	Decker and Menendez 1974
Brook trout (14 mo., 210 mm, 130 g), *Salvelinus fontinalis*	F, M, T	Aluminum sulfate	-	5.5	-	4,000	NA[e]	-	Decker and Menendez 1974
Brook trout (0.6 g, 4.4-7.5 cm), *Salvelinus fontinalis*	S, U, T	Aluminum sulfate	40	5.6	1.6[d]	6,530	**30,038**	-	Tandjung 1982
Brook trout (0.6 g, 4.4-7.5 cm), *Salvelinus fontinalis*	S, U, T	Aluminum sulfate	18	5.6	1.6[d]	3,400	**24,514**	-	Tandjung 1982
Brook trout (0.6 g, 4.4-7.5 cm), *Salvelinus fontinalis*	S, U, T	Aluminum sulfate	2	5.6	1.6[d]	370	**9,187**	18,913	Tandjung 1982

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Green sunfish (juvenile, 3 mo.), *Lepomis cyanellus*	S, M, T	Aluminum chloride	47.4 (±4.51)	7.55 (±0.13)	1.1[d]	>50,000	**>31,087**	>31,087	Call et al. 1984
Guppy, *Poecilia reticulata*	R, M, T	Aluminum sulfate	18.72 (±1.72)	6.68 (±0.2)	3.2	6,760	**9,061**	9,061	Shuhaimi-Othman et al. 2013
Rio Grande silvery minnow (larva, 3-5 dph), *Hybognathus amarus*	R, M, T	Aluminum chloride	140	8.1 (7.9-8.4)	0.5[d]	>59,100	**>21,779**	>21,779	Buhl 2002
Fathead minnow (adult), *Pimephales promelas*	S, U, NR	Aluminum sulfate	-	7.6	-	>18,900	NA[e]	-	Boyd 1979
Fathead minnow (juvenile, 32-33 d), *Pimephales promelas*	S, M, T	Aluminum chloride	47.4 (±4.51)	7.61	1.1[d]	>48,200	**>28,019**	-	Call et al. 1984
Fathead minnow (juvenile, 32-33 d), *Pimephales promelas*	S, M, T	Aluminum chloride	47.4 (±4.51)	8.05	1.1[d]	>49,800	**>17,678**	-	Call et al. 1984
Fathead minnow (juvenile, 11 mm, 3 mg dw), *Pimephales promelas*	F, U, T	Aluminum chloride	21.6 (±1.31)	6.5 (±0.2)	0.9	>400	>1,181[c]	-	Palmer et al. 1989
Fathead minnow (juvenile, 11 mm, 3 mg dw), *Pimephales promelas*	F, U, T	Aluminum chloride	21.6 (±1.31)	7.5 (±0.2)	0.9	>400	>304.5[c]	-	Palmer et al. 1989
Fathead minnow (larva, 7 mm, 0.31 mg, 12 dph), *Pimephales promelas*	F, U, T	Aluminum chloride	21.6 (±1.31)	7.5 (±0.2)	0.9	>400	>304.5[c]	-	Palmer et al. 1989
Fathead minnow (yolk-sac larva, 1 dph), *Pimephales promelas*	F, U, T	Aluminum chloride	21.6 (±1.31)	6.5 (±0.2)	0.9	>400	>1,181[c]	-	Palmer et al. 1989

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	LC_{50} / EC_{50} (µg/L)	Normalized Acute Value[b] (µg/L)	Species Mean Acute Value (µg/L)	Reference
Fathead minnow (yolk-sac larva, 1 dph), *Pimephales promelas*	F, U, T	Aluminum chloride	21.6 (±1.31)	7.5 (±0.2)	0.9	>400	>304.5[c]	-	Palmer et al. 1989
Fathead minnow (larva, 4-6 dph), *Pimephales promelas*	R, M, T	Aluminum chloride	140	8.1 (7.9-8.4)	0.5[d]	>59,100	**>21,779**	>22,095	Buhl 2002
Smallmouth bass (larva, 48 hr post hatch), *Micropterus dolomieui*	S, M, T	Aluminum sulfate	12.15 (12.1-12.2)	5.05 (4.7-5.4)	1.6[d]	130	**2,442**	-	Kane 1984; Kane and Rabeni 1987
Smallmouth bass (larva, 48 hr post hatch), *Micropterus dolomieui*	S, M, T	Aluminum sulfate	12.4 (12.0-12.8)	6.25 (6.0-6.5)	1.6[d]	>978.4	**>3,655**	-	Kane 1984; Kane and Rabeni 1987
Smallmouth bass (larva, 48 hr post hatch), *Micropterus dolomieui*	S, M, T	Aluminum sulfate	12.0	7.5 (7.2-7.8)	1.6[d]	>216.8	>153.4[c]	2,988	Kane 1984; Kane and Rabeni 1987
Green tree frog (tadpole, <1 dph), *Hyla cinerea*	R, M, T	Aluminum chloride	4.55	5.49 (5.48-5.50)	0.5[d]	>405.2	**>18,563**	>18,563	Jung and Jagoe 1995

[a] S=static, F=flow-through, U=unmeasured, M=measured, A=acid exchangeable aluminum, T=total aluminum, D=dissolved aluminum, NR=not reported.

[b] Normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1.0 mg/L (see Section 2.7.1). Values in bold are used in SMAV calculations.

[c] Not used to calculate SMAV because either a more definitive value is available or value is considered an outlier.

[d] When definitive DOC values were not reported by the authors: a DOC value of 0.5 mg/L was used when dilution water was reconstituted, 1.1 mg/L when dilution water was Lake Superior, MN water, 2.8 mg/L when dilution water was Liberty Lake, WA water, 1.6 mg/L when dilution water was tap or well water, or half the detection limit when the reported value was less than the detection limit, based on recommendations in the 2007 Freshwater Copper AWQC (U.S. EPA 2007b).

[e] Missing water quality parameters and/or dilution water type needed to estimate water quality parameters, so values cannot be normalized.

[f] Not used to calculate SMAV because flow-through measured test(s) available.

[g] Phosphate in exposure media is providing an ameliorating effect against aluminum.

Appendix B ACCEPTABLE ACUTE TOXICITY DATA OF ALUMINUM TO ESTUARINE/MARINE AQUATIC ANIMALS

Appendix B. Acceptable Acute Toxicity Data of Aluminum to Estuarine/Marine Aquatic Animals
(Bold values are used in SMAV calculation).
(Species are organized phylogenetically).

Species	Method[a]	Chemical	Salinity (g/kg)	pH	LC_{50} / EC_{50} (µg/L)	Species Mean Acute Value (µg/L)	Reference
Polychaete worm, *Capitella capitata*	S, U	Aluminum chloride	-	-	**404.8**	404.8	Petrich and Reish 1979
Polychaete worm, *Ctenodrilus serratus*	S, U	Aluminum chloride	-	-	**97.15**	97.15	Petrich and Reish 1979
Polychaete worm, *Neanthes arenaceodentata*	S, U	Aluminum chloride	-	-	**>404.8**	>404.8	Petrich and Reish 1979
Copepod (adult), *Nitokra spinipes*	S, U	Aluminum chloride	7	8	**10,000**	10,000	Bengtsson 1978
American oyster (fertilized eggs, ≤1 hr), *Crassostrea virginica*	S, U	Aluminum chloride	25	7.0-8.5	**>1,518**	>1,518	Calabrese et al. 1973

[a] S=static, F=flow-through, U=unmeasured, M=measured.

Appendix C ACCEPTABLE CHRONIC TOXICITY DATA OF ALUMINUM TO FRESHWATER AQUATIC ANIMALS

Appendix C. Acceptable Chronic Toxicity Data of Aluminum to Freshwater Aquatic Animals
(Bold values are used in SMCV calculation).
(Species are organized phylogenetically).

Species	Test[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	EC_{20} Endpoint	EC_{20} (µg/L)	Normalized Chronic Value[b] (µg/L)	Species Mean Chronic Value (µg/L)	Reference
Oligochaete (<24 hr), *Aeolosoma sp.*	17 d	Aluminum nitrate	48	5.95 (5.8-6.1)	<0.5[c]	Reproduction (population size)	1,235	**20,514**	20,514	OSU 2012e; Cardwell et al. 2018
Rotifer (newly hatched, <2 hr), *Brachionus calyciflorus*	48 hr	Aluminum nitrate	100	6.45 (6.4-6.5)	<0.5[c]	Reproduction (population size)	431.0	**1,845**	-	OSU 2012c; Cardwell et al. 2018
Rotifer (newly hatched, <2 hr), *Brachionus calyciflorus*	48 hr	Aluminum nitrate	63	6.3 (5.98-6.56)	1.39	Reproduction (population size)	1,751	**4,518**	-	OSU 2018e
Rotifer (newly hatched, <2 hr), *Brachionus calyciflorus*	48 hr	Aluminum nitrate	105	6.3 (6.02-6.55)	1.39	Reproduction (population size)	2,066	**3,844**	-	OSU 2018e
Rotifer (newly hatched, <2 hr), *Brachionus calyciflorus*	48 hr	Aluminum nitrate	114	6.2 (5.98-6.47)	2.63	Reproduction (population size)	3,061	**4,323**	-	OSU 2018e
Rotifer (newly hatched, <2 hr), *Brachionus calyciflorus*	48 hr	Aluminum nitrate	105	6.1 (5.89-6.63)	3.77	Reproduction (population size)	4,670	**6,653**	-	OSU 2018e
Rotifer (newly hatched, <2 hr), *Brachionus calyciflorus*	48 hr	Aluminum nitrate	185	6.3 (6.05-6.54)	1.33	Reproduction (population size)	1,604	**2,132**	3,539	OSU 2018e
Great pond snail (newly-hatched, <24 hr), *Lymnaea stagnalis*	30 d	Aluminum nitrate	117	6.0 (5.6-6.4)	<0.5[c]	Biomass	745.7	**5,945**	-	OSU 2012b; Cardwell et al. 2018

Species	Test[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	EC_{20} Endpoint	EC_{20} (µg/L)	Normalized Chronic Value[b] (µg/L)	Species Mean Chronic Value (µg/L)	Reference
Great pond snail (newly-hatched, <24 hr), *Lymnaea stagnalis*	30 d	Aluminum nitrate	121 (121-122)	6.15 (6.08-6.45)	1.37 (1.29-1.45)	Biomass	833.4	**1,812**	-	OSU 2018f
Great pond snail (newly-hatched, <24 hr), *Lymnaea stagnalis*	30 d	Aluminum nitrate	124 (121-127)	6.17 (6.06-6.41)	1.45 (1.38-1.51)	Biomass	1,951	**3,902**	-	OSU 2018f
Great pond snail (newly-hatched, <24 hr), *Lymnaea stagnalis*	30 d	Aluminum nitrate	117 (116-118)	5.98 (5.86-6.16)	3.85 (3.60-4.20)	Biomass	1,392	**2,251**	3,119	OSU 2018f
Fatmucket (6 wk, 1.97 mm), *Lampsilis siliquoidea*	28 d	Aluminum nitrate	105.5 (105-106)	6.04 (5.95-6.12)	0.40 (0.34-0.45)	Biomass	169	**1,026**	1,026	Wang et al. 2016, 2018
Cladoceran (≤16 hr), *Ceriodaphnia dubia*	LC	Aluminum chloride	50	7.15 (±0.05)	1.1[c]	Reproduction (young/adult)	1,780	**2,031**	-	McCauley et al. 1986
Cladoceran (≤16 hr), *Ceriodaphnia dubia*	LC	Aluminum chloride	50.5	7.61 (±0.11)	1.1[c]	Reproduction (young/adult)	<1,100 (MATC)	<925.5[f]	-	McCauley et al. 1986
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum chloride	25 (24-26)	7.65 (7.3-8.0)	0.5[c]	Reproduction (young/female)	1,557	**2,602**	-	ENSR 1992b
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum chloride	47 (46-48)	7.7 (7.3-8.1)	0.5[c]	Reproduction (young/female)	808.7	**1,077**	-	ENSR 1992b
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum chloride	94 (92-96)	8.2 (7.9-8.5)	0.5[c]	Reproduction (young/female)	647.2	**708.8**	-	ENSR 1992b
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum chloride	196 (194-198)	8.45 (8.1-8.8)	0.5[c]	Reproduction (young/female)	683.6	**746.8**	-	ENSR 1992b
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	6.34	0.5[c]	Reproduction	36.6	**291.7**	-	European Al Association 2010; Gensemer et al. 2018

Species	Test[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	EC_{20} Endpoint	EC_{20} (µg/L)	Normalized Chronic Value[b] (µg/L)	Species Mean Chronic Value (µg/L)	Reference
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	60	6.4	0.5[c]	Reproduction	160.3	**667.9**	-	European Al Association 2010; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	120	6.38	0.5[c]	Reproduction	221.6	**619.4**	-	European Al Association 2010; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	6.34	2	Reproduction	377.4	**1,315**	-	European Al Association 2010; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	60	6.38	2	Reproduction	631.3	**1,187**	-	European Al Association 2010; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	120	6.37	2	Reproduction	1,011.6	**1,254**	-	European Al Association 2010; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	6.33	4	Reproduction	622.6	**1,460**	-	European Al Association 2010; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	60	6.3	4	Reproduction	692.9	**981.4**	-	European Al Association 2010; Gensemer et al. 2018

Species	Test[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	EC_{20} Endpoint	EC_{20} (µg/L)	Normalized Chronic Value[b] (µg/L)	Species Mean Chronic Value (µg/L)	Reference
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	120	6.38	4	Reproduction	840.5	**678.9**	-	European Al Association 2010; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	6.37	2	Reproduction	353.0	**1,164**	-	Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	6.34	2	Reproduction	452.4	**1,576**	-	Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	6.35	2	Reproduction	439.7	**1,504**	-	Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	7.04	0.5	Reproduction (young/female)	250	**701.1**	-	CECM 2014; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	120	7.14	0.5	Reproduction (young/female)	860	**1,072**	-	CECM 2014; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	7.98	0.5	Reproduction (young/female)	700	**1,029**	-	CECM 2014; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	60	8.03	0.5	Reproduction (young/female)	1,010	**1,189**	-	CECM 2014; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	120	8.1	0.5	Reproduction (young/female)	870	**879.6**	-	CECM 2014; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	25	6.34	0.5	Reproduction (young/female)	260	**2,072**	-	CECM 2014; Gensemer et al. 2018
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	120	6.36	0.5	Reproduction (young/female)	390	**1,122**	-	CECM 2014; Gensemer et al. 2018

Species	Test[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	EC_{20} Endpoint	EC_{20} (µg/L)	Normalized Chronic Value[b] (µg/L)	Species Mean Chronic Value (µg/L)	Reference
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	64	6.42	1.87	Reproduction (young/female)	828.6	**1,463**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	133	6.325	8.71	Reproduction (young/female)	3,829	**1,973**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	138	6.395	12.3	Reproduction (young/female)	6,224	**2,308**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	428	6.295	1.64	Reproduction (young/female)	2,011	**1,388**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	125	7.205	6.57	Reproduction (young/female)	6,401	**1,614**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	127	7.185	12.01	Reproduction (young/female)	6,612	**1,170**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	263	8.17	1.3	Reproduction (young/female)	3,749	**1,854**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	425	8.21	1.2	Reproduction (young/female)	2,852	**1,372**	-	OSU 2018a
Cladoceran (<24 hr), *Ceriodaphnia dubia*	LC	Aluminum nitrate	125	8.7	1.04	Reproduction (young/female)	1,693	**1,530**	1,181	OSU 2018a
Cladoceran (<24 hr), *Daphnia magna*	LC	Aluminum nitrate	140	6.3	2	Reproduction (young/female)	791.0	**985.3**	985.3	European Al Association 2010; Gensemer et al. 2018
Amphipod (juvenile, 7-9 d), *Hyalella azteca*	28 d	Aluminum nitrate	95	6.35 (6.0-6.7)	0.51	Biomass	199.3	**665.9**	-	OSU 2012h; Cardwell et al. 2018
Amphipod (juvenile, 7 d, 1.31 mm), *Hyalella azteca*	28 d	Aluminum nitrate	106 (105-107)	6.04 (5.92-6.16)	0.33 (0.26-0.39)	Biomass	425	**2,890**	1,387	Wang et al. 2016, 2018

Species	Test[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	DOC (mg/L)	EC_{20} Endpoint	EC_{20} (µg/L)	Normalized Chronic Value[b] (µg/L)	Species Mean Chronic Value (µg/L)	Reference
Midge (1st instar larva, <24 hr), *Chironomus riparius*	30 d	Aluminum sulfate	11.8	5.58 (5.51-5.64)	1.8[e]	Adult midge emergence	29.55	**1,075**	-	Palawski et al. 1989
Midge (1st instar larva, <24 hr), *Chironomus riparius*	30 d	Aluminum sulfate	11.9	5.05 (4.99-5.1)	1.8[e]	Adult midge emergence	84.42	**15,069**	-	Palawski et al. 1989
Midge (1st instar larva, 3d), *Chironomus riparius*	28 d	Aluminum nitrate	91	6.6 (6.3-6.9)	0.51	Reproduction (# of eggs/case)	3,387	**8,181**	5,099	OSU 2012f; Cardwell et al. 2018
Atlantic salmon (embryo), *Salmo salar*	ELS	Aluminum sulfate	12.7	5.7 (5.6-5.8)	1.8[e]	Biomass	61.56	**434.4**	-	McKee et al. 1989
Atlantic salmon (fertilized eggs), *Salmo salar*	ELS	Aluminum sulfate	12.7	5.7 (5.6-5.8)	1.8[e]	Survival	154.2	1,088[d]	434.4	Buckler et al. 1995
Brook trout (eyed eggs), *Salvelinus fontinalis*	ELS	Aluminum sulfate	12.3	6.55 (6.5-6.6)	1.9	Biomass	164.4	**378.7**	-	Cleveland et al. 1989
Brook trout (eyed eggs), *Salvelinus fontinalis*	ELS	Aluminum sulfate	12.8	5.65 (5.6-5.7)	1.8	Biomass	143.5	**1,076**	638.2	Cleveland et al. 1989
Fathead minnow, *Pimephales promelas*	ELS	Aluminum sulfate	220	7.70 (7.27-8.15)	1.6[c]	Biomass	6,194	**2,690**	-	Kimball 1978
Fathead minnow (embryo, <24 hr), *Pimephales promelas*	ELS	Aluminum nitrate	96	6.20 (5.9-6.5)	<0.5[c]	Survival	428.6	**2,154**	2,407	OSU 2012g; Cardwell et al. 2018
Zebrafish (embryo, <36hpf), *Danio rerio*	ELS	Aluminum nitrate	83	6.15 (6.0-6.3)	<0.5[c]	Biomass	234.4	**1,342**	1,342	OSU 2013; Cardwell et al. 2018

[a] LC=Life cycle, ELS=Early life-stage.
[b] Normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1.0 mg/L (see Section 2.7.1). Values in bold are used in SMCV calculations.
[c] When definitive DOC values were not reported by the authors: a DOC value of 0.5 mg/L was used when dilution water was reconstituted, 1.1 mg/L when dilution water was Lake Superior water, 1.6 mg/L when dilution water was tap or well water, or half the detection limit when the reported value was less than the detection limit, based on recommendations in the 2007 Freshwater Copper AWQC (U.S. EPA 2007b).
[d] Buckler et al. (1995) appears to be a republication of McKee et al. (1989), but does not report the most sensitive endpoint and therefore only the most sensitive endpoint used for calculation of the SMCV.
[e] DOC was taken from reported values in Cleveland et al. (1989) for a similar pH; all studies are from the same lab and used the same procedures to make the dilution water (well water plus reverse osmosis water mixture).
[f] Value is a MATC, poor dose response prevented an EC_{20} from being calculated; not used in SMCV calculation.

Appendix D ACCEPTABLE CHRONIC TOXICITY DATA OF ALUMINUM TO ESTUARINE/MARINE AQUATIC ANIMALS

Appendix D. Acceptable Chronic Toxicity Data of Aluminum to Estuarine/Marine Aquatic Animals

Species	Duration	Chemical	Salinity (g/kg)	pH	Chronic Limits (µg/L)	Chronic Value (µg/L)	Effect	Species Mean Chronic Value (µg/L)	Reference
Estuarine/Marine Species									

There are no acceptable estuarine/marine chronic toxicity data for aluminum.

Appendix E ACCEPTABLE TOXICITY DATA OF ALUMINUM TO FRESHWATER AQUATIC PLANTS

Appendix E. Acceptable Toxicity Data of Aluminum to Freshwater Aquatic Plants

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	Duration	Effect	Chronic Limits (µg/L)	Concentration (µg/L)	Reference
Freshwater Species									
Green alga, *Arthrodesmus octocornus*	S, M	-	-	5.7	21 d	LOEC (number of semicells)	-	50	Pillsbury and Kingston 1990
Green alga, *Arthrodesmus indentatus*	S, M	-	-	5.7	21 d	LOEC (number of semicells)	-	50	Pillsbury and Kingston 1990
Green alga, *Arthrodesmus quiriferus*	S, M	-	-	5.7	21 d	LOEC (number of semicells)	-	50	Pillsbury and Kingston 1990
Green alga, *Dinobryon bavaricum*	S, M	-	-	5.7	21 d	NOEC (number of cells)	-	>200	Pillsbury and Kingston 1990
Green alga, *Elaktothrix sp.*	S, M	-	-	5.7	21 d	Number of cells	100-200	141.4	Pillsbury and Kingston 1990
Green alga, *Oedogonium sp.*	S, M	-	-	5.7	21 d	NOEC (number of cells)	-	>200	Pillsbury and Kingston 1990
Green alga, *Peridinium limbatum*	S, M	-	-	5.7	21 d	NOEC (number of cells)	-	>200	Pillsbury and Kingston 1990
Green alga, *Staurastrum arachne v. curvatum*	S, M	-	-	5.7	21 d	LOEC (number of semicells)	-	50	Pillsbury and Kingston 1990

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	Duration	Effect	Chronic Limits (µg/L)	Concentration (µg/L)	Reference
Green alga, *Staurastrum longipes v. contractum*	S, M	-	-	5.7	21 d	LOEC (number of semicells)	-	50	Pillsbury and Kingston 1990
Green alga, *Staurastrum pentacerum*	S, M	-	-	5.7	21 d	LOEC (number of semicells)	-	50.0	Pillsbury and Kingston 1990
Green alga, *Mougeotia sp.*	S, U	Aluminum sulfate	-	4.1	14 d	NOEC (chlorophyll a)	-	3,600	Graham et al. 1996
Green alga, *Monoraphidium dybowskii*	S, U	Aluminum chloride	-	5.0	12 d	EC50 (growth)	-	1,000	Claesson and Tornqvist 1988
Green alga, *Monoraphidium dybowskii*	S, U	Aluminum chloride	-	5.5	12 d	EC50 (growth)	-	1,000	Claesson and Tornqvist 1988
Green alga, *Monoraphidium dybowskii*	S, U	Aluminum chloride	-	6.0	12 d	EC50 (growth)	-	550	Claesson and Tornqvist 1988
Green alga, *Monoraphidium dybowskii*	S, U	Aluminum chloride	14.9	4.8	4 d	Growth	600-1,000	774.6	Hornstrom et al. 1995
Green alga, *Monoraphidium dybowskii*	S, U	Aluminum chloride	14.9	6.8	4 d	LOEC (growth)	-	200	Hornstrom et al. 1995
Green alga, *Monoraphidium griffithii*	S, U	Aluminum chloride	14.9	4.8	4 d	LOEC (growth)	-	100	Hornstrom et al. 1995
Green alga, *Monoraphidium griffithii*	S, U	Aluminum chloride	14.9	6.8	4 d	LOEC (growth)	-	100	Hornstrom et al. 1995

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	Duration	Effect	Chronic Limits (µg/L)	Concentration (µg/L)	Reference
Green alga, *Scenedesmus quadricauda*	S, U	Aluminum chloride	-	7.5	4 d	LOEC (growth inhibition)	-	1,500	Bringmann and Kuhn 1959b
Green alga, *Pseudokirchneriella subcapitata*	-	Sodium aluminate	15	7.0	14 d	Reduce cell counts and dry weight	990-1,320	1,143	Peterson et al. 1974
Green alga, *Pseudokirchneriella subcapitata*	S, U	Aluminum chloride	47.4 (±4.51)	7.6	4 d	EC50 (biomass)	-	570	Call et al. 1984
Green alga, *Pseudokirchneriella subcapitata*	S, U	Aluminum chloride	47.4 (±4.51)	8.2	4 d	EC50 (biomass)	-	460	Call et al. 1984
Green alga, *Pseudokirchneriella subcapitata*	S, U	Aluminum sulfate	-	5.5	4 d	LOEC (growth inhibition)	-	160	Kong and Chen 1995
Green alga, *Stichococcus sp.*	S, U	Aluminum chloride	-	5.0	9 d	IC50 (growth rate)	-	560	Tornqvist and Claesson 1987
Green alga, *Stichococcus sp.*	S, U	Aluminum chloride	-	5.0	9 d	EC50 (growth)	-	500	Claesson and Tornqvist 1988
Green alga, *Stichococcus sp.*	S, U	Aluminum chloride	-	5.5	9 d	EC50 (growth)	-	220	Claesson and Tornqvist 1988
Diatom, *Asterionella ralfsii var. americana*	S, M	Aluminum chloride	-	5.0	7-9 d	Growth	404.7-620.5	501.1	Gensemer 1989
Diatom, *Asterionella ralfsii var. americana*	S, M	Aluminum chloride	-	6.0	7-9 d	Growth	404.7-647.5	511.9	Gensemer 1989

Species	Method[a]	Chemical	Total Hardness (mg/L as $CaCO_3$)	pH	Duration	Effect	Chronic Limits (µg/L)	Concentration (µg/L)	Reference
Diatom, *Asterionella ralfsii var. americana*	S, M	-	-	5.7	21 d	LOEC (number of live cells)	-	50	Pillsbury and Kingston 1990
Diatom, *Cyclotella meneghiniana*	S, U	Aluminum chloride	-	7.9	16 d	Partially inhibit growth	-	809.6	Rao and Subramanian 1982
Diatom, *Cyclotella meneghiniana*	S, U	Aluminum chloride	-	7.9	16 d	Algistatic	-	3,238	Rao and Subramanian 1982
Diatom, *Cyclotella meneghiniana*	S, U	Aluminum chloride	-	7.9	16 d	Algicidal	-	6,477	Rao and Subramanian 1982
Eurasian watermilfoil, *Myriophyllum spicatum*	S, U	-	95.93	-	32 d	IC50 (root dry weight)	-	2,500	Stanley 1974
Duckweed, *Lemna minor*	S, M	Aluminum chloride	47.4 (±4.51)	7.6	4 d	NOEC (reduce frond production)	-	>45,700	Call et al. 1984
Duckweed, *Lemna minor*	S, M	Aluminum chloride	47.4 (±4.51)	8.2	4 d	NOEC (reduce frond production)	-	>45,700	Call et al. 1984

[a] S=static, F=flow-through, U=unmeasured, M=measured.

Appendix F ACCEPTABLE TOXICITY DATA OF ALUMINUM TO ESTUARINE/MARINE AQUATIC PLANTS

Appendix F. Acceptable Toxicity Data of Aluminum to Estuarine/Marine Aquatic Plants

Species	Method[a]	Chemical	Salinity (g/kg)	pH	Duration	Effect	Chronic Limits (µg/L)	Concentration (µg/L)	Reference
Estuarine/Marine Species									
Seagrass, *Halophila stipulacea*	R, U	-	35.0	6.5-7.0	12 d	Observed protoplast necrosis	0.02698-0.2698	0.08532	Malea and Haritonidis 1996
Seagrass, *Halophila stipulacea*	R, U	-	35.0	6.5-7.0	12 d	Greater than 50% mortality of teeth cells	-	269.8	Malea and Haritonidis 1996
Seagrass, *Halophila stipulacea*	R, U	-	35.0	6.5-7.0	12 d	Less than 50% mortality of teeth cells	-	26.98	Malea and Haritonidis 1996

[a] S=static, F=flow-through, U=unmeasured, M=measured.

Appendix G ACCEPTABLE BIOACCUMULATION DATA OF ALUMINUM BY AQUATIC ORGANISMS

Appendix G. Acceptable Bioaccumulation Data of Aluminum by Aquatic Organisms

Species	Lifestage	Chemical	Concentration in water (µg/L)	Total Hardness (mg/L as $CaCO_3$)	pH	Tissue	Duration	BCF or BAF	Reference
Freshwater Species									
Snail, *Lymnaea stagnalis*	-	Aluminum nitrate	242	208	7	Digestive gland	30 d	4.26	Dobranskyte et al. 2004
Snail, *Lymnaea stagnalis*	-	Aluminum nitrate	242	208	7	Soft tissue	15 d	2.29	Dobranskyte et al. 2004
Brook trout, *Salvelinus fontinalis*	30 d	Aluminum sulfate	214.0	~12.5	5.3	Whole body	14 d	142	Cleveland et al. 1991a
Brook trout, *Salvelinus fontinalis*	30 d	Aluminum sulfate	223.5	~12.5	6.1	Whole body	14 d	104	Cleveland et al. 1991a
Brook trout, *Salvelinus fontinalis*	30 d	Aluminum sulfate	267.6	~12.5	7.2	Whole body	56 d	14.2	Cleveland et al. 1991a
Atlantic salmon, *Salmo salar*	larva	Aluminum sulfate	33	12.8	5.5	Whole body	60 d (embryo to post-hatch)	76	Buckler et al. 1995
Atlantic salmon, *Salmo salar*	larva	Aluminum sulfate	71	12.8	5.5	Whole body	60 d (embryo to post-hatch)	154	Buckler et al. 1995
Atlantic salmon, *Salmo salar*	larva	Aluminum sulfate	124	12.8	5.5	Whole body	60 d (embryo to post-hatch)	190	Buckler et al. 1995
Species	**Lifestage**	**Chemical**	**Concentration in water (µg/L)**	**Salinity (g/kg)**	**pH**	**Tissue**	**Duration**	**BCF or BAF**	**Reference**
Estuarine/Marine Species									

There are no acceptable estuarine/marine bioaccumulation data for aluminum.

Appendix H OTHER DATA ON EFFECTS OF ALUMINUM TO FRESHWATER AQUATIC ORGANISMS

Appendix H. Other Data on Effects of Aluminum to Freshwater Aquatic Organisms

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Freshwater Species								
Planktonic communities	Aluminum sulfate	1 hr	-	6.1-6.9	Decreased phosphate uptake and photosynthesis	50	Nalewajko and Paul 1985	Community exposure
Algal community	Aluminum sulfate	28 d	-	4.8	Growth	100-500 (NOEC-LOEC)	Genter and Amyot 1994	Community exposure
Microcosm community	Aluminum chloride	21 d	-	-	Production rate	2,000-5,000 (NOEC-LOEC)	Sugiura 2001	Community exposure
Blue-green alga, *Aphanizomenon flos-aquae*	Aluminum sulfate	22 hr	12.6	8.0	IC50 (nitrogen fixation)	>3,942	Peterson et al. 1995	Duration
Green alga, *Dunaliella acidophila*	Aluminum chloride	4-5 d	-	1.0	IC50 (photosynthesis)	>269,800	Gimmler et al. 1991	Lack of exposure details
Green alga, *Dunaliella acidophila*	Aluminum chloride	4-5 d	-	7.0	IC50 (photosynthesis)	134,900	Gimmler et al. 1991	Lack of exposure details
Green alga, *Dunaliella acidophila*	Aluminum chloride	4-5 d	-	1.0	IC50 (growth)	>269,800	Gimmler et al. 1991	Lack of exposure details
Green alga, *Dunaliella parva*	Aluminum chloride	4-5 d	-	7.0	IC50 (photosynthesis)	26,980	Gimmler et al. 1991	Lack of exposure details
Green alga, *Dunaliella parva*	Aluminum chloride	4-5 d	-	5.5	IC50 (growth)	1,619	Gimmler et al. 1991	Lack of exposure details
Green alga, *Chlorella sp.*	Aluminum sulfate	72 hr	1.0 (DOC = 1 mg/L)	5.0	IC50 (growth)	275	Trenfield et al. 2012	Duration
Green alga, *Chlorella sp.*	Aluminum sulfate	72 hr	1.0 (DOC = 2 mg/L)	5.0	IC50 (growth)	613	Trenfield et al. 2012	Duration
Green alga, *Chlorella sp.*	Aluminum sulfate	72 hr	4.1 (DOC = 1 mg/L)	5.0	IC50 (growth)	437	Trenfield et al. 2012	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Chlorella sp.*	Aluminum sulfate	72 hr	4.1 (DOC = 2 mg/L)	5.0	IC50 (growth)	801	Trenfield et al. 2012	Duration
Green alga, *Chlorella pyrenoidosa*	Aluminum sulfate	26 d	-	4.6	Reduced growth	6,000-12,000 (NOEC-LOEC)	Foy and Gerloff 1972	pH too low
Green alga, *Chlorella pyrenoidosa*	Aluminum chloride	5 d	-	5.0	Growth	50-100 (NOEC-LOEC)	Parent and Campbell 1994	pH too low
Green alga, *Chlorella vulgaris*	Aluminum chloride	3-4 mo.	-	<7.0	Inhibited growth	4,000	De Jong 1965	Lack of exposure details
Green alga, *Chlorella vulgaris*	Aluminum chloride	15 d	-	6.8	LC50	107,952	Rai et al. 1998	Lack of exposure details
Green alga, *Chlorella vulgaris*	Aluminum chloride	15 d	-	6.0	LC50	5,937	Rai et al. 1998	Lack of exposure details
Green alga, *Chlorella vulgaris*	Aluminum chloride	3 d	-	4.5	LC50	4,048	Rai et al. 1998	Lack of exposure details
Green alga, *Monoraphidium dybowskii*	Aluminum chloride	9 d	-	5.0	IC56 (growth rate)	1,800	Tornqvist and Claesson 1987	Atypical endpoint
Green alga, *Monoraphidium dybowskii*	Aluminum chloride	9 d	-	5.0	IC42 (growth rate)	560	Tornqvist and Claesson 1987	Atypical endpoint
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (growth) - flask	2,206	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (growth) - flask	2,894	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (growth) - 24 well microplate	2,834	Eisentraeger et al. 2003	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (growth) - 24 well microplate	3,340	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (growth) - 96 well microplate	2,773	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (growth) - 96 well microplate	2,915	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (biomass) - flask	2,028	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (biomass) - flask	2,423	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (unidentified) - flask	2,605	Eisentraeger et al. 2003	Duration
Green alga, Desmo*desmus subspicatus*	Aluminum chloride	72 hr	-	-	EC50 (unidentified) - flask	2,467	Eisentraeger et al. 2003	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 0 mg/L)	6.25	EC50 (biomass)	28.3	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 0 mg/L)	7.23-7.26	EC50 (biomass)	155.5	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 0 mg/L)	8.05-8.12	EC50 (biomass)	851.4	European Al Association 2009; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 0 mg/L)	6.29-6.30	EC50 (biomass)	76.4	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 0 mg/L)	7.12-7.13	EC50 (biomass)	232.9	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 0 mg/L)	7.90-8.12	EC50 (biomass)	516.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 0 mg/L)	6.22-6.24	EC50 (biomass)	74.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 0 mg/L)	7.10-7.13	EC50 (biomass)	226.3	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 0 mg/L)	7.94-8.11	EC50 (biomass)	366.9	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 0 mg/L)	6.25	EC50 (growth rate)	72.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 0 mg/L)	7.23-7.26	EC50 (growth rate)	345.6	European Al Association 2009; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 0 mg/L)	8.05-8.12	EC50 (growth rate)	1,351.8	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 0 mg/L)	6.29-6.30	EC50 (growth rate)	206.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 0 mg/L)	7.12-7.13	EC50 (growth rate)	584.0	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 0 mg/L)	7.90-8.12	EC50 (growth rate)	1,607.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 0 mg/L)	6.22-6.24	EC50 (growth rate)	323.4	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 0 mg/L)	7.10-7.13	EC50 (growth rate)	550.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 0 mg/L)	7.94-8.11	EC50 (growth rate)	889.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 2 mg/L)	6.19-6.23	EC50 (biomass)	669.9	European Al Association 2009; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 2 mg/L)	6.96-7.05	EC50 (biomass)	1,815.8	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 2 mg/L)	7.74-7.96	EC50 (biomass)	2,157.0	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 2 mg/L)	6.13-6.19	EC50 (biomass)	1,030.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 2 mg/L)	6.97-7.04	EC50 (biomass)	2,266.7	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 2 mg/L)	7.82-8.04	EC50 (biomass)	927.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 2 mg/L)	6.09-6.18	EC50 (biomass)	1,451.5	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 2 mg/L)	6.94-7.12	EC50 (biomass)	2,591.7	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 2 mg/L)	7.87-8.05	EC50 (biomass)	774.2	European Al Association 2009; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 2 mg/L)	6.19-6.23	EC50 (growth rate)	1,181.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 2 mg/L)	6.96-7.05	EC50 (growth rate)	2,896.0	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 2 mg/L)	7.74-7.96	EC50 (growth rate)	4,980.9	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 2 mg/L)	6.13-6.19	EC50 (growth rate)	1,473.5	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 2 mg/L)	6.97-7.04	EC50 (growth rate)	4,332.3	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 2 mg/L)	7.82-8.04	EC50 (growth rate)	2,000.0	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 2 mg/L)	6.09-6.18	EC50 (growth rate)	2,100.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 2 mg/L)	6.94-7.12	EC50 (growth rate)	3,645.8	European Al Association 2009; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 2 mg/L)	7.87-8.05	EC50 (growth rate)	1,639.9	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 4 mg/L)	6.09-6.19	EC50 (biomass)	778.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 4 mg/L)	6.98-7.10	EC50 (biomass)	2,630.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 4 mg/L)	7.82-7.98	EC50 (biomass)	2,229.7	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 4 mg/L)	6.10-6.19	EC50 (biomass)	1,273.7	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 4 mg/L)	7.0-7.05	EC50 (biomass)	2,736.4	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 4 mg/L)	7.78-7.87	EC50 (biomass)	1,660.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 4 mg/L)	6.09-6.24	EC50 (biomass)	1,572.8	European Al Association 2009; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 4 mg/L)	7.0-7.09	EC50 (biomass)	3,546.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 4 mg/L)	7.77-7.81	EC50 (biomass)	1,521.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 4 mg/L)	6.09-6.19	EC50 (growth rate)	1,443.9	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 4 mg/L)	6.98-7.10	EC50 (growth rate)	3,845.9	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	24.3 (DOC = 4 mg/L)	7.82-7.98	EC50 (growth rate)	4,716.1	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 4 mg/L)	6.10-6.19	EC50 (growth rate)	1,890.7	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 4 mg/L)	7.0-7.05	EC50 (growth rate)	4,260.0	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	60 (DOC = 4 mg/L)	7.78-7.87	EC50 (growth rate)	2,905.0	European Al Association 2009; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 4 mg/L)	6.09-6.24	EC50 (growth rate)	2,429.3	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 4 mg/L)	7.0-7.09	EC50 (growth rate)	4,930.0	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	120 (DOC = 4 mg/L)	7.77-7.81	EC50 (growth rate)	2,556.3	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Solution aged 3 hr (DOC = 0 mg/L)	6.23-6.24	EC50 (growth)	196.2	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Solution aged 27 hr (DOC = 0 mg/L)	6.12-6.23	EC50 (growth)	182.7	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Solution aged 3 hr (DOC = 0 mg/L)	7.93-8.06	EC50 (growth)	1,762.4	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Solution aged 27 hr (DOC = 0 mg/L)	7.93-8.23	EC50 (growth)	1,328.0	European Al Association 2009; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium not buffered (DOC = 0 mg/L)	7.80-8.21	EC50 (growth rate)	1,282.1	European Al Association 2010; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium HEPES buffered (DOC = 0 mg/L)	8.05-8.12	EC50 (growth rate)	1,351.8	European Al Association 2010; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium HEPES buffered (DOC = 0 mg/L)	7.99-8.08	EC50 (growth rate)	1,476.6	European Al Association 2010; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium HEPES buffered (DOC = 0 mg/L)	7.65-7.70	EC50 (growth rate)	1,417.9	European Al Association 2010; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium not buffered (DOC = 0 mg/L)	7.80-8.21	EC50 (biomass)	626.6	European Al Association 2010; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium HEPES buffered (DOC = 0 mg/L)	8.05-8.12	EC50 (biomass)	851.4	European Al Association 2010; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium HEPES buffered (DOC = 0 mg/L)	7.99-8.08	EC50 (biomass)	717.9	European Al Association 2010; Gensemer et al. 2018	Duration
Green alga, *Pseudokirchneriella subcapitata*	Aluminum nitrate	72 hr	Medium HEPES buffered (DOC = 0 mg/L)	7.65-7.70	EC50 (biomass)	563.3	European Al Association 2010; Gensemer et al. 2018	Duration
Red alga, *Cyanidium caldarium*	Aluminum chloride	5-10 d	-	2	Reduced growth rate by 42%	5,396,000	Yoshimura et al. 1999	Lack of exposure details; pH too low
Protozoa, *Euglena gracilis*	Aluminum chloride	10 min	-	6.0-7.0	Some survival	111,800	Ruthven and Cairns 1973	Single-cell organism

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Protozoa (1 wk), *Euglena gracilis*	Aluminum chloride	7 d	-	-	Growth	10,000-15,000 (NOEC-LOEC)	Danilov and Ekelund 2002	Single-cell organism
Protozoa, *Chilomonas paramecium*	Aluminum chloride	10 min	-	5.5-7.4	Some survival	110	Ruthven and Cairns 1973	Single-cell organism
Protozoa, *Microregma heterostoma*	Aluminum chloride	28 hr	-	7.5-7.8	Incipient inhibition	12,000	Bringmann and Kuhn 1959a	Single-cell organism
Protozoa, *Peranema trichoporum*	Aluminum chloride	10 min	-	5.5-6.5	Some survival	62,600	Ruthven and Cairns 1973	Single-cell organism
Protozoa, *Tetrahymena pyriformis*	Aluminum chloride	10 min	-	5.5-6.5	Some survival	100	Ruthven and Cairns 1973	Single-cell organism
Protozoa, *Tetrahymena pyriformis*	Aluminum chloride	96 hr	-	6.5	IC50 (growth)	15,000	Sauvant et al. 2000	Single-cell organism
Protozoa, *Tetrahymena pyriformis*	Aluminum sulfate	96 hr	-	6.5	IC50 (growth)	10,000	Sauvant et al. 2000	Single-cell organism
Protozoa, *Tetrahymena pyriformis*	Aluminum nitrate	96 hr	-	6.5	IC50 (growth)	14,000	Sauvant et al. 2000	Single-cell organism
Rotifer (0-2 hr), *Brachionus calyciflorus*	Aluminum chloride	24 hr	90 (80-100)	7.5	LC50	>3,000	Snell et al. 1991	Lack of exposure details and effects
Nematode (3-4 d, adult), *Caenorhabditis elegans*	Aluminum nitrate	-	-	-	LC50	1,800	Williams and Dusenbery 1990	Test species fed
Nematode, *Caenorhabditis elegans*	Aluminum nitrate	24 hr	-	4.5-6.5	LC50	49,000	Dhawan et al. 2000	Duration; test species fed
Nematode, *Caenorhabditis elegans*	Aluminum nitrate	24 hr	-	4.5-6.5-	EC50 (movement)	3,000	Dhawan et al. 2000	Duration; test species fed
Nematode, *Caenorhabditis elegans*	Aluminum chloride	48 hr	-	-	LC50	18,150	Chu and Chow 2002	Duration
Nematode (adult), *Caenorhabditis elegans*	Aluminum chloride	4 hr	-	-	EC50 (rate of movement)	1,241	Anderson et al. 2004	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Tubificid worm, *Tubifex tubifex*	Aluminum ammonium sulfate	96 hr	245	7.6	EC50 (death and immobility)	50,230	Khangarot 1991	Inappropriate form of toxicant
Planarian (adult), *Dugesia tigrina*	Aluminum chloride	48 hr	47.4	7.48	Mortality	>16,600 (NOEC)	Brooke 1985	Duration
Planarian, *Dugesia tigrina*	-	48 hr	~47.42	7.48	LC50	>23,200	Lange 1985	Duration
Brown hydra, *Hydra oligactis*	Aluminum sulfate	72 hr	-	-	86% mortality	475,000	Kovacevic et al. 2007	Duration
Brown hydra, *Hydra oligactis*	Aluminum sulfate	72 hr	-	-	Tail growth	250,000 (LOEC)	Kovacevic et al. 2007	Duration; atypical endpoint
Green hydra, *Hydra viridissima*	Aluminum sulfate	72 hr	-	-	LC50	475,000-480,000	Kovacevic et al. 2007	Duration
Green hydra, *Hydra viridissima*	Aluminum sulfate	72 hr	-	-	Tail growth	250,000-475,000 (NOEC-LOEC)	Kovacevic et al. 2007	Duration; atypical endpoint
Green hydra, *Hydra viridissima*	Aluminum nitrate	7 d	1.0 (DOC = 1 mg/L)	5.0	IC50 (population growth rate)	56	Trenfield et al. 2012	Duration
Green hydra, *Hydra viridissima*	Aluminum nitrate	7 d	1.0 (DOC = 2 mg/L)	5.0	IC50 (population growth rate)	90	Trenfield et al. 2012	Duration
Green hydra, *Hydra viridissima*	Aluminum nitrate	7 d	4.1 (DOC = 1 mg/L)	5.0	IC50 (population growth rate)	152	Trenfield et al. 2012	Duration
Green hydra, *Hydra viridissima*	Aluminum nitrate	7 d	4.1 (DOC = 2 mg/L)	5.0	IC50 (population growth rate)	166	Trenfield et al. 2012	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Snail, *Amnicola limosa*	Aluminum	96 hr	15.3	3.5	LC50	>1,000	Mackie 1989	pH too low
Snail, *Amnicola limosa*	Aluminum	96 hr	15.3	4.0	LC50	>400	Mackie 1989	pH too low
Snail, *Amnicola limosa*	Aluminum	96 hr	15.3	4.5	LC50	>400	Mackie 1989	pH too low
Snail (adult, 3.5-5.6 g), *Lymnaea stagnalis*	Aluminum nitrate	30 d	-	7.0	Increase in number of granules	300	Elangovan et al. 2000	Unmeasured chronic exposure
Snail (Adult, 3.5-5.6 g), *Lymnaea stagnalis*	Aluminum nitrate	30 d	~74.0	7.0	BCF = 4,500 (whole soft tissue)	234	Elangovan et al. 1997	Steady state not reached
Snail (Adult, 3.5-5.6 g), *Lymnaea stagnalis*	Aluminum nitrate	30 d	~74.0	7.0	BCF = 15,000 (whole soft tissue)	285	Elangovan et al. 1997	Steady state not reached
Snail (25-35 mm), *Lymnaea stagnalis*	Aluminum nitrate	30 d	-	7.3	BCF = 444 (digestive gland)	500	Desouky et al. 2003	Steady state not reached
Zebra mussel (veliger larvae, 135-157 µm), *Dreissena polymorpha*	Aluminum sulfate	24 hr	137.1	7.42-7.48	LC50	130,500	Mackie and Kilgour 1995	Duration
Pea cockle, *Pisidium casertanum*	-	96 hr	15.3	3.5	LC50	>1,000	Mackie 1989	pH too low
Pea cockle, *Pisidium casertanum*	-	96 hr	15.3	4.0	LC50	>400	Mackie 1989	pH too low
Pea cockle, *Pisidium casertanum*	-	96 hr	15.3	4.5	LC50	>400	Mackie 1989	pH too low
Ridged-beak peaclam, *Pisidium compressum*	-	96 hr	15.3	3.5	LC50	>1,000	Mackie 1989	pH too low
Ridged-beak peaclam, *Pisidium compressum*	-	96 hr	15.3	4.0	LC50	>400	Mackie 1989	pH too low
Ridged-beak peaclam, *Pisidium compressum*	-	96 hr	15.3	4.5	LC50	>400	Mackie 1989	pH too low

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Cladoceran (<24 hr), *Ceriodaphnia sp.*	Aluminum chloride	8 d	47.4	7.68	LC50	8,600	Call et al. 1984	Duration
Cladoceran (<24 hr), *Ceriodaphnia sp.*	Aluminum chloride	48 hr	47.4	7.68	LC50	3,690	Call et al. 1984	Species not defined; other data available for the genus
Cladoceran (<24 hr), *Ceriodaphnia sp.*	Aluminum chloride	48 hr	47.4	7.36	LC50	2,300 (aged solution)	Call et al. 1984	Species not defined; other data available for the genus
Cladoceran (<24 hr), *Ceriodaphnia sp.*	Aluminum chloride	LC (3 broods)	47.4	7.68	Reproduction	4,900-12,100 (NOEC-LOEC)	Call et al. 1984	Species not defined; other data available for the genus
Cladoceran, *Ceriodaphnia dubia*	Aluminum chloride	LC (3 broods)	90 (80-100)	-	IC25 (reproduction)	566	Zuiderveen and Birge 1997	Unmeasured chronic exposure
Cladoceran, *Ceriodaphnia dubia*	Aluminum chloride	LC (3 broods)	90 (80-100)	-	IC25 (reproduction)	641	Zuiderveen and Birge 1997	Unmeasured chronic exposure
Cladoceran (<24 hr), *Ceriodaphnia dubia*	Aluminum nitrate	LC (3 broods)	10.6 Solution not filtered (DOC = 0 mg/L)	7.74-7.90	Reproduction - # of juveniles	10.0-100.0 (NOEC-LOEC)	European Al Association 2009	Unmeasured chronic exposure
Cladoceran (<24 hr), *Ceriodaphnia dubia*	Aluminum nitrate	LC (3 broods)	10.6 Solution filtered (DOC = 0 mg/L)	7.79-7.91	Reproduction - # of juveniles	500.0-1,000.0 (NOEC-LOEC)	European Al Association 2009	Unmeasured chronic exposure
Cladoceran (<24 hr), *Ceriodaphnia dubia*	Aluminum nitrate	LC (3 broods)	10.6 Solution not filtered (DOC = 0 mg/L)	6.62-7.03	Reproduction - # of juveniles	100.0-1,000.0 (NOEC-LOEC)	European Al Association 2009	Unmeasured chronic exposure
Cladoceran (<24 hr), *Ceriodaphnia dubia*	Aluminum nitrate	LC (3 broods)	10.6 Solution filtered (DOC = 0 mg/L)	6.66-7.04	Reproduction - # of juveniles	100.0-1,000.0 (NOEC-LOEC)	European Al Association 2009	Unmeasured chronic exposure

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Cladoceran (mature), *Daphnia catawba*	Aluminum chloride	72 hr	8.07	6.5	Reduced survival	1,020	Havas and Likens 1985b	Duration
Cladoceran (<8 hr), *Daphnia magna*	Aluminum sulfate	16 hr	-	-	Incipient immobilization	10,717	Anderson 1944	Duration
Cladoceran (<8 hr), *Daphnia magna*	Potassium aluminum sulfate	16 hr	-	-	Incipient immobilization	15,677	Anderson 1944	Duration, inappropriate form of toxicant
Cladoceran, *Daphnia magna*	Aluminum chloride	48 hr	-	7.5	Toxic effect	1,000,000	Bringmann and Kuhn 1959a	Endpoint not clearly defined
Cladoceran (≥12 hr), *Daphnia magna*	Aluminum chloride	21 d	45.3	7.74	EC16 (reduced reproduction)	320	Biesinger and Christensen 1972	Unmeasured chronic exposure
Cladoceran (≥12 hr), *Daphnia magna*	Aluminum chloride	21 d	45.3	7.74	EC50 (reduced reproduction)	680	Biesinger and Christensen 1972	Unmeasured chronic exposure
Cladoceran (≥12 hr), *Daphnia magna*	Aluminum chloride	21 d	45.3	7.74	LC50	1,400	Biesinger and Christensen 1972	Unmeasured chronic exposure
Cladoceran, *Daphnia magna*	Sodium aluminate	96 hr	27	7	Mortality	>40,000	Peterson et al. 1974	LC50 or EC50 endpoint not defined
Cladoceran (≥12 hr), *Daphnia magna*	Aluminum sulfate	28 d	220	8.3	Reproduction	4,260 (NOEC)	Kimball 1978	Control survival (70%)
Cladoceran (≥12 hr), *Daphnia magna*	Aluminum sulfate	28 d	220	8.3	Survival	540-1,020 (NOEC-LOEC)	Kimball 1978	Control survival (70%)
Cladoceran (0-24 hr), *Daphnia magna*	-	28 d	-	-	Survival and reproduction	1,890-4,260 (NOEC-LOEC)	Stephan 1978	Author reported that the results are considered questionable for one reason or another [not provided]

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Cladoceran (14 d), *Daphnia magna*	-	7 d	-	-	Survival and reproduction	3,300-8,400 (NOEC-LOEC)	Stephan 1978	Author reported that the results are considered questionable for one reason or another [not provided]
Cladoceran, *Daphnia magna*	-	28 d	-	-	LC50	38,000	Stephan 1978	Author reported that the results are considered questionable for one reason or another [not provided]
Cladoceran, *Daphnia magna*	Aluminum chloride	48 hr	45.4	7.61	EC50	>25,300	Brooke 1985	No dose response observed
Cladoceran, *Daphnia magna*	Aluminum chloride	48 hr	8.26	6.5	Mortality	320	Havas 1985	Dilution water is lake water, atypical endpoint
Cladoceran, *Daphnia magna*	Aluminum chloride	24 hr	8.26	6.5	BCF = 18,000	20	Havas 1985	Duration, lack of exposure details; dilution water is lake water
Cladoceran, *Daphnia magna*	Aluminum chloride	24 hr	8.26	6.5	BCF = 9,600	320	Havas 1985	Duration, lack of exposure details; dilution water is lake water
Cladoceran, *Daphnia magna*	Aluminum chloride	24 hr	8.26	6.5	BCF = 11,000	1,020	Havas 1985	Duration, lack of exposure details; dilution water is lake water

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Cladoceran, *Daphnia magna*	Aluminum chloride	24 hr	33.35	6.5	BCF = 18,000	20	Havas 1985	Duration, lack of exposure details; dilution water is lake water
Cladoceran, *Daphnia magna*	Aluminum chloride	24 hr	33.35	6.5	BCF = 14,700	1,020	Havas 1985	Duration, lack of exposure details; dilution water is lake water
Cladoceran, *Daphnia magna*	Aluminum chloride	48 hr	-	6.5	Loss of sodium	1,020	Havas and Likens 1985a	Dilution water is lake water, atypical endpoint
Cladoceran, *Daphnia magna*	Aluminum ammonium sulfate	48 hr	240	7.6	LC50	59,600	Khangarot and Ray 1989	Inappropriate form of toxicant
Isopod (7 mm), *Asellus aquaticus*	Aluminum sulfate	72 hr	50	6.75	LC50	4,370	Martin and Holdich 1986	Duration
Amphipod, *Gammarus pseudolimnaeus*	Aluminum chloride	96 hr	47.4	7.53	LC50	22,000	Call et al. 1984	Test species fed
Amphipod, *Hyalella azteca*	-	96 hr	15.3	5.0	LC50	>1,000	Mackie 1989	Not enough information in the paper to determine is acceptable test conditions are met
Amphipod, *Hyalella azteca*	-	96 hr	15.3	5.5	LC50	>400	Mackie 1989	Not enough information in the paper to determine is acceptable test conditions are met

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Amphipod, *Hyalella azteca*	-	96 hr	15.3	6.0	LC50	>400	Mackie 1989	Not enough information in the paper to determine is acceptable test conditions are met
Amphipod (1-11 d), *Hyalella azteca*	-	7 d	18	7.39-8.27	LC50	89	Borgmann et al. 2005	Duration, control mortality (≥80 %)
Amphipod (1-11 d), *Hyalella azteca*	-	7 d	-	8.21-8.46	LC50	>3,150	Borgmann et al. 2005	Duration, control mortality (≥80 %)
Crayfish (80-160 cm), *Pacifastacus leniusculus*	Aluminum nitrate	20 d	-	-	BCF = 3.44 (flexor muscle)	436	Alexopoulos et al. 2003	More accumulation in the controls than exposure
Crayfish (80-160 cm), *Pacifastacus leniusculus*	Aluminum nitrate	20 d	-	-	BCF = 527.5 (gill content)	436	Alexopoulos et al. 2003	Gill content not whole body
Crayfish (larvae), *Procambarus clarkii*	-	30 min	-	-	Oxygen consumption	>100,000 (NOEC)	Becker and Keller 1983	Duration
Caddisfly (larva, 5th instar), *Arctopsyche ladogensis*	Aluminum sulfate	4 d	-	5.0	EC50 (frequency of abnormalities)	938-1,089	Vuori 1996	Atypical endpoint, effect range reported
Damselfly, *Enallagma sp.*	-	96 hr	15.3	3.5	LC50	>1,000	Mackie 1989	pH too low
Damselfly, *Enallagma sp.*	-	96 hr	15.3	4.0	LC50	>400	Mackie 1989	pH too low
Damselfly, *Enallagma sp.*	-	96 hr	15.3	4.5	LC50	>400	Mackie 1989	pH too low
Midge (1st instar larva, 3d), *Chironomus riparius*	Aluminum nitrate	10 d	91	6.5-6.7	Survival	4,281.8->4,281.9 (NOEC-LOEC)	OSU 2012f; Cardwell et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Midge (1st instar larva, 3d), *Chironomus riparius*	Aluminum nitrate	10 d	91	6.5-6.7	Growth-dry weight	1,100.2-2,132.7 (NOEC-LOEC)	OSU 2012f; Cardwell et al. 2018	Duration
Midge, *Paratanytarsus dissimilis*	Aluminum sulfate	55 d	17.43	6.63	Survival	800 (LOEC)	Lamb and Bailey 1981, 1983	Not a flow-through chronic exposure
Dragonfly (last instar nymph), *Libellula julia*	Aluminum chloride	96 hr	-	4	Oxygen uptake inhibition	3,000-30,000 (NOEC-LOEC)	Rockwood et al. 1990	Atypical endpoint
Golden trout (alevin), *Oncorhynchus aguabonita aguabonita*	Aluminum sulfate	7 d	4.89	5.0	Survival	97-293 (NOEC-LOEC)	DeLonay 1991; DeLonay et al. 1993	Duration
Golden trout (swim-up larvae), *Oncorhynchus aguabonita aguabonita*	Aluminum sulfate	7 d	4.89	5.0	Survival	97-293 (NOEC-LOEC)	DeLonay 1991; DeLonay et al. 1993	Duration
Cutthroat trout (egg/embryo), *Oncorhynchus clarkii*	-	7 d	42.5	5	Survival	300->300 (NOEC-LOEC)	Woodward et al. 1989	Duration
Cutthroat trout (egg/embryo), *Oncorhynchus clarkii*	-	7 d	42.5	5	Growth	300->300 (NOEC-LOEC)	Woodward et al. 1989	Duration
Cutthroat trout (alevin, 2 d post hatch), *Oncorhynchus clarkii*	-	7 d	42.5	5	Survival	50-100 (NOEC-LOEC)	Woodward et al. 1989	Duration
Cutthroat trout (alevin/larvae), *Oncorhynchus clarkii*	-	7 d	42.5	5	Growth	50->50 (NOEC-LOEC)	Woodward et al. 1989	Duration
Cutthroat trout (swim-up larvae), *Oncorhynchus clarkii*	-	7 d	42.5	5	Survival	<50-50 (NOEC-LOEC)	Woodward et al. 1989	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Rainbow trout (fingerling), *Oncorhynchus mykiss*	Aluminum chloride	-	28.3	8.48	LT50=7.46 d	5,140	Freeman and Everhart 1971	Atypical endpoint, test species fed
Rainbow trout (fingerling), *Oncorhynchus mykiss*	Aluminum chloride	-	28.3	8.99	LT50=2.96 d	5,200	Freeman and Everhart 1971	Atypical endpoint, test species fed
Rainbow trout (fingerling), *Oncorhynchus mykiss*	Aluminum chloride	-	46.8	8.02	LT50=31.96 d	5,230	Freeman and Everhart 1971	Atypical endpoint, test species fed
Rainbow trout (fingerling), *Oncorhynchus mykiss*	Aluminum chloride	-	56.6	6.8	LT50=38.90 d	5,140	Freeman and Everhart 1971	Atypical endpoint, test species fed
Rainbow trout (fingerling), *Oncorhynchus mykiss*	Aluminum chloride	-	56.6	6.52	LT50=43.90 d	513	Freeman and Everhart 1971	Atypical endpoint, test species fed
Rainbow trout (embryo), *Oncorhynchus mykiss*	Aluminum chloride	Fert. to hatch	-	7.0-9.0	No reduced fertility	5,200	Freeman and Everhart 1971	Lack of exposure details
Rainbow trout (embryo/larvae), *Oncorhynchus mykiss*	Aluminum chloride	28 d	104	7.4	EC50 (death and deformity)	560	Birge 1978; Birge et al. 1978	Duration
Rainbow trout (juvenile), *Oncorhynchus mykiss*	Aluminum sulfate	10 d	25	7	0% dead	200,000	Hunter et al. 1980	Duration, test species fed
Rainbow trout (juvenile), *Oncorhynchus mykiss*	Aluminum sulfate	96 hr	25	8	40% dead	50,000	Hunter et al. 1980	Lack of exposure details
Rainbow trout (juvenile), *Oncorhynchus mykiss*	Aluminum sulfate	42 hr	25	8.5	100% dead	50,000	Hunter et al. 1980	Duration; lack of exposure details
Rainbow trout (juvenile), *Oncorhynchus mykiss*	Aluminum sulfate	42 hr	25	9	100% dead	50,000	Hunter et al. 1980	Duration; lack of exposure details
Rainbow trout, *Oncorhynchus mykiss*	-		-	5.0	LC50	160	Holtze 1983	pH too low
Rainbow trout, *Oncorhynchus mykiss*	-		-	4.5	LC50	120	Holtze 1983	pH too low

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Rainbow trout (embryo/larvae), *Oncorhynchus mykiss*	Aluminum sulfate	8 d	14.3	6.5	No effect	1,000	Holtze 1983	Duration, lack of exposure details
Rainbow trout (embryo/larvae), *Oncorhynchus mykiss*	Aluminum sulfate	8 d	14.3	7.2	No effect	1,000	Holtze 1983	Duration, lack of exposure details
Rainbow trout (eyed embryo), *Oncorhynchus mykiss*	Aluminum sulfate	8 d	14.3	6.5	14.2% dead	1,000	Holtze 1983	Duration, lack of exposure details
Rainbow trout (eyed embryo), *Oncorhynchus mykiss*	Aluminum sulfate	8 d	14.3	7.2	14.2% dead	1,000	Holtze 1983	Duration, lack of exposure details
Rainbow trout (juvenile, 5-8 cm), *Oncorhynchus mykiss*	Aluminum sulfate	24 hr	-	6	Opercula rate	200-500 (NOEC-LOEC)	Ogilvie and Stechey 1983	Duration, atypical endpoint
Rainbow trout (juvenile, 5-8 cm), *Oncorhynchus mykiss*	Aluminum sulfate	24 hr	-	6	Cough frequency	100-200 (NOEC-LOEC)	Ogilvie and Stechey 1983	Duration, atypical endpoint
Rainbow trout (3.5 g), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	11.2	5.09-5.31	LC50	175	Orr et al. 1986	Duration
Rainbow trout (alevin, 23-26 dph), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	10.3	5.8	LC50	>1,050	Hickie et al. 1993	Duration
Rainbow trout (alevin, 16-19 dph), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	10.3	4.9	LC50	88	Hickie et al. 1993	Duration, pH too low
Rainbow trout (alevin, 23-26 dph), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	10.3	4.9	LC50	91	Hickie et al. 1993	Duration, pH too low
Rainbow trout (92-220 g), *Oncorhynchus mykiss*	-	1 hr	-	5.4	Gill content (50 µg/g)	954	Handy and Eddy 1989	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	Aluminum chloride	16 d	20.3	8.3	LC50	1,940	Gundersen et al. 1994	Duration
Rainbow trout (juvenile, 1-3 g), *Oncorhynchus mykiss*	Aluminum chloride	16 d	103.0	8.3	LC50	3,910	Gundersen et al. 1994	Duration
Rainbow trout (embryo), *Oncorhynchus mykiss*	Aluminum chloride	7-12 d	100	7.0-7.8	LC50	560	Birge et al. 2000	Duration
Chinook salmon (juvenile), *Oncorhynchus tshawytscha*	Sodium aluminate	96 hr	28.0	7.00	LC50	>40,000	Peterson et al. 1974	Inappropriate form of toxicant
Atlantic salmon (eggs), *Salmo salar*	Aluminum sulfate	60 d	13.5	5.5	RNA/DNA content	33-264 (NOEC-LOEC)	McKee et al. 1989	Atypical endpoint
Atlantic salmon (>1 yr, 5.9 g), *Salmo salar*	-	7 d	10.4	4.5	LC50	88	Wilkinson et al. 1990	Duration
Atlantic salmon (eggs), *Salmo salar*	Aluminum sulfate	60 d	12.8	5.5	Time to hatch	>264 (NOEC)	Buckler et al. 1995	Atypical endpoint
Atlantic salmon (larva), *Salmo salar*	Aluminum sulfate	60 d	12.8	5.5	Behavior-swimming & feeding activity	<33 (NOEC)	Buckler et al. 1995	Atypical endpoint
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	4.47	LC50	259	Roy and Campbell 1995	Duration
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	4.42	LC50	283	Roy and Campbell 1995	Duration
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	4.83	LC50	121	Roy and Campbell 1995	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	5.26	LC50	54	Roy and Campbell 1995	Duration
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	5.24	LC50	51	Roy and Campbell 1995	Duration
Atlantic salmon (juvenile, 6.8 g), *Salmo salar*	Aluminum sulfate	96 hr	10.6	4.86	LC50	75.54	Roy and Campbell 1995	pH too low
Atlantic salmon (juvenile, 1.8 g), *Salmo salar*	Aluminum sulfate	96 hr	10.6	4.99	LC50	79.60	Roy and Campbell 1997	pH too low
Atlantic salmon (juvenile, 1.8 g), *Salmo salar*	Aluminum sulfate	96 hr	10.6	4.96	LC50	124.1	Roy and Campbell 1997	pH too low
Brook trout (alevins, 23.6 mm, 13.4 mg), *Salvelinus fontinalis*	-	15 min	7.2	6.9	Avoidance	389	Gunn and Noakes 1986	Duration, atypical endpoint
Brook trout (juvenile), *Salvelinus fontinalis*	Aluminum hydroxide	24 d	8-10	4.4	Survival	<200-200 (NOEC-LOEC)	Siddens et al. 1986	Duration
Brook trout (juvenile), *Salvelinus fontinalis*	Aluminum hydroxide	24 d	8-10	4.9	Survival	<200-200 (NOEC-LOEC)	Siddens et al. 1986	Duration
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 8 mg/L	5.2	100% survival	54	Mount 1987	Unmeasured chronic exposure
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 8 mg/L	5.2	93% survival	162	Mount 1987	Unmeasured chronic exposure
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 8 mg/L	4.8	100% survival	162	Mount 1987	Unmeasured chronic exposure
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 8 mg/L	4.8	50% survival	486	Mount 1987	Unmeasured chronic exposure
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 0.5 mg/L	5.2	93% survival	54	Mount 1987	Unmeasured chronic exposure

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 0.5 mg/L	5.2	86% survival	162	Mount 1987	Unmeasured chronic exposure
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 0.5 mg/L	4.8	86% survival	162	Mount 1987	Unmeasured chronic exposure
Brook trout (1.5 yr), *Salvelinus fontinalis*	-	147 d	Ca = 0.5 mg/L	4.8	36% survival	486	Mount 1987	Unmeasured chronic exposure
Brook trout (eggs), *Salvelinus fontinalis*	Aluminum sulfate	60 d	12.5	5.5	Strike frequency	142-292 (NOEC-LOEC)	Cleveland et al. 1989	Atypical endpoint
Brook trout (eggs), *Salvelinus fontinalis*	Aluminum sulfate	60 d	12.5	6.5	Strike frequency	350->350 (NOEC-LOEC)	Cleveland et al. 1989	Atypical endpoint
Brook trout (1 yr), *Salvelinus fontinalis*	Aluminum chloride	28 d	250	4.4	Survival	131-332 (NOEC-LOEC)	Ingersoll et al. 1990a	Duration; pH too low
Goldfish (60-90 mm), *Carassius auratus*	Aluminum potassium sulfate	96 hr	-	6.8	Reduced survival time	5,700	Ellis 1937	Atypical endpoint; no LC50 reported
Goldfish (eggs), *Carassius auratus*	Aluminum chloride	7 d	195	7.4	EC50 (death and deformity)	150	Birge 1978	Duration
Goldfish (embryo), *Carassius auratus*	Aluminum chloride	7-12 d	100	7.0-7.8	LC50	330	Birge et al. 2000	Duration
Common carp (95 g), *Cyprinus carpio*	Aluminum sulfate	4 hr	-	5.2	Ca 2+ flux	30-100 (NOEC-LOEC)	Verbost et al. 1992	Duration, atypical endpoint
Rio Grande silvery minnow (larva, 3-5 dph), *Hybognathus amarus*	Aluminum chloride	96 hr	140	8.1	EC50 (death and immobility)	>59,100	Buhl 2002	Atypical endpoint
Fathead minnow (juvenile), *Pimephales promelas*	Aluminum sulfate	8 d	220	7.3	LC50	22,400	Kimball 1978	Duration, test species fed

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Fathead minnow (juvenile), *Pimephales promelas*	Aluminum sulfate	96 hr	220	7.34	LC50	35,000	Kimball 1978	Test species fed
Fathead minnow (adult), *Pimephales promelas*	Aluminum chloride	-	-	-	50% reduction in AChE	18,000	Olson and Christensen 1980	Duration unknown, atypical endpoint
Fathead minnow (juvenile, 11 mm), *Pimephales promelas*	Aluminum chloride	96 hr	21.6	5.5	LC50	>50	Palmer et al. 1989	Measured dissolved total Al greater than (unmeasured) nominal total Al.
Fathead minnow (larvae, <24 hr), *Pimephales promelas*	Aluminum chloride	7 d	46	7.5	Growth (weight)	400-740 (NOEC-LOEC)	ENSR 1992a	Duration
Fathead minnow (larvae, <24 hr), *Pimephales promelas*	Aluminum chloride	7 d	194	8.2	Growth (weight)	630-700 (NOEC-LOEC)	ENSR 1992a	Duration
Fathead minnow (≤7 d), *Pimephales promelas*	Aluminum chloride	96 hr	25 (24-26)	8.05 (7.2-8.9)	LC50	1,160	ENSR 1992c	Test species fed
Fathead minnow (≤7 d), *Pimephales promelas*	Aluminum chloride	96 hr	44 (42-46)	8.1 (7.5-8.7)	LC50	8,180	ENSR 1992c	Test species fed
Fathead minnow (≤7 d), *Pimephales promelas*	Aluminum chloride	96 hr	97 (96-98)	8.05 (7.6-8.5)	LC50	20,300	ENSR 1992c	Test species fed
Fathead minnow (≤7 d), *Pimephales promelas*	Aluminum chloride	96 hr	193 (192-194)	8.2 (7.8-8.6)	LC50	44,800	ENSR 1992c	Test species fed
Fathead minnow (larva, 4-6 dph), *Pimephales promelas*	Aluminum chloride	96 hr	140	8.1	EC50 (death and immobility)	>59,100	Buhl 2002	Atypical endpoint
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	12 (DOC = <0.08 mg/L)	6.0	EC20 (mean dry biomass)	127.2	OSU 2012a; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	12 (DOC = 0.92 mg/L)	6.1	EC20 (mean dry biomass)	425.7	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	12 (DOC = 1.73 mg/L)	6.1	EC20 (mean dry biomass)	632.8	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	16 (DOC = 3.35 mg/L)	6.0	EC20 (mean dry biomass)	828.8	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	24 (DOC = 0.19 mg/L)	6.1	EC20 (mean dry biomass)	135.8	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	60 (DOC = 0.22 mg/L)	6.0	EC20 (mean dry biomass)	314.3	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	60 (DOC = 0.86 mg/L)	6.1	EC20 (mean dry biomass)	633.9	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	56 (DOC = 1.74 mg/L)	6.0	EC20 (mean dry biomass)	1,325.8	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	60 (DOC = 3.51 mg/L)	6.0	EC20 (mean dry biomass)	2,523	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	116 (DOC = 0.088 mg/L)	6.1	EC20 (mean dry biomass)	624.1	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	116 (DOC = 0.88 mg/L)	6.1	EC20 (mean dry biomass)	773.4	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	108 (DOC = 1.56 mg/L)	6.0	EC20 (mean dry biomass)	1,493.7	OSU 2012a; Gensemer et al. 2018	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	112 (DOC = 3.27 mg/L)	6.0	EC20 (mean dry biomass)	2,938	OSU 2012a; Gensemer et al. 2018	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	134 (DOC = 7.0 mg/L)	6.0	EC20 (mean dry biomass)	4,618	OSU 2018b	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	131 (DOC = 11.5 mg/L)	6.0	EC20 (mean dry biomass)	9,511	OSU 2018b	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	422 (DOC = 1.1 mg/L)	6.8	EC20 (mean dry biomass)	2,969	OSU 2018b	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	135 (DOC = 7.2 mg/L)	7.0	EC20 (mean dry biomass)	8,047	OSU 2018b	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	125 (DOC = 11.6 mg/L)	7.0	EC20 (mean dry biomass)	12,542	OSU 2018b	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	288 (DOC = 1.1 mg/L)	8.1	EC20 (mean dry biomass)	5,634	OSU 2018b	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	396 (DOC = 1.6 mg/L)	8.1	EC20 (mean dry biomass)	13,274	OSU 2018b	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	49 (DOC = 0.8 mg/L)	6.1	EC20 (mean dry biomass)	885	OSU 2018d	Duration
Fathead minnow (larva, <24 hr), *Pimephales promelas*	Aluminum nitrate	7 d	94 (DOC = 1.6 mg/L)	6.0	EC20 (mean dry biomass)	1,817	OSU 2018d	Duration
Zebrafish (egg, 1 d), *Danio rerio*	Aluminum chloride	24 hr	40	5	Median day to hatch	16,400 (NOEC)	Dave 1985	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Zebrafish (egg, 1 d), *Danio rerio*	Aluminum chloride	24 hr	40	6	Median day to hatch	16,400 (NOEC)	Dave 1985	Duration
Zebrafish (egg, 1 d), *Danio rerio*	Aluminum chloride	24 hr	40	7	Median day to hatch	16,400 (NOEC)	Dave 1985	Duration
Zebrafish (egg, 1 d), *Danio rerio*	Aluminum chloride	24 hr	40	8	Median day to hatch	16,400 (NOEC)	Dave 1985	Duration
Zebrafish (egg, 1 d), *Danio rerio*	Aluminum chloride	24 hr	40	9	Median survival time	<500-500 (NOEC-LOEC)	Dave 1985	Duration
Zebrafish (larva, 7-8 d), *Danio rerio*	Aluminum chloride	48 hr	40	7	LC50	106,000	Dave 1985	Duration
Zebrafish (larva, 7-8 d), *Danio rerio*	Aluminum chloride	48 hr	40	7.4-7.9	LC50	80,000	Dave 1985	Duration
Zebrafish (3 cm, 5g), *Danio rerio*	Aluminum chloride	4 d	-	-	LC50	56,920	Anandhan and Hemalatha 2009	Lack of exposure details (assumed fed too)
Zebrafish (adult, female), *Danio rerio*	Aluminum chloride	48 hr	142	8.2	LC50	>7,920	Griffitt et al. 2008	Duration
Zebrafish (fry, <24 hr), *Danio rerio*	Aluminum chloride	48 hr	142	8.2	LC50	>10,000	Griffitt et al. 2008	Duration
Zebrafish (adult, female), *Danio rerio*	Aluminum chloride	48 hr	142	6.8	100% mortality	12,500	Griffitt et al. 2011	Duration
Zebrafish (adult, female), *Danio rerio*	Aluminum chloride	48 hr	142	6.8	No mortality	5,000	Griffitt et al. 2011	Duration
Smallmouth bass (eyed egg), *Micropterus dolomieu*	F, M	11 d	15.7	4.8	Survival	100-200 (NOEC-LOEC)	Holtze and Hutchinson 1989	Duration; pH too low
Largemouth bass (juvenile), *Micropterus salmoides*	Aluminum sulfate	7 d	64-80	6.6-7.4	0% dead	50,000	Sanborn 1945	Duration
Largemouth bass (eggs/fry), *Micropterus salmoides*	Aluminum chloride	8 d	93-105	7.2-7.8	LC50	170	Birge et al. 1978	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Largemouth bass (embryo), *Micropterus salmoides*	Aluminum chloride	7-12 d	100	7.0-7.8	LC50	190	Birge et al. 2000	Duration
Striped bass (160 d), *Morone saxatilis*	Aluminum sulfate	7 d	12.5-12.8	7.2	Survival	174-348.8 (NOEC-LOEC)	Buckler et al. Manuscript, 1987	Duration
Striped bass (160 d), *Morone saxatilis*	Aluminum sulfate	7 d	12.5-12.8	6.5	Survival	87.2-174.4 (NOEC-LOEC)	Buckler et al. Manuscript, 1987	Duration
Striped bass (160 d), *Morone saxatilis*	Aluminum sulfate	7 d	12.5-12.8	6	Survival	21.8-43.6 (NOEC-LOEC)	Buckler et al. Manuscript, 1987	Duration
Pike (yolk-sac fry), *Esox lucius*	Aluminum sulfate	10 d	18	4	LC50	~160	Vuorinen et al. 1993	Duration, pH too low
Pike (yolk-sac fry), *Esox lucius*	Aluminum sulfate	10 d	18	4.25	LC50	~325	Vuorinen et al. 1993	Duration, pH too low
Pike (yolk-sac fry), *Esox lucius*	Aluminum sulfate	10 d	18	4.5	LC50	~600	Vuorinen et al. 1993	Duration, pH too low
Pike (yolk-sac fry), *Esox lucius*	Aluminum sulfate	10 d	18	4.75	LC50	~1,000	Vuorinen et al. 1993	Duration, pH too low
White sucker (eyed egg), *Catostomus commersoni*	-	96 hr	15.7	4.8	Survival	100-200 (NOEC-LOEC)	Holtze and Hutchinson 1989	Atypical endpoint; pH too low
Lake whitefish (cleavage egg), *Coregonus clupeaformis*	-	12 d	15.7	4.8	Survival	300 (NOEC)	Holtze and Hutchinson 1989	Duration; pH too low
Bullfrog (embryo), *Rana catesbeiana*	Aluminum chloride	10-12 d	100	7.0-7.8	LC50	80	Birge et al. 2000	Duration
Leopard frog (embryo, 3 hr, Gosner stage 3-4), *Rana pipiens*	Aluminum chloride	4-5 d	2.0	4.6	LC50	811	Freda and McDonald 1990	pH too low

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Leopard frog (embryo, 3 hr, Gosner stage 3-4), *Rana pipiens*	Aluminum chloride	4-5 d	2.0	4.8	LC50	403	Freda and McDonald 1990	pH too low
Leopard frog (tadpole, Gosner stage 20), *Rana pipiens*	Aluminum chloride	96 hr	2.0	4.4	LC50	>250	Freda and McDonald 1990	pH too low
Leopard frog (tadpole, Gosner stage 20), *Rana pipiens*	Aluminum chloride	96 hr	2.0	4.6	LC50	>250	Freda and McDonald 1990	pH too low
Leopard frog (tadpole, 3 wk), *Rana pipiens*	Aluminum chloride	96 hr	2.0	4.2	LC50	>1,000	Freda and McDonald 1990	pH too low
Leopard frog (tadpole, 3 wk), *Rana pipiens*	Aluminum chloride	96 hr	2.0	4.4	LC50	>1,000	Freda and McDonald 1990	pH too low
Leopard frog (tadpole, 3 wk), *Rana pipiens*	Aluminum chloride	96 hr	2.0	4.6	LC50	>1,000	Freda and McDonald 1990	pH too low
Leopard frog (tadpole, 3 wk), *Rana pipiens*	Aluminum chloride	96 hr	2.0	4.8	LC50	>1,000	Freda and McDonald 1990	pH too low
Leopard frog (embryos), *Rana pipiens*	Aluminum chloride	96 hr	2.0	4.8	LC50	471	Freda et al. 1990	pH too low
Leopard frog (embryo), *Rana pipiens*	Aluminum chloride	10-11 d	100	7.0-7.8	LC50	90	Birge et al. 2000	Duration
Wood frog (eggs), *Rana sylvatica*	-	24 hr	7.78	5.75	Hatch success	20->20 (NOEC-LOEC)	Clark and LaZerte 1985	Duration
Wood frog (eggs), *Rana sylvatica*	-	24 hr	7.78	4.75	Hatch success	100->100 (NOEC-LOEC)	Clark and LaZerte 1985	Duration
Wood frog (eggs), *Rana sylvatica*	-	24 hr	7.78	4.415	Hatch success	10-20 (NOEC-LOEC)	Clark and LaZerte 1985	Duration
Wood frog (eggs), *Rana sylvatica*	-	24 hr	7.78	4.14-5.75	Survival	200 (NOEC)	Clark and LaZerte 1985	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Wood frog (larva, Gosner stage 25), *Rana sylvatica*	Aluminum sulfate	43-102 d	109.9-119.5	4.68-4.70	Survival and growth	2,000 (NOEC)	Peles 2013	pH too low
Spring peeper (embryo), *Pseudacris crucifer*	Aluminum chloride	7 d	100	7.0-7.8	LC50	90	Birge et al. 2000	Duration
Green tree frog (tadpole), *Hyla cinerea*	Aluminum chloride	96 hr	1.5	5.5	Growth	<150-150 (NOEC-LOEC)	Jung and Jagoe 1995	Atypical endpoint
Green tree frog (tadpole), *Hyla cinerea*	Aluminum chloride	96 hr	1.5	4.5	Growth	<150-150 (NOEC-LOEC)	Jung and Jagoe 1995	Atypical endpoint
Green tree frog (tadpole), *Hyla cinerea*	Aluminum chloride	96 hr	1.5	4.5	LC50	277	Jung and Jagoe 1995	pH too low
American toad (eggs), *Bufo americanus*	-	24 hr	7.78	5.75	Hatch success	20->20 (NOEC-LOEC)	Clark and LaZerte 1985	Duration
American toad (eggs), *Bufo americanus*	-	24 hr	7.78	4.75	Hatch success	100->100 (NOEC-LOEC)	Clark and LaZerte 1985	Duration
American toad (eggs), *Bufo americanus*	-	24 hr	7.78	4.14	Hatch success	5-10 (NOEC-LOEC)	Clark and LaZerte 1985	Duration
American toad (eggs), *Bufo americanus*	-	24 hr	7.78	4.14	Hatch success	<10-10 (NOEC-LOEC)	Clark and LaZerte 1985	Duration
American toad (eggs), *Bufo americanus*	-	24 hr	7.78	4.14-5.75	NOEC (survival)	200	Clark and LaZerte 1985	Duration
American toad (tadpoles, Gosner stage 26), *Bufo americanus*	Aluminum chloride	96 hr	2.0	4.5	LC50	672	Freda et al. 1990	pH too low
Common toad (spawn, 0-48 hr), *Bufo bufo*	Aluminum nitrate	7 d	50	6.0	Survival	>320 (NOEC)	Gardner et al. 2002	Duration
Common toad (spawn, 0-48 hr), *Bufo bufo*	Aluminum nitrate	7 d	50	7.5	Survival	>320 (NOEC)	Gardner et al. 2002	Duration

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Fowler's toad (embryo), *Bufo fowleri*	Aluminum chloride	7 d	100	7.0-7.8	LC50	280	Birge et al. 2000	Duration
Narrow-mouthed toad (eggs), *Gastrophryne carolinensis*	Aluminum chloride	7 d	195	7.4	EC50 (death and deformity)	50	Birge 1978	Duration
Narrow-mouthed toad (eggs), *Gastrophryne carolinensis*	Aluminum chloride	7 d	100	7.0-7.8	LC50	50	Birge et al. 2000	Duration
Marbled salamander (eggs), *Ambystoma opacum*	Aluminum chloride	8 d	93-105	7.2-7.8	EC50 (death and deformity)	2,280	Birge et al. 1978	Duration
Marbled salamander (embryo), *Ambystoma opacum*	Aluminum chloride	9-10 d	100	7.0-7.8	LC50	2,280	Birge et al. 2000	Duration

Appendix I OTHER DATA ON EFFECTS OF ALUMINUM TO ESTUARINE/MARINE AQUATIC ORGANISMS

Appendix I. Other Data on Effects of Aluminum to Estuarine/Marine Aquatic Organisms

Species	Chemical	Duration	Salinity (g/kg)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Estuarine/Marine Species								
Phytoplankton, *Dunaliella tertiolecta*	Aluminum nitrate	72 hr	-	8.2	IC25 (inhibit growth)	18,160	Sacan et al. 2007	Duration
Phytoplankton, *Dunaliella tertiolecta*	Aluminum nitrate	72 hr	-	8.2	SC20 (stimulate growth)	4,660	Sacan et al. 2007	Duration
Diatom, *Nitzschia closterium*	Aluminum chloride	72 hr	-	8.2	IC50 (growth rate)	190	Harford et al. 2011	Duration
Polychaete worm, *Ctenodrilus serratus*	Aluminum chloride	21 d	-	7.6-8	Reproduction	20-40 (NOEC-LOEC)	Petrich and Reish 1979	Unmeasured chronic exposure
Sea urchin (embryo), *Paracentrotus lividus*	Aluminum sulfate	72 hr	-	-	69.7% developmental effects	539.6	Caplat et al. 2010	Difficult to determine effect concentration
Bay mussel (28.0 mm), *Mytilus edulis*	Alum (potassium)	24 hr	30	4.4-7.3	LC50	>6,400,000	Robinson and Perkins 1977	Duration
Common winkle (13.3 mm), *Littorina littorea*	Alum (potassium)	24 hr	30	4.4-7.3	LC50	>6,400,000	Robinson and Perkins 1977	Duration
European shore crab (12.6 mm), *Carcinus maenas*	Alum (potassium)	24 hr	30	4.4-7.3	LC50	2,500,000	Robinson and Perkins 1977	Duration
Hermit crab (11.4 mm), *Eupagurus bernhardus*	Alum (potassium)	24 hr	30	4.4-7.3	LC50	250,000	Robinson and Perkins 1977	Duration

Species	Chemical	Duration	Salinity (g/kg)	pH	Effect	Concentration (µg/L)	Reference	Reason Other Data
Yellow crab (embryo, 4-lobed stage), *Cancer anthonyi*	-	7 d	34	7.8	Survival	<10,000-10,000 (NOEC-LOEC)	MacDonald et al. 1988	Duration, unmeasured chronic exposure
Yellow crab (embryo, 4-lobed stage), *Cancer anthonyi*	-	7 d	34	7.8	Hatching of embryos	<10,000-10,000 (NOEC-LOEC)	MacDonald et al. 1988	Duration, unmeasured chronic exposure
Daggerblade grass shrimp (embryo, 3 d), *Palaemonetes pugio*	-	12 d	20	7.6-8.1	LC50	1,079	Rayburn and Aladdin 2003	Duration

Appendix J LIST OF ALUMINUM STUDIES NOT USED IN DOCUMENT ALONG WITH REASONS

Appendix J. List of Aluminum Studies Not Used in Document Along with Reasons

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Aarab et al.	Histopathology alterations and histochemistry measurements in mussel, *Mytilus edulis* collected offshore from an aluminum smelter industry (Norway)	2008	Bay mussel, *Mytilus edulis*	-	Not applicable; no aluminum toxicity data
Abdelhamid and El-Ayouty	Effect of catfish (*Clarias lazera*) composition of ingestion rearing water contaminated with lead or aluminum compounds	1991	Catfish, *Clarias lazera*	6 wk 50,000 0.33 corrected mortality	Not North American species; dilution water not characterized
Abdel-Latif	The influence of calcium and sodium on aluminum toxicity in Nile tilapia (*Oreochromis niloticus*)	2008	Nile tilapia, *Oreochromis niloticus*	96 hr LC50=175.9	Dilution water not characterized; lack of exposure details
Abraham et al.	Quantified elemental changes in *Aspidisca cicada* and *Vorticella convallaria* after exposure to aluminum, copper, and zinc	1997	Protozoa, *Aspidisca cicada* Protozoa, *Vorticella convallaria*	-	Mixture
Adokoh et al.	Statistical evaluation of environmental contamination, distribution and source assessment of heavy metals (aluminum, arsenic, cadmium, and mercury) in some lagoons and an estuary along the coastal belt of Ghana	2011	-	-	Survey
Ahsan et al.	Comparative proteomic study of arsenic-induced differentially expressed proteins in rice roots reveals glutathione plays a central role during As stress	2008	-	-	Not applicable; no aluminum toxicity data
Al-Aarajy and Al-Saadi	Effect of heavy metals on physiological and biochemical features of *Anabaena cylindrica*	1998	Blue-green alga, *Anabaena cylindrica*	-	Only one exposure concentration; lack of exposure details (duration not reported)
Alessa and Oliveira	Aluminum toxicity studies in *Vaucheria longicaulis* var. *macounii* (Xanthophyta, Tribophyceae). I. Effects on cytoplasmic organization	2001a	Alga, *Vaucheria longicaulis* var. m*acouni*	10 hr 2,159 growth ceased	Only one exposure concentration
Alessa and Oliveira	Aluminum toxicity studies in *Vaucheria longicaulis* var. *macounii* (Xanthophyta, Tribophyceae). II. Effects on the F-Actin array	2001b	Alga, *Vaucheria longicaulis* var. m*acouni*	-	Lack of exposure details; dilution water not characterized; only one exposure concentration
Allin and Wilson	Behavioural and metabolic effects of chronic exposure to sublethal aluminum in acidic soft water in juvenile rainbow trout (*Oncorhynchus mykiss*)	1999	Rainbow trout, *Oncorhynchus mykiss*	6 wk 29.2 Reduced appetite	Only one exposure concentration

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Allin and Wilson	Effects of pre-acclimation to aluminum on the physiology and swimming behaviour of juvenile rainbow trout (*Oncorhynchus mykiss*) during a pulsed exposure	2000	Rainbow trout, *Oncorhynchus mykiss*	-	Pulsed exposures to pollutant
Alquezar et al.	Metal accumulation in the smooth toadfish, *Tetractenos glaber*, in estuaries around Sydney, Australia	2006	Toadfish, *Tetractenos glaber*	-	Not North American species; exposed to mixture
Alstad et al.	The significance of water ionic strength on aluminum in brown trout (*Salmo trutta* L.)	2005	Brown trout, *Salmo trutta*	650 Survival time =16-34 hr	No acclimation to test water; only one exposure concentration
Amato et al.	Concentrations, sources and geochemistry of airborne participate matter at a major European airport	2010	-	-	Not applicable; no aluminum toxicity data
Amenu	A comparative study of water quality conditions between heavily urbanized and less urbanized watersheds of Los Angeles Basin	2011	-	-	Not applicable; no aluminum toxicity data
Anderson	The apparent thresholds of toxicity to *Daphnia magna* for chlorides of various metals when added to Lake Erie water	1948	Cladoceran, *Daphnia magna*	64 hr 6,700 LOEC (mortality)	Lack of exposure details; control data not reported
Andersson	Toxicity and tolerance of aluminum in vascular plants	1988	-	-	Review
Andren and Rydin	Toxicity of inorganic aluminum at spring snowmelt-in-stream bioassays with brown trout (*Salmo trutta* L.)	2012	Brown trout, *Salmo trutta*	-	Mixture; dilution water is river water
Andren et al.	Effects of pH and aluminum on embryonic and early larval stages of Swedish brown frogs Rana arvalis, R. temporaria and R. dalmatina	1988	Brown frog, *Rana arvalis* Brown frog, *Rana temporaria* Brown frog, *Rana dalmatina*	15 d NOEC (mortality) =800, 800, & <800, respectively	Not North American species
Andrews et al.	Selected metals in sediments and streams in the Oklahoma part of the Tri-State Mining District, 2000-2006	2009	-	-	Survey
Annicchiarico et al.	PCBs, PAHs and metal contamination and quality index in marine sediments of the Taranto Gulf	2011	-	-	Survey; sediment

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Appelberg	Changes in haemolymph ion concentrations of *Astacus astacus* L. and *Pacifastacus leniusclus* (Dana) after exposure to low pH and aluminium	1985	Signal crayfish, *Pacifastacus leniusculus*	14 d 250 Decrease Na^+ haemolymph concentrations	Too few organisms per treatment (4 per treatmen); only 3 exposure concentrations
Arain et al.	Total dissolved and bioavailable elements in water and sediment samples and their accumulation in *Oreochromis mossambicus* of polluted Manchar Lake	2008	Mozambique tilapia, *Oreochromis mossambicus*	-	Survey
Arenhart et al.	Involvement of ASR genes in aluminium tolerance mechanisms in rice	2013	Rice	-	Scientific name not given
Arthur D. Little Inc.	Water quality criteria data book, volume 2; Inorganic chemical pollution of freshwater	1971	-	-	Review; results of previously published papers
AScI Corp.	Aluminum water-effect ratio for the 3M Middleway plant effluent discharge, Middleway, West virginia	1994	Cladoceran, *Daphnia magna* Rainbow trout, *Oncorhynchus mykiss*	-	Mixture
AScI Corp.	Aluminum water-effect ratio for Georgia-Pacific Corporation Woodland, Maine; Pulp and paper operations discharge and St. Croix River	1996	-	-	Review; results of previously published papers
Atland	Behavioural responses of brown trout, *Salmo trutta*, juveniles in concentration gradients of pH and Al - a laboratory study	1998	Atlantic salmon, *Salmo salar*	1 hr 200=avoidance, 70=no avoidance	Only two exposure concentrations
Atland and Barlaup	Avoidance behaviour of Atlantic salmo (*Salmo salar* L.) fry in waters of low pH and elevated aluminum concentration: laboratory experiments	1996	Atlantic salmon, *Salmo salar*	1 hr LC20=85, LC40=160	Only two exposure concentrations
Avis et al.`	Ultrastructural alterations in *Fusarium sambucinum* and *Heterobasidion annosum* treated with aluminum chloride and sodium metabisulfite	2009	Fungus, *Fusarium sambucinum* Fungus, *Heterobasidion annosum*	60 min LOEC (dead conidia) =269,880 for both species	Only two exposure concentrations
Baba and Gunduz	Effect of alteration zones on water quality: a case study from Biga Peninsula, Turkey	2010	-	-	Survey
Bailey et al.	Application of toxicity identification procedures to the echinoderm fertilization assay to identify toxicity in a municipal effluent	1995	Sand dollar, *Dendraster excentricus* Purple urchin, *Stronglocentrotus purpuratus*	-	Mixture; effluent

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Baker	Aluminum toxicity to fish as related to acid precipitation and Adirondack surface water quality	1981	Brook trout, *Salvelinus fontinalis* White sucker, *Catostomus commersoni*	14 d 46.7% survival=180, 43.4% survival=110	Only two exposure concentrations
Baker	Effects on fish metals associated with acidification	1982	-	-	Review; results of previously published papers
Baker and Schofield	Aluminum toxicity to fish in acidic waters	1982	-	-	Only two exposure concentrations; review of Baker 1982
Baldigo and Murdoch	Effect of stream acidification and inorganic aluminum on mortality of brook trout (Salvelinus fontinalis) in the Catskill Mountains, New York	1997	Brook trout, *Salvelinus fontinalis*	-	Mixture; fluctuating Catskill mountain stream chemical exposure
Ball et al.	Water-chemistry data for selected springs, geysers, and streams in Yellowstone National Park, Wyoming, 2006-2008	2010	-	-	Survey; occurrence
Ballance et al.	Influence of sediment biofilm on the behaviour of aluminum and its bioavailability to the snail *Lymnaea stagnalis* in neutral freshwater	2001	Snail, *Lymnaea stagnalis*	-	Not applicable; no aluminum toxicity data
Barbiero et al.	The effects of a continuous application of aluminum sulfate on lotic benthic invertebrates	1988	-	-	Exposure concentration not known; field dosing of Al sulfate to a reservoir
Barbour and Paul	Adding value to water resource management through biological assessment of rivers	2010	-	-	Not applicable; no aluminum toxicity data
Barcarolli and Martinez	Effects of aluminum in acidic water on hematological and physiological parameters of the neotropical fish *Leporinus macrocephalus* (Anostomidae)	2004	Neotropical fish, *Leporinus macrocephalus*	24 hr 15 Increase hematocrit %; decrease plasma Na, Cl; Increase plasma glucose	Not North American species; only one exposure concentration
Bargagli	Environmental contamination in Antarctic ecosystems	2008	-	-	Survey; occurrence
Barnes	The determination of specific forms of aluminum in natural water	1975	-	-	Not applicable; no aluminum toxicity data
Battram	The effects of aluminum and low pH on chloride fluxes in the brown trout, *Salmo trutta* L.	1988	Brown trout, *Salmo trutta*	-	Acclimation too short; too few organisms per concentration
Beattie and Tyler-Jones	The effects of low pH and aluminum on breeding success in the frog *Rana temporaria*	1992	European common frog, *Rana temporaria*	-	Not North American species; only 3 exposure concentrations

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Beattie et al.	The effects of pH, aluminum concentration and temperature on the embryonic development of the European common frog, *Rana temporaria*	1992	European common frog, *Rana temporaria*	-	Not North American species; cannot determine effect concentration; dose-response not well defined
Becker and Keller	The effects of iron and sulfate compounds on the growth of *Chlorella*	1973	Green alga, *Chlorella vulgaris*	30 d 163,972 Reduced growth	Too few exposure concentrations, lack of exposure details
Belabed et al.	Evaluation de la toxicite de quelques metaux lourds a l'aide du test daphnie	1994	-	-	Text in foreign language
Berg	Aluminum and manganese toxicities in acid coal mine wastes	1978	-	-	Review; results of previously published papers
Berg and Burns	The distribution of aluminum in the tissues of three fish species	1985	Channel catfish, *Ictalurus punctatus* Largemouth bass, *Micropterus salmoides* Gizzard shad, *Dorosoma cepedianum*	-	Exposure concentration not known; field accumulation study
Bergman	Development of biologically relevant methods for determination of bioavailable aluminum in surface waters	1992	Rainbow trout, *Oncorhynchus mykiss* Brook trout, *Salvelinus fontinalis*	-	Mixture; Al and organic acids
Bergman and Mattice	Lake acidification and fisheries project: adult brook trout (*Salvelinus fontinalis*) early life stages	1990	Brook trout, *Salvelinus fontinalis*	-	Review; results of previously published papers
Bergman et al.	Lake acidification and fisheries project: adult brook trout (*Salvelinus fontinalis*)	1988	Brook trout, *Salvelinus fontinalis*	-	Review; results of previously published papers
Berntssen et al.	Responses of skin mucous cells to aluminum exposure at low pH in Atlantic salmon (*Salmo salar*) smolts	1997	Atlantic salmon, *Salmo salar*	55.6, LT50=>80 hr, 91.0, LT50= 29 hr	Dilution water not characterized; not true control group
Bervoets et al.	Use of transplanted zebra mussels (*Dreissena polymorpha*) to assess the bioavailability of microcontaminants in Flemish surface waters	2005	Zebra mussel, *Dreissena polymorpha*	-	Exposure concentration not known; mixture; field accumulation study
Bexfield et al.	Potential chemical effects of changes in the source of water supply for the Albuquerque Bernalillo County Water Utility Authority	2008	-	-	Not applicable; no aluminum toxicity data
Birge et al.	Evaluation of aquatic pollutants using fish and amphibian eggs as bioassy organisms	1979	-	-	Results of previously published papers; review of Birge 1978
Birge et al.	Aquatic toxicity tests on inorganic elements occurring in oil shale	1980	-	-	Results of previously published papers; review of Birge 1978

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Birge et al.	The reproductive toxicology of aquatic contaminants	1981	-	-	Review; results of previously published papers
Birge et al.	Effects of chemical stresses on behavior of larval and juvenile fishes and amphibians	1993	Fathead minnow, *Pimephales promelas*	50 Reduced feeding	Only two exposure concentrations
Bjerknes et al.	Aluminum in acidic river water causes mortality of farmed Atlantic salmon (*Salmo salar* L.) in Norwegian fjords	2003	Atlantic salmon, *Salmo salar*	-	Exposure concentration not known; field study with run-off to fjord-based farms
Boniardi et al.	Effect of dissolved metals on the organic load removal efficiency of Lemna gibba	1999	Duckweed, *Lemna gibba*	7 d NOEC(growth)= >29,000	Excessive EDTA used (>200 µg/L)
Booth et al.	Effects of aluminum and low pH on net ion fluxes and ion balance in the brook trout (*Salvelinus fontinalis*)	1988	Brook trout, *Salvelinus fontinalis*	-	Mixture; low pH and Al
Bowry	Relative toxicity of different fumigants against the adults of lesser grain borer *Rhizopertha dominica* Fabr. and rice moth *Corcyra cephalonica* Staint	1985	-	-	Not applicable; terrestrial species
Bradford et al.	Effects of low pH and aluminum on two declining species of amphibians in the Sierra Nevada, California	1992	Mountain yellow-legged frog, *Rana muscosa* Yosemite toad, *Bufo canorus*	No effect on hatch time or growth at 75; Effect on hatch time and decrease growth at 75	Only one exposure concentration
Bradford et al.	Effects of low pH and aluminum on amphibians at high elevation in the Sierra Nevada, California	1994	Pacific chorus frog, *Pseudacris regilla* Long-toed salamander, *Ambystoma macrodactylum*	-	Only one exposure concentration at each pH level
Brady and Griffiths	Effects of pH and aluminum on the growth and feeding behaviour of smooth and palmate newt larvae	1995	Newt, *Triturus helveticus* Newt, *Triturus vulgaris*	14 d Reduce growth for both species at 222 and pH=7.0	Only one exposure concentration
Brodeur et al.	Increase of heart rate without elevation of cardiac output in adult Atlantic salmon (*Salmo salar*) exposed to acidic water and aluminum	1999	Atlantic salmon, *Salmo salar*	-	Mixture; dilution water is river water
Brodeur et al.	Effects of subchronic exposure to aluminum in acidic water on bioenergetics of Atlantic salmon (*Salmo salar*)	2001	Atlantic salmon, *Salmo salar*	36 d Decrease growth, but not food consumption at 50	Only one exposure concentration

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Brown	The effects of various cations on the survival of brown trout, *Salmo trutta* at low pHs	1981a	Brown trout, *Salmo trutta*	18 d Increase survival time at 250	Only two exposure concentrations
Brown	Effect of calcium and aluminum concentrations on the survival of brown trout (*Salmo trutta*) at low pH	1983	Brown trout, *Salmo trutta*	16 d 30% survival at 500 (Ca=2 mg/L); 0% survival at 500 (Ca=0.25 mg/L)	Only two exposure concentrations
Brown and Bruland	Dissolved and particulate aluminum in the Columbia River and coastal waters of Oregon and Washington: behavior in near-field and far-field plumes	2009	-	-	Survey; occurrence
Brown et al.	Report on a large fish kill resulting from natural acid water conditions in Australia	1983	-	-	Mixture; Al and low pH
Brown et al.	Effects of low ambient pH and aluminum on plasma kinetics of cortisol, T3, and T4 in rainbow trout (*Oncorhynchus mykiss*)	1990	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Brown et al.	Contaminant effects on the tleost fish thyroid	2004	-	-	Review; results of previously published papers
Brumbaugh and Kane	Variability of aluminum concentrations in organs and whole bodies of smallmouth bass (*Micropterus dolomieui*)	1985	Smallmouth bass, *Micropterus dolomieui*	-	Exposure concentration not known; field accumulation study
Budambula and Mwachiro	Metal status of Nairobo river waters and their bioaccumulation in *Labeo cylindricus*	2006	Fish, *Labeo cylindricus*	-	Not North American species; exposure concentration not known; field accumulation study
Buergel and Soltero	The distribution and accumulation of aluminum in rainbow trout following a whole-lake alum treatment	1983	-	-	Exposure concentration not known; field accumulation study
Burrows	Aquatic aluminum: chemistry, toxicology, and environmental prevalence	1977	-	-	Review; results of previously published papers

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Burton and Allan	Influence of pH, aluminum, and organic matter on stream invertebrates	1986	Stonefly, *Nemoura sp.* Isopod, *Asellus intermedius* Snail, *Physella heterostropha* Caddisfly, *Lepidostoma liba* Caddisfly, *Pycnopsyche guttifer*	28 d 35% survival at 500; 20% survival at 500; 55% survival at 500; 50% survival at 500; 70% survival at 500	Only two exposure concentrations
Cai et al.	Developmental characteristics and aluminum resistance of root border cells in rice seedlings	2011	Rice, *Oryza sativa*	-	Dilution water is distilled water
Calevro et al.	Toxic effects of aluminum, chromium and cadmium in intact and regenerating freshwater planarians	1998a	Planarian, *Dugesia etrusca*	15 d NOEC (mortality) =250; LOEC=500	Not North American species
Calevro et al.	Tests of toxicity and teratogenicity in biphasic vertebrates treated with heavy metals (Cr3+, Al3+, Cd2+)	1998b	Newt, *Triturus vulgaris meridionalis* Frog, *Rana esculenta*	170 hr NOEC (embryo development)=404.7; 120 hr NOEC (embryo development)=404.7	Not North American species, unmeasured chronic exposure
Calevro et al.	Bioassays for testing effects of Al, Cr and Cd using development in the amphibian *Pleurodeles waltl* and regeneration in the planarian *Dugesia etrusca*	1999	Planarian, *Dugesia etrusca*	14 d 100% mortality at 13,490 NOEC (regeneration)=1,349	Not North American species
Camargo et al.	Osmo-ionic alterations in a neotropical fish acutely exposed to aluminum	2007	Neotropical fish, *Prochilodus lineatus*	-	Not North American species; lack of exposure details; only one exposure concentration; abstract only
Camargo et al.	How aluminum exposure promotes osmoregulatory disturbances in the neotropical freshwater fish *Prochilus lineatus*	2009	Neotropical fish, *Prochilodus lineatus*	96 hr Increase hemoglobin; increase hematocrit %; decrease plasma ions and osmolarity at 438	Not North American species; only one exposure concentration
Camilleri et al.	Silica reduces the toxicity of aluminum to a Tropical Freshwater Fish (*Mogurnda mogurnda*)	2003	Australian spotted gudgeon, *Mogurnda mogurnda*	96 hr LC50=374; LC50=547	Not North American species

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Campbell et al.	Effect of aluminum and silica acid on the behavior of the freshwater snail *Lymnaea stagnalis*	2000	Snail, *Lymnaea stagnalis*	7 d Reduce behavioral state score (BSS) at 500	Only two exposure concentrations
Capdevielle and Scanes	Effect of dietary acid or aluminum on growth and growth-related hormones in mallard ducklings (*Anas platyrhynchos*)	1995	Mallard duck, *Anas platyrhynchos*	-	Dietary exposure; only two exposure concentrations
Capdevielle et al.	Aluminum and acid effects on calcium and phosphorus metabolism in young growing chickens (*Gallus gallus domesticus*) and mallard ducks (*Anas platyrhynchos*)	1998	Mallard duck, *Anas platyrhynchos*	-	Dietary exposure; only two exposure concentrations
Carballeira et al.	Biomonitoring of sporadic acidification of rivers on the basis of release of preloaded cadmium from the aquatic bryophyte *Fontinalis antipyretica* Hedw	2001	Bryophyte, *Fontinalis antipyretica*	-	Mixture; species prior exposed to Cd
Cardwell et al.	Toxic substances and water quality effects on larval marine organisms, technical report no. 45	1979	-	-	Not applicable; no aluminum toxicity data
Carter and Porter	Trace-element accumulation by *Hygrohypnum ochraceum* in the Upper Rio Grande Basin, Colorado and New Mexico, USA	1997	Bryophyte, *Hygrohypnum ochraceum*	-	Exposure concentration not known (not measured over time); field exposure with transplanted plants
Chakravorty et al.	Primary and secondary stress response of *Channa punctatus* to sublethal aluminium toxicity	2012	Snakehead catfish, *Channa punctatus*	96 hr LC50=220,000	Not North American species
Chamier and Tipping	Effects of aluminum in acid streams on growth and sporulation of aquatic hyphomycetes	1997	Fungi, *Tricladium splendens* Fungi, *Alatospora constricta* Fungi, *Varicosporium elodea*	-	Mixture; low pH and Al
Chang et al.	Response of the mussel *Anadonta grandi* to acid and aluminum. Comparison of blodd ions from laboratory and field results	1988	Mussel, *Anadonta grandi*	-	Mixture; aluminum sulphate added to a lake
Chapman et al.	Concentration factors of chemical elements in edible aquatic organisms	1968	-	-	Review; results of previously published papers
Chapman et al.	Why fish mortality in bioassays with aluminum reduction plant wastes don't always indicate chemical toxicity	1987	-	-	Not applicable; no aluminum toxicity data
Chen	Ecological risk assessment for aquatic species exposed to contaminants in Kelung River, Taiwan	2005	-	-	Not applicable; occurrence; no aluminum toxicity data

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Chen et al.	Environmental factors affecting settlement of quagga mussel (*Dreissena rostriformis bugensis*) veligers in Lake Mead, Nevada-Arizona, USA	2011	Quagga mussel, *Dreissena rostriformis bugensis*	-	Not applicable; no aluminum toxicity data
Chevalier et al.	Acidity and aluminum effects on osmo-iono-regulation in the brook trout	1987	Brook trout, *Salvelinus fontinalis*	7 d Addition of Al kept fish alive compared to control at 500 and pH=5.5	Only one exposure concentration
Christensen	Effects of metal cations and other chemicals upon the in vitro activity of twp enzymes in the blood plasma of the white sucker, *Catostomus commersoni* (lacepede)	1971/ 1972	White sucker, *Catostomus commersoni*	-	In vitro experiment
Christensen and Tucker	Effects of selected water toxicants on the in vitro activity of fish carbonic anhydrase	1976	Channel catfish, *Ictalurus punctatus*	-	Excised cells
Clark and Hall	Effects of elevated hydrogen ion and aluminum concentrations on the survival of amphibian embryos and larvae	1985	Toad, *Bufo americanus* Wood frog, *Rana sylvatica* Spotted salamander, *Ambystoma maculatum*	-	Exposure concentration not known; field experiment: dosed stream pools
Clark and LaZerte	Intraspecific variation in hydrogen ion and aluminum toxicity in *Bufo americanus* and *Ambystoma maculatum*	1987	Toad, *Bufo americanus* Spotted salamander, *Ambystoma maculatum*	-	Pre-exposure to pollutant
Cleveland et al.	Interactive toxicity of aluminum and acidity to early life stages of brook trout	1986	Brook trout, *Salvelinus fontinalis*	30 d Increase egg mortality at 318	Only one exposure concentration
Cleveland et al.	Sensitivity of brook trout to low pH, low calcium and elevated aluminum concentrations during laboratory pulse exposures	1991b	Brook trout, *Salvelinus fontinalis*	-	Only one exposure concentration; mixture; Al and acid pulses
Colman et al.	Determination of dilution factors for discharge of aluminum-containing wastes by public water-supply treatment facilities into lakes and reservoirs in Massachusetts	2011	-	-	Not applicable; no aluminum toxicity data

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Conklin et al.	Comparative toxicity of drilling muds: Role of chromium and petroleum hydrocarbons	1983	Grass shrimp, *Palaemonetes pugio* Sheepshead minnow, *Cyrpinodon variegatus*	-	Mixture; drilling mud
Cook and Haney	The acute effects of aluminum and acidity upon nine stream insects	1984	Five caddisflies, two mayflies, stonefly and beetle	-	Mixture; dilution water is river water
Correa et al.	Changes in oxygen consumption and nitrogen metabolism in the dragonfly *Somatochlora cingulata*	1985	Dragonfly, *Somatochlora cingulata*	96 hr No change in respiratory rate at 30	Lack of exposure details; dilution water not characterized; too few exposure concentration
Correa et al.	Oxygen consumption and ammonia excretion in the detritivore caddisfly *Limnephillus sp.* exposed to low pH and aluminum	1986	Caddisfly, *Limnephillus sp.*	-	Only one exposure concentration; mixture; low pH and Al
Correia et al.	Aluminum as an endocrine disruptor in female Nile tilapia (*Oreochromis niloticus*)	2010	Nile tilapia, *Oreochromis niloticus*	96 hr Increase gonad and decrease liver lipids at 1,600	Only one exposure concentration
Craig et al.	Water quality objectives development document: aluminum	1985	-	-	Review; results of previously published papers
Cravotta et al.	Abandoned mine drainage in the Swatara Creek Basin, southern anthracite coalfield, Pennsylvania, USA: 1. Stream water quality trends coinciding with the return of fish	2010	-	-	Mixture; dilution water is river water
Crawford et al.	A survey of metal and pesticide levels in stormwater retention pond sediments in coastal South Carolina	2010	-	-	Survey; occurrence
Crist et al.	Interaction of metal protons with algae. 3. Marine algae, with emphasis on lead and aluminum	1992	-	-	Bioaccumulation: steady state not reached
Cummins	Effects of aluminum and low pH on growth and development in *Rana temporaria* tadpoles	1986	Brown frog, *Rana temporaria*	18 d Decrease body mass and increase time to metamorph at 800	Not North American species; only two exposure concentrations
Dalziel et al.	The effects of low pH, low calcium concentrations and elevated aluminum concentrations on sodium fluxes in brown trout, *Salmo trutta* L.	1986	Brown trout, *Salmo trutta*	8 hr No effect on Na influx at 215.8	Only one exposure concentration
Delaune et al.	Total Hg, methyl Hg and other toxic heavy metals in a northern Gulf of Mexico Estuary: Louisiana Pontchartrain Basin	2008	-	-	Survey; occurrence

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Desouky	Tissue distribution and subcellular localization of trace metals on the pond snail *Lymnaea stagnalis* with special reference to the role of lysosomal granules in metal sequestration	2006	Snail, *Lymnaea stagnalis*	-	Bioaccumulation: exposure concentration not measured; inadequate exposure methods
Desouky	Metallothionein is up-regulated in molluscan responses to cadmium, but not aluminum, exposure	2012	Snail, *Lymnaea stagnalis* Zebra mussel, *Dreissena polymorpha*	-	Only one exposure concentration; possible prior exposure due to location collected in field
Desouky et al.	Influence of oligomeric silica and humic acids on aluminum accumulation in a freshwater grazing invertebrate	2002	Snail, *Lymnaea stagnalis*	-	Bioaccumulation: steady state not reached
DeWalle et al.	Episodic flow-duration analysis: a method of assessing toxic exposure of brook trout (*Salvelinus fontinalis*) to episodic increases in aluminum	1995	-	-	Not applicable; no aluminum toxicity data
Dickson	Liming toxicity of aluminum to fish	1983	-	-	Not applicable; no aluminum toxicity data
Dietrich and Schlatter	Aluminum toxicity to rainbow trout at low pH	1989a	Rainbow trout, *Oncorhynchus mykiss*	MT50=64 hrs at 200; MT50=45.5 hrs at 400 (pH=5.4); MT50=52 hrs at 400 (pH=5.6)	Only two exposure concentrations
Dietrich and Schlatter	Low levels of aluminum causing death of brown trout (*Salmo trutta fario*, L.) in a Swiss alpine lake	1989b	Brown trout, *Salmo trutta fario*	-	Mixture; exposure concentration varied over time; dilution water is lake water
Dietrich et al.	Aluminum and acid rain: mitigating effects of NaCl on aluminum toxicity to brown trout (*Salmo trutta fario*) in acid water	1989	Brown trout, *Salmo trutta fario*	-	No acclimation to test water; no aluminum toxicity data
Doke et al.	Habitat availability and benthic invertebrate population changes following alum treatment and hypolimnetic oxygenation in Newman Lake, Washington	1995	-	-	Mixture; alum added to lake; no species listed
Doudoroff and Katz	Critical review of literature on the toxicity of industrial wastes and their components to fish. II. The metals, as salts	1953	-	-	Review; results of previously published papers
Driscoll	A procedure for the fractionation of aqueous aluminum in dilute acidic waters	1984	-	-	Not applicable; no aluminum toxicity data
Driscoll	Aluminum in acidic surface waters: chemistry, transport, and effects	1985	-	-	Not applicable; no aluminum toxicity data

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Driscoll et al.	Effect of aluminum speciation on fish in dilute acidified waters	1980	Brook trout, *Salvelinus fontinalis*	14 d 28% survival at 420, pH=5.2; 42% survival at 480, pH=4.4	Lack of exposure details; only two exposure concentrations
Duis and Oberemm	Aluminum and calcium - Key factors determining the survival of vendace embryos and larvae in post-mining lakes?	2001	Vendace, *Coregonus albula*	Decrease hatch % at 2,100, pH=5.0	Not North American species; only one exposure concentration
Durrett et al.	The FRD3-mediated efflux of citrate into the root vasculature is necessary for efficient iron translocation	2007	-	-	Not applicable; no aluminum toxicity data
Dussault et al.	Effects of sublethal, acidic aluminum exposure on blood ions and metabolites, cardiac output, heart ratem and stroke volume of rainbow trout, *Oncorhynchus mykiss*	2001	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Dussault et al.	Effects of chronic aluminum exposure on swimming and cardiac performance in rainbow trout, *Oncorhynchus mykiss*	2004	Rainbow trout, *Oncorhynchus mykiss*	6 wk 75% survival at 32	Too few exposure concentrations; too few organisms per concentration
Dwyer et al.	Use of surrogates species in assessing contaminant risk to endangered and threatened species; final report - September 1995	1995	-	-	Not applicable; no aluminum toxicity data
Dwyer et al.	Assessing contaminant sensitivity of endangered and threatened aquatic species: part III. Effluent toxicity tests	2005	-	-	Not applicable; no aluminum toxicity data
Eaton et al.	A field and laboratory investigation of acid effects on largemouth bass, rock bass, black crappie, and yellow perch	1992	Rockbass, *Ambloplites rupestris* Largemouth bass, *Micropterus salmoides* Yellow perch, *Perca flavescens*	Hatch + 7 d NOEC (survival)=44.0; NOEC=44.0; NOEC=25.2	Too few exposure concentrations; control survival issues
Ecological Analysts, Inc.	Study on metals in food fish near the abandoned Vienna fly ash disposal area	1984	-	-	Field exposure, exposure concentrations not measured adequately
Eddy and Talbot	Formation of the perivitelline fluid in Atlantic salmon eggs (*Salmo salar*) in fresh water and in solutions of metal ions	1983	Atlantic salmon, *Salmo salar*	1 hr Inhibit perivitelline fluid formation at 26,980	Dilution water not characterized

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Eddy and Talbot	Sodium balance in eggs and dechlorinated embryos of the Atlantic salmon *Salmo salar* L. exposed to zinc, aluminum and acid waters	1985	Atlantic salmon, *Salmo salar*	-	Too few exposure concentrations; no true control group
Eisler et al.	Fourth annotated bibliography on biological effects of metals in aquatic environments (No. 2247-3132)	1979	-	-	Review
Elangovan et al.	Accumulation of aluminum by the freshwater crustacean Asellus aquaticus in neutral water	1999	Crustacean, *Asellus aquaticus*	-	Bioaccumulation: unmeasured concentration in exposure media
Elsebae	Comparative susceptibility of the Alareesh Marine Culture Center shrimp *Penaeus japonicus* and the brine shrimp *Artemia salina* to different insecticides and heavy metals	1994	Shrimp, *Penaeus japonicus*	96 hr LC50=0.001; LC50=0.0045; LC50=0.1	Not North American species; dilution water not characterized
Elwood et al.	Contribution of gut contents to the concentration and body burden of elements in Tipula spp. from a spring-fed stream	1976	-	-	Field exposure, exposure concentrations not measured adequately
Eriksen et al.	Short-term effects on riverine Ephemeroptera, Plecoptera, and Trichoptera of rotenone and aluminum sulfate treatment to eradicate *Gyrodactylus salaris*	2009	-	-	Mixture; mixed species exposure; dilution water is river water
Ernst et al.	Effects of habitat characteristics and water quality on macroinvertebrate communities along the Neversink River in southeastern New York, 1991-2001	2008	-	-	Not applicable; no aluminum toxicity data
Evans et al.	The effects of aluminum and acid on the gill morphology in rainbow trout, *Salmo gairdneri*	1988	Rainbow trout, *Oncorhynchus mykiss*	14 d LOEC (epithelial hyperplasma) = 269.8 (pH 5.2)	Only three exposure concentrations
Everhart and Freeman	Effect of chemical variations in aquatic environments. Vol. II. Toxic effects of aqueous aluminum to rainbow trout	1973	Rainbow trout, *Oncorhynchus mykiss*	45 d Reduced growth at 514 (pH=8 and pH=6.85)	Too few exposure concentrations; unmeasured chronic exposure
Exley	Avoidance of aluminum by rainbow trout	2000	Rainbow trout, *Oncorhynchus mykiss*	45 min. Avoidance at 33.73	No acclimation to test water
Exley et al.	Silicon, aluminium and the biological availability of phosphorus in algae	1993	Diatom, *Navicula pelliculosa* Green alga, *Chlorella vulgaris*	24 hr 269.8 inhibited growth rate; 24 hr 1,295 inhibited growth rate	Only one exposure concentration
Exley et al.	Polynuclear aluminum and acute toxicity in the fish	1994	-	-	Inappropriate form of toxicant; polynuclear aluminum

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Exley et al.	Kinetic constraints in acute aluminum toxicity in the rainbow trout (*Oncorhynchus mykiss*)	1996	Rainbow trout, *Oncorhynchus mykiss*	-	Only one exposure concentration; no control group
Exley et al.	Hydroxyaluminosilicates and acute aluminum toxicity to fish	1997	Rainbow trout, *Oncorhynchus mykiss*	-	Mixture; Al and Si
Fageria	Influence of aluminum in nutrient solutions on chemical composition in two rice cultivars at different growth stages	1985	Rice, *Oryza sativa*	-	Bioaccumulation study: exposure concentrations not measured
Famoso et al.	Development of a novel aluminum tolerance phenotyping platform used for comparisons of cereal aluminum tolerance and investigations into rice aluminum tolerance mechanisms	2010	Sorghum, *Sorghum bicolor* Wheat, *Triticum aestivum* Rice, *Oryza sativa*	-	Excessive EDTA in growth media (25 mg/L)
Farag et al. 1993	The effects of low pH and elevated aluminum on yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*)	1993	Yellowstone cutthroat trout, *Oncorhynchus clarki bouvieri*	7 d No effect on survival at 50	Too few exposure concentrations; poor control survival
Farringer	The determination of the acute toxicity of rotenone and Bayer 73 to selected aquatic organisms	1972	-	-	Not applicable; no aluminum toxicity data
Fernandez-Davila et al.	Aluminum-induced oxidative stress and neurotoxicity in grass carp (Cyprinidae-*Ctenopharingodon idella*)	2012	Grass carp, *Ctenopharingodon idella*	96 hr Increase lipid peroxidation, dopamine levels, SOD activity and decrease CAT activity in brain tissue at 100	Only one exposure concentration
Finn	The physiology and toxicology of salmonid eggs and larvae in relation to water quality criteria	2007	-	-	Review; results of previously published papers
Fischer and Gode	Toxicological studies in natural aluminum silicates as additives to detergents using freshwater organisms	1977	-	-	Text in foreign language
Fivelstad and Leivestad	Aluminum toxicity to Atlantic salmon (*Salmo salar* L.) and brown trout (*Salmo trutta* L.): Mortality and physiological response	1984	Atlantic salmon, *Salmo salar*	LT50=26 hr at 84.18; LT50=41 hr at 84.72; LT50=62 hr at 45.06	Lack of exposure details; dilution water not characterized
Fjellheim et al.	Effect of aluminium at low pH on the mortality of elvers (*Anguilla anguilla* L.), a laboratory experiment	1985	Eel, *Anguilla anguilla*	-	Only two exposure concentrations; dilution water not characterized

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Fok et al.	Determination of 3,5,3"-triiodo-L-thyronine (T3) levels in tissues of rainbow trout (*Salmo gairdneri*) and the effects of low ambient pH and aluminum.	1990	Rainbow trout, *Oncorhynchus mykiss*	-	Inappropriate form of toxicant ($AlKSO_4$); surgically altered test species
Folsom et al.	Comparative study of aluminum and copper transport and toxicity in an acid-tolerant freshwater green alga	1986	Green alga, *Chlorella saccarophila*	-	Lack of details; cannot determine effect concentration
France and Stokes	Influence of manganese, calcium, and aluminum on hydrogen ion toxicity to the amphipod *Hyalella azteca*	1987	Amphipod, *Hyalella azteca*	-	Mixture; Mn, Ca, pH and Al
Freda	The effects of aluminum and other metals on amphibians	1991	-	-	Review; results of previously published papers
Freeman	Recovery of rainbow trout from aluminum poisoning	1973	Rainbow trout, *Oncorhynchus mykiss*	-	Pre-exposure to pollutant
Frick and Herrmann	Aluminum accumulation in a lotic mayfly at low pH - a laboratory study	1990	Mayfly, *Heptagenia sulphurea*	-	Not North American species; lack of exposure details; cannot determine effect concentration
Fuma et al.	Ecological effects of various toxic agents on the aquatic microcosm in comparison with acute ionizing radiation	2003	Bacteria, *Escherichia coli* Protozoa, *Tetrahymena thermophila* Protozoa, *Euglena gracilis*	-	Mixture; radiation and Al
Gagen	Aluminum toxicity and sodium loss in three salmonid species along a pH gradient in a mountain stream	1986	-	-	Exposure concentration not known; field exposure
Gagen et al.	Mortality of brook trout, mottled sculpins, and slimy sculpins during acidic episodes	1993	Brook trout, *Salvelinus fontinalis* Mottled sculpin, *Cottus bairdi* Slimy sculpin, *Cottus cognatus*	-	Mixture; exposure concentration varied over time; dilution water is river water
Galindo et al.	Genotoxic effects of aluminum on the neotropical fish *Prochilodus lineatus*	2010	Neotropical fish, *Prochilodus lineatus*	96 hr Increase COMET score and number of damaged necleoids at 438	Not North American species, only one exposure concentration
Gallon et al.	Hydrophonic study of aluminum accumulation by aquatic plants: effects of fluoride and pH	2004	Five aquatic plants	-	Bioaccumulation: steady state not reached

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Galloway et al.	Water quality and biological characteristics of the Middle Fork of the Saline River, Arkansas, 2003-06	2008	-	-	Not applicable; no aluminum toxicity data
Garcia-Garcia et al.	Impact of chromium and aluminum pollution on the diversity of zooplankton: a case study in the Chimaliapan wetland (Ramsar Site) (Lerma Basin, Mexico)	2012	-	-	Mixture; dilution water is wetland water
Garcia-Medina et al.	Aluminum-induced oxidative stress in lymphocytes of common carp (*Cyprinus carpio*)	2010	Common carp, *Cyprinus carpio*	96 hr Increase lipid peroxidation and decrease SOD activity at 50	Too few exposure concentrations, dilution water not characterized
Garcia-Medina et al.	Genotoxic and cytotoxic effects induced by aluminum in the lymphocytes of the common carp (*Cyprinus carpio*)	2011	Common carp, *Cyprinus carpio*	96 hr DNA damage: T/N index at 50	Too few exposure concentrations, dilution water not characterized
Garcia-Medina et al.	The relationship of cytotoxic and genotoxic damage with blood aluminum levels and oxidative stress induced by this metal in common carp (*Cyprinus carpio*) erythrocytes	2013	Common carp, *Cyprinus carpio*	96 hr LOEC (reduced USOD and NADPH on erythocytes) = 50	Only three exposure concentrations
Gardner and Al-Hamdani	Interactive effects of aluminum and humic substances on Salvinia	1997	-	-	Not applicable; no aluminum toxicity data
Gardner et al.	Towards the establishment of an environmental quality standard for aluminum in surface waters	2008	-	-	Not applicable; no aluminum toxicity data
Gascon et al.	The interaction of pH, calcium and aluminum concentrations on the survival and development of wood frog (*Rana sylvatica*) eggs and tadpoles	1987	Wood frog, *Rana sylvatica*	100% mortality at 200	Only two exposure concentrations; lack of exposure details; duration not reported
Geiger et al.	Acute toxicities of organic chemicals to fathead minnows (*Pimephales promelas*) Volume V	1990	Fathead minnows, *Pimephales promelas*	-	Not applicable; no aluminum toxicity data
Gensemer	Role of aluminum and growth rate on changes in cell size and silica content of silica-limited populations of *Asterionella ralfsii* var. *americana* (Bacillariophyceae).	1990	Diatom, *Asterionella ralfsii* var. *americana*	21 d Decrease mean cell length, total surface area and biovolume at 75.54	Only two exposure concentrations
Gensemer	The effects of pH and aluminum on the growth of the acidophilic diatom *Asterionella ralfsii* var. *americana*	1991a	Diatom, *Asterionella ralfsii* var. *americana*	-	Review of Gensemer 1989 thesis
Gensemer	The effects of aluminum on phosphorus and silica-limited growth in *Asterionella ralfsii* var. *americana*	1991b	Diatom, *Asterionella ralfsii* var. *americana*	-	Growth stimulation study, not toxicity

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Gensemer and Playle	The bioavailability and toxicity of aluminum in aquatic environments	1999	-	-	Review; results of previously published papers
Gensemer et al.	Comparative effects of pH and aluminum on silica-limited growth and nutrient uptake in *Asterionella ralfsii* var. *americana* (Bacillariophyceae)	1993	Diatom, *Asterionella ralfsii* var. *americana*	-	Only one exposure concentration; cannot determine effect concentration
Gensemer et al.	Interactions of pH and aluminum on cell length reduction in *Asterionella ralfsii* var. *americana* Korn	1994	Diatom, *Asterionella ralfsii* var. *americana*	25 d No effect on cell length at 539.6	Only one exposure concentration; dilution water not characterized
Genter	Benthic algal populations respond to aluminum, acid, and aluminum-acid mixture in artificial streams	1995	Green alga, *Cosmarium melanosporum* Blue-green alga, *Schizothrix calcicola* Diatom, *Achnanthes minutissima* Diatom, *Naviculoids*	28 d Increased growth at 200	Only one exposure concentration
Gibbons et al.	Effects of multiphase restoration, particularly aluminum sulfate application, on the zooplankton community of a eutrophic lake in eastern Washington	1984	-	-	Exposure concentration not known; population/ community changes of a lake exposed to Al over a series of years
Gill et al.	Assessment of water-quality conditions in Fivemile Creek in the vicinity of the Fivemile Creek Greenway, Jefferson County, Alabama, 2003-2005	2008	-	-	Not applicable; no aluminum toxicity data
Gladden	The effect of aluminum on cortisol levels in goldfish (*Carassius auratus*)	1987	Goldfish, *Carassius auratus*	-	Surgically altered test species
Goossenaerts et al.	A microanalytical study of the gills of aluminum-exposed rainbow trout (*Salmo gairdneri*)	1988	Rainbow trout, *Oncorhynchus mykiss*	72 hr Increase the Al-content of the gills at 190	Duration too short, only one exposure concentration
Gopalakrishnan et al.	Toxicity of heavy metals on embryogenesis and larvae of the marine sedentary polychaete *Hydroides elegans*	2007	Polychaete, *Hydroides elegans*	-	Pre-exposure to pollutant
Goss and Wood	The effects of acid and acid/aluminum exposure on circulating plasma cortisol levels and other blood parameters in the rainbow trout, *Salmo gairdneri*	1988	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered fish
Greger et al.	Aluminum effects on *Scenedesmus obtusiusculus* with different phosphorus status. I. Mineral uptake	1992a	Green alga, *Scenedesmus obtusiusculus*	-	Excessive EDTA in growth media (108 µm Na_2EDTA)

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Greger et al.	Aluminum effects on *Scenedesmus obtusiusculus* with different phosphorus status. II. Growth, photosynthesis and pH	1992b	Green alga, *Scenedesmus obtusiusculus*	-	Excessive EDTA in growth media (108 µm Na_2EDTA)
Gregor et al.	Growth assays with mixed cultures of cyanobacteria and algae assessed by in vivo fluorescence: One step closer to real ecosystems?	2008	Green alga, *Pseudokirchneriella subcapitata* Blue-green alga, *Aphanothece clathrata*	-	Inappropriate form of toxicant (PAX-18)
Guerold et al.	Occurrence of aluminum in chloride cells of *Perla marginata* (Plecoptera) after exposure to low pH and elevated aluminum concentration	1995	Stonefly, *Perla marginata*	-	Not North American species; Bioaccumulation: steady state not reached
Gunn and Keller	Spawning site water chemistry and lake trout (*Salvelinus namaycush*) sac fry survival during spring snow melt	1984	Lake trout, *Salvelinus namaycush*	-	Mixture, Al and low pH
Gunn and Noakes	Latent effects of pulse exposure to aluminum and low pH on size, ionic composition, and feeding efficiency of lake trout (*Salvelinus namaycush*) alevins	1987	Lake trout, *Salvelinus namaycush*	5 d LOEC (growth)=<100	Only two exposure concentrations
Guo et al.	Involvement of antioxidative defense system in rice seedlings exposed to aluminum toxicity and phosphorus deficiency	2012	Rice, *Oryza sativa*	-	Excessive chelator in growth media (5 mg/L Fe-citrate)
Guthrie et al.	Aquatic bacterial populations and heavy metals-II. Influence of chemical content of aquatic environments on bacterial uptake of chemical elements	1977	Bacterial population	-	Exposure concentration not known; field accumulation study
Guzman et al.	Implementing *Lecane quadridentata* acute toxicity tests to assess the toxic effects of selected metals (Al, Fe and Zn)	2010	Rotifer, *Lecane quadridentata*	48 hr LC50=1,572	Not North American species
Hackett	Ecological aspects of the nutrition of *Deschampsia flexuosa* (L.) Trin. III. Investigation of phosphorus requirement and response to aluminium in water culture, and a study of growth in soil	1967	Wavy hair grass, *Deschampsia flexuosa*	-	Not applicable; terrestrial species
Hall et al.	Mortality of striped bass larvae in relation to contaminants and water quality in a Chesapeake Bay tributary	1985	Striped bass, *Morone saxatilis*	-	Exposed to mixture, high control mortality (15-25%); dilution water is river water
Hamilton-Taylor et al.	Depositional fluxes of metals and phytoplankton in Windermere as measured by sediment traps	1984	-	-	Effluent or mixture

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Handy and Eddy	Surface absorption of aluminium by gill tissue and body mucus of rainbow trout, *Salmo gairdneri*, at the onset of episodic exposure	1989	Rainbow trout, *Oncorhynchus mykiss*	1 hr Gill content=50 µg/g at 954	Only one exposure concentration
Hanks	Effect of metallic aluminum particles on oysters and clams	1965	Soft-shell clam, *Mya arenaria* American oyster, *Crassostrea virginica*	-	Dilution water not characterized, inappropriate form of Al
Harper et al.	In vivo biodistribution and toxicity depends on nanomaterial composition, size, surface functionalisation and route of exposure	2008	Zebrafish, *Danio rerio*	-	Inappropriate form of toxicant (Al-oxide)
Harry and Aldrich	The distress syndrome in *Taphtus glabratus* (Say) as a reaction to toxic concentrations of inorganic ions	1963	Snail, *Taphius glabratus*	24 hr LOEC (distress, inability to move)=5,000	Dilution water is distilled water
Havas	Effects of aluminum on aquatic biota	1986a	-	-	Review
Havas and Hutchinson	Aquatic invertebrates from the Smoking Hills, N.W.T.: effect of pH and metals on mortality	1982	-	-	Mixture
Havas and Hutchinson	Effect of low pH on the chemical composition of aquatic invertebrates from tundra ponds at the Smoking Hills, N.W.T., Canada	1983	-	-	Pre-exposure to pollutant
Havens	Aluminum binding to ion exchange sites in acid-sensitive versus acid tolerant cladocerans	1990	Cladoceran, *Daphnia galeata mendotae* Cladoceran, *Daphnia retrocurva* Cladoceran, *Bosmina longirostris*	24 hr 98% mortality at 200; 94% mortality at 200; 6% mortality at 200	Only one exposure concentration
Havens	Littoral zooplankton response to acid and aluminum stress during short-term laboratory bioassays	1991	-	-	Only one exposure concentration; mixture; low pH and Al
Havens	Acid and aluminum effects on sodium homeostasis and survival of acid-sensitive and acid-tolerant cladocera	1992	Cladoceran, *Daphnia galeata mendotae* Cladoceran, *Bosmina longirostris*	24 hr NOEC (survival)=100; NOEC=200	Only two exposure concentrations
Havens	Acid and aluminum effects on the survival of littoral macro-invertebrates during acute bioassays	1993a	-	-	Only one exposure concentration; control survival issues or mixed species exposure

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Havens	Acid and aluminum effects on osmoregulation and survival of the freshwater copepod *Skistodiaptomus oregonensis*	1993b	Copepod, *Skistodiaptomus oregonensis*	48 hr NOEC (survival)=200 at pH=7.5; LOEC=100 at pH=6.0	Only two exposure concentrations
Havens and Decosta	The role of aluminum contamination in determining phytoplankton and zooplankton responses to acidification	1987	-	-	Mixture; exposure concentration varied over time; Dilution water is lake water
Havens and Heath	Acid and aluminum effects on freshwater zooplankton: and in situ mesocosm study	1989	Zooplankton community	-	Mixture (low pH and Al); only one exposure concentration
Havens and Heath	Phytoplankton succession during acidification with and without increasing aluminum levels	1990	Phytoplankton community	-	Mixture (low pH and Al); only one exposure concentration
Heier et al.	Sublethal effects in Atlantic salmon (*Salmo salar*) exposed to mixtures of copper, aluminum and gamma radiation	2012	Atlantic salmon, *Salmo salar*	48 hr No mortality, but increase plasma glucose and decrease plasma sodium at 267	Only one exposure concentration, too few animals per concentration
Helliwell	Speciation and toxicity of aluminum in a model fresh water	1983	-	-	Lack of details; cannot determine effect concentration
Heming and Blumhagen	Plasma acid-base and electrolyte states of rainbow trout exposed to alum (aluminum sulphate) in acidic and alkaline environments	1988	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered fish
Herkovits et al.	Identification of aluminum toxicity and aluminum-zinc interaction in amphibian *Bufo arenarum* embryos	1997	Toad, *Bufo arenarum*	96 hr LC50=460	Not North American Species
Herrmann and Andersson	Aluminum impact on respiration of lotic mayflies at low pH	1986	-	-	Mixture; dilution water is stream water
Herrmann and Frick	Do stream invertebrates accumulate aluminum at low pH conditions?	1995	-	-	Survey
Hesse	Phosphorus relationships in a mangrove-swamp mud with particular reference to aluminum toxicity	1963	-	-	Sediment
Hill et al.	Zebrafish as a model vertebrate for investigating chemical toxicity	2005	Zebrafish, *Danio rerio*	-	Review
Hockett and Mount	Use of metal chelating agents to differentiate among sources of acute aquatic toxicity	1996	Cladoceran, *Ceriodaphnia dubia*	-	Mixture; EDTA, thiosulfate and Al
Hofler	Action of aluminum salts on Spirogyra and Zygnema	1958	-	-	Text in foreign language

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Horne and Dunson	Exclusion of the Jefferson salamander, *Ambystoma jeffersonianum*, from some potential breeding ponds in Pennsylvania: effects of pH, temperature, and metals on embryonic development	1994	Jefferson salamander, *Ambystoma jeffersonianum*	-	Lack of details; mixture; low pH and AL; duration not reported
Horne and Dunson	Toxicity of metals and low pH to embryos and larvae of the Jefferson salamander, Ambystoma jeffersonianum	1995a	Jefferson salamander, *Ambystoma jeffersonianum*	No effect values presented	No effect values presented
Horne and Dunson	Effects of low pH, metals, and water hardness on larval amphibians	1995b	Wood frog, *Rana sylvatica* Jefferson salamander, *Ambystoma jeffersonianum*	Percent survival depended on hardness, duration and species	Only one exposure concentration
Hornstrom et al.	Effects of pH and different levels of aluminum on lake plankton in the Swedish west coast area	1984	-	-	Survey; mixture; dilution water is lake water
Howells et al.	Effects of acidity, calcium, and aluminum on fish survival and productivity - a review	1983	-	-	Review; results of previously published papers
Howells et al.	EIFAC water quality criteria for European freshwater fish: Report on aluminum	1990	-	-	Review
Huebner and Pynnonen	Viability of glochidia of two species of *Anodonta* exposed to low pH and selected metals	1992	Swan mussel, *Anodonta cygnea*	24 hr glochidia EC50=18,000 at pH 4.5	Not North American species
Hunn et al.	Influence of pH and aluminum on developing brook trout in a low calcium water	1987	Brook trout, *Salvelinus fontinalis*	45 d Reduced growth and some behaviors at 283	Only one exposure concentration
Husaini and Rai	pH dependent aluminum toxicity to *Nostoc linckia*: Studies on phosphate uptake, alkaline and acid phosphatase activity, ATP content, and photosynthesis and carbon fixation	1992	Blue-green alga, *Nostoc linckia*	14 d Reduce photosynthetic O_2 evolution at 53,336	Only three exposure concentrations
Husaini et al.	Impact of aluminum, fluoride and flouroaluminate on ATPase activity og *Nostoc linckia* and *Chlorella vulgaris*	1996	Blue-green alga, *Nostoc linckia* Green alga, *Chlorella vulgaris*	-	Mixture
Hutchinson and Sprague	Toxicity of trace metal mixtures to American flagfish (*Jordanella floridae*) in soft, acidic water and implications for cultural acidification	1986	American flagfish, *Jordanella floridae*	-	Mixture; heavy metals
Hutchinson et al.	Lethal responses of salmonid early life stages to H+ and Al in dilute waters	1987	-	-	Review
Hwang	Lysosomal responses to environmental contaminants in bivalves	2001	American oyster, *Crassostrea virginica*	-	Exposure concentration not known; field accumulation study

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Hyne and Wilson	Toxicity of acid-sulphate soil leachate and aluminum to the embryos and larvae of Australian bass (*Macquaria novemaculeata*) in estuarine water	1997	Australian bass, *Macquaria novemaculeata*	No effect on survival at 1,000 and pH=1,000; Reduce survival by 63% at 500 and pH=4.0	Not North American species; dilution water not characterized
Ingersoll	The effects of pH, aluminum, and calcium on survival and growth of brook trout (*Salvelinus fontinalis*) early life stages	1986	Brook trout, *Salvelinus fontinalis*	-	Survival problems; low fertility success
Ingersoll et al.	Epidermal response to pH, aluminum, and calcium exposure in brook trout (*Salvelinus fontinalis*) fry	1990b	Brook trout, *Salvelinus fontinalis*	-	Only two exposure concentrations; too few test organisms per concentration
Jagoe and Haines	Changes in gill morphology of Atlantic salmon (*Salmo salar*) smolts due to addition of acid and aluminum to stream water	1997	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration, increasing Al concentration over time
Jain et al.	Acute and chronic toxicity of aluminium fluoride to flora and fauna in a microcosm	2012	Duckweed, *Lemna aequinoctialis* Cladoceran, *Daphnia similis* Western mosquitofish, *Gambusis affinis*	-	Inapporiate form of toxicant (Aluminum floride)
Jan and Matsumoto	Early effects of aluminium on nutrient (K, Ca, and Mg) status of different root zones of two rice cultivars	1999	Rice, *Oryza sativa*	-	No control group; only one exposure concentration
Jan and Pettersson	Effects of low aluminium levels on growth and nutrient relations in three rice cultivars with different tolerances to aluminium	1993	Rice, *Oryza sativa*	-	Bioaccumulation study: exposure concentrations not measured
Jancula et al.	Effects of polyaluminium chloride on the freshwater invertebrate Daphnia magna	2011	-	-	Inappropriate form of toxicant; PAX-18 (9% Al)
Jaworska and Tomasik	Metal-metal interactions in biological systems. Part VI. Effects of some metal ions on mortality, pathogenicity and reproductivity of Steinernema carpocapsae and Heterohabditis bacteriophora entomopathogenic nematodes under laboratory conditions	1999	Nematode, *Steinernema carpocapsae*	-	Distilled water without proper salts added
Jaworska et al.	Effect of metal ions under laboratory conditions on the entomopathogenic Steinernema carpocapsae (Rhabditida: sterinernematidae)	1996	Nematode, *Steinernema carpocapsae*	-	Distilled water without proper salts added; infected test organism

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Jaworska et al.	Effect of metal ions on the entomopathogenic nematode *Heterorhabditis bacteriophora* Poinar (Nematode: Heterohabditidae) under laboratory conditions	1997	Nematode, *Heterorhabditis bacteriophora*	-	Distilled water without proper salts added
Jay and Muncy	Toxicity to channel catfish of wastewater from an Iowa coal beneficiation plant	1979	-	-	Effluent
Jensen and Malte	Acid-base and electrolyte regulation, and haemolymph gas transport in crayfish, *Astacus astacus*, exposed to soft, acid water with and without aluminum	1990	Crayfish, *Astacus astacus*	21 d No effect on haemolymph haemocyanin concentration at 675	Not North American species, only one exposure concentration
Jensen and Weber	Internal hypoxia-hypercapnia in tench exposed to aluminum in acid water: Effects on blood gas transport, acid-base status and electrolyte composition in arterial blood	1987	Tench, *Tinca tinca*	-	Surgically altered test species
Ji et al.	Toxicity of oxide nanoparticles to the green algae *Chlorella sp.*	2011	Green alga, *Chlorella sp.*	-	Inappropriate form of toxicant (aluminum oxide)
Jones	The relation between the electrolytic solution pressures on the metals and their toxicity to the stickleback (*Gasterosteus aculeatus* L.)	1939	Threespine stickleback, *Gasterosteus aculeatus*	-	Lack of details; review
Jones	A further study of the relation between toxicity and solution prssure, with *Polycelis nigra* as test animals	1940	Planarian, *Polycelis nigra*	48 hr Survival time affected at 100,000	Not North American species; distilled water without proper salts
Jones et al.	Comparison of observed and calculated concentrations of dissolved Al and Fe in stream water	1974	-	-	Not applicable; no aluminum toxicity data
Jonsson et al.	Metals and linear alkylbenzene sulphonate as inhibitors of the algae *Pseudokirchneriella subcapitata* acid phosphatase activity	2009	Green alga, *Pseudokirchneriella subcapitata*	7 d Decrease relative acitivity at 53,960	Only two exposure concentrations
Juhel et al.	Alumina nanoparticles enhance growth of Lemna minor	2011	Duckweed, *Lemna minor*	-	Inappropriate form of toxicant; nanoparticles
Kadar et al.	Avoidance responses to aluminum in the freshwater bivalve, *Anodonta cygnea*	2001	Swan mussel, *Anodonta cygnea*	15 d Decrease in duration of shell gape at 516.3	Not North American species
Kadar et al.	Effect of sub-lethal concentrations of aluminum on the filtration activity of the freshwater mussel *Anodonta cygnea* L. At Neutral Ph	2002	Swan mussel, *Anodonta cygnea*	15 d Duration of siphon activity at 241.3	Not North American species, only two exposure concentrations

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Kaiser	Correlation and prediction of metal toxicity to aquatic biota	1980	-	-	Review; results of previously published papers
Karlsson-Norrgren et al.	Acid water and aluminum exposure: experimentally induced gill lesions in brown trout, *Salmo trutta* L	1986a	Brown trout, *Salmo trutta*	21 d Alteration in secondary gill lamellae at 200	Too few exposure concentrations, atypical endpoint
Karlsson-Norrgren et al.	Acid water and aluminum exposure: Gill lesions and aluminum accumulation in farmed brown trout, *Salmo trutta* L.	1986b	Brown trout, *Salmo trutta*	-	Bioaccumulation: survey; exposure concentration not measured over time
Keinanen et al.	Ion regulation in whitefish (*Coregonus lavaretus* L.) yolk-sac fry exposed to low pH and aluminum at low and moderate ionic strength	1998	Whitefish, *Coregonus lavaretus*	-	Not North American species; cannot determine effect concentration
Keinanen et al.	Comparison of the responses of the yolk-sac fry of pike (*Esox lucius*) and roach (*Rutilus rutilus*) to low pH and aluminum: sodium influx, development and activity	2000	Pike, *Esox lucius* Roach, *Rutilus rutilus*	10 d NOEC (growth)=600 at pH=5.0; 9 d LOEC (survival)=100 at pH=5.25	Too few exposure concentrations
Keinanen et al.	Fertilization and embryonic development of whitefish (*Coregonus lavaretus lavaretus*) in acidic low-ionic strength water with aluminum	2003	Whitefish, *Coregonus lavaretus lavaretus*	Decrease fertilization % and fertilization rate at 250	Not North American species; only one exposure concentration, duration, exposure methods unknown
Keinanen et al.	The susceptibility of early developmental phases of an acid-tolerant and acid-sensitive fish species to acidity and aluminum	2004	Pike, *Esox lucius*	-	Mixture; dilution water is lake water
Khangarot and Das	Acute toxicity of metals and reference toxicants to a freshwater ostracod, *Cypris subglobosa* Sowerby, 1840 and correlation to EC50 values of other test models	2009	Ostracod, *Cypris subglobosa*	-	Inappropriate form of toxicant (aluminum ammonia sulfate)
Kinross et al.	The influence of pH and aluminum on the growth of filamentous algae in artificial streams	2000	Alga (various species)	~3 d Decrease growth rate at 199.6	Only one exposure concentration
Kitamura	Relation between the toxicity of some toxicants to the aquatic animals (*Tanichthys albonubes* and *Neocaridina denticulata*) and the hardness of the test solution	1990	White cloud mountain minnow, *Tanichthys albonubes*	48 hr LC50=>100,000	Not North American species; text in foreign language
Klaprat et al.	The effect of low pH and aluminum on the olfactory organ of rainbow trout, *Salmo gairdneri*	1988	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Klauda and Palmer	Responses of bluback herring eggs and larvae to pulses of aluminum	1987	Blueback herring, *Alosa aestivalis*	-	Pulsed exposures to pollutant

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Klauda et al.	Sensitivity of early life stages of blueback herring to moderate acidity and aluminum in soft freshwater	1987	Blueback herring, *Alosa aestivalis*	-	Poor control survival (>10%)
Klimek et al.	The toxicity of aluminium salts to *Lecane inermis* rotifers: Are chemical and biological methods used to overcome activated sludge bulking mutually exclusive?	2013	Rotifer, *Lecane inermis*	24 hr EC50=12	Dilution water not characterized
Kline	The effects of organic complexation on aluminum toxicity to rainbow trout (*Oncorhynchus mykiss*)	1992	Rainbow trout, *Oncorhynchus mykiss*	-	Only two exposure concentrations; effect for inorganic Al not total Al
Klusek et al.	Trace element concentrations in the soft tissue of transplanted freshwater mussels near a coal-fired power plant	1993	Eastern lampmussel, *Lampsilis radiata*	-	Field exposure, exposure concentrations not measured
Knapp and Soltero	Trout-zooplankton relationships in Medical Lake, WA following restoration by aluminum sulfate treatment	1983	-	-	Field study, exposure concentration unknown
Kobbia et al.	Studies on the effects of some heavy metals in the biological activities of some phytoplankton species. I. differential tolerance of some Nile phytoplanktonic populations in cultures to the effects of some heavy metals	1986	-	-	Mixed species exposure
Kovacevic et al.	The effect of aluminum on the planarian *Polycelis felina* (Daly.)	2009a	Planarian, *Polycelis felina*	5 d No mortality at 200,000 and pH=6.14	Not North American species
Kovacevic et al.	Aluminum deposition in hydras	2009b	Hydra	-	Bioaccumulation: steady state not reached; static, unmeasured exposure
Krishnasamy and Seshu	Phosphine fumigation influence on rice seed germination and vigor	1990	Rice, *Oryza sativa*	-	Not applicable; no aluminum toxicity data
Kroglund et al.	Exposure to moderate acid water and aluminum reduces Atlantic salmon post-smolt survival	2007	Atlantic salmon, *Salmo salar*	-	Dilution water not characterized; mixture
Kroglund et al.	Water quality limits for Atlantic salmon (*Salmo salar* L.) exposed to short term reductions in pH and increased aluminum simulating episodes	2008	Atlantic salmon, *Salmo salar*	-	Review; results of previously published papers
Kroglund et al.	Recovery of Atlantic salmon smolts following aluminum exposure defined by changes in blood physiology and seawater tolerance	2012	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration; no control group

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Kudlak et al.	Determination of EC50 in toxicity data of selected heavy metals toward *Heterocypris incongruens* and their comparison to "direct-contact" and microbiotests	2011	Ostracod, *Heterocypris incongruens*	-	Sediment contact test; dilution water is distilled water
Kure et al.	Molecular responses to toxicological stressors: Profiling microRNAs in wild Atlantic salmon (*Salmo salar*) exposed to acidic aluminum-rich water	2013	Atlantic salmon, *Salmo salar*	72 hr Decrease sodium and chloride and increase glucose in blood plasma at 123-128	Only one exposure concentration; no true control group
Lacroix et al.	Aluminum dynamics on gills of Atlantic salmon fry in the presence of citrate and effects on integrity of gill structures	1993	Atlantic salmon, *Salmo salar*	-	Mixture; Al and citrate
Laitinen and Valtonen	Cardiovascular, ventilatory and haematological responses of brown trout (*Salmo trutta* L.), to the combined effects of acidity and aluminum in humic water at winter temperatures	1995	Brown trout, *Salmo trutta*	-	Mixture; dilution water is river water
Lange	Toxicity of aluminum to selected freshwater invertebrates in water of pH 7.5	1985	Fingernail clam, *Sphaerium sp.*	4 d LC50=2,360	High control mortality (26.7%)
Lee and Hughes	A plant bioassay protocol for sediment heavy metal toxicity studies using wild rice as an indicator species	1998	Rice, *Oryza sativa*	-	Exposure medium not defined; hard to determine effect concentration
Lee et al.	Zebrafish transgenic line huORFZ is an effective living bioindicator for detecting environmental toxicants	2014	Zebrafish, *Danio rerio*	-	Distilled water without proper salts added
Leino and McCormick	Response of juvenile largemouth bass to different pH and aluminum levels at overwintering temperatures: effects on gill morphology, electrolyte balance, scale calcium, liver glycogen, and depot fat	1993	Largemouth bass, *Micropterus salmoides*	84 d Increase respiratory barrier thickness and interlamellar epithelial thickness in gills at 29.2	Only one exposure concentration; too few animals per concentration
Leino et al.	Effects of acid and aluminum on swim bladder development and yolk absorption in the fathead minnow, *Pimephales promelas*	1988	Fathead minnow, *Pimephales promelas*	38 % hatching success at 25	Only two exposure concentrations, lack of details
Leino et al.	Multiple effects of acid and aluminum on brood stock and progeny of fathead minnows, with emphasis on histopathology	1990	Fathead minnow, *Pimephales promelas*	-	Repeat of used paper (Leino et al. 1989)
Li and Zhang	Toxic effects of low pH and elevated Al concentration on early life stages of several species of freshwater fishes	1992	Grass carp, *Ctenopharingodon idella*	4 d LC50=260	Lack of exposure details; text in foreign language

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Li et al.	Responses of *Ceriodaphnia dubia* to TiO_2 and Al_2O_3 nanoparticles: A dynamic nano-toxicity assessment of energy budget distribution	2011	Cladoceran, *Ceriodaphnia dubia*	-	Inappropriate form of toxicant (nanoparticles)
Li et al.	Surface interactions affect the toxicity of engineered metal oxide nanoparticles toward *Paramecium*	2012	Protozoa, *Paramecium micronucleatum*	-	Inappropriate form of toxicant (nanoparticles)
Lincoln et al.	Quality-assurance data for routine water analyses by the U.S. Geological Survey laboratory in Troy, New York - July 2005 through June 2007	2009	-	-	Not applicable; no aluminum toxicity data
Lindemann et al.	The impact of aluminum on green algae isolated from two hydrochemically different headwater streams, Bavaria, Germany	1990	Green alga, *Chlorella sp.* Green alga, *Scenedesmus sp.*	-	Exposure concentration varied over time
Linnik	Aluminum in natural waters: content, forms of migration, toxicity	2007	-	-	Review; results of previously published papers
Lithner et al.	Bioconcentration factors for metals in humic waters at different pH in the Ronnskar area (N. Sweden)	1995	-	-	Exposure concentration not known; field accumulation study
Lockard and McWalter	Effects of toxic levels of sodium, arsenic, iron and aluminium on the rice plant	1956	Rice	-	Scientific name not provided
Macova et al.	Polyaluminium chloride (PAX-18) - acute toxicity and toxicity for early development stages of common carp (*Cyprinus carpio*)	2009	Common carp, *Cyprinus carpio*	-	Inappropriate form of toxicant, PAX-18 (9% Al)
Macova et al.	Acute toxicity of the preparation PAX-18 for juvenile and embryonic stages of zebrafish (*Danio rerio*)	2010	Zebrafish, *Danio rerio*	-	Inappropriate form of toxicant, PAX-18 (9% Al)
Madigosky et al.	Concentrations of aluminum in gut tissue of crayfish (*Procambarus clarkii*), purged in sodium chloride	1992	Crayfish, *Procambarus clarkii*	-	Exposure concentration not known; field accumulation study
Maessen et al.	The effects of aluminum/calcium and pH on aquatic plants from poorly buffered environments	1992	-	-	Only one exposure concentration; sediment
Malcolm et al.	Relationships between hydrochemistry and the presence of juvenile brown trout (*Salmo trutta*) in headwater streams recovering from acidification	2012	Brown trout, *Salmo trutta*	-	Survey
Malecki-Brown et al.	Alum application to improve water quality in a municipal wastewater treatment wetland: Effects on macrophyte growth and nutrient uptake	2010	Aquatic vegatation	-	Only one exposure concentration; dilution water not characterized; mixture
Malley and Chang	Effects of aluminum and acid on calcium uptake by the crayfish *Orconectes virilis*	1985	Crayfish, *Orconectes virilis*	-	No aluminum toxicity data; calcium uptake with Al treatment

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Malley et al.	Changes in the aluminum content of tissues of crayfish held in the laboratory and in experimental field enclosures	1986	Crayfish, *Orconectes virilis*	-	Mixture; sediment
Malley et al.	Effects on ionic composition of blood tissues of *Anodonta grandis grandis* (Bivalvia) of an addition of aluminum and acid to a lake	1988	Mussel, *Anodonta grandis grandis*	-	Exposure concentrations not known; Al dosed in a lake
Malte	Effects of aluminum in hard, acid water on metabolic rate, blood gas tensions and ionic status in the rainbow trout	1986	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Malte and Weber	Respiratory stress in rainbow trout dying from aluminum exposure in soft, acid water, with or without added sodium chloride	1988	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Mao et al.	Assessment of sacrificial anode impact by aluminum accumulation in mussel *Mytilus edulis*: a large-scale laboratory test	2011	Bay mussel, *Mytilus edulis*	-	Inappropriate form of toxicant; Al anode
Markarian et al.	Toxicity of nickel, copper, zinc and aluminum mixtures to the white sucker (*Catostomus commersoni*)	1980	White sucker, *Catostomus commersoni*	-	Mixture; industrial effluent streams
Marquis	Aluminum neurotoxicity: An experimental perspective	1982	-	-	Cannot determine effect concentration
Martin et al.	Relationships between physiological stress and trace toxic substances in the bay mussel, *Mytilus edulis*, from San Fransico Bay, California	1984	Bay mussel, *Mytilus edulis*	-	Exposure concentration not known; field accumulation study
Mayer and Ellersieck	Manual of acute toxicity: interpretation and data base for 410 chemicals and 66 species of freshwater animals	1986	-	-	Review; results of previously published papers
McCahon and Pascoe	Short-term experimental acidification of a Welsh stream: Toxicity of different forms of aluminum at low pH to fish and invertebrates	1989	-	-	Mixture; dilution water is stream water
McComick and Jensen	Osmoregulatory failure and death of first-year largemouth bass (*Micropterus salmoides*) exposed to low pH and elevated aluminum at low temperature in soft water	1992	Largemouth bass, *Micropterus salmoides*	84 d 56% survival at 53.9	Only one exposure concentration; duration too short
McCormick et al.	Chronic effects of low pH and elevated aluminum on survival, maturation, spawning and embryo-larval development of the fathead minnow in soft water	1989	Fathead minnow, *Pimephales promelas*	4 d 38% hatch at 49 and pH=5.5; 94% hatch at 66 and pH=7.5	Only two exposure concentrations

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McCormick et al.	Thresholds for short-term acid and aluminum impacts on Atlantic salmon smolts	2012	Atlantic salmon, *Salmo salar*	48 hr No mortality at 169 and pH=6.0; 100% mortality at 184 and pH=5.3	Too few exposure concentrations; duration too short
McCrohan et al.	Bioaccumulation and toxicity of aluminum in the pond snail at neutral pH	2000	Snail, *Lymnaea stagnalis*	-	Dilution water not characterized; lack of exposure details
McDonald and Milligan	Sodium transport in the brook trout, Salvelinus fontinalis: effects of prolonged low pH exposure in the presence and absence of aluminum	1988	Brook trout, *Salvelinus fontinalis*	-	Only one exposure concentration; pre-exposure to pollutant
McDonald et al.	Nature and time course of acclimation to aluminum in juvenile brook trout (*Salvelinus fontinalis*). I. Physiology	1991	Brook trout, *Salvelinus fontinalis*	-	Exposure concentration varied over time; changed dose mid experiment
McKee and Wolf	Water quality criteria. 2nd Edition	1963	-	-	Review; results of previously published papers
McLeish et al.	Skin exposure to micro- and nano-particles can cause haemostasis in zebrafish larvae	2010	Zebrafish, *Danio rerio*	-	Inappropriate form of toxicant (nanoparticles)
Mehta et al.	Relative toxicity of some non-insecticidal chemicals to the free living larvae guinea-worm (*Dracunuculus medinensis*)	1982	Guinea worm (larvae), *Dracunculus medinensis*	24 hr LC50=16,218	Lack of details; dilution water not characterized; exposure methods unknown
Meili and Wills	Seasonal concentration changes of Hg, Cd, Cu and Al in a population of roach	1985	Roach, *Rutilus rutilus*	-	Not North American species; exposure concentration not known; field accumulation study
Meland et al.	Exposure of brown trout (Salmo trutta L.) to tunnel wash water runoff -- Chemical characterisation and biological impact	2010	Brown trout, *Salmo trutta*	-	Mixture; run-off
Mendez	Water-quality data from storm runoff after the 2007 fires, San Diego County, California	2010	-	-	Survey; occurrence
Merrett et al.	The response of macroinvertebrates to low pH and increased aluminum concentrations in Welsh streams: Multiple episodes and chronic exposure	1991	-	-	Mixture; exposure concentration varied over time; dilution water is stream water
Mersch et al.	Transplanted aquatic mosses for monitoring trace metal mobilization in acidified streams of the Vosges Mountains, France	1993	Moss, *Amblystegium riparium*	-	Field exposure, exposure concentrations not measured
Michailova et al.	Functional and structural rearrangements of salivary gland polytene chromosomes of *Chironomus riparius* Mg. (Diptera, Chironomidae) in response to freshly neutralized aluminum	2003	Midge, *Chironomus riparius*	24-25 d Higher frequency of numerous somatic aberrations at 500	Only one exposure concentration

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Minzoni	Effects of aluminum on different forms of phosphorus and freshwater plankton	1984	Zooplankton community	-	Only one exposure concentration
Mitchell	The effects of aluminum and acidity on algal productivity: a study of an effect of acid deposition	1982	Green alga, *Selenastrum capricornutum*	4 hr Productivity drops at 5,000	Lack of details; abstract only
Mo et al.	A study of the uptake by duckweed of aluminum, copper, and lead from aqueous solution	1988	Duckweed	-	No scientific name of test species provided
Mohanty et al.	Effect of a low dose of aluminum on mitotic and meiotic activity, 4C DNA content, and pollen sterility in rice, *Oryza sativa* L	2004	Rice, *Oryza sativa*	-	Only one exposure concentration; distilled water without proper salts added
Monette	Impacts of episodic acid and aluminum exposure on the physiology of Atlantic salmon, *Salmo salar*, smolt development	2007	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration; pulse exposures
Monette and McCormick	Impacts of short-term acid and aluminum exposure on Atlantic salmon (*Salmo salar*) physiology: a direct comparison of parr and smolts	2008	Atlantic salmon, *Salmo salar*	-	Review of Monette 2007
Monette et al.	Effects of short-term acid and aluminum exposure on the parr-smolt transformation in the Atlantic slamon (*Salmo salar*): disruption of seawater tolerance and endocrine status	2008	Atlantic salmon, *Salmo salar*	-	Review of Monette 2007
Monette et al.	Physiological, molecular, and cellular mechanisms of impaired seawater tolerance following exposure of Atlantic salmon, *Salmo salar*, smolts to acid and aluminum	2010	Atlantic salmon, *Salmo salar*	6 d NOEC (mortality)=43; LOEC=71	Only two exposure concentrations;
Morgan et al.	A plant toxicity test with the moss *Physcomitrella patens* (Hedw.) B.S.G.	1990	Moss, *Physcomitrella patens*	-	Lack of details; toxicity information not discernible
Morgan et al.	An aquatic toxicity test using the moss *Physcomitrella patens* (Hedw) B.S.G.	1993	Moss, *Physcomitrella patens*	-	Lack of details; toxicity information not discernible
Mothersill et al.	Multiple stressor effects of radiation and metals in salmon (*Salmo salar*)	2007	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration; too few fish per exposure concentration (3 per treatment)
Mount et al.	Effect of long-term exposure to acid, aluminum, and low calcium in adult brook trout (*Salvelinus fontinalis*). 1. survival, growth, fecundity, and progeny survival	1988a	Brook trout, *Salvelinus fontinalis*	-	Mixture; low pH and Al
Mount et al.	Effect of long-term exposure to acid, aluminum, and low calcium in adult brook trout (*Salvelinus fontinalis*). 2. vitellogenesis and osmoregulation	1988b	Brook trout, *Salvelinus fontinalis*	-	Mixture; low pH and Al

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Mount et al.	Response of brook trout (*Salvelinus fontinalis*) fry to fluctuating acid, aluminum, and low calcium exposures	1990	Brook trout, *Salvelinus fontinalis*	-	Pre-exposure to pollutant; only two exposure concentrations
Mueller et al.	Nature and time course of acclimation to aluminum in juvenile brook trout (*Salvelinus fontinalis*). II. Gill histology	1991	Brook trout, *Salvelinus fontinalis*	-	Only one exposure concentration; exposure concentration varied over time
Mukai	Effects of chemical pretreatment on the germination of statoblasts of the freshwater bryozoan, *Pectinatella gelatinosa*	1977	Bryozoa, *Pectinatella gelatinosa*	-	Not applicable; no aluminum toxicity data
Mulvey et al.	Effects of potassium aluminium sulphate (alum) used in an *Aeromonas salmonicida* bacterin on Atlantic salmon, *Salmo salar*	1995	Atlantic salmon, *Salmo salar*	-	Inject toxicant; inappropriate form of toxicant (potassium aluminum sulphate)
Muniz and Leivestad	Toxic effects of aluminum on the brown trout, *Salmo trutta* L.	1980b	Brown trout, *Salmo trutta*	-	Mixture; dilution water is breakwater
Muniz et al.	Physiological response of brown trout (*Salmo trutta*) spawners and postspawners to acidic alminum-rich stream water	1987	Brown trout, *Salmo trutta*	-	Field exposure, exposure concentrations not measured
Muramoto	Influence of complexans (NTA, EDTA) on the toxicity of aluminum chloride and sulfate to fish at high concentrations	1981	Common carp, *Cyprinus carpio*	48 hr 30% mortality at 8,000 and pH=6.3	Dilution water not characterized
Murungi and Robinson	Synergistic effects of pH and aluminum concentrations on the life expectancy of tilapia (Mozambica) fingerlings	1987	-	-	Scientific name not given
Murungi and Robinson	Uptake and accumulation of aluminum by fish - the modifying effect of added ions	1992	Shiners, *Notropis sp.*	96 hr Whole fish tissue = 0.78 mg/g (dry weight) at 5,000	Lack of details, exposure methods unknown
Musibono and Day	Active uptake of aluminum, copper, and manganese by the freshwater amphipod *Paramelita nigroculus* in acidic waters	2000	Amphipod, *Paramelita nigroculus*	-	Not North American species; mixture
Nagasaka et al.	Novel iron-storage particles may play a role in aluminum tolerance of *Cyanidium caldarium*	2002	Red alga, *Cyanidium caldarium*	-	Only one exposure concentration; mixture (low pH and Al)
Naskar et al.	Aluminum toxicity induced poikilocytosis in an air-breathing telost, *Clarias batrachus* (Linn.)	2006	Catfish, *Clarias batrachus*	5 d Some membrane abnormalities with red blood cells at 165,000	Only two exposure concentrations; non-wild population test animals

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Neave et al.	The transcriptome and proteome are altered in marine polychaetes (Annelida) exposed to elevated metal levels	2012	Polychaete, *Ophelina sp.*	-	Mixture; field study: exposure concentration not known
Negri et al.	Effects of alumina refinery wastewater and signature metal constituents at the upper thermal tolerance of: 2. The early life stages of the coral *Acropora tenuis*	2011	Coral, *Acropora tenuis*	-	Not North American species
Neville	Physiological response of juvenile rainbow trout, Salmo gairdneri, to acid and aluminum - prediction of field responses from laboratory data	1985	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Neville and Campbell	Possible mechanisms of aluminum toxicity in a dilute, acidic environment to fingerlings and older life stages of salmonids	1988	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Nilsen et al.	Effects of acidic water and aluminum exposure on gill Na+, K+ -ATPase α-subunit isoforms, enzyme activity, physiology and return rates in Atlantic salmon (*Salmo salar* L.)	2010	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration; dilution water not characterized
Nilsen et al.	Atlantic salmon (*Salmo salar* L.) smolts require more than two weeks to recover from acidic water and aluminum exposure	2013	Atlantic salmon, *Salmo salar*	7 d, 86 Gill content=26.6 µg/g dw at pH=5.7	Only one exposure concentration; not whole body or muscle
Norrgren and Degerman	Effects of different water qualities on the early development of Atlantic salmon and brown trout exposed in situ	1993	-	-	Mixture; no control group; dilution water is river water
Norrgren et al.	Accumulation and effects of aluminum in the minnow (*Phoxinus phoxinus* L.) at different pH levels	1991	Minnow, *Phoxinus phoxinus*	48 d No effect on mortality at 174 and pH=7.1; Increase mortality at 168 and pH=5.9	Only one exposure concentration
Nyberg et al.	Labile inorganic manganese - An overlooked reason for fish mortality in acidified streams?	1995	Brown trout, *Salmo trutta*	-	Field exposure, exposure concentrations not measured
Odonnell et al.	A review of the toxicity of aluminum in fresh water	1984	-	-	Review
Olaveson and Nalewajko	Effects of acidity on the growth of two *Euglena* species	2000	Alga, *Euglena mutabilis* Alga, *Euglena gracilis*	-	Mixture (low pH and Al)
Ormerod et al.	Short-term experimental acidification of Welsh stream: Comparing the biological effects of hydrogen ions and aluminum	1987	-	-	Mixture; dilution water is river water

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
OSU (Oregon State University)	Chronic toxicity of aluminum, at pH6, to the freshwater duckweed, *Lemna minor*	2012d	Duckweed, *Lemna minor*	-	Excessive EDTA used
Pagano et al.	Use of sea urchin sperm and embryo bioassay in testing the sublethal toxicity of realistic pollutant levels	1989	-	-	Mixture; effluent
Pagano et al.	Cytogenetic, developmental, and biochemical effects of aluminum, iron, and their mixture in sea urchins and mussels	1996	-	-	Lack of details; exposure duration not reported; cannot determine effect concentration
Pakrashi et al.	Cytotoxicity of aluminium oxide nanoparticles towards fresh water algal isolate at low exposure concentrations	2013a	Alga, *Chlorella ellipsoids*	-	Inappropriate form of toxicant (nanoparticles)
Pakrashi et al.	*Ceriodaphnia dubia* as a potential bio-indicator for assessing acute aluminum oxide nanoparticle toxicity in fresh water environment	2013b	Cladoceran, *Ceriodaphnia dubia*	-	Inappropriate form of toxicant (nanoparticles)
Paladino and Swartz	Interactive and synergistic effects of temperature, acid and aluminum toxicity on fish critical thermal tolerance	1984	-	-	Scientific name not given; lack of exposure details; abstract only
Palmer et al.	Comparative sensitivities of bluegill, channel catfish and fathead minnow to pH and aluminum	1988	Bluegill, *Lepomis macrochirus* Fathead minnow, *Pimephales promelas* Channel catfish, *Ictalurus punctatus*	Exposure concentrations overlapped (all over the place)	Exposure concentrations overlapped
Panda and Khan	Lipid peroxidation and oxidative damage in aquatic duckweed (*Lemna minor* L.) in response to aluminum toxicity	2004	Duckweed, *Lemna minor*	-	Cannot determine effect concentration, dilution media not defined; no statistical analysis
Pandey et al.	Salicylic acid alleviates aluminum toxicity in rice seedlings better than magnesium and calcium by reducing aluminum uptake, suppressing oxidative damage and increasing antioxidative defense	2013	Rice, *Oryza sativa*	12 d Reduced root and shoot length at 13,494	Only one exposure concentration
Papathanasiou et al.	Toxicity of aluminium in natural waters controlled by type rather than quantity of natural organic matter	2011	Snail, *Lymnaea stagnalis*	24 d Decrease mean eggs/day at 500	Only one exposure concentration
Parent et al.	Influences of natural dissolved organic matter on the interaction of aluminum with the microalga Chlorella: a test of free-ion model of trace metal toxicity	1996	Green alga, *Chlorella pyrenoidosa*	-	Mixture; Al and soil fluvic acid

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Parkhurst et al.	Inorganic monomeric aluminum and pH as predictors of acidic water toxicity to brook trout (Salvelinus fontinalis)	1990	Brook trout, *Salvelinus fontinalis*	-	Only three exposure concentrations, difficult to determine effect concentration
Parsons Engineering Science, Inc.	Aluminum water-effect ratio study for the calculation of a site-specific water quality standard in Welsh reservoir	1997	Cladoceran, *Ceriodaphnia dubia* Fathead minnow, *Pimephales promela*	-	Mixture; power plant effluent
Pauwels	Some effects of exposure to acid and aluminum on several lifestages of the Atlantic salmon (*Salmo salar*)	1990	Atlantic salmon, *Salmo salar*	24 d Mortality increased faster at 106 and pH=5.25	Only one exposure concentration
Payton and Greene	A comparison of the effect of aluminum on a single species algal assay and indigenous community algal toxicity bioassay	1980	Green alga, *Scenedesmus bijgua*	-	Lack of details; duration and exposure methods not provided
Peterson et al.	Responses of Atlantic salmon (*Salmo salar*) alevins to dissolved organic carbon and dissolved aluminum at low pH	1989	Atlantic salmon, *Salmo salar*	-	Poor control survival; only two exposure concentrations
Pettersson et al.	Physiological and structural responses of the cyanobacterium *Anabaena cylindrica* to aluminum	1985a	Blue-green alga, *Anabaena cylindrica*	-	Excessive EDTA used (672.52 µg/L)
Pettersson et al.	Accumulation of aluminum by *Anabaena cylindrica* into polyphosphate granules and cell walls: an X-ray energy-dispersive microanalysis study	1985b	Blue-green alga, *Anabaena cylindrica*	-	Bioaccumulation: not renewal or flow-through
Pettersson et al.	Aluminum uptake by A*nabaena cylindrica*	1986	Blue-green alga, *Anabaena cylindrica*	-	Bioaccumulation: not renewal or flow-through; steady state not reached
Pettersson et al.	Aluminum effects on uptake and metabolism of phosphorus by the cyanobacterium *Anabaena cylindrica*	1988	Blue-green alga, *Anabaena cylindrica*	-	Only two exposure concentrations; cannot determine effect concentration; no statistical analysis
Peuranen et al.	Effects of acidity and aluminum on fish gills in laboratory experiments and in the field	1993	Whitefish, *Coregonus lavaretus*	143 d Decrease of respiratory diffusing capacity at 150 and pH=4.75	Not North American species; dilution water not characterized; only one exposure concentration
Phillips and Russo	Metal bioaccumulation in fishes and aquatic invertebrates: A literature review	1978	-	-	Review
Piasecki and Zacharzewski	Influence of coagulants used for lake restoration on *Daphnia magna* Straus (Crustacea, Cladocera)	2010	Cladoceran, *Daphnia magna*	-	Inappropriate form of toxicant, PIX 113 and PAX 18

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Playle	Physiological effects of aluminum on rainbow trout in acidic soft water, with emphasis on the gill micro-environment	1989	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Playle and Wood	Water pH and aluminum chemistry in the gill micro-environment of rainbow trout during acid and aluminum exposures	1989	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Playle and Wood	Mechanisms of aluminum extraction and accumulation at the gills of rainbow trout, *Oncorhynchus mykiss* (Walbaum), in acidic soft water	1991	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Playle et al.	Physiological disturbances in rainbow trout during acid and aluminum exposures	1988	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Playle et al.	Physiological disturbances in rainbow trout (*Salmo gairdneri*) during acid and aluminum exposures in soft water of two calcium concentrations	1989	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Poleo	Temperature as a major factor concerning fish mortality in acidic Al-rich waters: Experiments with young stage Atlantic salmon (*Salmo salar* L.)	1992	Atlantic salmon, *Salmo salar*	-	Text in foreign language
Poleo	Aluminum polymerization: a mechanism of acute toxicity of aqueous aluminum to fish	1995	-	-	Review
Poleo and Muniz	Effect of aluminum in soft water at low pH and different temperatures on mortality, ventilation frequency and water balance in smoltifying Atlantic salmon, *Salmo salar*	1993	Atlantic salmon, *Salmo salar*	LT50=49 hr at 271 (1˚C); LT50=21 hr at 272 (10˚C)	Only one exposure concentration; no control group
Poleo et al.	The influence of temperature on aqueous aluminum chemistry and survival of Atlantic salmon (*Salmo salar* L.) fingerlings	1991	Atlantic salmon, *Salmo salar*	LT50=170 hr at 403 (1˚C); LT50=46 hr at 402 (10˚C)	Only one exposure concentration; no control group
Poleo et al.	Survival of crucian carp, *Carassius carassius*, exposed to a high low-molecular weight inorganic aluminum challenge	1995	Crucian carp, *Carassius carassius*	-	Not North American species; only two exposure concentrations; no true control group
Poleo et al.	Toxicity of acid aluminum-rich water to seven freshwater fish species: a comparative laboratory study	1997	-	-	Too few organisms per treatment, 1-2 fish per treatment
Poleo et al.	The effect of various metals on *Gyrodactylus salaris* (Plathyrlminthes, Monogenea) infections in Atlantic salmon (*Salmo salar*)	2004	Parasite, *Gyrodactylus salaris* Atlantic salmon, *Salmo salar*	-	Two species tested with one exposure; not sure how much exposure to the parasite

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Pond et al.	Downstream effects of mountaintop coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools	2008	-	-	Field survey, mixture
Poor	Effect of lake management efforts on the trophic state of a subtropical shallow lake in Lakeland, Florida, USA	2010	-	-	Survey
Poston	Effects of dietary aluminum on growth and composition of young Atlantic salmon	1991	Atlantic salmon, *Salmo salar*	-	Fed pollutant
Prange and Dennison	Physiological responses of five seagrass species to trace metals	2000	Seagrass	-	Exposure concentration not known; field accumulation study
Pribyl et al.	Cytoskeletal alterations in interphase cells of the green alga Spirogyra decimina in response to heavy metals exposure: I. the effect of cadmium	2005	Green alga, *Spirogyra decimina*	-	Not applicable; cadmium study
Pynnonen	Aluminum accumulation and distribution in the freshwater clams (Unionidae)	1990	Mussel, *Anodonta anatina* Mussel, *Unio pictorum*	-	Not North American species; exposure concentrations varied too much over time
Quiroz-Vazquez et al.	Bioavailability and toxicity of aluminum in a model planktonic food chain (*Chlamydomonas-Daphnia*) at neutral pH	2010	-	-	Bioaccumulation: not renewal or flow-through; steady state not reached
Radic et al.	Ecotoxicological effects of aluminum and zinc on growth and antioxidants in *Lemna minor* L.	2010	Duckweed, *Lemna minor*	15 d NOEC (relative growth rate)=4,047; LOEC=8,094	
Rahman et al.	Varietal differences in the growth of rice plants in response to aluminum and silicon	1998	Rice	-	Scientific name not given
Rai et al.	Physiological and biochemical responses of *Nostoc linckia* to combined effects of aluminium, fluoride and acidification	1996	Cyanobacteria, *Nostoc linckia*	15 d pH 7.5 LC50=121.4, pH 6.0 LC50=11.13, pH4.5 LC50=3.643	Only three exposure concentrations
Rajesh	Toxic effect of aluminum in *Oreochromis mossambicus* (Peters)	2010	Mozambique tilapia, *Oreochromis mossambicus*	4 d LC50=8,000	Dilution water not characterized
Ramamoorthy	Effect of pH on speciation and toxicity of aluminum to rainbow trout (*Salmo gairdneri*)	1988	Rainbow trout, *Oncorhynchus mykiss*	-	Mixture
Razo-Estrada et al.	Aluminum-induced oxidative stress and apoptosis in liver of the common carp, *Cyprinus carpio*	2013	Common carp, *Cyprinus carpio*	96 hr Increase lipid peroxidation at 50	Only three exposure concentrations

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Reader and Morris	Effects of aluminium and pH on calcium fluxes, and effects of cadmium and manganese on calcium and sodium fluxes in brown trout (*Salmo trutta* L.)	1988	Brown trout, *Salmo trutta*	-	Only one exposure concentration; too few fish per exposure concentration
Reader et al.	Growth, mineral uptake and skeletal calcium deposition in brown trout, *Salmo trutta* L., yolk-sac fry exposed to aluminum and manganese in soft acid water	1988	Brown trout, *Salmo trutta*	-	Mixture, Al, NH_3, NH_4
Reader et al.	The effects of eight trace metals in acid soft water on survival, mineral uptake and skeletal calcium deposition in yolk-sac fry of brown trout, *Salmo trutta* L.	1989	Brown trout, *Salmo trutta*	30 d 0% survival at 178.1 and pH=4.5; No effect on survival at 170.0 at pH=6.5	Only one exposure concentration
Reader et al.	Episodic exposure to acid and aluminum in soft water: survival and recovery of brown trout, *Salmo trutta* L.	1991	Brown trout, *Salmo trutta*	-	No control group
Reid et al.	Acclimation to sublethal aluminum: modification of metal - gill surface interactions of juvenile rainbow trout (*Oncorhynchus mykiss*)	1991	Rainbow trout, *Oncorhynchus mykiss*	-	Only two exposure concentrations; pre-exposure to pollutant
Reznikoff	Micrurgical studies in cell physiology. II. The action of chlorides of lead, mercury, copper, iron, and aluminum on the protoplasm of Amoeba proteus	1926	-	-	Lack of exposure details; dilution water not characterized
Riseng et al.	The effect of pH, aluminum, and chelator manipulations on the growth of acidic and circumneutral species of *Asterionella*	1991	Diatom, *Asterionella ralfsii* Diatom, *Asterionella formosa*	-	Mixture; EDTA and Al
Rizzo et al.	Removal of THM precursors from a high-alkaline surface water by enhanced coagulation and behaviour of THMFP toxicity on *D. magna*	2005	Cladoceran, *Daphnia magna*	-	Not applicable; no aluminum toxicity data
Robertson and Liber	Bioassays with caged *Hyalella azteca* to determine in situ toxicity downstream of two Saskatchewan, Canada, uranium operations	2007	Amphipod, *Hyalella azteca*	-	Mixture; downstream exposure of uranium mining operation
Robertson et al.	Survival of *Cryptosporidium parvum* oocysts under various environmental pressures	1992	Parasite, *Cryptosporidium parvum*	-	Poor control survival; only two exposure concentrations
Robinson and Deano	The synergistic effects of acidity and aluminum on fish (Golden shiners) in Louisiana	1985	Golden shiner, *Notemigonus crystoleucas*	-	Dilution water not characterized; high control mortality (10-20%)

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Robinson and Deano	Acid rain: the effect of pH, aluminum, and leaf decomposition products on fish survival	1986	Golden shiner, *Notemigonus crystoleucas*	-	Only two exposure concentrations
Rosemond et al.	Comparative analysis of regional water quality in Canada using the water quality index	2009	-	-	Survey; no aluminum toxicity data
Rosseland and Skogheim	Comparative study on salmonid fish species in acid aluminum-rich water II. Physiological stress and mortality of one- and two-year-old fish	1984	-	-	Mixture; dilution water is lake water
Rosseland et al.	Mortality and physiological stress of year-classes of landlocked and migratory Atlantic salmon, brown trout and brook trout in acidic aluminium-rich soft water	1986	Atlantic salmon, *Salmo salar* Brown trout, *Salmo trutta* Brook trout, *Salvelinus fontinalis*	83 hr, pH=5.14, 228 100% mortality; 0% mortality; 0% mortality	Dilution water not characterized; only one exposure concentration
Rosseland et al.	Environmental effects of aluminum	1990	-	-	Review of previously published literature
Rosseland et al.	The mixing zone between limed and acidic river waters: Complex aluminum and extreme toxicity for salmonids	1992	-	-	Mixture; exposure concentration varied over time; dilution water is river water
Roy and Bhadra	Hematoxylin staining of seedling roots is a potential phenotypic index for screening of aluminium tolerance in rice (*Oryza sativa* L.)	2013	Rice, *Oryza sativa*	-	Not applicable, no aluminum toxicity information
Royset et al.	Diffusive gradients in thin films sampler predicts stress in brown trout (*Salmo trutta* L.) exposed to aluminum in acid fresh waters	2005	Brown trout, *Salmo trutta*	-	Mixture; dilution water is river water
Rueter et al.	Indirect aluminum toxicity to the green alga *Scenedesmus* through increased cupric ion activity	1987	Green alga, *Scenedesmus quadricauda*	-	Mixture; Al and Cu
Sacan and Balcioglu	Bioaccumulation of aluminium in *Dunaliella tertiolecta* in natural sewater: Aluminium-metal (Cu, Pb, Se) interactions and influence of pH	2001	Phytoplankton, *Dunaliella tertiolecta*	-	Bioaccumulation, steady state not documented
Sadler and Lynam	Some effects on the growth of brown trout from exposure to aluminum at different pH levels	1987	Brown trout, *Salmo trutta*	7 d NOEC (specific growth rate)=18.87 at pH=5.5; LOEC=30.04 at pH=5.5	Too few exposure concentrations; duration

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Sadler and Lynam	The influence of calcium on aluminum-induced changes in the growth rate and mortality of brown trout, *Salmo trutta* L.	1988	Brown trout, *Salmo trutta*	42 d Increase mortality at 54 and hardness from 3-6 mg/L as $CaCO_3$, but not greater than 9 mg/L	Only one exposure concentration
Salbu et al.	Environmentally relevant mixed exposures to radiation and heavy metals induce measurable stress responses in Atlantic salmon	2008	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration; mixture
Sauer	Heavy metals in fish scales: accumulation and effects on cadmium regulation in the mummichog, *Fundulus heteroclitus* L.	1986	Mummichog, *Fundulus heteroclitus*	-	Not applicable; no aluminum toxicity data
Sayer	Survival and subsequent development of brown trout, *Salmo trutta* L., subjected to episodic exposures of acid, aluminum and copper in soft water during embryonic and larval stages	1991	Brown trout, *Salmo trutta*	-	Only one exposure concentration; mixture; low pH and Al
Sayer et al.	Embryonic and larval development of brown trout, Salmo trutta L.: exposure to aluminum, copper, lead or zinc in soft, acid water	1991a	Brown trout, *Salmo trutta*	700 d 13% mortality at 161.8	Only one exposure concentration
Sayer et al.	Embryonic and larval development of brown trout, *Salmo trutta* L.: exposure to trace metal mixtures in soft water	1991b	Brown trout, *Salmo trutta*	-	Only two exposure concentrations; mixture
Sayer et al.	Effects of six trace metals on calcium fluxes in brown trout (*Salmo trutta* L.) in soft water	1991c	Brown trout, *Salmo trutta*	-	Only two exposure concentrations; mixture
Sayer et al.	Mineral content and blood parameters of dying brown trout (*Salmo trutta* L.) exposed to acid and aluminum in soft water	1991d	Brown trout, *Salmo trutta*	4 d Increase haematocrit and decrease plasma sodium levels and whole body sodium and potassium content at 273.6	Only one exposure concentration; too few organisms per concentration
Schindler and Turner	Biological, chemical and physical responses of lakes to experimental acidification	1982	-	-	Mixture, Al and low pH
Schofield and Trojnar	Aluminum toxicity to brook trout (*Salvelinus fontinalis*) in acidified waters	1980	Brook trout, *Salvelinus fontinalis*	-	Mixture; dilution water not characterized
Schumaker et al.	Zooplankton responses to aluminum sulfate treatment of Newman Lake, Washington	1993	-	-	Exposure concentrations not known
Segner et al.	Growth, aluminum uptake and mucous cell morphometrics of early life stages of brown trout, *Salmo trutta*, in low pH water	1988	Brown trout, *Salmo trutta*	5d Decrease weight and length at 230	Only one exposure concentration

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Senger et al.	Aluminum exposure alters behavioral parameters and increases acetylcholinesterase activity in zebrafish (*Danio rerio*) brain	2011	Zebrafish, *Danio rerio*	4 d Increase AChE activity in brain at 10.12 at pH=5.8 but not pH=6.8	Only one exposure concentration
Shabana et al.	Studies on the effects of some heavy metals on the biological activities of some phytoplankton species. II. The effects of some metallic ions on the growth criteria and morphology of *Anabaena oryzae* and *Aulosira fertilissima*	1986a	-	-	Lack of details; cannot determine effect concentration
Shabana et al.	Studies on the effects of some heavy metals on the biological activities os some phytoplankton species. III. Effects Al3+, Cr3+, Pb2+ and Zn 2+ on heterocyst frequency, nitrogen and phosphorus metabolism of *Anabaena oryzae* and *Aulosira fertilissima*	1986b	-	-	Lack of details; cannot determine effect concentration
Sharma et al.	Protective effect of *Spirulina* and tamarind fruit pulp diet supplement in fish (*Gambusia affinis* Baird & Girard) exposed to sublethal concentration of fluoride, aluminum and aluminum fluoride	2012	Western mosquitofish, *Gambusia affinis*	-	Only one exposure concentration; poor control survival
Shuhaimi-Othman et al.	Toxicity of eight metals to Malaysian freshwater midge larvae *Chironomus javanus* (Diptera, Chironomidae)	2011b	Midge, *Chironomus javanus*	4 d LC50=1,430	Not North American species
Shuhaimi-Othman et al.	Toxicity of metals to tadpoles of the commone Sunda toad, *Duttaphrynus melanostictus*	2012c	Sunda toad, *Duttaphrynus melanostictus*	4 d LC50=1,900	Not North American species
Siebers and Ehlers	Heavy metal action on transintegumentary absorption of glycine in two annelid species	1979	-	-	Not applicable; no aluminum toxicity data
Simon	Sediment and interstitial water toxicity to freshwater mussels and the ecotoxicological recovery of remediated acrid mine drainage streams	2005	-	-	Sediment exposure
Sivakumar and Sivasubramanian	FT-IR study of the effect of aluminum on the muscle tissue of *Cirrhinus mrigala*	2011	Carp hawk, *Cirrhinus mrigala*	4 d LC50=8,200	Not North American species; dilution water not characterized
Skogheim and Rosseland	A comparative study on salmonid fish species in acid aluminum-rich water I. Mortality of eggs and alevins	1984	Trout	-	Mixture; dilution water is lake water
Skogheim and Rosseland	Mortality of smolt of Atlantic salmon, *Salmo salar* L., at low levels of aluminum in acidic softwater	1986	Atlantic salmon, *Salmo salar*	-	Mixture; dilution water is lake water

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Skogheim et al.	Addition of NaOH, limestone slurry and finegrained limestone to acidified lake water and the effects on smolts of Atlantic salmon (*Salmo salar* L.)	1987	Atlantic salmon, *Salmo salar*	-	Prior exposure; stressed organisms
Soleng et al.	Toxicity of aqueous aluminum to the ectoparasitic monogenean *Gyrodactylus salaris*	2005	-	-	Only two exposure concentrations; two species tested with one exposure; not sure how much exposure to the parasite
Sonnichsen	Toxicity of a phosphate-reducing agent (aluminum sulphate) on the zooplankton in the lake Lyngby So	1978	-	-	Not applicable; no aluminum toxicity data
Sparling	Conditioned aversion of aluminum sulfate in black ducks	1990	Black ducks, *Anas rubripes*	-	Dietary exposure
Sparling and Lowe	Environmental hazards of aluminum to plants, invertebrates, fish and wildlife	1996a	-	-	Review; results of previously published papers
Sparling and Lowe	Metal concentrations of tadpoles in experimental ponds	1996b	-	-	Exposed through soil
Sparling and Lowe	Metal concentrations in aquatic macrophytes as influenced by soil and acidification	1998	Macrophytes	-	Exposed through soil
Sparling et al.	Responses of amphibian populations to water and soil factors in experimentally-treated aquatic macrocosms	1995	-	-	Exposed through soil
Sparling et al.	Ecotoxicology of aluminum to fish and wildlife	1997	-	-	Review
Staurnes et al.	Reduced carbonic anhydrase and Na-K-ATPase activity in gills of salmonids exposed to aluminium-containing acid water	1984	-	-	Mixture, Al and low pH
Staurnes et al.	Effects of acid water and aluminum on parr-smolt transformation and seawater tolerance in Atlantic salmon, *Salmo salar*	1993	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration; high control mortality (>40%)
Stearns et al.	Occurrence of cyanide-resistant respiration and of increased concentrations of cytochromes in Tetrahymena cells grown with various metals	1978	-	-	Cannot determine effect concentration
Storey et al.	An appraisal of some effects of simulated acid rain and aluminum ions on *Cyclops viridis* (Crustacea, Copepoda) and *Gammarus pulex* (Crustacea, Amphipoda)	1992	Copepod, *Cyclops viridis* Amphipod, *Gammarus pulex*	168 hr LC50=>26,980; LC50=>26,980	Dilution water not characterized
Strigul et al.	Acute toxicity of boron, titanium dioxide, and aluminum nanoparticles to Daphnia magna and Vibrio fischeri	2009	Cladoceran, *Daphnia magna*	-	Inappropriate form of toxicant, nanoparticles

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Sudo and Aiba	Effect of some metals on the specific growth rate of Ciliata isolated from activated sludge	1975	-	-	Pre-exposure to pollutant; isolated from activated sludge
Tabak and Gibbs	Effects of aluminum, calcium and low pH on egg hatching and nymphal survival of *Cloeon triangulifer* McDunnough (Ephemeroptera: Baetidae)	1991	Mayfly, *Cloeon triangulifer*	No effect on hatch success at 100 and pH=5.5	Only two exposure concentrations
Takano and Shimmen	Effects of aluminum on plasma membrane as revealed by analysis of alkaline band formation in internodal cells of *Chara corallina*	1999	Alga, *Chara corallina*	-	Excised cells
Tanaka and Navasero	Aluminum toxicity of the rice plant under water culture conditions	1966	-	-	Species not given
Taneeva	Toxicity of some heavy metals for hydrobionts	1973	Barnacle, *Balanus eburneus*	LC50=240	Text in foreign language
Taskinen et al.	Effect of pH, iron and aluminum on survival of early life history stages of the endangered freshwater pearl mussel, *Margaritifera margaritifera*	2011	Pearl mussel, *Margaritifera margaritifera*	-	Mixture; dilution water is river water
Tease and Coler	The effect of mineral acids and aluminum from coal leachate on substrate periphyton composition and productivity	1984	-	-	Mixture, Al and low pH
Teien et al.	Sodium silicate as alternative to liming-reduced aluminium toxicity for Atlantic salmon (*Salmo salar* L.) in unstable mixing zones	2006b	Atlantic salmon, *Salmo salar*	-	Only one exposure concentration; dilution water not characterized
Terhaar et al.	Toxicity of photographic processing chemicals to fish	1972	-	-	Mixture; no aluminum toxicity data
Thawornwong and Van Diest	Influences of high acidity and aluminum on the growth of lowland rice	1974	Rice	-	Scientific name not provided
Thomas	Effects of certain metallic salts upon fishes	1915	Mummichog, *Fundulus heteroclitus*	36 hr 100% mortality at 2,208; 120 hr 100% mortality at 1,104	Dilution water not characterized; lack of exposure details
Thompson et al.	Concentration factors of chemical elements in edible aquatic organisms	1972	-	-	Review

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Thomsen et al.	Effect of aluminum and calcium ions on survival and physiology of rainbow trout *Salmo gairdneri* (Richardson) eggs and larvae exposed to acid stress	1988	Rainbow trout, *Oncorhynchus mykiss*	25 d LC50=3,800 (soft water); LC50=71,000 (hard water)	Dilution water not characterized; unmeasured chronic exposure
Thorstad et al.	Reduced marine survival of hatchery-reared Atlantic salmon post-smolts exposed to aluminium and moderate acidification in freshwater	2013	Atlantic salmon, *Salmo salar*	-	Only two exposure concentrations; surgically altered test species (outfitted with acoustic transmitters)
Tietge et al.	Morphometric changes in gill secondary lamellae of brook trout (Salvelinus fontinalis) after long-term exposure to acid and aluminum	1988	Brook trout, *Salvelinus fontinalis*	147 d No effect on survival, but decrease growth at 393	Only one exposure concentration
Tipping et al.	Metal accumulation by stream bryophytes, related to chemical speciation	2008	Bryophytes	-	Exposure concentration not known; field accumulation study
Tomasik et al.	The metal-metal interactions in biological systems. Part III. *Daphnia magna*	1995a	Cladoceran, *Daphnia magna*	24 hr 10% mortality at 7,500	High control mortality (10-20%)
Tomasik et al.	The metal-metal interactions in biological systems. Part IV. Freshwater snail *Bulinus globosus*	1995b	Snail, *Bulinus globosus*	96 hr 100% mortality at 10,000; 1% mortality at 3,000	Not North American species
Troilo et al.	Biochemical responses of *Prochilodus lineatus* after 24-h exposure to aluminum	2007	Sabalo, *Prochilodus lineatus*	24 hr Increase in liver GST and increase in gill CAT at 100	Not North American species; lack of details; exposure methods unknown; abstract only
Truscott et al.	Effect of aluminum and lead on activity in the freshwater pond snail *Lymnaea stagnalis*	1995	Snail, *Lymnaea stagnalis*	45 hr Reduce activity at 500	Only two exposure concentrations
Tunca et al.	Tissue distribution and correlation profiles of heavy-metal accumulation in the freshwater crayfish *Astacus leptodactylus*	2013	Crayfish, *Astacus leptodactylus*	-	Field bioaccumulation study: exposure concentration not know; not North American species
Tyler-Jones et al.	The effects of acid water and aluminium on the embryonic development of the common frog, *Rana temporaria*	1989	Common frog, *Rana temporaria*	-	Not North American species; only three exposure concentrations
Umebese and Motajo	Accumulation, tolerance and impact of aluminium, copper and zinc on growth and nitrate reductase activity of *Ceratophyllum demersum* (Hornwort)	2008	Hornwort, *Ceratophyllum demersum*	15 d Decrease dry biomass at 3,000	Only two exposure concentrations

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Upreti et al.	Toxic effects of aluminium and fluoride on planktonic community of the microcosms	2013	Microcosms	-	Only one exposure concentration; dilution water not characterized
van Coillie and Rousseau	Mineral composition of the scales of *Catostomus commersoni* from two different waters: Studies using electron microprobe analysis	1974	White sucker, *Catostomus commersoni*	-	Exposure concentration not known; field accumulation study
van Dam et al.	Impact of acidification on diatoms and chemistry of Dutch moorland pools	1981	-	-	Mixture, Al and low pH
Van Hoecke et al.	Influence of alumina coating on characteristics and effects of SiO2 nanoparticles in algal growth inhibition assays at various pH and organic matter contents	2011	Alga	-	Inappropriate form of toxicant (nanoparticles)
Vazquez et al.	Effects of water acidity and metal concentrations on accumulation and within-plant distribution of metals in the aquatic bryophyte *Fontinalis antipyretica*	2000	Bryophyte, *Fontinalis antipyretica*	-	Exposure concentration not known; field accumulation study
Velzeboer et al.	Release of geosmin by *Anabaena circinalis* following treatment with aluminium sulphate	1995	Cyanobacteria, *Anabaena circinalis*	-	Only two exposure concentrations; dilution water not characterized
Velzeboer et al.	Aquatic ecotoxicity tests of some nanomaterials	2008	-	-	Inappropriate form of toxicant, nanoparticles
Verbost et al.	The toxic mixing zone of neutral and acidic river water: acute aluminum toxicity in brown trout (*Salmo trutta* L.)	1995	Brown trout, *Salmo trutta*	-	Mixture; dilution water is lake water
Vieira et al.	Effects of aluminum on the energetic substrates in neotropical freshwater *Astyanax bimaculatus* (Teleostei: Characidae) females	2013	Two spot astyanax, *Astyanax bimaculatus*	96 hr Decrease T_4 levels and increase T_3 levels at 600	Not North American species; only one exposure concentration
Vinay et al.	Toxicity and dose determination of quillaja saponin, aluminum hydroxide and squalene in olive flounder (*Paralichthys olivaceus*)	2013	Olive flounder, *Paralichthys olivaceus*	-	Injected toxicant
Vincent et al.	Accumulation of Al, Mn, Fe, Cu, Zn, Cd, and Pb by the bryophyte *Scapania undulata* in three upland waters of different pH	2001	Bryophyte, *Scapania undulata*	-	Exposure concentration not known; field accumulation study
Vuorinen et al.	The sensitivity to acidity and aluminum of newly-hatched perch (*Perca fluviatilis*) originating from strains from four lakes with different degrees	1994a	Perch, *Perca fluviatilis*	7 d LC50=>1,000	Not North American species

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Vuorinen et al.	The sensitivity to acidification of pike (*Esox lucius*), whitefish (*Coregonus lavaretus*) and roach (*Rutilus rutilus*): a comparison of field and laboratory studies	1994b	-	-	Review of Vuorineu et al. 1993
Vuorinen et al.	Reproduction, blood and plasma parameters and gill histology of vendace (*Coregonus albula* L.) in long-term exposure to acidity to aluminum	2003	Vendace, *Coregonus albula*	60 d Decrease growth at 168 and pH=5.25; Decrease growth at 213 and pH=4.75	Not North American species; only one exposure concentration
Wakabayashi et al.	Relative lethal sensitivity of two Daphnia species to chemicals	1988	-	-	Text in foreign language
Walker et al.	Effects of low pH and aluminum on ventilation in the brook trout (*Salvelinus fontinalis*)	1988	Brook trout, *Salvelinus fontinalis*	-	Surgically altered fish; only one exposure concentration
Walker et al.	Effects of long-term preexposure to sublethal concentrations of acid and aluminum on the ventilatory response to aluminum challenge in brook trout (*Salvelinus fontinalis*)	1991	Brook trout, *Salvelinus fontinalis*	-	Pre-exposure to pollutant
Wallen et al.	Toxicity to *Gambusia affinis* of certain pure chemicals in turbid waters	1957	Western mosquitofish, *Gambusia affinis*	96 hr LC50=26,919 ($AlCl_3$); LC50=37,062 ($Al_2(SO_4)_3$	Dilution water not characterized; farm pond with high turbidity and poor fish population
Walton et al.	Tissue accumulation of aluminum is not a predictor of toxicity in the freshwater snail, *Lymnaea stagnalis*	2009	Snail, *Lymnaea stagnalis*	steady state not reached	Lack of details; steady state not reached
Walton et al.	Trophic transfer of aluminium through an aquatic grazer-omnivore food chain	2010a	Snail, *Lymnaea stagnalis* Crayfish, *Pacifasticus leniusculus*	-	Bioaccumulation: steady state not reached
Walton et al.	The suitability of gallium as a substitute for aluminum in tracing experiments	2010b	Snail, *Lymnaea stagnalis*	-	Bioaccumulation: steady state not reached
Wang et al.	Optimising indoor phosphine fumigation of paddy rice bag-stacks under sheeting for control of resistant insects	2006	-	-	Not applicable, no aluminum toxicity information
Wang et al.	Toxicity of nanoparticulate and bulk ZnO, Al2O3 and TiO2 to the nematode *Caenorhabditis elegans*	2009	Nematode, *Caenorhabditis elegans*	-	Inappropriate form of toxicant (nanoparticles)
Wang et al.	Synergistic toxic effect of nano-Al2O3 and As(V) on *Ceriodaphnia dubia*	2011	Cladoceran, *Ceriodaphnia dubia*	-	Inappropriate form of toxicant (nanoparticles)

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Ward et al.	Influences of aqueous aluminum on the immune system of the freshwater crayfish Pacifasticus leniusculus	2006	Crayfish, *Pacifasticus leniusculus*	-	Only one exposure concentration; test organism injected with bacteria
Waring and Brown	Ionoregulatory and respiratory responses of brown trout, *Salmo trutta*, exposed to lethal and sublethal aluminum in acidic soft waters	1995	Brown trout, *Salmo trutta*	5 d NOEC (survival)=12.5; LOEC=25	Too few exposure concentrations
Waring et al.	Plasma prolactin, cortisol, and thyroid response of the brown trout (*Salmo trutta*) exposed to lethal and sublethal aluminum in acidic soft waters	1996	Brown trout, *Salmo trutta*	-	Surgically altered test species
Waterman	Effect of salts of heavy meatls on development of the sea urchin, *Arbacia punctulata*	1937	Sea urchin, *Arbacia punctulata*	-	Dilution water not characterized; cannot determine effect concentration
Wauer and Teien	Risk of acute toxicity for fish during aluminum application to hardwater lakes	2010	-	-	Survey
Weatherley et al.	The response of macroinvertebrates to experimental episodes of low pH with different forms of aluminum, during a natural spate	1988	-	-	Mixture; dilution water is stream water
Weatherley et al.	The survival of early life stages of brown trout (*Salmo trutta* L.) in relation to aluminum speciation in upland Welsh streams	1990	Brown trout, *Salmo trutta*	-	Mixture; dilution water is stream water
Weatherley et al.	Liming acid streams: aluminum toxicity to fish in mixing zones	1991	-	-	Mixture; dilution water is stream water
White et al.	Avoidance of aluminum toxicity on freshwater snails involves intracellular silicon-aluminum biointeraction	2008	Snail, *Lymnaea stagnalis*	-	Mixture, Al and Si
Whitehead and Brown	Endocrine responses of brown trout, Salmo trutta L., to acid, aluminum and lime dosing in a Welsh hill stream	1989	Brown trout, *Salmo trutta*	-	Mixture, field experiment-dosed stream
Wilkinson and Campbell	Aluminum bioconcentration at the gill surface of juvenile Atlantic salmon in acidic media	1993	Atlantic salmon, *Salmo salar*	-	Bioaccumulation: steady state not reached
Wilkinson et al.	Surface complexation of aluminum on isolated fish gill cells	1993	Largemouth bass, *Micropterus salmoides*	-	Exposed cells only
Williams et al.	Assessment of surface-water quantity and quality, Eagle River Watershed, Colorado, 1947-2007	2011	-	-	Not applicable; no aluminum toxicity data
Wilson	Physiological and metabolic costs of acclimation to chronic sub-lethal acid and aluminum exposure in rainbow trout	1996	-	-	Review

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Wilson and Hyne	Toxicity of acid-sulfate soil leachate and aluminum to embryos of the Sydney Rock Oyster	1997	Sydney rock oyster, *Accostrea commercialis*	48 hr EC50 (development)=222; EC50=227	Not North American species
Wilson and Wood	Swimming performance, whole body ions, and gill A1 accumulation during acclimation to sublethal aluminum in juvenile rainbow trout (*Oncorhynchus mykiss*)	1992	Rainbow trout, *Oncorhynchus mykiss*	22 d No effect on mortality, but decrease weight at 31.4	Only one exposure concentration
Wilson et al.	Metabolic costs and physiological consequences of acclimation to aluminum in juvenile rainbow trout (*Oncorhynchus mykiss*). 1: Acclimation specificity, resting physiology, feeding, and growth	1994a	Rainbow trout, *Oncorhynchus mykiss*	34 d 5.5% mortality at 38.1	Only one exposure concentration
Wilson et al.	Metabolic costs and physiological consequences of acclimation to aluminum in juvenile rainbow trout (*Oncorhynchus mykiss*). 2: Gill morphology, swimming performance, and aerobic scope	1994b	Rainbow trout, *Oncorhynchus mykiss*	34 d Decrease # of mucous cells in gills, oxygen consumption rates, swimming performance at 38	Only one exposure concentration
Wilson et al.	Growth and protein turnover during acclimation to acid and aluminum in juvenile rainbow trout (*Oncorhynchus mykiss*)	1996	Rainbow trout, *Oncorhynchus mykiss*	-	Only one exposure concentration; pre-exposure to pollutant
Winter et al.	Influences of acidic to basic water pH and natural organic matter on aluminum accumulation by gills of rainbow trout (*Oncorhynchus mykiss*)	2005	Rainbow trout, *Oncorhynchus mykiss*	-	Bioaccumulation: not renewal or flow-through exposure; high control mortality
Witters	Acute acid exposure of rainbow trout, S*almo gairdneri* Richardson: effects of aluminum and calcium on ion balance and haematology	1986	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species; only one exposure concentration
Witters et al.	Interference of aluminum and pH on the Na-influx in an aquatic insect *Corixa punctata* (Illig.)	1984	Waterbug, *Corixa punctata*	-	Mixture; low pH and Al
Witters et al.	Ionoregulatory and haematological responses of rainbow trout *Salmo gairdneri* Richardson to chronic acid and aluminum stress	1987a	Rainbow trout, *Oncorhynchus mykiss*	48 hr ~50% mortality at 200	Only one exposure concentration
Witters et al.	Physiological study on the recovery of rainbow trout (*Salmo gairdneri* Richardson) from acid and Al stress	1987b	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species; only one exposure concentration
Witters et al.	Haematological disturbances and osmotic shifts in rainbow trout, *Oncorhynchus mykiss* (Walbaum) under acid and aluminum exposure	1990a	Rainbow trout, *Oncorhynchus mykiss*	2.5 d ~53% mortality at 200	Only one exposure concentration

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Witters et al.	The effect of humic substances on the toxicity of aluminum to adult rainbow trout, *Oncorhynchus mykiss* (Walbaum)	1990b	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Witters et al.	Adrenergic response to physiological disturbances in rainbow trout, Oncorhynchus mykiss, exposed to aluminum at acid pH	1991	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Witters et al.	Physicochemical changes of aluminum in mixing zones: Mortality and physiological disturbances in brown trout (*Salmo trutta* L.)	1996	Brown trout, *Salmo trutta*	48 hr 60% mortality at 184.0	Only one exposure concentration
Wold	Some effects of aluminum sulfate and arsenic sulfide on *Daphnia pulex* and *Chironomus tentans*	2001	Cladoceran, *Daphnia pulex* Midge, *Chironomus tentans*	-	Inadequate exposure methods; chronic was a static, unmeasured exposure; pre-exposure to pollutant
Wold et al.	Life-history responses of *Daphnia pulex* with exposure to aluminum sulfate	2005	Cladoceran, *Daphnia pulex*	Increased survivorship in clones that were prior-exposed to alum treated lakes	Only three exposure concentrations; dilution water not characterized
Wood and McDonald	The physiology of acid/aluminum stress in trout	1987	Trout	-	Too few exposure concentrations, cannot determine effect concentration
Wood et al.	Blood gases, acid-base status, ions, and hematology in adult brook trout (*Salvelinus fontinalis*) under acid/aluminum exposure	1988a	Brook trout, *Salvelinus fontinalis*	-	Only one exposure concentration; surgically altered test species
Wood et al.	Physiological evidence of acclimation to acid/aluminum stress in adult brook trout (*Salvelinus fontinalis*). 1. Blood composition and net sodium fluxes	1988b	Brook trout, *Salvelinus fontinalis*	10 wk 28% mortality at 77	Only two exposure concentrations
Wood et al.	Physiological evidence of acclimation to acid/aluminum stress in adult brook trout (*Salvelinus fontinalis*). 2. Blood parameters by cannulation	1988c	Brook trout, *Salvelinus fontinalis*	-	Only one exposure concentration; surgically altered test species
Wood et al.	Whole body ions of brook trout (*Salvelinus fontinalis*) alevins: responses of yolk-sac and swim-up stages to water acidity, calcium, and aluminum, and recovery effects	1990a	Brook trout, *Salvelinus fontinalis*	-	Lack of details; cannot determine effect concentration

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Wood et al.	Effects of water acidity, calcium, and aluminum on whole body ions of brook trout (*Salvelinus fontinalis*) continuously exposed from fertilization to swim-up: a study by instrumental neutron activation analysis	1990b	Brook trout, *Salvelinus fontinalis*	-	Lack of details; cannot determine effect concentration
Woodburn et al.	Accumulation and toxicity of aluminium-contaminated food in the freshwater crayfish, *Pacifastacus leniusculus*	2011	Crayfish, *Pacifastacus leniusculus*	-	Dietary exposure
Wooldridge and Wooldridge	Internal damage in an aquatic beetle exposed to sublethal concentrations of inorganic ions	1969	Aquatic beetle, *Tropistermus lateralis nimbatus*	14 d Change the body fat at 26,981	Only one exposure concentration
Wren et al.	Examination of bioaccumulation and biomagnification of metlas in a Precambrian Shield Lake	1983	-	-	Field exposure, exposure concentrations not measured adequately
Wu et al.	QTLs and epistasis for aluminum tolerance in rice (*Oryza sativa* L.) at different seedling stages	2000	Rice, *Oryza sativa*	-	Only one exposure concentration; difficult to determine effect concentration
Wu et al.	Aluminum nanoparticle exposure in L1 larvae results in more severe lethality toxicity than in L4 larvae or young adults by strengthening the formation of stress response and intestinal lipofuscin accumulation in nematodes	2011	-	-	Inappropriate form of toxicant, nanoparticles
Yang and van den Berg	Metal complexation by humic substances in seawater	2009	-	-	Not applicable; no aluminum toxicity data
Yang et al.	Identification of aluminum-responsive proteins in rice roots by a proteomic approach: Cysteine synthase as a key player in Al response	2007	Rice, *Oryza sativa*	3 d Decreased root length at 53,960	Only two exposure concentrations
Youson and Neville	Deposition of aluminum in the gill epithelium of rainbow trout (*Salmo gairdneri* Richardson) subjected to sublethal concentrations of the metal	1987	Rainbow trout, *Oncorhynchus mykiss*	-	Surgically altered test species
Ytrestoyl et al.	Swimming performance and blood chemistry in Atlantic salmon spawners exposed to acid river water with elevated aluminium concentrations	2001	Atlantic salmon, *Salmo salar*	-	Only one exposure concentrations; dilution water not characterized; no true control group
Zaifnejad et al.	Aluminum and water stress effects on growth and proline of sorghum	1997	Sorghum, *Sorghum bicolor*	-	Inappropriate form of toxicant (aluminum potassium sulfate)
Zaini and Mercado	Calcium-aluminum interaction on the growth of two rice cultivars in culture solution	1984	Rice	-	Scientific name not provided

Author	Title	Date	Organism(s)	Concentration (µg/L)	Reason Unused
Zarini et al.	Effects produced by aluminum in freshwater communities studied by "enclosure" method	1983	-	-	Mixed species exposure; no species names provided; dilution water not characterized
Zhou and Yokel	The chemical species of aluminum influences its paracellular flux across and uptake into Caco-2 cells, a model of gastrointestinal absorption	2005	-	-	Excised cells, in vitro
Zhu et al.	Comparative toxicity of several metal oxide nanoparticle aqueous suspensions to Zebrafish (*Danio rerio*) early developmental stage	2008	Zebrafish, *Danio rerio*	-	Inappropriate form of toxicant, nanoparticles
Zhu et al.	Acute toxicities of six manufactured nanomaterial suspensions to *Daphnia magna*	2009	Cladoceran, *Daphnia magna*	-	Inappropriate form of toxicant (nanoparticles)

Appendix K RECOMMENDED CRITERIA FOR VARIOUS WATER CHEMISTRY CONDITIONS

Table K-1. Freshwater CMC at DOC of 0.1 mg/L and Various Water Total Hardness Levels and pHs.

Total Hardness	Acute Criterion (CMC) (µg/L total aluminum) (DOC=0.1 mg/L)																											
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*1.0*	a	*4.8*	b	18	d	**51**	d	**120**	b	**210**	a	**290**	a	**310**	a	***320***	a	***270***	a	*180*	a	*95*	a	*39*	a
25	*2.4*	a	*10*	h	32	d	**84**	d	**180**	d	**290**	a	**370**	a	**380**	a	***360***	a	***280***	a	*170*	a	*78*	a	*29*	a
50	*4.6*	a	*17*	d	50	d	**120**	d	**240**	d	**380**	a	**440**	a	**430**	a	***400***	a	***280***	a	*160*	a	*67*	a	*23*	a
75	*6.7*	b	*24*	d	64	d	**150**	d	**290**	d	**440**	b	**480**	a	**470**	a	***420***	a	***280***	a	*150*	a	*62*	a	*20*	a
100	*8.8*	b	*30*	d	76	d	**170**	d	**320**	d	**490**	i	**520**	a	**500**	a	***430***	a	***280***	a	*150*	a	*58*	a	*18*	a
150	*13*	c	*40*	d	96	d	**200**	d	**380**	d	**560**	h	**580**	a	**540**	a	***460***	a	***290***	a	*140*	a	*53*	a	*16*	a
200	*17*	c	*49*	d	110	d	**230**	d	**420**	d	**610**	d	**620**	a	**570**	a	***480***	a	***290***	a	*140*	a	*50*	a	*15*	a
250	*20*	d	*58*	d	130	d	**250**	d	**460**	d	**660**	d	**650**	a	**600**	a	***490***	a	***290***	a	*130*	a	*48*	a	*14*	a
300	*24*	d	*66*	d	140	d	**270**	d	**490**	d	**700**	d	**680**	a	**620**	a	***500***	a	***290***	a	*130*	a	*46*	a	*13*	a
350	*28*	d	*73*	d	150	d	**290**	d	**510**	d	**730**	d	**710**	a	**640**	a	***510***	a	***290***	a	*130*	a	*45*	a	*12*	a
400	*31*	d	*80*	d	160	d	**310**	d	**540**	d	760	d	**730**	a	**660**	a	***520***	a	***290***	a	*130*	a	*43*	a	*12*	a
430	*33*	d	*84*	d	170	d	**320**	d	**550**	d	780	d	750	a	**670**	a	***530***	a	***290***	a	*130*	a	*43*	a	*11*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4).

a Daphnia, Ceriodaphnia, Stenocypris, Nais
b Daphnia, Ceriodaphnia, Stenocypris, Micropterus
c Daphnia, Micropterus, Ceriodaphnia, Stenocypris
d Daphnia, Micropterus, Oncorhynchus, Ceriodaphnia
e Daphnia, Micropterus, Oncorhynchus, Salmo
f Micropterus, Daphnia, Oncorhynchus, Salmo
g Micropterus, Oncorhynchus, Daphnia, Salmo
h Daphnia, Micropterus, Ceriodaphnia, Oncorhynchus
i Daphnia, Ceriodaphnia, Micropterus, Stenocypris

Table K-2. Freshwater CCC at DOC of 0.1 mg/L and Various Water Hardness Levels and pHs.

Total Hardness	Chronic Criterion (CCC) (µg/L total aluminum) (DOC=0.1 mg/L)																							
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*0.63*	a	*3.1*	b	12	d	**33**	e	**77**	b	130	a	180	a	200	a	*200*	a	*170*	a	*110*	a	*59*	a	*24*	a
25	*1.5*	a	*6.7*	c	19	f	**48**	f	120	d	180	a	230	a	240	a	*230*	a	*170*	a	*100*	a	*49*	a	*18*	a
50	*2.9*	a	*11*	e	26	h	**63**	g	140	e	240	b	270	a	270	a	*250*	a	*180*	a	*97*	a	*42*	a	*14*	a
75	*4.3*	b	*14*	f	31	g	**71**	g	160	f	290	b	300	a	290	a	*260*	a	*180*	a	*94*	a	*39*	a	*13*	a
100	*5.8*	b	*17*	f	35	g	**77**	g	180	f	320	c	330	a	310	a	*270*	a	*180*	a	*91*	a	*36*	a	*11*	a
150	*8.6*	c	*21*	h	42	g	87	g	190	g	370	c	360	a	340	a	*290*	a	*180*	a	*88*	a	*33*	a	*10*	a
200	*11*	c	*25*	g	47	g	94	g	200	g	400	e	390	a	360	a	*300*	a	*180*	a	*85*	a	*31*	a	*9.1*	a
250	*13*	d	*28*	g	51	g	100	g	210	g	420	e	410	a	380	a	*310*	a	*180*	a	*83*	a	*30*	a	*8.5*	a
300	*16*	e	*31*	g	55	g	100	g	220	g	430	e	430	a	390	a	*320*	a	*180*	a	*82*	a	*29*	a	*8.0*	a
350	*17*	e	*33*	g	58	g	110	g	220	g	440	e	440	a	400	a	*320*	a	*180*	a	*81*	a	*28*	a	*7.6*	a
400	*19*	e	*36*	g	61	g	110	g	230	g	450	e	460	a	410	a	*330*	a	*180*	a	*80*	a	*27*	a	*7.3*	a
430	*20*	e	*37*	g	63	g	120	g	230	g	450	e	470	a	420	a	*330*	a	*180*	a	*79*	a	*27*	a	*7.1*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4)

a Daphnia, Lampsilis, Ceriodaphnia, Hyalella
b Daphnia, Lampsilis, Ceriodaphnia, Salmo
c Daphnia, Lampsilis, Salmo, Ceriodaphnia
d Daphnia, Salmo, Lampsilis, Ceriodaphnia
e Salmo, Daphnia, Lampsilis, Ceriodaphnia
f Salmo, Daphnia, Lampsilis, Salvelinus
g Salmo, Salvelinus, Daphnia, Lampsilis
h Salmo, Daphnia, Salvelinus, Lampsilis
i Salmo, Salvelinus, Daphnia, Danio
j Salmo, Salvelinus, Danio, Daphnia
k Salmo, Salvelinus, Danio, Pimephales

Table K-3. Freshwater CMC at DOC of 0.5 mg/L and Various Water Total Hardness Levels and pHs.

Total Hardness	Acute Criterion (CMC) (µg/L total aluminum) (DOC=0.5 mg/L)																									
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*2.6*	a	*13*	a	46	c	**130**	d	**300**	d	**550**	i	770	a	820	a	*830*	a	***710***	a	*470*	a	*250*	a	*100*	a
25	*6.3*	a	*27*	a	86	d	**210**	d	**430**	d	750	d	960	a	980	a	*940*	a	***720***	a	*430*	a	*200*	a	*75*	a
50	*12*	a	*47*	b	130	d	**300**	d	**560**	d	920	d	1,100	b	1,100	a	*1,000*	a	***730***	a	*410*	a	*180*	a	*60*	a
75	*18*	a	*66*	c	170	d	**360**	d	**650**	d	1,000	d	1,300	b	1,200	a	*1,100*	a	***740***	a	*390*	a	*160*	a	*52*	a
100	*23*	a	*82*	d	210	d	**410**	d	**720**	d	1,100	d	1,400	c	1,300	a	*1,100*	a	***740***	a	*380*	a	*150*	a	*48*	a
150	*34*	a	*110*	d	260	d	**480**	d	820	d	1,200	d	1,500	d	1,400	b	*1,200*	a	*750*	a	*370*	a	*140*	a	*42*	a
200	*44*	a	*140*	d	310	d	**550**	d	890	d	1,300	d	1,600	d	1,500	b	*1,200*	a	*750*	a	*360*	a	*130*	a	*38*	a
250	*54*	a	*170*	d	350	d	**600**	d	950	d	1,400	d	1,600	d	1,600	i	*1,300*	a	*760*	a	*350*	a	*130*	a	*35*	a
300	*65*	a	*190*	d	390	d	**650**	e	1,000	e	1,500	d	1,700	d	1,600	c	*1,300*	a	*760*	a	*340*	a	*120*	a	*33*	a
350	*75*	a	*220*	d	420	d	**700**	e	1,100	e	1,500	d	1,800	d	1,700	c	*1,300*	a	*760*	a	*340*	a	*120*	a	*32*	a
400	*85*	a	*240*	d	450	d	**740**	e	1,100	e	1,500	d	1,800	d	1,700	h	*1,400*	a	*760*	a	*330*	a	*110*	a	*30*	a
430	*90*	a	*250*	d	470	d	770	e	1,100	e	1,600	d	1,800	d	1,700	d	*1,400*	a	*760*	a	*330*	a	*110*	a	*30*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4).

a Daphnia, Ceriodaphnia, Stenocypris, Nais
b Daphnia, Ceriodaphnia, Stenocypris, Micropterus
c Daphnia, Micropterus, Ceriodaphnia, Stenocypris
d Daphnia, Micropterus, Oncorhynchus, Ceriodaphnia
e Daphnia, Micropterus, Oncorhynchus, Salmo
f Micropterus, Daphnia, Oncorhynchus, Salmo
g Micropterus, Oncorhynchus, Daphnia, Salmo
h Daphnia, Micropterus, Ceriodaphnia, Oncorhynchus
i Daphnia, Ceriodaphnia, Micropterus, Stenocypris

Table K-4. Freshwater CCC at DOC of 0.5 mg/L and Various Water Hardness Levels and pHs.

Total Hardness	Chronic Criterion (CCC) (µg/L total aluminum) (DOC=0.5 mg/L)																									
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*1.7*	a	*7.9*	a	31	c	**78**	e	180	e	370	c	480	a	510	a	*520*	a	*440*	a	*300*	a	*150*	a	*63*	a
25	*3.9*	a	*17*	b	52	e	110	h	230	f	470	e	600	a	620	a	*590*	a	*450*	a	*270*	a	*130*	a	*47*	a
50	*7.5*	a	*31*	c	74	f	140	g	270	g	520	f	740	b	710	a	*650*	a	*460*	a	*250*	a	*110*	a	*37*	a
75	*11*	a	*44*	c	89	g	160	g	290	g	560	f	840	c	770	a	*680*	a	*460*	a	*240*	a	*100*	a	*33*	a
100	*14*	a	*54*	d	100	g	170	g	300	g	580	g	910	c	820	a	*710*	a	*460*	a	*240*	a	*95*	a	*30*	a
150	*21*	a	*70*	e	120	g	190	g	320	g	600	g	970	d	910	b	*750*	a	*470*	a	*230*	a	*87*	a	*26*	a
200	*28*	a	*84*	e	130	g	200	g	340	g	610	g	990	e	990	b	*780*	a	*470*	a	*220*	a	*82*	a	*24*	a
250	*34*	a	*96*	f	150	g	220	g	350	g	610	g	1,000	e	1,000	c	*800*	a	*470*	a	*220*	a	*78*	a	*22*	a
300	*40*	a	*110*	f	160	g	230	g	360	g	620	g	1,000	e	1,100	c	*820*	a	*470*	a	*210*	a	*75*	a	*21*	a
350	*47*	a	*120*	f	170	g	240	g	370	g	620	g	1,000	e	1,100	c	*840*	a	*480*	a	*210*	a	*73*	a	*20*	a
400	*53*	a	*130*	f	180	g	250	g	370	g	630	g	1,000	f	1,100	c	*860*	a	*480*	a	*210*	a	*71*	a	*19*	a
430	*57*	a	*140*	f	180	g	250	g	380	g	630	g	1,000	f	1,100	d	*860*	a	*480*	a	*210*	a	*70*	a	*19*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4)

a Daphnia, Lampsilis, Ceriodaphnia, Hyalella
b Daphnia, Lampsilis, Ceriodaphnia, Salmo
c Daphnia, Lampsilis, Salmo, Ceriodaphnia
d Daphnia, Salmo, Lampsilis, Ceriodaphnia
e Salmo, Daphnia, Lampsilis, Ceriodaphnia
f Salmo, Daphnia, Lampsilis, Salvelinus
g Salmo, Salvelinus, Daphnia, Lampsilis
h Salmo, Daphnia, Salvelinus, Lampsilis
i Salmo, Salvelinus, Daphnia, Danio
j Salmo, Salvelinus, Danio, Daphnia
k Salmo, Salvelinus, Danio, Pimephales

Table K-5. Freshwater CMC at DOC of 1.0 mg/L and Various Water Total Hardness Levels and pHs.

Total Hardness	Acute Criterion (CMC) (µg/L total aluminum) (DOC=1.0 mg/L)																									
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*4.0*	a	*19*	a	70	c	**190**	d	**430**	d	810	d	1,200	a	1,200	a	*1,300*	a	*1,100*	a	*720*	a	*370*	a	*150*	a
25	*9.5*	a	*40*	a	130	d	**310**	d	**620**	d	1,100	d	1,400	c	1,500	a	*1,400*	a	*1,100*	a	*660*	a	*310*	a	*110*	a
50	*18*	a	*72*	b	210	d	**430**	d	790	d	1,300	d	1,700	d	1,700	b	*1,600*	a	*1,100*	a	*610*	a	*270*	a	*90*	a
75	*27*	a	*100*	b	260	d	**520**	d	900	d	1,400	d	1,800	d	1,800	c	*1,700*	a	*1,100*	a	*590*	a	*240*	a	*79*	a
100	*35*	a	*130*	c	320	d	**590**	d	980	d	1,500	d	1,900	d	1,900	d	*1,700*	a	*1,100*	a	*570*	a	*230*	a	*72*	a
150	*51*	a	*170*	d	400	d	**700**	d	1,100	d	1,600	d	2,100	d	2,100	d	*1,800*	a	*1,100*	a	*550*	a	*210*	a	*63*	a
200	*67*	a	*220*	d	470	d	790	d	1,200	e	1,700	d	2,200	d	2,200	d	*1,900*	b	*1,100*	a	*540*	a	*200*	a	*57*	a
250	*82*	a	*260*	d	540	d	870	e	1,300	e	1,800	d	2,200	d	2,200	d	*1,900*	b	*1,100*	a	*530*	a	*190*	a	*53*	a
300	*98*	a	*300*	d	600	d	950	e	1,400	f	1,900	d	2,300	d	2,300	d	*2,000*	b	*1,100*	a	*520*	a	*180*	a	*50*	a
350	*110*	a	*340*	d	650	d	1,000	e	1,500	f	1,900	e	2,300	d	2,300	d	*2,000*	c	*1,200*	a	*510*	a	*180*	a	*48*	a
400	*130*	a	*380*	d	700	d	1,100	f	1,600	f	2,000	e	2,400	d	2,400	d	*2,100*	c	*1,200*	a	*500*	a	*170*	a	*46*	a
430	*140*	a	*400*	d	730	d	1,100	f	1,600	f	2,000	e	2,400	d	2,400	d	*2,100*	c	*1,200*	a	*500*	a	*170*	a	*45*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).
Ranking of four most sensitive genera (Rank 1-Rank 4).
a Daphnia, Ceriodaphnia, Stenocypris, Nais
b Daphnia, Ceriodaphnia, Stenocypris, Micropterus
c Daphnia, Micropterus, Ceriodaphnia, Stenocypris
d Daphnia, Micropterus, Oncorhynchus, Ceriodaphnia
e Daphnia, Micropterus, Oncorhynchus, Salmo
f Micropterus, Daphnia, Oncorhynchus, Salmo
g Micropterus, Oncorhynchus, Daphnia, Salmo
h Daphnia, Micropterus, Ceriodaphnia, Oncorhynchus
i Daphnia, Ceriodaphnia, Micropterus, Stenocypris

Table K-6. Freshwater CCC at DOC of 1.0 mg/L and Various Water Hardness Levels and pHs.

Total Hardness	Chronic Criterion (CCC) (µg/L total aluminum) (DOC=1.0 mg/L)																							
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*2.5*	a	*12*	a	47	c	110	e	240	f	500	e	730	b	770	a	*790*	a	*670*	a	*450*	a	*230*	a	*95*	a
25	*5.9*	a	*25*	a	81	e	160	g	300	g	580	f	970	c	930	a	*890*	a	*680*	a	*410*	a	*190*	a	*71*	a
50	*11*	a	*46*	b	110	f	200	g	340	g	620	g	1,100	e	1,100	c	*980*	a	*690*	a	*380*	a	*170*	a	*56*	a
75	*17*	a	*66*	b	140	h	220	g	360	g	640	g	1,100	e	1,200	c	*1,000*	a	*700*	a	*370*	a	*150*	a	*49*	a
100	*22*	a	*85*	c	160	g	240	g	380	g	650	g	1,100	f	1,300	d	*1,100*	a	*700*	a	*360*	a	*140*	a	*45*	a
150	*32*	a	*120*	d	190	g	260	g	400	g	660	g	1,100	f	1,300	e	*1,100*	a	*710*	a	*350*	a	*130*	a	*39*	a
200	*42*	a	*140*	e	210	g	290	g	420	g	670	g	1,100	g	1,300	e	*1,200*	b	*710*	a	*340*	a	*120*	a	*36*	a
250	*51*	a	*160*	e	230	g	300	g	430	g	670	g	1,100	g	1,300	f	*1,300*	b	*720*	a	*330*	a	*120*	a	*33*	a
300	*61*	a	*180*	e	250	g	320	g	440	g	680	g	1,100	g	1,300	f	*1,300*	c	*720*	a	*320*	a	*110*	a	*31*	a
350	*71*	a	*200*	e	260	g	330	g	450	i	680	g	1,100	g	1,300	f	*1,400*	c	*720*	a	*320*	a	*110*	a	*30*	a
400	*80*	a	*220*	e	280	g	340	g	470	j	680	g	1,100	g	1,300	f	*1,400*	c	*720*	a	*310*	a	*110*	a	*29*	a
430	*86*	a	*230*	f	290	g	350	i	470	j	680	g	1,100	g	1,300	f	*1,400*	c	*720*	a	*310*	a	*110*	a	*28*	a

(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4)

a Daphnia, Lampsilis, Ceriodaphnia, Hyalella
b Daphnia, Lampsilis, Ceriodaphnia, Salmo
c Daphnia, Lampsilis, Salmo, Ceriodaphnia
d Daphnia, Salmo, Lampsilis, Ceriodaphnia
e Salmo, Daphnia, Lampsilis, Ceriodaphnia
f Salmo, Daphnia, Lampsilis, Salvelinus
g Salmo, Salvelinus, Daphnia, Lampsilis
h Salmo, Daphnia, Salvelinus, Lampsilis
i Salmo, Salvelinus, Daphnia, Danio
j Salmo, Salvelinus, Danio, Daphnia
k Salmo, Salvelinus, Danio, Pimephales

Table K-7. Freshwater CMC at DOC of 2.5 mg/L and Various Water Total Hardness Levels and pHs.

Total Hardness	Acute Criterion (CMC) (µg/L total aluminum) (DOC=2.5 mg/L)																											
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*6.9*	a	*33*	a	120	i	**330**	d	**700**	d	1,300	d	1,900	d	2,100	c	*2,200*	a	*1,900*	a	*1,200*	a	*650*	a	*260*	a
25	*16*	a	*70*	a	230	d	**520**	d	960	d	1,600	d	2,300	d	2,500	d	*2,500*	b	*1,900*	a	*1,100*	a	*530*	a	*200*	a
50	*31*	a	*120*	a	360	d	**720**	d	1,200	d	1,800	d	2,500	d	2,700	d	*2,700*	h	*1,900*	a	*1,100*	a	*460*	a	*160*	a
75	*46*	a	*170*	a	460	d	850	d	1,300	e	2,000	d	2,700	d	2,800	d	*2,800*	d	*1,900*	a	*1,000*	a	*420*	a	*140*	a
100	*60*	a	*220*	b	550	d	970	d	1,500	e	2,100	e	2,700	d	2,900	d	*2,900*	d	*1,900*	a	*990*	a	*400*	a	*120*	a
150	*88*	a	*310*	b	710	d	1,100	d	1,700	f	2,300	e	2,900	d	3,000	d	*3,000*	d	*2,000*	a	*960*	a	*360*	a	*110*	a
200	*120*	a	*390*	c	840	d	1,300	e	1,900	g	2,500	f	2,900	d	3,100	d	*3,000*	d	*2,000*	a	*930*	a	*340*	a	*99*	a
250	*140*	a	*460*	c	960	d	1,500	e	2,100	g	2,600	g	3,000	d	3,100	d	*3,000*	d	*2,000*	a	*910*	a	*330*	a	*92*	a
300	*170*	a	*530*	d	1,100	d	1,600	f	2,200	g	2,700	g	3,000	e	3,100	d	*3,100*	d	*2,000*	a	*890*	a	*320*	a	*87*	a
350	*190*	a	*600*	d	1,200	d	1,700	f	2,300	g	2,800	g	3,100	e	3,200	d	*3,100*	d	*2,000*	a	*880*	a	*310*	a	*83*	a
400	*220*	a	*670*	d	1,200	d	1,800	f	2,400	g	2,900	g	3,100	e	3,200	d	*3,100*	d	*2,000*	a	*870*	a	*300*	a	*79*	a
430	*240*	a	*710*	d	1,300	d	1,900	g	2,400	g	2,900	g	3,100	e	3,200	d	*3,100*	d	*2,000*	a	*860*	a	*290*	a	*77*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).
Ranking of four most sensitive genera (Rank 1-Rank 4).
a Daphnia, Ceriodaphnia, Stenocypris, Nais
b Daphnia, Ceriodaphnia, Stenocypris, Micropterus
c Daphnia, Micropterus, Ceriodaphnia, Stenocypris
d Daphnia, Micropterus, Oncorhynchus, Ceriodaphnia
e Daphnia, Micropterus, Oncorhynchus, Salmo
f Micropterus, Daphnia, Oncorhynchus, Salmo
g Micropterus, Oncorhynchus, Daphnia, Salmo
h Daphnia, Micropterus, Ceriodaphnia, Oncorhynchus
i Daphnia, Ceriodaphnia, Micropterus, Stenocypris

Table K-8. Freshwater CCC at DOC of 2.5 mg/L and Various Water Hardness Levels and pHs.

Total Hardness	Chronic Criterion (CCC) (µg/L total aluminum) (DOC=2.5 mg/L)																									
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*4.3*	a	*21*	a	81	c	180	f	340	g	650	g	1,200	e	1,400	c	*1,400*	a	*1,200*	a	*780*	a	*400*	a	*160*	a
25	*10*	a	*44*	a	140	e	250	g	400	g	690	g	1,200	f	1,500	e	*1,600*	b	*1,200*	a	*710*	a	*330*	a	*120*	a
50	*20*	a	*77*	a	200	f	310	g	450	g	710	g	1,200	g	1,500	f	*1,800*	c	*1,200*	a	*660*	a	*290*	a	*98*	a
75	*29*	a	*110*	a	250	f	340	g	480	g	720	g	1,200	g	1,500	g	*1,800*	e	*1,200*	a	*640*	a	*260*	a	*86*	a
100	*38*	a	*140*	b	290	g	370	g	500	g	730	g	1,200	g	1,400	g	*1,700*	e	*1,200*	a	*620*	a	*250*	a	*78*	a
150	*55*	a	*200*	b	340	g	410	g	530	i	740	g	1,100	g	1,400	g	*1,700*	f	*1,200*	a	*600*	a	*230*	a	*68*	a
200	*72*	a	*260*	c	390	g	440	g	560	j	750	j	1,100	g	1,300	g	*1,700*	f	*1,200*	a	*580*	a	*210*	a	*62*	a
250	*89*	a	*310*	c	420	g	470	g	580	j	760	j	1,100	g	1,300	g	*1,600*	f	*1,200*	a	*570*	a	*210*	a	*58*	a
300	*110*	a	*350*	d	460	g	490	i	600	j	770	j	1,100	g	1,300	g	*1,600*	h	*1,200*	a	*560*	a	*200*	a	*54*	a
350	*120*	a	*390*	e	480	g	520	i	610	j	780	j	1,100	g	1,200	g	*1,600*	g	*1,200*	a	*550*	a	*190*	a	*52*	a
400	*140*	a	*430*	e	510	g	540	j	630	j	780	j	1,000	g	1,200	g	*1,500*	g	*1,300*	b	*540*	a	*190*	a	*50*	a
430	*150*	a	*450*	e	520	g	550	j	640	j	790	j	1,000	g	1,200	g	*1,500*	g	*1,300*	b	*540*	a	*180*	a	*48*	a

(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4)

a Daphnia, Lampsilis, Ceriodaphnia, Hyalella
b Daphnia, Lampsilis, Ceriodaphnia, Salmo
c Daphnia, Lampsilis, Salmo, Ceriodaphnia
d Daphnia, Salmo, Lampsilis, Ceriodaphnia
e Salmo, Daphnia, Lampsilis, Ceriodaphnia
f Salmo, Daphnia, Lampsilis, Salvelinus
g Salmo, Salvelinus, Daphnia, Lampsilis
h Salmo, Daphnia, Salvelinus, Lampsilis
i Salmo, Salvelinus, Daphnia, Danio
j Salmo, Salvelinus, Danio, Daphnia
k Salmo, Salvelinus, Danio, Pimephales

Table K-9. Freshwater CMC at DOC of 5.0 mg/L and Various Water Total Hardness Levels and pHs.

Total Hardness	Acute Criterion (CMC) (µg/L total aluminum) (DOC=5.0 mg/L)																											
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*10*	a	*50*	a	180	b	**490**	d	970	d	1,700	d	2,600	d	3,000	d	*3,300*	d	*2,800*	a	*1900*	a	*980*	a	*400*	a
25	*25*	a	*110*	a	350	d	760	d	1,300	d	2,000	d	3,000	d	3,300	d	*3,500*	d	*2,900*	a	*1,700*	a	*810*	a	*300*	a
50	*47*	a	*190*	a	550	d	1,000	d	1,600	e	2,400	e	3,100	d	3,400	d	*3,700*	d	*2,900*	a	*1,600*	a	*700*	a	*240*	a
75	*69*	a	*260*	a	710	d	1,200	d	1,900	f	2,600	f	3,200	d	3,500	d	*3,700*	d	*2,900*	b	*1,500*	a	*640*	a	*210*	a
100	*91*	a	*330*	a	850	d	1,400	d	2,100	f	2,800	g	3,300	e	3,500	d	*3,700*	d	*2,900*	b	*1,500*	a	*600*	a	*190*	a
150	*130*	a	*460*	a	1,100	d	1,700	e	2,400	g	3,000	g	3,500	f	3,600	e	*3,700*	d	*2,900*	c	*1,400*	a	*550*	a	*160*	a
200	*170*	a	*590*	b	1,300	d	1,900	e	2,600	g	3,200	g	3,600	f	3,700	e	*3,700*	d	*2,900*	d	*1,400*	a	*520*	a	*150*	a
250	*210*	a	*700*	b	1,500	d	2,100	f	2,800	g	3,400	g	3,700	g	3,700	e	*3,700*	d	*2,900*	d	*1,400*	a	*500*	a	*140*	a
300	*260*	a	*820*	i	1,600	d	2,300	f	3,000	g	3,500	g	3,800	g	3,800	f	*3,700*	d	*2,900*	d	*1,400*	a	*480*	a	*130*	a
350	*290*	a	*930*	c	1,800	d	2,500	g	3,100	g	3,600	g	3,800	g	3,800	f	*3,600*	d	*2,900*	d	*1,300*	a	*460*	a	*130*	a
400	*330*	a	*1,000*	c	1,900	d	2,600	g	3,200	g	3,700	g	3,900	g	3,800	g	*3,600*	d	*2,900*	d	*1,300*	a	*450*	a	*120*	a
430	*360*	a	*1,100*	c	2,000	d	2,700	g	3,300	g	3,700	g	3,900	g	3,900	g	*3,600*	d	*2,900*	d	*1,300*	a	*440*	a	*120*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4).

a Daphnia, Ceriodaphnia, Stenocypris, Nais
b Daphnia, Ceriodaphnia, Stenocypris, Micropterus
c Daphnia, Micropterus, Ceriodaphnia, Stenocypris
d Daphnia, Micropterus, Oncorhynchus, Ceriodaphnia
e Daphnia, Micropterus, Oncorhynchus, Salmo
f Micropterus, Daphnia, Oncorhynchus, Salmo
g Micropterus, Oncorhynchus, Daphnia, Salmo
h Daphnia, Micropterus, Ceriodaphnia, Oncorhynchus
i Daphnia, Ceriodaphnia, Micropterus, Stenocypris

Table K-10. Freshwater CCC at DOC of 5.0 mg/L and Various Water Hardness Levels and pHs.

Total Hardness	Chronic Criterion (CCC) (µg/L total aluminum) (DOC=5.0 mg/L)																							
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*6.5*	a	*31*	a	120	b	260	f	430	g	740	g	1,300	g	1,700	f	*2,200*	d	*1,800*	a	*1200*	a	*610*	a	*250*	a
25	*15*	a	*66*	a	220	e	350	g	500	g	760	g	1,300	g	1,600	g	*2,000*	e	*1,800*	a	*1,100*	a	*500*	a	*190*	a
50	*30*	a	*120*	a	320	e	430	g	550	g	780	g	1,200	g	1,500	g	*1,900*	h	*1,800*	b	*1,000*	a	*440*	a	*150*	a
75	*43*	a	*160*	a	390	f	480	g	590	i	790	j	1,200	g	1,400	g	*1,800*	g	*1,900*	b	*970*	a	*400*	a	*130*	a
100	*57*	a	*210*	a	450	h	520	g	620	j	810	j	1,100	g	1,300	g	*1,700*	g	*2,000*	c	*940*	a	*380*	a	*120*	a
150	*83*	a	*290*	b	540	g	570	g	660	j	830	j	1,100	i	1,300	g	*1,600*	g	*2,000*	c	*900*	a	*350*	a	*100*	a
200	*110*	a	*380*	b	610	g	620	g	700	j	840	j	1,100	j	1,200	g	*1,500*	g	*1,900*	e	*880*	a	*330*	a	*94*	a
250	*130*	a	*470*	b	670	g	660	i	720	j	850	j	1,100	j	1,200	g	*1,500*	g	*1,800*	e	*860*	a	*310*	a	*87*	a
300	*160*	a	*550*	c	720	g	690	j	750	j	860	j	1,100	j	1,200	i	*1,400*	g	*1,800*	e	*850*	a	*300*	a	*82*	a
350	*180*	a	*620*	c	760	g	730	j	770	j	860	j	1,000	j	1,100	j	*1,400*	g	*1,700*	e	*830*	a	*290*	a	*78*	a
400	*210*	a	*690*	c	800	g	760	j	780	j	870	j	1,000	j	1,100	j	*1,300*	g	*1,700*	e	*820*	a	*280*	a	*75*	a
430	*220*	a	*730*	c	830	g	770	j	790	j	870	j	1,000	j	1,100	j	*1,300*	g	*1,700*	e	*820*	a	*280*	a	*73*	a

(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4)

a Daphnia, Lampsilis, Ceriodaphnia, Hyalella
b Daphnia, Lampsilis, Ceriodaphnia, Salmo
c Daphnia, Lampsilis, Salmo, Ceriodaphnia
d Daphnia, Salmo, Lampsilis, Ceriodaphnia
e Salmo, Daphnia, Lampsilis, Ceriodaphnia
f Salmo, Daphnia, Lampsilis, Salvelinus
g Salmo, Salvelinus, Daphnia, Lampsilis
h Salmo, Daphnia, Salvelinus, Lampsilis
i Salmo, Salvelinus, Daphnia, Danio
j Salmo, Salvelinus, Danio, Daphnia
k Salmo, Salvelinus, Danio, Pimephales

Table K-11. Freshwater CMC at DOC of 10.0 mg/L and Various Water Total Hardness Levels and pHs.

Total Hardness	Acute Criterion (CMC) (µg/L total aluminum) (DOC=10.0 mg/L)																													
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*16*	a	*75*	a	280	b	**720**	d	1,300	d	2,200	d	3,300	d	3,800	d	*4,400*	d	*4,300*	b	*2,800*	a	*1,500*	a	*600*	a
25	*37*	a	*160*	a	530	d	1,100	d	1,800	e	2,700	f	3,600	e	4,000	d	*4,500*	d	*4,300*	d	*2,600*	a	*1,200*	a	*450*	a
50	*72*	a	*280*	a	830	d	1,500	d	2,300	f	3,100	g	3,900	f	4,100	e	*4,400*	d	*4,200*	d	*2,400*	a	*1,100*	a	*360*	a
75	*100*	a	*400*	a	1,100	d	1,800	d	2,600	g	3,400	g	4,100	g	4,200	f	*4,300*	d	*4,100*	d	*2,300*	a	*970*	a	*310*	a
100	*140*	a	*500*	a	1,300	d	2,000	e	2,900	g	3,600	g	4,200	g	4,300	g	*4,300*	e	*4,000*	d	*2,300*	a	*910*	a	*280*	a
150	*200*	a	*700*	a	1,700	d	2,500	e	3,300	g	3,900	g	4,300	g	4,400	g	*4,300*	e	*3,900*	d	*2,200*	a	*840*	a	*250*	a
200	*260*	a	*890*	a	2,000	d	2,800	f	3,600	g	4,100	g	4,400	g	4,500	g	*4,300*	f	*3,800*	d	*2,100*	a	*790*	a	*230*	a
250	*330*	a	*1,100*	a	2,300	d	3,100	f	3,800	g	4,200	g	4,500	g	4,500	g	*4,300*	g	*3,700*	d	*2,100*	b	*750*	a	*210*	a
300	*390*	a	*1,200*	b	2,500	d	3,400	g	4,000	g	4,300	g	4,500	g	4,500	g	*4,300*	g	*3,600*	d	*2,000*	b	*720*	a	*200*	a
350	*450*	a	*1,400*	b	2,700	d	3,600	g	4,200	g	4,400	g	4,500	g	4,500	g	*4,300*	g	*3,500*	d	*2,000*	b	*700*	a	*190*	a
400	*510*	a	*1,600*	b	3,000	d	3,900	g	4,300	g	4,500	g	4,600	g	4,500	g	*4,300*	g	*3,500*	d	*2,000*	b	*680*	a	*180*	a
430	*540*	a	*1,700*	b	3,100	d	4,000	g	4,400	g	4,500	g	4,600	g	4,500	g	*4,300*	g	*3,400*	d	*2,000*	i	*670*	a	*180*	a

Bolded values indicate where the 2018 criteria are lower than the 1988 criteria magnitude within the 1988 pH range applied of 6.5-9.0.
(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).
Ranking of four most sensitive genera (Rank 1-Rank 4).
a Daphnia, Ceriodaphnia, Stenocypris, Nais
b Daphnia, Ceriodaphnia, Stenocypris, Micropterus
c Daphnia, Micropterus, Ceriodaphnia, Stenocypris
d Daphnia, Micropterus, Oncorhynchus, Ceriodaphnia
e Daphnia, Micropterus, Oncorhynchus, Salmo
f Micropterus, Daphnia, Oncorhynchus, Salmo
g Micropterus, Oncorhynchus, Daphnia, Salmo
h Daphnia, Micropterus, Ceriodaphnia, Oncorhynchus
i Daphnia, Ceriodaphnia, Micropterus, Stenocypris

Table K-12. Freshwater CCC at DOC of 10.0 mg/L and Various Water Hardness Levels and pHs.

Total Hardness	Chronic Criterion (CCC) (µg/L total aluminum) (DOC=10.0 mg/L)																							
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*9.9*	a	*47*	a	180	b	370	g	540	g	810	g	1,300	g	1,700	g	*2,300*	g	*2,800*	b	*1,800*	a	*930*	a	*380*	a
25	*23*	a	*100*	a	340	d	490	g	610	g	830	i	1,200	g	1,500	g	*2,000*	g	*2,800*	e	*1,600*	a	*760*	a	*280*	a
50	*45*	a	*180*	a	490	e	600	g	690	j	870	j	1,200	j	1,300	g	*1,700*	g	*2,400*	e	*1,500*	a	*660*	a	*220*	a
75	*66*	a	*250*	a	610	f	670	g	740	j	890	j	1,100	j	1,300	j	*1,600*	g	*2,300*	f	*1,500*	a	*600*	a	*200*	a
100	*86*	a	*310*	a	700	f	720	g	780	j	900	j	1,100	j	1,200	j	*1,500*	g	*2,100*	h	*1,400*	a	*570*	a	*180*	a
150	*130*	a	*440*	a	850	g	800	g	830	j	910	j	1,100	j	1,200	j	*1,400*	g	*1,900*	g	*1,400*	a	*520*	a	*160*	a
200	*160*	a	*560*	a	960	g	860	i	870	j	920	j	1,100	j	1,200	j	*1,300*	i	*1,800*	g	*1,300*	b	*490*	a	*140*	a
250	*200*	a	*670*	b	1,100	g	930	j	900	j	930	j	1,100	j	1,100	j	*1,300*	j	*1,700*	g	*1,300*	b	*470*	a	*130*	a
300	*240*	a	*800*	b	1,100	g	980	j	920	j	930	j	1,000	j	1,100	j	*1,200*	j	*1,600*	g	*1,300*	b	*450*	a	*120*	a
350	*280*	a	*920*	b	1,200	g	1,000	j	950	j	950	k	1,000	j	1,100	j	*1,200*	j	*1,500*	g	*1,300*	b	*440*	a	*120*	a
400	*320*	a	*1,000*	b	1,300	g	1,100	j	960	j	970	k	1,000	j	1,100	j	*1,200*	j	*1,500*	g	*1,300*	c	*420*	a	*110*	a
430	*340*	a	*1,100*	b	1,300	g	1,100	j	970	j	970	k	1,000	j	1,100	j	*1,200*	j	*1,400*	g	*1,300*	c	*420*	a	*110*	a

(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4)

a Daphnia, Lampsilis, Ceriodaphnia, Hyalella
b Daphnia, Lampsilis, Ceriodaphnia, Salmo
c Daphnia, Lampsilis, Salmo, Ceriodaphnia
d Daphnia, Salmo, Lampsilis, Ceriodaphnia
e Salmo, Daphnia, Lampsilis, Ceriodaphnia
f Salmo, Daphnia, Lampsilis, Salvelinus
g Salmo, Salvelinus, Daphnia, Lampsilis
h Salmo, Daphnia, Salvelinus, Lampsilis
i Salmo, Salvelinus, Daphnia, Danio
j Salmo, Salvelinus, Danio, Daphnia
k Salmo, Salvelinus, Danio, Pimephales

Table K-13. Freshwater CMC at DOC of 12.0 mg/L and Various Water Total Hardness Levels and pHs.

Total Hardness	Acute Criterion (CMC) (µg/L total aluminum) (DOC=12.0 mg/L)																											
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*18*	a	*84*	a	310	b	800	d	1,500	d	2,300	e	3500	d	4,000	d	*4,700*	d	*4,700*	c	*3,200*	a	*1,600*	a	*670*	a
25	*42*	a	*180*	a	590	d	1,200	d	2,000	e	2,900	f	3,800	f	4,100	e	*4,700*	d	*4,600*	d	*2,900*	a	*1,400*	a	*500*	a
50	*80*	a	*320*	a	930	d	1,700	d	2,500	g	3,400	g	4,100	g	4,400	f	*4,500*	d	*4,500*	d	*2,700*	a	*1,200*	a	*400*	a
75	*120*	a	*440*	a	1,200	d	2,000	d	2,900	g	3,600	g	4,300	g	4,500	g	*4,500*	e	*4,300*	d	*2,600*	a	*1,100*	a	*350*	a
100	*150*	a	*560*	a	1,500	d	2,200	e	3,100	g	3,800	g	4,400	g	4,500	g	*4,500*	e	*4,200*	d	*2,500*	a	*1,000*	a	*320*	a
150	*220*	a	*780*	a	1,900	d	2,700	e	3,500	g	4,100	g	4,500	g	4,600	g	*4,500*	f	*4,100*	d	*2,400*	b	*930*	a	*280*	a
200	*290*	a	*990*	a	2,200	d	3,100	f	3,900	g	4,300	g	4,600	g	4,600	g	*4,500*	g	*3,900*	d	*2,400*	b	*880*	a	*250*	a
250	*360*	a	*1,200*	a	2,500	d	3,500	g	4,100	g	4,400	g	4,600	g	4,700	g	*4,500*	g	*3,800*	d	*2,300*	c	*840*	a	*240*	a
300	*430*	a	*1,400*	b	2,800	d	3,700	g	4,300	g	4,500	g	4,700	g	4,700	g	*4,500*	g	*3,700*	d	*2,300*	c	*800*	a	*220*	a
350	*500*	a	*1,600*	b	3,100	d	4,000	g	4,500	g	4,600	g	4,700	g	4,700	g	*4,500*	g	*3,600*	d	*2,200*	h	*780*	a	*210*	a
400	*560*	a	*1,800*	b	3,300	d	4,300	g	4,700	g	4,700	g	4,700	g	4,700	g	*4,400*	g	*3,500*	d	*2,200*	d	*760*	a	*200*	a
430	*600*	a	*1,900*	b	3,500	d	4,400	g	4,800	g	4,800	g	4,700	g	4,700	g	*4,400*	g	*3,500*	d	*2,200*	d	*750*	a	*200*	a

(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4).

a Daphnia, Ceriodaphnia, Stenocypris, Nais
b Daphnia, Ceriodaphnia, Stenocypris, Micropterus
c Daphnia, Micropterus, Ceriodaphnia, Stenocypris
d Daphnia, Micropterus, Oncorhynchus, Ceriodaphnia
e Daphnia, Micropterus, Oncorhynchus, Salmo
f Micropterus, Daphnia, Oncorhynchus, Salmo
g Micropterus, Oncorhynchus, Daphnia, Salmo
h Daphnia, Micropterus, Ceriodaphnia, Oncorhynchus
i Daphnia, Ceriodaphnia, Micropterus, Stenocypris

Table K-14. Freshwater CCC at DOC of 12.0 mg/L and Various Water Hardness Levels and pHs.

Total Hardness	Chronic Criterion (CCC) (µg/L total aluminum) (DOC=12.0 mg/L)																							
	pH 5.0		pH 5.5		pH 6.0		pH 6.5		pH 7.0		pH 7.5		pH 8.0		pH 8.2		pH 8.5		pH 9.0		pH 9.5		pH 10.0		pH 10.5	
10	*11*	a	*52*	a	200	b	410	g	570	g	820	g	1,300	g	1,600	g	*2,200*	g	*3,200*	c	*2,000*	a	*1,000*	a	*420*	a
25	*26*	a	*110*	a	390	d	540	g	650	g	860	j	1,200	g	1,400	g	*1,900*	g	*2,800*	e	*1,800*	a	*850*	a	*310*	a
50	*50*	a	*200*	a	560	e	650	g	730	j	890	j	1,200	j	1,300	j	*1,700*	g	*2,400*	f	*1,700*	a	*730*	a	*250*	a
75	*73*	a	*280*	a	680	f	730	g	780	j	910	j	1,100	j	1,300	j	*1,500*	g	*2,200*	g	*1,600*	a	*670*	a	*220*	a
100	*96*	a	*350*	a	790	f	780	g	820	j	920	j	1,100	j	1,200	j	*1,400*	g	*2,100*	g	*1,600*	a	*630*	a	*200*	a
150	*140*	a	*490*	a	950	g	870	g	880	j	940	j	1,100	j	1,200	j	*1,300*	j	*1,800*	g	*1,600*	b	*580*	a	*170*	a
200	*180*	a	*620*	a	1,100	g	950	i	920	j	940	j	1,100	j	1,100	j	*1,300*	j	*1,700*	g	*1,600*	b	*550*	a	*160*	a
250	*230*	a	*740*	a	1,200	g	1,000	j	950	j	950	k	1,000	j	1,100	j	*1,200*	j	*1,600*	g	*1,500*	c	*520*	a	*150*	a
300	*270*	a	*880*	b	1,300	g	1,100	j	980	j	980	k	1,000	j	1,100	j	*1,200*	j	*1,500*	g	*1,500*	c	*500*	a	*140*	a
350	*310*	a	*1,000*	b	1,400	g	1,100	j	1,000	j	990	k	1,000	j	1,100	j	*1,200*	j	*1,400*	g	*1,500*	c	*490*	a	*130*	a
400	*350*	a	*1,100*	b	1,400	g	1,200	j	1,000	j	1,000	k	1,000	k	1,000	j	*1,100*	j	*1,400*	g	*1,400*	d	*470*	a	*130*	a
430	*380*	a	*1,200*	b	1,500	g	1,200	j	1,000	j	1,000	k	1,000	k	1,000	j	*1,100*	j	*1,300*	g	*1,400*	e	*470*	a	*120*	a

(Italicized and underlined values are outside the pH limits of the empirical data used to generate the MLR models and should be used with caution).

Ranking of four most sensitive genera (Rank 1-Rank 4)

a Daphnia, Lampsilis, Ceriodaphnia, Hyalella
b Daphnia, Lampsilis, Ceriodaphnia, Salmo
c Daphnia, Lampsilis, Salmo, Ceriodaphnia
d Daphnia, Salmo, Lampsilis, Ceriodaphnia
e Salmo, Daphnia, Lampsilis, Ceriodaphnia
f Salmo, Daphnia, Lampsilis, Salvelinus
g Salmo, Salvelinus, Daphnia, Lampsilis
h Salmo, Daphnia, Salvelinus, Lampsilis
i Salmo, Salvelinus, Daphnia, Danio
j Salmo, Salvelinus, Danio, Daphnia
k Salmo, Salvelinus, Danio, Pimephales

Appendix L EPA's MLR Model Comparison of DeForest et al. (2018b) Pooled and Individual-Species Model Options

Background

The EPA conducted a comparison of the DeForest et al. (2018b) pooled MLRs (fish and invertebrate data pooled) and individual-species MLRs (fish and invertebrates regressed separately) in order to determine which approach would be most appropriate for use in the Final 2018 Aluminum Aquatic Life AWQC. This appendix describes the EPA's analysis.

DeForest et al. (2018b) updated the individual-species MLR models to incorporate new toxicity data, with the addition of nine *Ceriodaphnia dubia* and nine *Pimephales promelas* toxicity tests under water chemistry conditions that were largely not addressed in the 2017 EPA Draft Aluminum AWQC or the DeForest et al. (2018a) publication. These toxicity tests were conducted by Oregon State University (OSU) and provided to the EPA and DeForest et al. as a courtesy in 2018. These new toxicity tests included fish and invertebrate testing under higher DOC concentration, higher hardness, and slightly higher pH conditions that were not included in the original publication and MLR database (DeForest et al. 2018a). DeForest et al. provided the MLR analyses, using both the new and older datasets in an memorandum to the EPA (DeForest et al. 2018b).

In addition to the analyses described in this appendix, the EPA subjected the DeForest et al. (2018b) memorandum to independent, external expert peer review in 2018. Several of the external peer reviewers noticed trends in the data and criteria derived using the pooled model. (See EPA's website for the Aluminum AWQC [https://www.epa.gov/wqc/aquatic-life-criteria-aluminum] for supporting documentation including the external peer review reports and EPA's responses to the external peer reviewer comments).

The conditions addressed in these new toxicity tests expanded the water quality conditions for model development (**Table L-1**). All conditions and effect concentrations for the 32 *Ceriodaphnia dubia* and 31 *Pimephales promelas* tests are presented in **Table L-2**.

Table L-1. Range of Water Quality Conditions Tested for MLR Model Development.

		Number of test	Range of Water Quality Conditions Tested		
			DOC (mg/L)	pH	Total Hardness (mg/L as $CaCO_3$)
Expanded database	*Ceriodaphnia dubia*	32	0.1-12.3	6.3-8.7	9.8-428
Former database	*Ceriodaphnia dubia*	23	0.1-4	6.3-8.1	9.8-123
Expanded database	*Pimephales promelas*	31	0.08-11.6	6.0-8.12	10.2-422
Former database	*Pimephales promelas*	22	0.08-5.0	6.0-8.0	10.2-127

Table L-2. Database Used for MLR Model Development.

Species	Endpoint	Duration	DOC (mg/L)	pH	Total Hardness (mg/L)	EC_{20} (µg Al/L)	Lower 95% CI	Upper 95% CI	Reference	
Ceriodaphnia dubia	Reproduction	7 d	0.1	6.92	9.8	124	12	1259	CIMM 2009	
Ceriodaphnia dubia	Reproduction	7 d	0.1	7.84	9.8	379	141	1020	CIMM 2009	
Ceriodaphnia dubia	Reproduction	7 d	0.1	6.34	25	37	22	62	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.1	6.4	60	160	123	209	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.1	6.38	121	222	105	466	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	2	6.34	25	377	159	895	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	2	6.38	61	631	362	1101	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	2	6.37	121	1012	692	1479	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	4	6.33	25	623	532	729	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	4	6.3	61	693	618	777	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	4	6.38	121	841	773	914	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.3	7.15	50	1780			Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.3	7.61	51	426	249	727	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	2	6.37	25	353	268	465	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	2	6.34	25	452	401	511	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	2	6.35	25	440	357	523	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.5	6.34	26	260	170	310	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.5	6.36	122	390	170	450	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.5	7	26	250	150	340	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.5	7.1	123	860	590	1090	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.5	8	25	700	510	830	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.5	8	62	1010	740	1180	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	0.5	8.1	123	870	710	1130	Gensemer et al. 2018	
Ceriodaphnia dubia	Reproduction	7 d	1.87	6.42	64	829	437	1572	OSU 2018a	new
Ceriodaphnia dubia	Reproduction	7 d	8.71	6.33	133	3829			OSU 2018a	new
Ceriodaphnia dubia	Reproduction	7 d	12.3	6.40	138	6224	3866	10022	OSU 2018a	new
Ceriodaphnia dubia	Reproduction	7 d	1.64	6.30	428	2011	1539	2628	OSU 2018a	new
Ceriodaphnia dubia	Reproduction	7 d	6.57	7.21	125	6401	4274	9588	OSU 2018a	new
Ceriodaphnia dubia	Reproduction	7 d	12.01	7.19	127	6612			OSU 2018a	new
Ceriodaphnia dubia	Reproduction	7 d	1.3	8.17	263	3749	2904	4838	OSU 2018a	new
Ceriodaphnia dubia	Reproduction	7 d	1.2	8.21	425	2852	1647	4939	OSU 2018a	new

Species	Endpoint	Duration	DOC (mg/L)	pH	Total Hardness (mg/L)	EC_{20} (µg Al/L)	Lower 95% CI	Upper 95% CI	Reference	
Ceriodaphnia dubia	Reproduction	7 d	1.04	8.7	125	1693			OSU 2018a	new
Pimephales promelas	Mean Dry Biomass	7 d	0.3	8	48	10753	1458	79301	Parametrix 2009	
Pimephales promelas	Mean Dry Biomass	7 d	0.08	6	10.6	127	-	-	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.19	6.1	25.8	136	98	188	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.22	6	60.8	314	200	495	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.09	6	123.9	624	410	951	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.92	6.1	10.2	426	402	451	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.86	6.1	61	634	338	1190	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.88	6.1	123.7	773	559	1070	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	1.73	6.1	10.6	633	497	805	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	1.74	6	59.9	1326	1119	1571	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	1.56	6	118.2	1494	1116	1999	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	3.35	6	11.8	829	691	995	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	3.51	6	64.8	2523	1971	3230	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	3.27	6	119.6	2938	2288	3772	Gensemer et al. 2018	
Pimephales promelas	Larval Survival	33 d	0.3	6	93.9	429			Cardwell et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.7	6.1	25.9	660	364	1197	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.9	6	116	824	393	1729	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	2.9	6.1	122	2210	1640	2978	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.8	7.1	26.5	1534	932	2522	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	2.5	7	123	5411	3144	9313	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	0.7	8	28.8	7262	4714	11187	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	5	7.9	127	6795	3161	14607	Gensemer et al. 2018	
Pimephales promelas	Mean Dry Biomass	7 d	7	6.04	134	4618	3281	6499	OSU 2018b	new
Pimephales promelas	Mean Dry Biomass	7 d	11.5	6.04	131	9511	7291	12408	OSU 2018b	new
Pimephales promelas	Mean Dry Biomass	7 d	1.1	6.82	422	2969	2010	4386	OSU 2018b	new
Pimephales promelas	Mean Dry Biomass	7 d	7.2	7.00	135	8047	6273	10322	OSU 2018b	new
Pimephales promelas	Mean Dry Biomass	7 d	11.6	6.96	125	12542	6598	23842	OSU 2018b	new
Pimephales promelas	Mean Dry Biomass	7 d	1.1	8.06	288	5634	1768	17957	OSU 2018b	new
Pimephales promelas	Mean Dry Biomass	7 d	1.6	8.12	396	13274	6674	26401	OSU 2018b	new
Pimephales promelas	Mean Dry Biomass	7 d	0.8	6.1	49	885	574	1365	OSU 2018d	new
Pimephales promelas	Mean Dry Biomass	7 d	1.6	6	94	1817	1444	2287	OSU 2018d	new

DeForest et al. (2018b) developed a pooled MLR model that combined the two datasets, fish and invertebrate, with common slopes for the multiple linear regression test parameters. Deforest et al. (2018b) provided the EPA with a memorandum that presented four new MLR models: 1) a *C. dubia* Individual-species MLR Model; 2) *C. dubia* Pooled MLR Model (*C. dubia* and *P. promelas* data pooled, but using *C. dubia* intercept); 3) *P. promelas* Individual-species MLR Model; and 4) *P. promelas* Pooled MLR Model (*C. dubia* and *P. promelas* data pooled, but using *P. promelas* intercept).

Note: the species-specific intercepts in the pooled model account for the difference in sensitivity of the two test organisms, but slopes for each test parameter are the same. To incorporate these models into AWQC, the EPA evaluated the most appropriate approach to normalize the freshwater aluminum toxicity data by comparing model performance. The DeForest et al. reported models from their 2018 memorandum were:

Invertebrate-focused models

C. dubia Individual-species MLR Model:

$$C.dubia\ EC_{20} = e^{\left[-32.523+[0.597\times\ln(DOC)]+[2.089\times\ln(hard)]+(8.802\times pH)-(0.491\times pH^2)-[0.230\times pH:\ln(hard)]\right]}$$

C. dubia Pooled MLR Model (*C. dubia* and *P. promelas* data pooled, but using *C. dubia* intercept):

$$C.dubia\ EC_{20} = e^{[-8.555+[0.592\times\ln(DOC)]+[2.188\times\ln(hard)]+(1.998\times pH)-[0.268\times pH:\ln(hard)]]}$$

Vertebrate-focused models

P. promelas Individual-species MLR Model:

$$P.promelas\ EC_{20} = e^{[-7.371+[2.209\times\ln(DOC)]+[1.862\times\ln(hard)]+(2.041\times pH)-[0.232\times pH:\ln(hard)]-[0.261\times pH:\ln(DOC)]]}$$

P. promelas Pooled MLR Model (*C. dubia* and *P. promelas* data pooled, but using *P. promelas* intercept):

$$P.promelas\ EC_{20} = e^{[-7.550+[0.592\times\ln(DOC)]+[2.188\times\ln(hard)]+(1.998\times pH)-[0.268\times pH:\ln(hard)]]}$$

The EPA Analysis of the DeForest et al. (2018b) MLR Models

The EPA analyzed model performance to determine if it was more appropriate to normalize the freshwater toxicity data using the two individual models applied to vertebrate and invertebrates separately or to use the common pooled slope model to normalize all the data regardless of taxonomy. As DeForest et al. (2018b) suggested in the memorandum, both the pooled model and the individual models performed similarly when comparing observed versus predicted values, with predicted values within a factor of two being a benchmark to determine performance. **Figure L-1** show that 31/32 (97%) of the predicted values for the *C. dubia* tests for both MLR models were within a factor of two (DeForest et al. 2018b). The individual model for *P. promelas* had a similar level of performance with 30/31 (97%) of the tests within a factor of two, while the pooled model was only slightly less with 29/31 (94%) of the predicted values within a factor of two of the observed values (**Figure L-2**) (DeForest et al. 2018b).

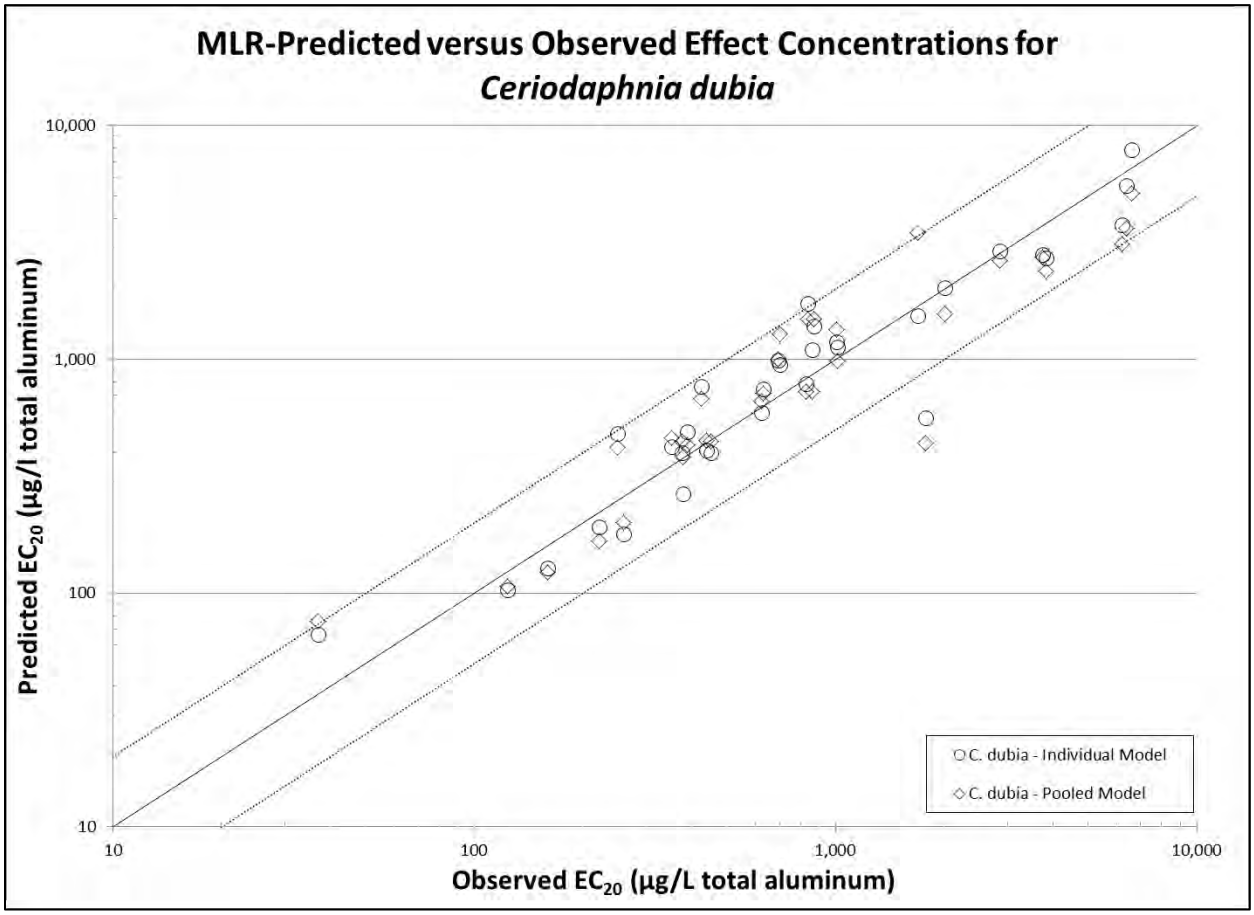

Figure L-1. Predicted versus Observed Values for the *C. dubia* MLR models.
(The solid diagonal line represents a 1:1 relationship while the dotted diagonal lines represent a factor of two).

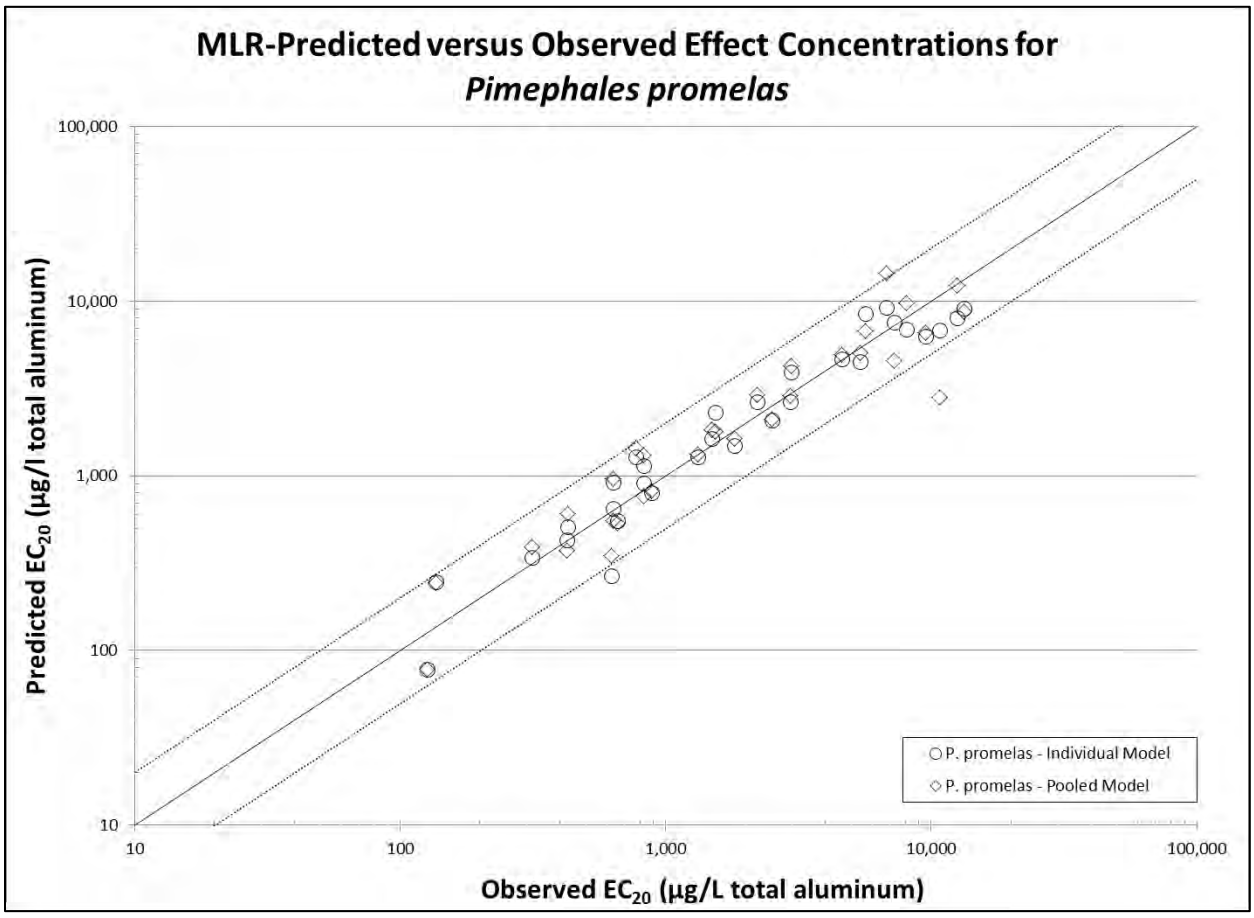

Figure L-2. Predicted versus Observed Values for the *P. promelas* MLR models.
(The solid diagonal line represents a 1:1 relationship while the dotted diagonal lines represent a factor of two).

In order to refine the analysis, the EPA looked at the residuals (observed value minus the predicted value) to determine if one model fit the data better. This analysis is similar to the approach in DeForest et al. (2018a). The residuals were plotted against each individual water quality parameter (pH, total hardness and DOC) to determine if either model generated a biased predicted value. All parameters were natural log transformed for clarity of presentation except pH.

The results of these plots revealed that the *C. dubia* pooled MLR model was over predicting test concentrations (higher predicted EC_{20}s than observed values) as pH increased, and under predicting test concentrations as DOC and total hardness increased (lower predicted EC_{20}s than observed values) (**Figure L-3**, **Figure L-4** and **Figure L-5**). Conversely, the *C. dubia* individual-species MLR model showed no trends in the residuals over any of the test parameters.

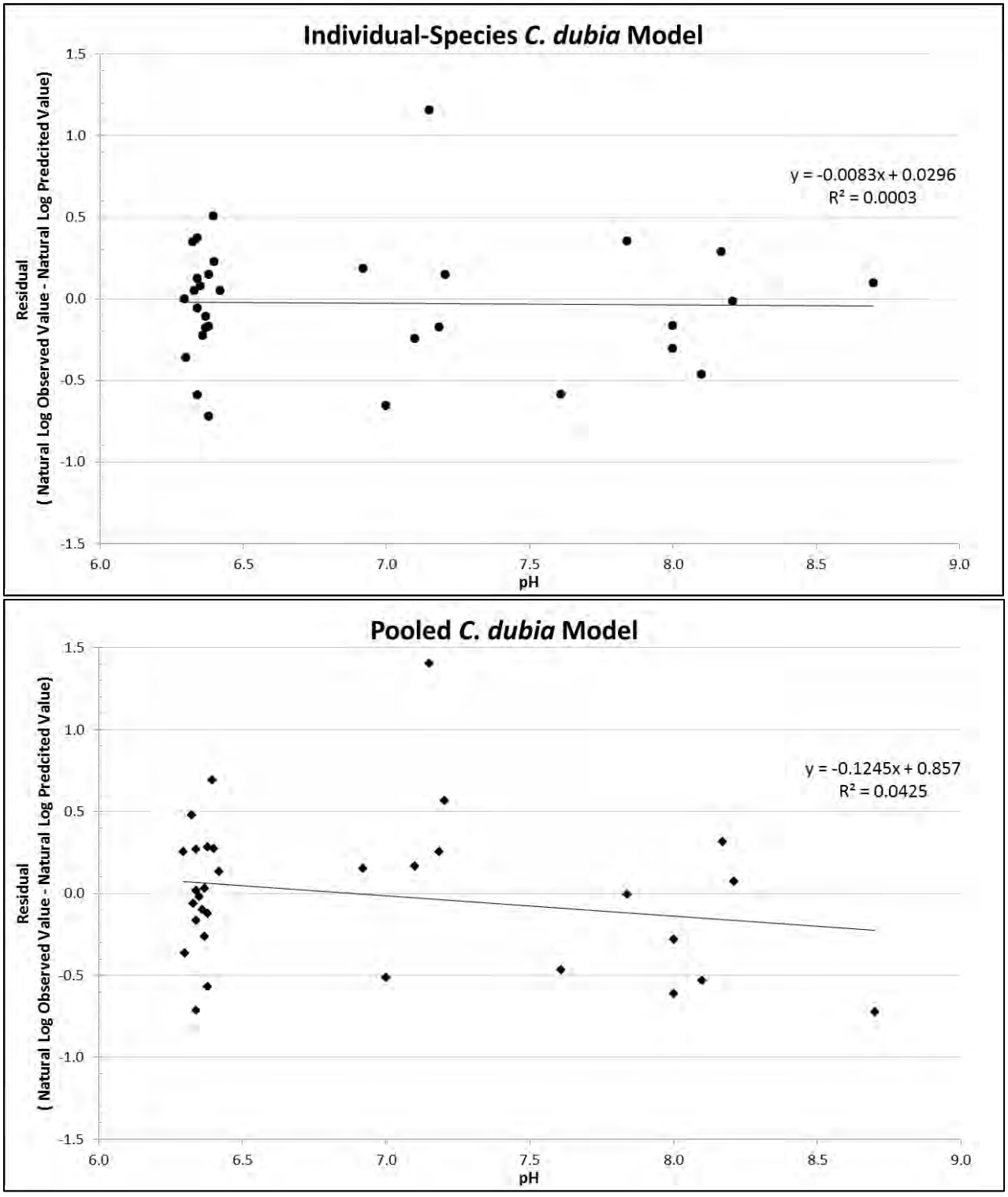

Figure L-3. Residual Plots for the *Ceriodaphnia dubia* models versus pH.

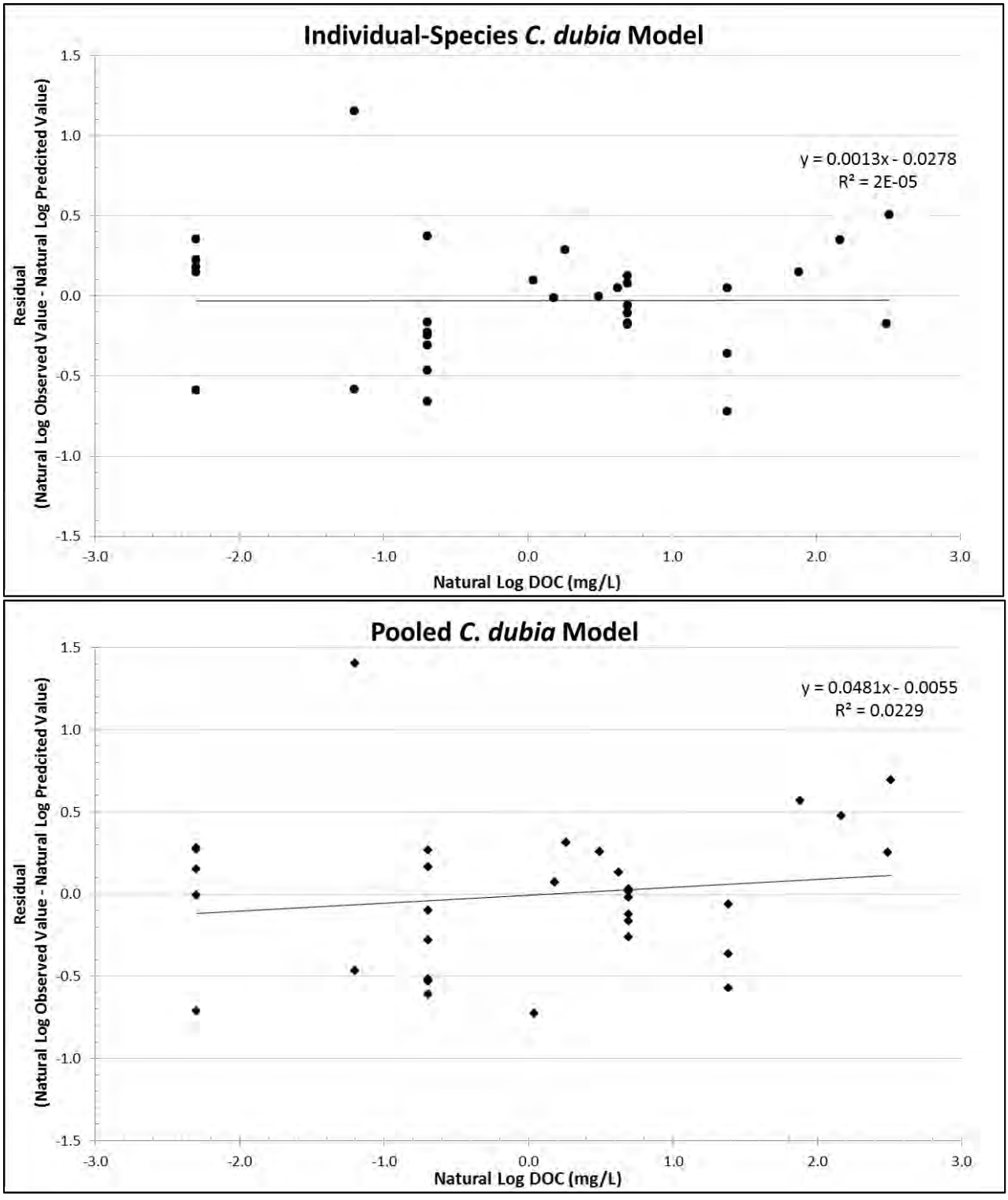

Figure L-4. Residual Plots for the *Ceriodaphnia dubia* models versus DOC

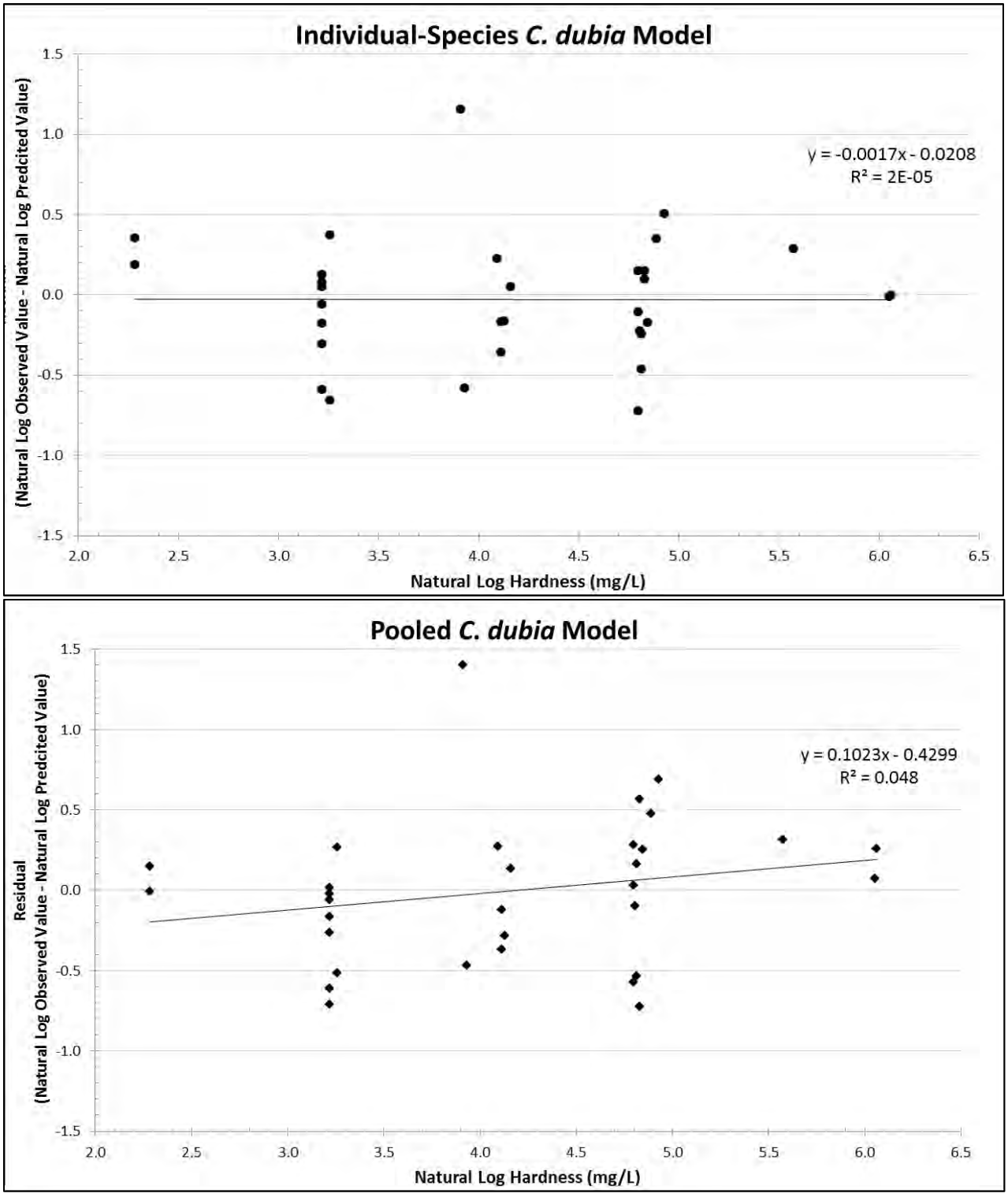

Figure L-5. Residual Plots for the *Ceriodaphnia dubia* models versus Total Hardness.

Similarly, a comparison of the residuals plots for the individual-species *P. promelas* showed no trends in the residuals over any of the test parameters (**Figure L-6**, **Figure L-7** and

Figure L-8). Likewise, there were also trends in the residuals for the pooled *P. promelas* MLR model. The predicted values were over predicting (higher predicted EC_{20}s than observed) as total hardness and DOC increased and under predicting (lower predicted EC_{20}s than observed) as pH increased.

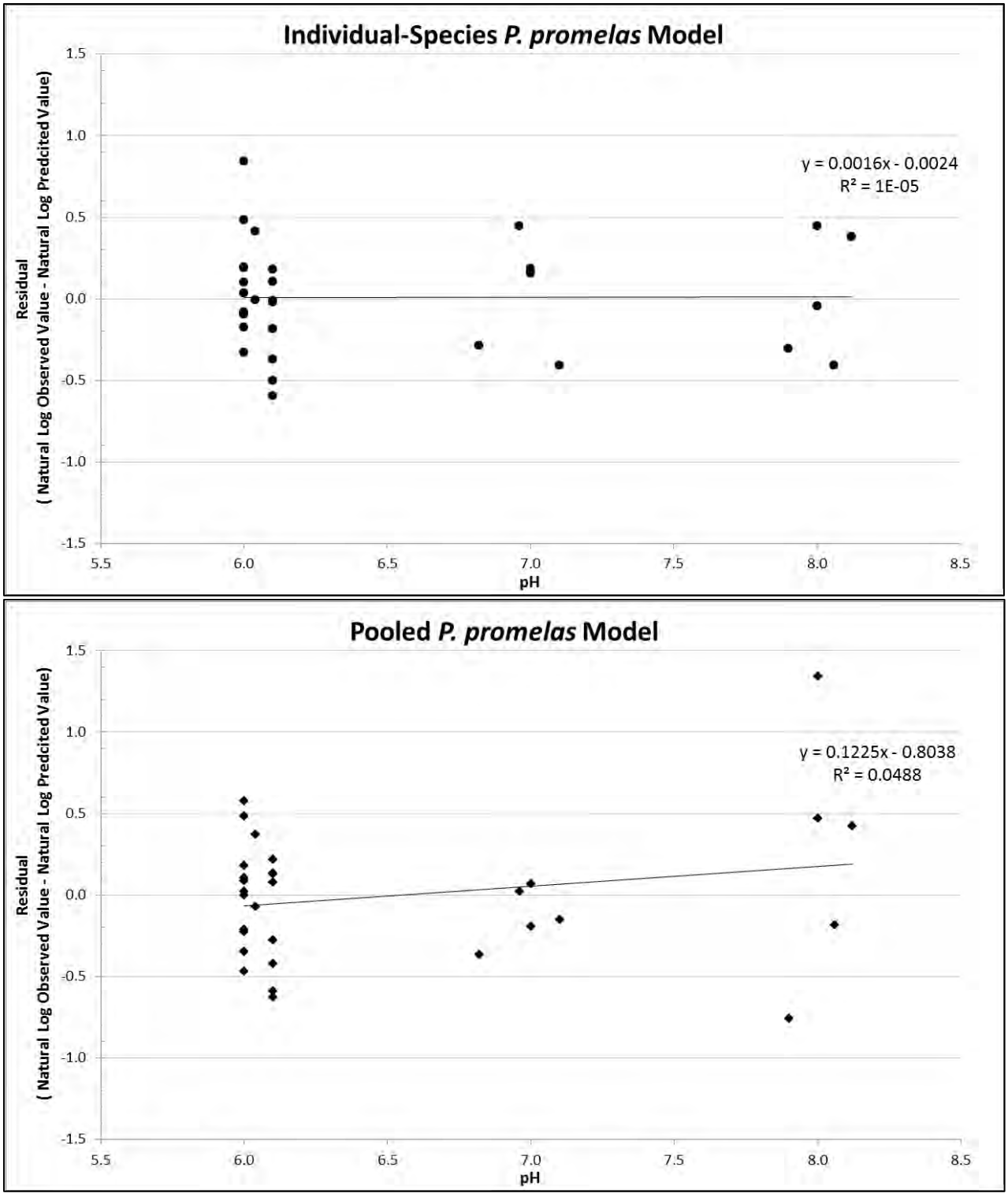

Figure L-6. Residual Plots for the *Pimephales promelas* models versus pH.

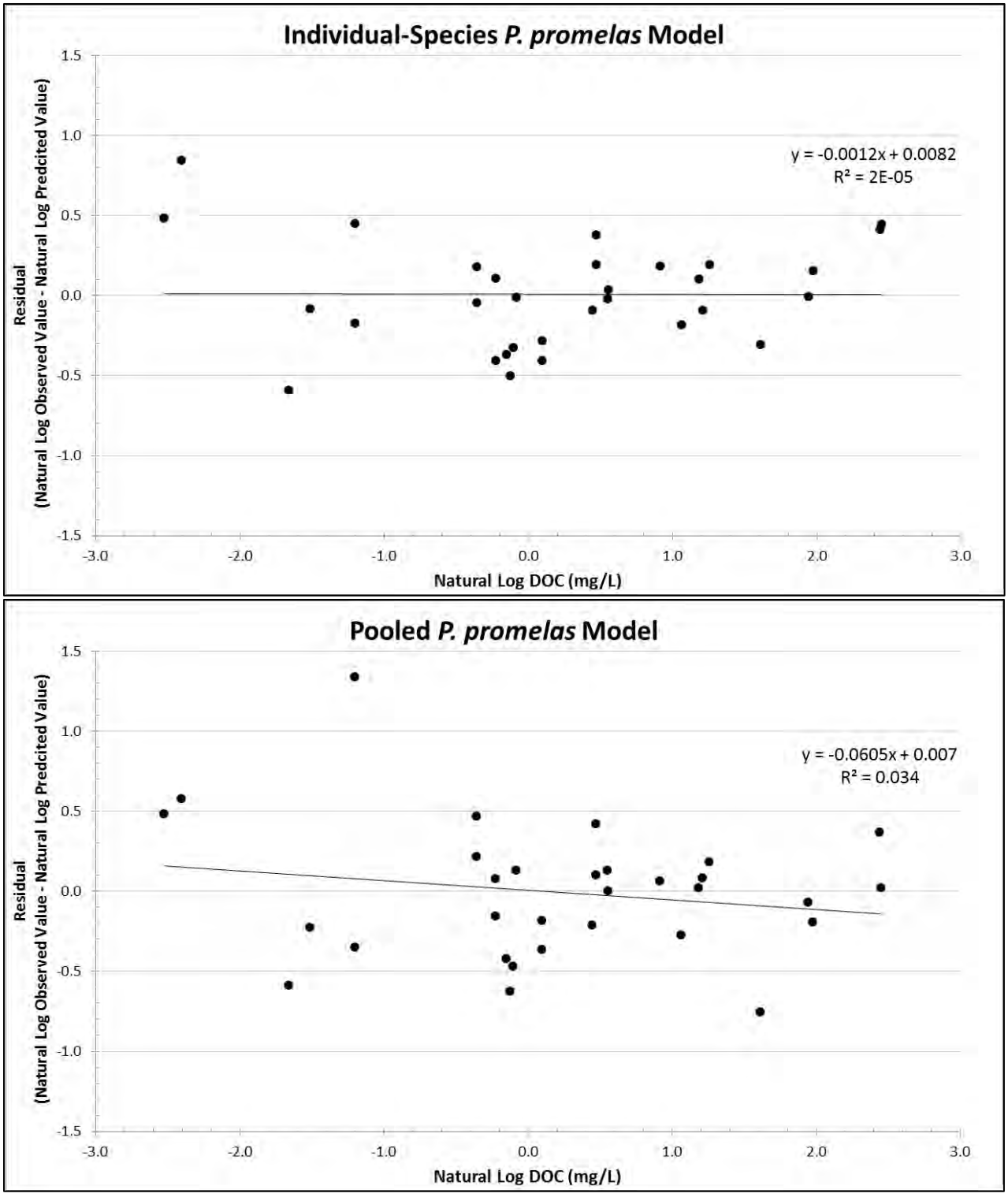

Figure L-7. Residual Plots for the *Pimephales promelas* models versus DOC.

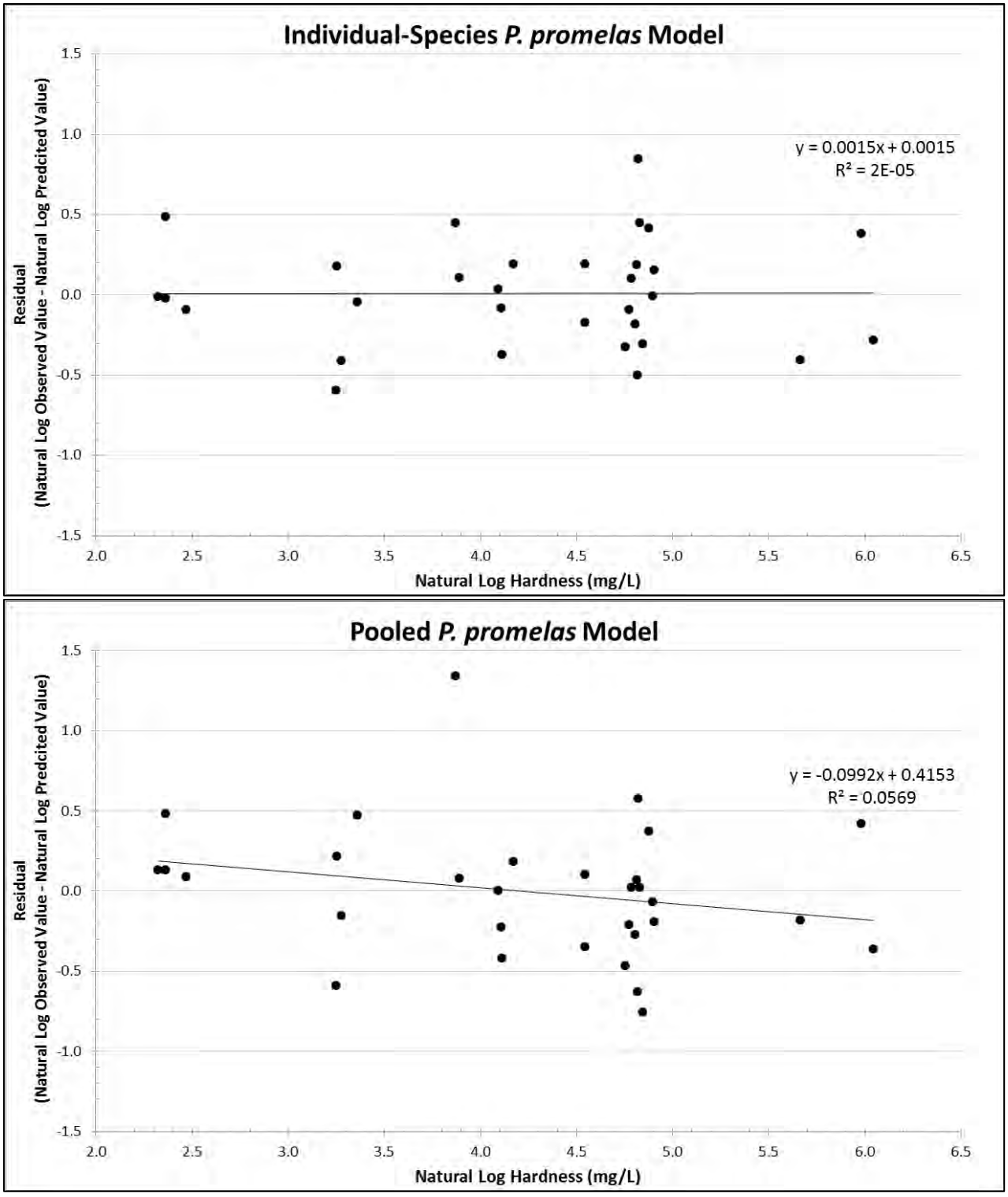

Figure L-8. Residual Plots for the *Pimephales promelas* models versus Total Hardness.

In addition to these residual trends for the pooled model, a poorer fit for the pooled model is indicated by higher standard deviations of the residuals than for the individual-species models.

For the natural logarithm transformed observed and predicted EC_{20}s, the residual standard deviation for the *C. dubia* dataset was 0.45 for the pooled model versus 0.38 for the individual-species model (18% higher). For *P. promelas*, the difference was 0.41 versus 0.32 (27% higher). The statistical significance of this poorer fit was evaluated using an F-test on the merged data across both species. The residual sum-of-squares for the pooled models (SS=11.618, df=57) was reduced 33% by applying the individual-species models (SS=7.814, df=51). For the null hypothesis of no improvement from applying the individual-species models, this translates into a F statistic of 4.14 with 6 and 51 degrees of freedom, rejecting the null hypothesis at $p<0.002$.

Based on these analyses, the EPA decided to use the updated individual-species MLR models presented in DeForest et al. (2018b) to normalize the freshwater aluminum toxicity data in developing the Final 2018 Aluminum Aquatic Life AWQC.

United States Environmental Protection Agency

Office of Water
EPA- 823-D-19-001
July 2019

Draft Technical Support Document:

Implementing the 2018 Recommended Aquatic Life Water Quality Criteria for Aluminum

Notice: This draft question and answer document is intended for states and authorized tribes that wish to adopt and implement the U.S. Environmental Protection Agency's (EPA's) recommended *Final Aquatic Life Ambient Water Quality Criteria for Aluminum*. Pursuant to 40 CFR 131.11(b), when establishing numerical criteria designed to protect designated uses, states and authorized tribes should base those criteria on (i) 304(a) guidance; (ii) 304(a) guidance modified to reflect site-specific conditions; or (iii) other scientifically defensible methods. Because the EPA's Section 304(a) aluminum aquatic life criteria are recommendations, and not requirements, states and authorized tribes should consider the advantages and potential challenges of each approach to adopting the recommended criteria, as well as other approaches that may not be described in this document. This document addresses state and tribal adoption of numeric aluminum criteria under 131.11(b)(1)(i) and (ii).

The national 304(a) recommended aluminium criteria are water-chemistry dependent, and criteria values will vary from site to site based on the values of water chemistry parameters at the site. States and authorized tribes may choose to adopt these criteria into their water quality standards using a performance-based approach.[1] This approach involves a two-step process. First, the state or authorized tribe would adopt criteria and a methodology for deriving site-specific criteria values (both of which the EPA must first approve). Then, after EPA approval, the state or authorized tribe would apply the methodology at specific waterbodies to derive site-specific criteria values for each waterbody. After the EPA approves the state's or authorized tribe's methodology, additional approval of the site-specific criteria values derived on a waterbody-by-waterbody basis would not be required. That is, once the state's or authorized tribe's chosen method of adopting the criteria is approved by the EPA, the state or authorized tribe may use the method to derive site-specific criteria values that are used for other Clean Water Act purposes[2] without additional Agency review. In some cases, more than one method may be appropriate, as explained in Question 1 of this document.

The EPA could update this document as new information becomes available. While this document cites statutes and regulations that contain requirements applicable to water quality standards, it does not impose legally binding requirements on the EPA, states, authorized tribes, other regulatory authorities, or the regulated community and its content might not be appropriate in a particular situation based upon the circumstances. The EPA, state, tribal, and other decision makers retain the discretion to adopt approaches on a case-by-case basis that differ from those provided in this document as appropriate and consistent with statutory and regulatory requirements. In addition to this document, the EPA has other documents which provide considerations and recommendations on implementing the aluminum criteria and can be found at the Agency's aluminum website: https://www.epa.gov/wqc/aquatic-life-criteria-aluminum.

[1] The EPA first described the performance-based approach in the preamble to *EPA Review and Approval of State and Tribal Water Quality Standards* (65 FR 24641, April 27, 2000). "A performance-based approach relies on adoption of a process (i.e., a criterion derivation methodology) rather than a specific outcome (i.e., concentration limit for a pollutant) consistent with 40 CFR 131.11 & 131.13. When such a "performance-based'' approach is sufficiently detailed and has suitable safeguards to ensure predictable, repeatable outcomes, EPA approval of such an approach can also serve as approval of the outcomes as well. If a particular State or Tribe's approach is not sufficiently detailed or lacks appropriate safeguards, then EPA review of a specific outcome is still necessary." (65 FR 24648).

[2] For example, serving as the basis to derive water quality-based effluent limits for National Pollutant Discharge Elimination System (NPDES) permits, identifying impaired and threatened waters for waterbody assessments under Section 303(d) of the Act, and developing total maximum daily loads (TMDLs) for impaired or threatened waters.

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Background: Adopting and Implementing the 2018 Recommended Aquatic Life Water Quality Criteria for Aluminum

On December 21, 2018, the EPA issued the final updated ambient water quality criteria national recommendations to protect aquatic life from the toxic effects of aluminum.[3] The criteria document recommends two primary methods for deriving instantaneous site-specific acute and chronic concentration values for aluminum that would be considered protective of aquatic life, given the conditions of pH, total hardness and dissolved organic carbon (DOC) at the site. For states or authorized tribes that have chosen to adopt the recommended criteria, the EPA recommends two methods to derive the acute and chronic numeric criterion values:

(1) Calculate the criteria values for each waterbody or waterbody segment that has aquatic life as a designated use by entering the pH, total hardness and DOC values into the EPA's Aluminum Criteria Calculator V2.0;[4] or,
(2) Use the lookup tables provided in the criteria document to find the values associated with the specific conditions of pH, total hardness and DOC.

The calculator was derived using a multiple linear regression (MLR) technique to model the interactive effects of three parameters on the bioavailability and toxicity of aluminum to aquatic life,[5] and the lookup tables were created using results from the calculator. Throughout this document, we use the term *input parameters* to refer to site-specific concurrently measured values of pH, total hardness and DOC that a state or authorized tribe may use to derive numeric values of the criteria magnitude (*outputs*) that represent local conditions, using the aluminum criteria calculator or the lookup tables.

Regardless of the method used to derive site-specific criteria values, the state or authorized tribe will need input parameters for pH, total hardness and dissolved organic carbon at each site of interest.[6] These parameters affect the bioavailability of aluminum and its toxicity to aquatic life; however, the interactive effect of these three parameters—pH, total hardness and dissolved organic carbon (DOC)—is not linear.[7]

- For example, if the concentrations of total hardness and DOC are held constant, aluminum is most bioavailable (and therefore, most toxic) at values of high and low pH; and aluminum is least bioavailable near values of neutral pH (again, when total hardness and DOC are held constant).

[3] In accordance with the provisions of Section 304(a) of the Clean Water Act, the EPA periodically revises ambient water-quality criteria to reflect the latest scientific knowledge. For information related to EPA's December 2018 recommended aquatic life criteria for aluminum in freshwater, see: https://www.epa.gov/wqc/2018-final-aquatic-life-criteria-aluminum.
[4] For a link to the criteria document (with lookup tables in Appendix K) and the aluminum criteria calculator (v2.0), see https://www.epa.gov/wqc/aquatic-life-criteria-aluminum#2018.
[5] Two models, one for invertebrates and one for vertebrates, were used to normalize freshwater aluminum toxicity values. These separate models correspond to effects on invertebrates and vertebrates due to differing effects of pH, total hardness and DOC on aluminum bioavailability and toxicity, and therefore enable instantaneous criteria magnitude values to be calculated as a function of the unique chemistry conditions at a given site, at the time at which pH, total hardness and DOC were measured.
[6] Methods using local data are preferred over other methods of deriving site-specific criteria; however, estimated values for DOC may be used in the absence of local data, as described in Question 2.
[7] For more information about the relationships between the pH, total hardness, DOC, and the bioavailability of aluminum to aquatic life, please see the 2018 aluminum criterion document at: https://www.epa.gov/sites/production/files/2018-12/documents/aluminum-final-national-recommended-awqc.pdf.

- Likewise, as the concentration of DOC increases (and pH and total hardness are held constant), aluminum becomes less toxic because the aluminum binds to the DOC, making the metal less bioavailable.

Although pH and DOC are the main factors driving aluminum bioavailability and toxicity, total hardness also has an effect. By knowing the pH, total hardness and DOC in a waterbody, one may derive the numeric criterion values for aluminum, for the acute exposure (i.e., the criterion maximum concentration, CMC) and the chronic exposure (i.e., the criterion continuous concentration, CCC), that will be protective of aquatic life.

1. What flexibility does a state or authorized tribe have when adopting the EPA's recommended aluminum criteria into its water quality standards, and what are the advantages and potential challenges of each approach?

A state or authorized tribe may adopt an EPA-recommended method to establish numeric aluminum criteria protective of aquatic life or may modify the Agency's recommendations to propose an alternative method that reflects site-specific conditions that are not already incorporated into the MLR models which underpin the aluminum criteria calculator that the EPA developed. The EPA regulations also allow States and authorized tribes to adopt scientifically defensible criteria that differ from the EPA's recommendations, if the criteria are protective of designated uses (in the case of these recommended criteria, the designated use is aquatic life).[8] The EPA's Section 304(a) aluminum aquatic life criteria are recommendations, and not requirements. States and authorized tribes should consider the advantages and potential challenges of each approach, as well as other approaches that may not be described in this document.

The EPA's national 304(a) recommended aluminum criteria to protect aquatic life in freshwater ecosystems will vary from site to site based on the values of water chemistry parameters at the site. The derivation of site-specific criteria values relies on the adoption of both the criteria and implementation of a site-specific criterion derivation *methodology* rather than a specific *outcome* (i.e., a constant concentration or criteria magnitude value for a pollutant). Using a performance-based approach, the state or authorized tribe would adopt the water-chemistry dependent criteria and a derivation methodology (both of which EPA must first approve). With sufficient data inputs for pH, total hardness and DOC, the site-specific criteria magnitude values generated by following the methodology are predictable and repeatable. Therefore, once a performance-based approach is approved under CWA Section 303(c), the state or authorized tribe may derive and implement site-specific criteria values without additional Agency review. See question 3 below for information on adoption of implementation methods to support these criteria. States or authorized tribes may consider adopting any of the approaches below as a methodology. More information on performance-based approaches to water quality standards—including the elements the EPA expects to see in any submission of such an approach—may be found in *EPA Review and Approval of State and Tribal Water Quality Standards* (65 FR 24641, April 27, 2000).[9]

[8] In establishing numerical criteria to protect designated uses, states and authorized tribes should base those values on: 304(a) guidance; 304(a) guidance modified to reflect site-specific conditions; or other scientifically defensible methods. (40 CFR 131.11(b)(1)). Additionally, if a waterbody has multiple designated uses with different criteria for the same pollutant, states and authorized tribes should protect the most sensitive use, in accordance with 40 CFR 131.11(a).

[9] For more information, see 40 CFR 131.21(c) State and Tribal Water Quality Standards, "Alaska Rule" (proposed June 1999 and effective May 30, 2000) and EPA Review and Approval of State and Tribal Water Quality Standards, 65 Fed. Reg. 24641 (April 27, 2000) (codified at 45 C.F.R. 131.21(c)).

If a state or authorized tribe chooses to adopt the recommended aluminum criteria into its water quality standards, the EPA recommends choosing one, or a combination,[10] of the following approaches (see Table 1):

(1) adopting by reference to the applicable sections of the 304(a) criteria document (e.g., Section 4.1, Appendix K);
(2) adopting by reference to the Aluminum Criteria Calculator [V2.0] (note: Future versions of the calculator may require the state or authorized tribe to update their standards to incorporate the revised calculator by reference.);
(3) adopting the criteria value lookup tables[11]; or,
(4) adopting relevant ecoregional criteria default values.

[10] For example, approaches (2) and (3) can be adopted together (the calculator can be used to derive numeric criteria values (*outputs*) when input parameters (i.e., pH, total hardness or DOC measures or estimates) are not displayed in the lookup tables). Also, ecoregional criteria default values may be adopted in addition to any of the first three listed approaches. When adopting a combination of approaches, the state or authorized tribe should specify the conditions under which each method should be applied.

[11] If the state or authorized tribe chooses to adopt lookup tables into their water quality standards, they should include information on interpolating or rounding data when input parameter values for pH, total hardness, or DOC fall between values listed in the lookup tables.

Table 1: Comparison of approaches to adopting the recommended aluminum criteria

Approach to adopting the recommended criteria	Advantages	Potential Challenges
(1) adopting the applicable sections of the 304(a) criteria document (which includes the criteria value calculator and criteria value lookup tables).	This approach may provide the greatest amount of background information and context for the criteria and may also provide the greatest flexibility to states and authorized tribes because it allows multiple ways to calculate criteria values.	This approach may be difficult to implement due to individual states' or authorized tribes' legal and administrative requirements (e.g., whether a state's or tribe's regulations allow them to use incorporation by reference).
(2) adopting the aluminum criteria calculator [V2.0] (or similar method to calculate outputs based on the underlying MLR model equations).	This is likely the most adaptable and concise approach. Future updates to the criteria value calculator would likely be accompanied by guidance.	The calculator may be viewed as a "black box," compared to the lookup tables that might be more familiar to some users. Future versions of the calculator may require the state or authorized tribe to update their standards to incorporate the revised calculator by reference.
(3) adopting the criteria value lookup tables.	This may be more transparent than adopting only the criteria value calculator because the lookup tables do not require access to a computer. The tables are included in Appendix K of the publicly-available criteria document and may be helpful when communicating to the public about the criteria implemented at a given site.	The state or authorized tribe may need to develop a standard procedure to determine which values for pH, total hardness, or DOC to use if measured values are *between* the range of input parameter values provided in the lookup table. For example, a state or authorized tribe may decide to always use the nearest value in the table for each input parameter or may decide to use the value which would yield the most protective criteria.
(4) adopting relevant ecoregional[12] criteria default values.	This would allow states and authorized tribes to apply consistent criteria throughout an ecoregion. This approach does not require a state to identify site-specific input parameters because the criteria values are the same for all sites within the ecoregion, calculated based on representative water quality parameters. Defaults may also help to increase transparency of criteria for the public if they are adequately explained. This approach may be used in combination with the criteria calculator or lookup tables, for example as backup criteria for waterbodies with insufficient input parameters.	This approach may be too general for areas with complex geology. That is, the approach, used without the calculator or lookup tables, does not allow for the use of site-specific ambient data (input parameters for a specific site of interest) even though there may be site-specific exceptions within a region.

[12] For more information on how ecoregions are defined, see https://www.epa.gov/eco-research/ecoregions

After adopting the recommended aluminum criteria methodology, and obtaining EPA's approval, the state or authorized tribe may derive site-specific criteria values (i.e., criteria magnitude values) for acute and chronic criteria that correspond to a given ecoregion or set of site-specific conditions for pH, total hardness and DOC. The EPA recommends that states and authorized tribes derive criteria values that will protect aquatic life over the full range of conditions throughout the year (i.e., accounting for the variability of pH, total hardness and DOC). Question 3 provides more information about how to derive final criteria values that will protect aquatic life throughout the range of seasonal and flow conditions at a site, including those conditions when aluminum is most toxic.

To promote transparent and repeatable outcomes, the EPA recommends that states and authorized tribes consider making the input parameters, along with the calculations and the resulting values for aluminum criteria, publicly available on-line. A map showing the extent of the site to which the criteria values apply would also be helpful, if available, to communicate those criteria values to the public. States and authorized tribes may also include detailed methods in an appendix to their water quality standards or in a Continuing Planning Process document, as required by 40 CFR 130.5(b)(6). This provision requires that states include a process for establishing and assuring implementation for new or revised water quality standards.

2. How often and over what time period should a state or authorized tribe collect input parameter data? What if DOC data are insufficient?

The EPA recommends that states and authorized tribes concurrently collect water chemistry data for pH, total hardness and DOC throughout the year to ensure that the collected data are representative of the temporal and spatial variability for each waterbody or waterbody segment. Conditions of pH, total hardness and DOC may vary within a waterbody throughout the year, thereby affecting the bioavailability of aluminum over time.[13] To ensure the criteria will be protective during the times when aluminum is most bioavailable (and most toxic), the EPA recommends that the state or authorized tribe collect, if possible, *24 months of monthly sampling data* for the three input parameters. This approach will help to account for both intra- and inter-annual variability of the input parameters. The EPA recognizes that not all states and authorized tribes will collect this amount of data prior to calculating site-specific aluminum criteria values. Data may be collected for greater than or less than 24 months, and at a frequency greater than or less than monthly. However, higher quality and more consistently collected data will provide more information to establish scientifically defensible criteria that are protective of the designated use. If data collected is insufficient to establish local DOC values, the EPA recommends that default DOC values may be used with concurrently collected data for pH and total hardness. Lastly, ecoregional default values may be used in the absence of local data.

A state or authorized tribe may use the collected or default parameter values as inputs to the aluminum criteria calculator or when using lookup tables to determine the recommended aluminum concentrations for acute and chronic criteria values which correspond to each set of input parameters. That is, if monthly data are provided as inputs to the calculator or lookup tables, then corresponding instantaneous criteria values will be generated for each month data was provided. Then, for permitting or assessment purposes, the state or authorized tribe can analyze the range of monthly instantaneous criteria values to identify criteria values that will be protective under conditions when aluminum is most bioavailable and most toxic to aquatic life. A state or authorized tribe

[13] For example, as DOC increases and pH is constant, aluminum is less bioavailable because it binds to the DOC instead of being bioavailable for uptake into aquatic organisms. Similarly, as total hardness increases and pH is constant, aluminum is less bioavailable because other ions in solution compete with the aluminum ions for uptake into aquatic organisms. That is, as concentrations of calcium and magnesium (i.e., the components of hardness) increase, the relative bioavailability of aluminum decreases.

may find that, depending on the amount and quality of the data, different time periods and conditions throughout the year are best protected by different criteria values. That is, the state or authorized tribe may choose a single set of acute and chronic criteria values which are protective throughout the year or may decide to apply two or more different sets of acute and chronic criteria values appropriate to the different time periods and conditions for permitting and assessment purposes (e.g., seasonal criteria).

Among the input parameters to the aluminum criteria calculator (or lookup tables), DOC data are least likely to be available. That is, states and authorized tribes may collect data for pH and total hardness more routinely than they collect data for DOC. For waterbodies that lack sufficient DOC data, but for which pH and total hardness data are available, the EPA recommends states and authorized tribes use suitable estimates of DOC concentrations, in combination with concurrently measured data for pH and total hardness. The estimated DOC and the measured pH and total hardness values can then be used together to calculate corresponding site-specific criteria values. Because aluminum is most bioavailable and most toxic at low levels of DOC, the EPA recommends using conservative estimates for DOC concentrations which will yield corresponding criteria values that are more likely to ensure protection of aquatic life from the toxic effects of aluminum. States and authorized tribes may also adopt relevant ecoregional criteria default values, as explained in Question 1, or may use other scientifically defensible approaches, including those similar to the approach used to develop the EPA's 2016 draft missing parameters document[14] to estimate input parameter values or to generate default criteria values.

3. What methods can be used to reconcile model outputs and derive criteria values that will result in protection of aquatic life at a site?

The EPA recommends that the final criteria values for each site be derived in a way that will protect aquatic life throughout the range of seasonal and flow conditions at a site, including those conditions of pH, total hardness and DOC, when aluminum is most bioavailable and toxic. There are three methods that the EPA recommends using to derive criteria values that will protect aquatic life throughout the range of seasonal and flow conditions at a given site. Method 1 needs the greatest amount of input parameter data and Method 3 needs the least.

Method 1: Identify protective criteria values by selecting one or more individual model outputs based upon spatially and temporally representative site-specific measured values for model inputs. Method 1 can be used where input datasets are complete and inputs are measured frequently enough to statistically represent changes in the toxicity of aluminum, including conditions under which aluminum is most toxic. In this case, the criteria values are determined by selecting one or more individual outputs that will be protective of aquatic life under the full range of ambient conditions, including conditions of high aluminum toxicity. Method 1 could be used to also establish criteria values to apply on a seasonal basis where the data are sufficient.

Method 2: Calculate protective criteria values from the lowest 10th percentile of the distribution of individual model outputs, based upon spatially and temporally representative site-specific measured model input values. While the 10th percentile of outputs should be protective in a majority of cases, certain circumstances may warrant use of a more stringent model output (e.g. consideration of listed species). Sufficient data to characterize the appropriate distribution of model outputs are necessary to derive a protective percentile so that the site is protected under conditions of high aluminum toxicity.

[14] *Draft Technical Support Document: Recommended Estimates for Missing Water Quality Parameters for Application in EPA's Biotic Ligand Model*, EPA-820-R-15-106, March 2016 (https://www.epa.gov/sites/production/files/2016-02/documents/draft-tsd-recommended-blm-parameters.pdf)

Method 3: Select the lowest model outputs (the lowest CMC and the lowest CCC) calculated from spatially and temporally representative input datasets that capture the most toxic conditions at a site as the criteria values. EPA recommends Method 3 be used where ten or fewer individual model outputs are available.

Either Method 2 or 3 is particularly useful when values of acute and chronic criteria need to be protective of the more toxic of site conditions to be implemented, such as for National Pollutant Discharge Elimination System (NPDES) permitting (as discussed under Question 4). In order to maximize transparency, defensibility, and regulatory certainty, states and authorized tribes should consider developing written implementation methods and make these documents available to the public. To the same end, the EPA recommends states or authorized tribes make publicly available the following on the state's or authorized tribe's website:

- Site-specific water chemistry data, including the inputs used in the aluminum criteria value calculations and resultant criteria values, and
- The geographic extent of each site.

As mentioned in Question 1, a state or authorized tribe may choose to adopt ecoregional default criteria for all or some of the waterbodies within the state or tribal jurisdiction. Where a state or authorized tribe chooses to adopt ecoregional default criteria values, the state or authorized tribe does not need a method for reconciling criteria calculator outputs (because the ecoregional default criteria values are constants that are independent of the criteria calculator).

4. How can a state or authorized tribe implement the aluminum criteria in its Clean Water Act programs?

For NPDES permitting, waterbody assessments and development of total maximum daily loads (TMDLs), states and authorized tribes can use different methods to derive site-specific criteria values (as discussed in the answer to Question 3 above). States and authorized tribes should ensure that the methods used are transparent and predictable, and that they produce repeatable outcomes.

Making information available to the public, the regulated community and other stakeholders is important to ensuring regulatory certainty and clarity, particularly when a state or authorized tribe adopts a performance-based approach. For example,

- states or authorized tribes may wish to describe how they derived the criteria values, including the data and data source used;
- the permitting authority may wish to describe in the permit factsheet or statement of basis how it used the numeric criteria values to determine reasonable potential and derive water quality-based effluent limits (WQBELs), if needed;
- states and authorized tribes may wish to describe in TMDL documents how they derived the criteria values and used them to determine TMDL targets; or,
- states and authorized tribes may wish to describe how they derived site-specific values for the aluminum criteria in assessment methodologies and integrated reports for each assessed waterbody.

Pursuant to 40 CFR 131.10(b), states and authorized tribes must take into consideration water quality standards of downstream waters when designating uses and adopting criteria for instream waters to ensure its water quality standards provide for the attainment and maintenance of the water quality standards for downstream waters. The EPA recommends that states and authorized tribes consider water chemistry conditions

downstream from a point of discharge when deriving aluminum criteria values on a site-specific basis. Because metals are generally persistent and can travel long distances downstream, calculating a criterion value using values for pH, total hardness and DOC from a location at or near a discharge point could result in a criterion value that may not be protective of other areas or downstream waters that have different values of pH, total hardness and DOC. The EPA recommends that states and authorized tribes also account for tributary sources of pH, total hardness or DOC that might affect protectiveness downstream for the aluminum discharged at an upstream point; hence it is recommended that criteria concentration calculations take into account the range of downstream effects of the discharge.

For NPDES permitting purposes, the EPA recommends that states and authorized tribes collect sufficiently representative data for pH, total hardness and DOC to ensure that conditions in the waterbody are being adequately captured downstream from the point of discharge. If a discharge is controlled so that it does not cause water quality standards to be exceeded in the receiving water under critical conditions, then it is reasonable to conclude that water quality standards should be attained under all other conditions. Criteria that will be protective for the more toxic of site conditions should be used to develop WQBELs. Once criteria values protective for the more toxic conditions are calculated, critical low flows—for the purposes of dilution of the pollutant concentration in effluent, combined with effluent concentrations of the pollutant—may be used to establish whether there is reasonable potential for the discharge to cause or contribute to an impairment and therefore a need to establish WQBELs. The *U.S. Environmental Protection Agency's NPDES Permit Writers' Manual*[15] describes the importance of characterizing critical conditions for the effluent and the receiving water. Section 4.5.1 of the *Technical Support Document for Water Quality-based Toxics Control* explains that, where adequate data exist, dynamic modeling techniques may be used in lieu of steady-state modeling using critical conditions.[16] Permit writers may also choose to establish tiered effluent limits. The EPA recommends that, in the context of permit renewals, WQBELs be reevaluated when changes to water chemistry are evident or suspected. Aluminum toxicity in receiving waters could change as the result of a newly permitted discharge or modification of an existing discharge, land-use changes or changes to hydrologic conditions; all of which may affect pH, total hardness and DOC. Additionally, site characterization is important: as the size of a site increases, the spatial and temporal variability are likely to increase. Thus, more water samples may be needed to adequately characterize the entire site.[17]

TMDL and NPDES analysis generally includes considerations for critical conditions. Implementation procedures should clearly define how permit writers will consider critical conditions related to critical low flows and the greatest bioavailability and toxicity of aluminum. This should ensure that reasonable potential is properly assessed and, if needed, appropriate permit limits are established that fully protect aquatic-life beneficial uses under the full range of environmental conditions.

[15] USEPA. 2010. *NPDES Permit Writers' Manual*. U.S. Environmental Protection Agency, Office of Water, Washington, D.C. EPA-833-K-10-001. September 2010.

[16] USEPA. 1991. Technical Support Document for Water Quality-based Toxics Control. U.S Environmental Protection Agency, Office of Water, Washington, D.C. EPA/505/2-90-001. March 1991.

[17] For information on site characterization, see: USEPA. 1994. *Interim Guidance on Determination and Use of Water-Effect Ratios for Metals*. U.S. Environmental Protection Agency, Office of Water, Washington, D.C. EPA-823-B-94-001. February 1994.

United States Environmental Protection Agency

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Draft Technical Support Document: Recommended Estimates for Missing Water Quality Parameters for Application in EPA's Biotic Ligand Model

Disclaimer Page

This technical support document (herein referred to as the "Missing Parameters TSD") summarizes data analysis approaches EPA used to develop recommendations for default values for water quality parameters used in the copper BLM when data are lacking. When published in final form, this document will provide information to states, tribes, and the regulated community interested in using the Biotic Ligand Model to protect aquatic life from toxic effects of copper. Under the CWA, states and tribes are to establish water quality criteria to protect designated uses. State and tribal decision makers retain the discretion to adopt approaches on a case-by-case basis when appropriate. This document does not substitute for the CWA or EPA's regulations; nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, states, tribes, or the regulated community, and might not apply to a particular situation based upon the specific circumstances. EPA may change this document in the future. This document has been approved for publication by the Office of Science and Technology, Office of Water, U.S. Environmental Protection Agency.

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http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/copper/2007_index.cfm

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Acronyms or Abbreviations

Acronym/Abbreviation	Definition
ACR	Acute to Chronic Ratio
ASE	Average Standard Error
BLM	Biotic Ligand Model
BOD	Biochemical Oxygen Demand
CCC	Criterion Continuous Concentration
CMC	Criterion Maximum Concentration
CWA	Clean Water Act
df	Degrees of Freedom
DOC	Dissolved Organic Carbon
Ecoregion	Areas in which ecosystems and the type, quality, and quantity of environmental resources are generally similar. In this document is represented by EPA Level III Ecoregions
EMAP	Environmental Monitoring and Assessment Program
EPA	United States Environmental Protection Agency
FSC	Fixed Site Criteria
GI	Geochemical Ions; ion parameters for the Biotic Ligand Model (Ca, Mg, Cl, Na, K, SO_4, alkalinity)
GIS	Geographic Information System
GLEC	Great Lakes Environmental Center
HUC	Hydrologic Unit Code (a two to eight digit code that identifies each hydrologic unit)
IQR	Interquartile range
IWQC	Instantaneous Water Quality Criteria
LCL	Lower Confidence Limit
LDC	Legacy Data Center (EPA historical STORET database, recently renamed)
LR	Linear Regression
mg/L	Milligrams per liter
NHD	National Hydrography Dataset
NHD-Plus	National Hydrography Dataset Plus
NOCD	National Organic Carbon Database (combined organic carbon data from two databases: USGS WATSTORE and EPA STORET)
NRC	National Research Council
NRSA	National River and Stream Assessment
NWIS	National Waters Information System
PCS	Permit Compliance System
RPDs	Relative Percent Differences
PCA	Principal Components Analysis
pH	Negative logarithm of the hydrogen ion concentration (in moles per liter); scale range from 0 to 14
POC	Particulate Organic Carbon
POTWs	Publicly Owned Treatment Works
RMSE	Root Mean Square Error

Acronym/Abbreviation	Definition
RSS	Residual Sum of Squares
SPLOM	Scatter Plot Matrices
SO	Strahler Stream Order
STORET	EPA STOrage and RETrieval Data Warehouse (recently renamed Legacy Data Center, LDC)
TDS	Total Dissolved Solids
TSD	Technical support document
TSS	Total Suspended Solids
TOC	Total Organic Carbon
UCL	Upper Confidence Limit
U.S.	United States
USGS	United States Geological Survey
WATSTORE	USGS National WATer Data STOrage and REtrieval System (predecessor of NWIS)
WSA	Wadeable Stream Assessment
WQC	Water Quality Criteria
µS/cm	Microsiemens per centimeter
µg/L	Micrograms per liter

Executive Summary

The United States (U.S.) Environmental Protection Agency (EPA) developed revised freshwater aquatic life criteria for copper using the Biotic Ligand Model (BLM) in 2007. The 2007 Freshwater Copper BLM predicts acute copper toxicity based on site-specific water quality parameters, and calculates aquatic life criteria based on the predicted copper toxicity. The current freshwater copper BLM requires 10 input parameters to calculate copper criteria: temperature, pH, dissolved organic carbon (DOC), alkalinity, calcium, magnesium, sodium, potassium, sulfate, and chloride, the last seven of which are also referred to as geochemical ions (GI). Previously available hardness-based copper criteria incorporated consideration only of the effects of hardness on bioavailability, while the BLM incorporates consideration of all of the water chemistry parameters that have a major influence on metal bioavailability. This allows the BLM-based criteria to be customized to the particular water body under consideration. However, given the broad geographical range over which the BLM is likely to be applied, and the limited availability of data for input parameters in many areas, a practical method to estimate missing water quality parameters was needed to successfully run the BLM. This technical support document (herein referred to as the "Missing Parameters TSD") summarizes data analysis approaches EPA used to develop recommendations for default values for water quality parameters used in the Freshwater Copper BLM when data are lacking. These default values could also be used to fill in missing water quality input parameters in the application of other metal BLM models as well, when data are lacking. EPA used three approaches to develop these default value recommendations:

- Conducted geostatistics and conductivity analyses to predict GI parameters
- Applied stream order to refine prediction of GI parameters
- Mined the National Organic Carbon Database (NOCD) to estimate DOC

In brief, EPA found that an approach that used correlation (with conductivity and discharge as explanatory variables), combined with geostatistical techniques (kriging), and a consideration of stream order produced the best estimates for BLM GI parameters. Tables 8, 9, and 10 present estimated inputs for each GI and water hardness in each ecoregion categorized by stream order for low, medium, and high order streams, respectively. Recommended GI values are based on the 10^{th} percentile of ecoregional Level III values for the appropriate stream order (size) and are expected to yield appropriately protective criteria values when applied in the BLM model. In Table 20 of Section 4, EPA provides estimates for DOC by ecoregion based on an analysis of a compilation of national organic carbon databases. The 10^{th} percentile of ecoregional Level III values are recommended for DOC. There was insufficient data to refine the DOC estimates by stream order. EPA recommends measurement of pH and temperature directly to use as an input in the BLM. Temperature is a commonly measured parameter, and should be easily obtainable for use in the BLM. The following paragraphs summarize the contents of each section in this report.

Section 1 provides an introduction to this study, including background on the BLM. In developing the approaches outlined in this study, EPA relied upon several previous studies that attempted to estimate values for BLM input water quality parameters; these studies are outlined briefly in Section 1 and are described in detail in Appendices A through D. This earlier work demonstrates that protective water

quality criteria (WQC) for copper generally corresponded to a low percentile of the distribution of instantaneous water quality criteria (IWQC) predicted by the BLM.

Section 2 provides a discussion of the approach taken by EPA to estimate BLM GI parameter values using geostatistics, which are a suite of statistical methodologies that use spatial coordinates to formulate models used in estimation and prediction. Section 2 also describes how EPA supplemented the geostatistical approach with conductivity as an explanatory variable, because conductivity data are abundant and correlate well to the BLM GI parameters.

Section 3 provides an analysis and discussion of the EPA approach to estimation of BLM GI parameters incorporating stream order as a variable, with a goal of providing BLM users with tables of recommended GI parameter estimates based upon both ecoregion and stream order. For each Level III ecoregion, we recalculated the 10^{th} percentiles of the distributions of all daily water quality parameters measured at all NWIS stations taking into account stream orders or ranges (groups) of stream orders within each ecoregion. Values of the BLM GI parameters generally increased with stream order. Based upon this trend, we grouped the estimates for each parameter by stream order: 1 through 3 (headwater streams), 4 through 6 (mid-reaches), and 7 through 9 (rivers).

Section 4 discusses the estimation of DOC based on the NOCD and two other databases. The NOCD was compiled from a number of sources, including EPA's Storage and Retrieval Data Warehouse (STORET) and the United States Geological Survey's National Water Data Storage and Retrieval System (WATSTORE) (the predecessor of the National Waters Information System (NWIS)). The two other databases, the Wadeable Stream Assessment (WSA) and the National River and Stream Assessment (NRSA), were used to supplement and update the DOC analysis. Section 4 summarizes the data sources, analysis, and uncertainty associated with ecoregional statistics for the NOCD and outlines how tests for bias in the data influence selection of 10^{th} percentile DOC concentrations from either the NOCD or the WSA or NRSA databases. The importance of field sampling for DOC is highlighted in Section 4 because of limitations of the NOCD and the importance of DOC in criteria calculation.

Section 5 provides a summary of the three approaches used to develop EPA's recommendations. Taken together, the approaches presented in this TSD describe EPA's recommendations for default input parameters in the BLM to derive protective freshwater aquatic life criteria when data are lacking. However, it should be noted that site-specific data are always preferable for developing criteria based on the BLM and should be used when possible. Users of the BLM are encouraged to sample their water body of interest, and to analyze the samples for the constituent (parameter) concentrations as a basis for determining BLM inputs where possible.

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1 INTRODUCTION

1.1 Background and Objective

The United States (U.S.) Environmental Protection Agency (EPA) has a congressional mandate to develop and publish criteria for water quality that reflects the effects of pollutants on aquatic life and human health under 304(a)(1) of the Clean Water Act (CWA). The CWA was intended to protect the chemical, physical, and biological integrity of the Nation's waters. Section 304(a)(1) of the CWA, 33 U.S.C. § 1314(a)(1), directs the Administrator of EPA to publish water quality criteria that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that might be expected from the presence of pollutants in any body of water, including ground water. Under this authority, EPA developed revised aquatic life criteria for copper that are based on the Biotic Ligand Model (BLM) in 2007. The BLM predicts metal toxicity based on site-specific water quality parameters, and derives acute and chronic criteria from the predicted toxicity. Derivation of water quality criteria using the BLM requires 10 input parameters (temperature, pH, dissolved organic carbon (DOC), alkalinity, calcium, magnesium, sodium, potassium, sulfate, and chloride). Data regarding these parameters may not be available for many receiving waters. Given that the BLM is likely to be applied over a broad geographical range, and that limited data are available for many areas, a practical method to estimate missing water quality parameters was needed to facilitate full use of the BLM in water quality standards across the U.S. This technical support document (herein referred to as the "Missing Parameters TSD") summarizes data analysis approaches EPA used to develop recommendations for default values for water quality parameters that may be used in the BLM when data are lacking. The section of the CWA related to the development of the information presented in this technical support document is CWA Section 304(a)(2). CWA Section 304(a)(2) generally requires EPA to develop and publish information on the factors necessary to restore and maintain the chemical, physical, and biological integrity of navigable waters. Section 304(a)(2) also allows EPA to provide information on the conditions necessary for the protection and propagation of shellfish, fish, and wildlife in receiving waters and for allowing recreational activities in and on the water.

The objective of this report is to summarize recommendations that BLM users can apply to estimate values for missing input water quality parameters.

1.2 Input Data and the BLM

The BLM calculates metal toxicity to aquatic organisms as a function of the concentrations of certain chemical constituents of water, including, for example, ions that can complex with metals and limit biological availability, and ions that compete with metals for binding sites at the ion exchange tissues of aquatic organisms (e.g., at the fish gill). The BLM predicts the metal criteria concentrations, such as copper in freshwater, which will vary according to changes in the associated water quality parameters.

An appropriately protective acute and chronic copper (or other metals) criteria must reflect the variability of water quality parameters at the site. In previous analyses, EPA found that protective water quality criteria for copper generally correspond to approximately the 2.5th percentile of the

distribution of instantaneous water quality criteria (IWQC) predicted by the BLM[1] (USEPA, 2002). Thus, predictions made for a site using the corresponding low percentile of the water quality parameter distributions are appropriately protective. Copper BLM predictions are most sensitive to the following five important parameters: DOC, pH, and calcium, magnesium, and sodium concentrations (taken together). Estimates are most sensitive to DOC, and vary in direct proportion to a change in value (i.e., they are 100% sensitive to DOC). Estimates are 50% sensitive to a change in pH, and 20% sensitive to the combined concentrations of calcium, magnesium, and sodium.

1.3 Previous Studies

EPA has conducted previous studies to develop tools to estimate BLM water quality parameters for sites where there may be few (or no) water quality data available. Brief summaries of these previous studies are provided below, and more detailed descriptions are provided in Appendices A through D.

1.3.1 An Examination of Spatial Trends in Surface Water Chemistry in the Continental United States: Implications for the Use of Default Values as Inputs to the BLM for Prediction of Acute Metal Toxicity to Aquatic Organisms (Carlton, 2006)

A large database of surface water chemistry monitoring data was examined to look for spatial trends in five chemical constituents that are key inputs to a model for predicting metal toxicity to aquatic organisms: pH, dissolved organic carbon, alkalinity, calcium, and sodium. Continuous prediction maps of concentrations were generated using various kriging techniques to interpolate between site-median values measured at several thousand separate locations throughout the continental U.S. Continuous concentration surfaces were then averaged over 8-digit Hydrologic Unit (HUC) polygons to produce block-averaged mean estimates of site-median concentrations. Pairwise comparisons indicated distinct trends between various HUC-averaged predicted constituents. The same analyses performed on data from 772 locations where all five constituents had been measured revealed similar relationships between monitored constituents. Principal components analyses (PCA) performed on these data sets showed that 80 to 90 percent of the variance in both cases could be explained by a single component with loadings on three of the five constituents. The use of kriging to produce appropriate quantile maps for block-averaging is suggested as a possible approach for developing regional values to use as default model inputs, when site-specific monitoring data are lacking. Refer to Appendix A for more information.

1.3.2 Approaches for Estimating Missing BLM Input Parameters: Correlation Approaches to Estimate BLM Input Parameters Using Conductivity and Discharge as Explanatory Variables (USEPA, 2007)

In this 2007 report, EPA developed regression models to project BLM water quality parameters from conductivity data. EPA assessed supplementing the geostatistical approach with classical estimation methods, such as regression and correlation by assessing the degree of correlation between conductivity and each of the BLM water quality parameters using National Water Information System

[1] This was the median for 17 sites; the range was 1 to 36%.

(NWIS) data from three states (Colorado, Utah, and Wyoming). These states were selected because of the large spatial and temporal variability observed.

EPA concluded that conductivity is significantly correlated with BLM water quality parameters between sites, especially for the low-end distribution statistics of interest for parameter estimation. Since conductivity data are abundant and correlate well with BLM water quality parameters, EPA determined it is reasonable to incorporate conductivity in spatial projections of BLM parameters. Correlation coefficients were lower for pH and DOC than for the geochemical ions (GIs) and alkalinity, but were also significant. Refer to Appendix B for more information.

1.3.3 Copper Biotic Ligand Model (BLM) Software and Supporting Documents Preparation: Development of Tools to Estimate BLM Parameters (USEPA, 2008)

In order to predict parameters based on geographic location, this 2008 report investigated how to project BLM water quality parameters for a given site based on other site data using geostatistical methods. There are a number of ways in which the conductivity regressions can be used to project BLM water quality inputs. The regressions allow estimates of the BLM water quality inputs from either: (1) a limited number of conductivity measurements, or (2) a low-end conductivity value estimated by geostatistical methods.

The first approach, projecting BLM water quality inputs from conductivity measurements, was demonstrated for a limited number of test sites. Regression models were developed to project 10^{th} percentiles of BLM water quality input parameters from the 10^{th} percentile of measured conductivity distributions at sites in Colorado, Utah, and Wyoming. The 10^{th} percentile is the value below which 10 percent of the observations may be found. The regression models were tested using data and copper BLM predictions for four sites, and produced highly consistent results. The regression models for pH and DOC, the most sensitive of the BLM water quality parameters, were not sufficiently accurate to make reliable BLM parameter predictions. However, regression models for the GI parameters (calcium, magnesium, sodium, potassium, chloride, sulfate, and alkalinity) were reasonably accurate, as judged by comparing model predictions made using projected values of the these BLM input parameters to model predictions made using measured input data. No estimate for site-specific pH was superior to the observed weak conductivity regression. To improve upon this estimate, it was necessary to use actual site-specific pH data. For DOC, the Level III ecoregion (referred to herein as simply “ecoregion”) and water body type-specific DOC concentration percentiles tabulated by EPA for the National Bioaccumulation Factors Technical Support Document (USEPA, 2003) appear to be far better estimates of lower-percentile DOC concentrations than the estimates made using the conductivity regression.

EPA also provided a proof of concept for the second approach, which was to see whether combining the kriged conductivities with the conductivity-hardness regression would project the 10^{th} percentiles of hardness better than direct kriging of the hardness data. EPA found that both approaches produce estimates of hardness that correlate significantly with the measured data (correlation coefficient $r=0.80$ for direct kriging of hardness; $r=0.95$ for conductivity kriging + regression). However, the kriging + regression approach fits the hardness data substantially better than direct kriging. Refer to Appendix C for more information.

1.3.4 Approaches for Estimating Missing BLM Input Parameters: Projections of Total Organic Carbon as a Function of Biochemical Oxygen Demand (USEPA, 2006a)

DOC concentrations downstream of an effluent discharge are necessary inputs for the BLM to predict toxicity associated with a wastewater discharge. Effluent DOC is monitored by very few publicly-owned treatment works (POTWs) according to data retrieved from EPA's Permit Compliance System (PCS). Biochemical oxygen demand (BOD) is monitored by most POTWs. EPA developed regressions to project total organic carbon (TOC) concentrations from BOD values using effluent samples at all POTWs reporting data for both parameters in EPA's PCS. EPA concluded that this regression gives reasonable estimates of TOC in POTWs effluents and are likely the best available estimates of effluent TOC to determine DOC concentrations for the BLM. Refer to Appendix D for more information.

1.4 Approaches to Estimate Water Quality Parameters for the BLM

Building upon the studies described above, this report uses three approaches to develop default estimates for parameters needed for the BLM when empirical data are lacking. The three approaches are listed below and are detailed in the following sections:

- Section 2: Using Geostatistics and Conductivity to Predict GI Parameters
- Section 3: Using Stream Order to Refine Prediction of GI Parameters
- Section 4: Using the National Organic Carbon Database to estimate DOC

EPA recommends that temperature and pH be measured directly in the field.

2 USING GEOSTATISTICS AND CONDUCTIVITY TO PREDICT GI PARAMETERS

The following section describes studies that demonstrate how geostatistical techniques, coupled with conductivity correlations, can be used to predict BLM input parameters for GIs when site-specific data are unavailable. In a previous study (USEPA, 2008) EPA demonstrated that combining kriging with regressions to estimate inputs based on conductivity improves the accuracy of GI estimates. In this section EPA has expanded on this approach and developed national estimates at the Level III ecoregion in the continental U.S.

The current freshwater copper BLM requires 10 input parameters that reflect water chemistry in order to calculate copper criteria: temperature, pH, DOC, alkalinity, calcium, magnesium, sodium, potassium, sulfate, and chloride, the last seven of which are GIs. The concentrations of GIs vary in surface waters due to dissolution, weathering, ground water-surface water interactions, and other geologic processes in the watershed, in addition to dilution by snowmelt and precipitation. Consequently, the concentrations of GI parameters tend to vary according to the regional geology. For example, alkalinity has noticeable geographic trends. Areas dominated by carbonate rocks, such as limestone as in the prairie states, tend toward high alkalinity. Areas dominated by igneous rocks, such as granite, such as parts of the northeast, tend toward low hardness and alkalinity.

In this section we expand on the EPA 2008 proof of concept (in Appendix C) using geostatistics to develop default missing GI parameter values based on geography. Geostatistics are statistical methodologies that use spatial coordinates to help formulate models used in estimation and prediction (ESRI, 2003). Geostatistical techniques are attractive because they explain parameter variation arising from spatial correlations, which are not used in conventional statistics. We have supplemented the geostatistical approach by adding conductivity as an additional explanatory variable. Conductivity is one of the most widely monitored water quality indicators in the U.S. Because conductivity data are abundant and correlate well to the BLM GI parameters, we incorporated conductivity in spatial projections of BLM parameters. Based on the proof of concept described above (and in Appendix C), we expected that this approach, which can be implemented by co-kriging (i.e., an interpolation technique that allows for better estimates by the incorporation of well-sampled, correlated secondary data) in geostatistical software, would allow more robust spatial projections of BLM water quality parameters.

2.1 Data Source and Processing

Water quality data for conductivity and BLM GI water quality parameters were retrieved from the United States Geological Survey (USGS) National Water Information System (NWIS). NWIS contains data from millions of sampling events at tens of thousands of individual sampling locations (stations) in the continental U.S. (Figure 1). Not all water quality parameters of relevance to the BLM were monitored at each location. The numbers of sampling events at individual locations also range widely, with a mean of 15, and a mode of one (i.e., most sites were only sampled once). Examination of the spatial distribution of numbers of sampling events per site reveals that the Midwestern and Western states tended to be sampled most intensively (Carleton, 2006). Because environmental sampling data tend to be lognormally distributed, disparities in numbers of samples may tend to produce higher mean and median values at locations that have been sampled more frequently. As spatial distributions of representative (e.g., median) concentrations are examined, it should be kept in mind that apparent

geographic trends in concentration may be in part simply the result of uneven sampling intensity (Carleton, 2006).

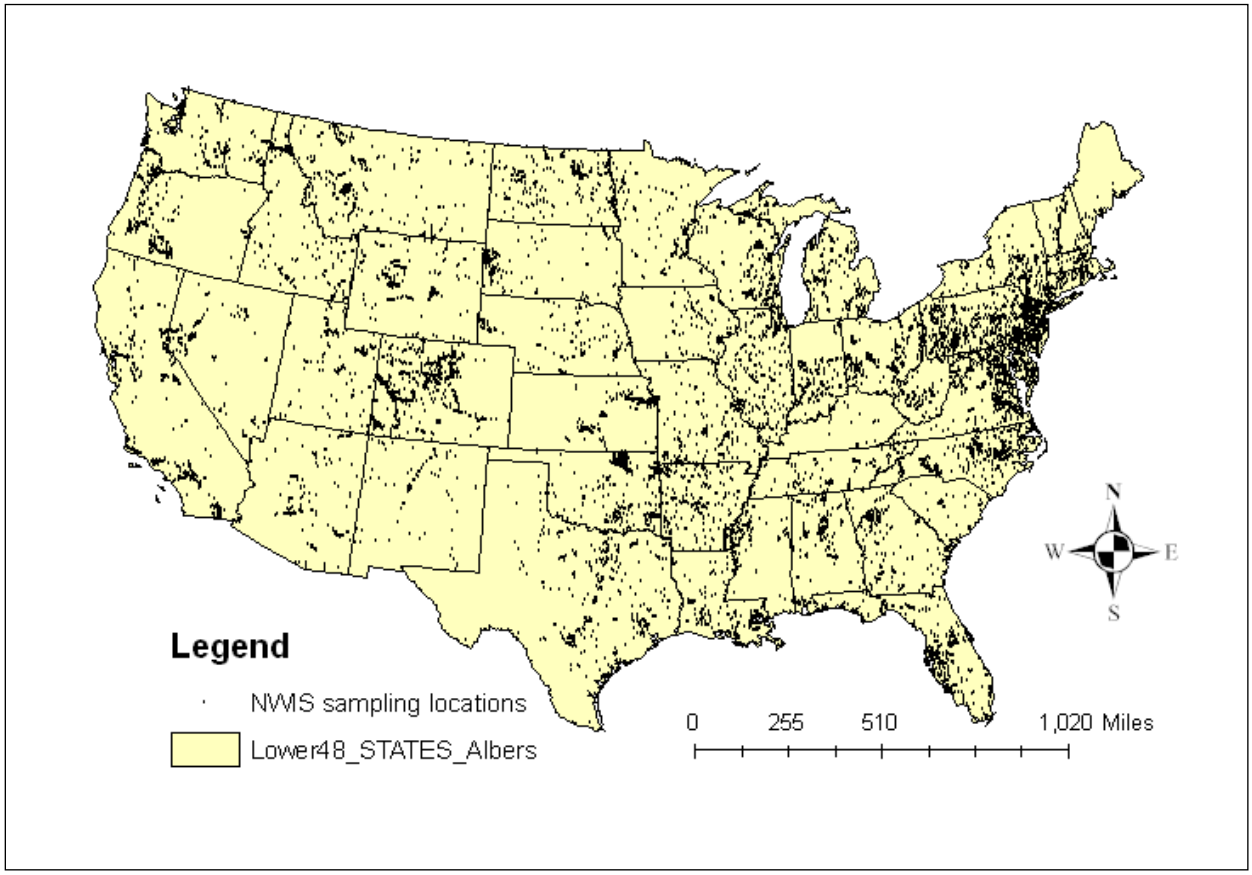

Figure 1. NWIS sample collection locations in the continental U.S. (Carleton, 2006)

We focused our efforts on data collected from rivers and streams between 1984 and 2009. Data collected prior to 1984 were excluded because a number of the analytical methods used by USGS prior to that date have been replaced by methods with improved precision and lower detection limits. Furthermore, only sites with 40 or more samples were included in the analysis. With support from USGS staff, we obtained a complete download of national water quality data from NWIS, which totaled 4,714,165 measurements from 959,946 samples, collected at 5,901 sites. These data included measurements for BLM water quality input parameters required to calculate copper criteria using the BLM: pH, DOC, alkalinity, calcium, magnesium, sodium, potassium, sulfate, and chloride. Data were also collected on filtered (dissolved) copper, and the spatial coordinates (latitude and longitude) of each sampling station were also retrieved. No data were collected on temperature. Only the GI data were included in the geostatistical analysis. A summary of the water quality data retrieved from NWIS is provided in Table 1.

Table 1. Summary of water quality data retrieved from NWIS			
BLM Water Quality Parameter	**NWIS Parameter Code**	**Parameter Description**	**Number of Observations**
Conductivity[1]	00094	Specific conductance, water, unfiltered, field, microsiemens per centimeter at 25 degrees Celsius	553,700
	00095	Specific conductance, water, unfiltered, laboratory, microsiemens per centimeter at 25 degrees Celsius	799
pH	00400	pH, water, unfiltered, field, standard units	352,336
	00403	pH, water, unfiltered, laboratory, standard units	151,161
Dissolver Organic Carbon	00681	Organic carbon, water, filtered, laboratory, milligrams per liter	30,008
Alkalinity	00410	Acid neutralizing capacity, water, unfiltered, fixed endpoint (pH 4.5) titration, field, milligrams per liter as calcium carbonate	35,232
	00417	Acid neutralizing capacity, water, unfiltered, fixed endpoint (pH 4.5) titration, laboratory, milligrams per liter as calcium carbonate	15,264
	00419	Acid neutralizing capacity, water, unfiltered, incremental titration, field, milligrams per liter as calcium carbonate	10,198
	00418	Alkalinity, water, filtered, fixed endpoint (pH 4.5) titration, field, milligrams per liter as calcium carbonate	2,686
Calcium	00915	Calcium, water, filtered, laboratory, milligrams per liter	146,608
Magnesium	00925	Magnesium, water, filtered, laboratory, milligrams per liter	145,938
Sodium	00930	Sodium, water, filtered, laboratory, milligrams per liter	136,310
Potassium	00935	Potassium, water, filtered, laboratory, milligrams per liter	132,659
Sulfate	00945	Sulfate, water, filtered, laboratory, milligrams per liter	147,824
Chloride	00940	Chloride, water, filtered, laboratory, milligrams per liter	146,601

[1] Conductivity is not a BLM parameter, but was used as an explanatory variable for the other water quality parameters.

The data were screened using established quality assurance procedures. All data were checked to confirm that they contained numerical values without null (missing) records and remark codes were

identified and reviewed. Minimum and maximum values for each parameter were confirmed to be within expected ranges and frequency distributions were plotted and examined for each of the parameters to identify outliers. We also confirmed that the spatial coordinate data placed each sampling location within the continental U.S. Additional data processing included the following steps:

- For the data at each station, the observations for each variable were averaged on a daily basis. This was done to reduce the influence of high frequency sampling at a few stations.
- Data were edited by censoring parameter(s) with fewer than 10 to 20 daily values at a station. The 10^{th} percentile for that parameter at that station was censored to improve the reliability of the lower-tail (i.e., 10^{th}) percentile statistics.
- Tenth (rank order/nonparametric) percentiles of the distributions of all water quality parameters measured at each station were calculated.

It should be emphasized that all of the statistical and geostatistical analyses and predictions presented in this report are based on the 10^{th} percentiles of the concentration distribution measured for each parameter at every station. The estimates of water quality parameter values for "missing" data are therefore also 10^{th} percentile concentrations. We selected the 10^{th} percentile of the site parameter distributions as a statistic that is a practical compromise between a lower-bound concentration and a percentile that can be reliably determined from small sample sizes. Initial testing with the BLM suggested that protective water quality criteria (WQC) for copper generally corresponded to approximately the 2.5^{th} percentile of the distribution of instantaneous water quality criteria (IWQC) predicted by the BLM. Thus, BLM predictions made for a site using the corresponding low percentiles of the water quality parameter distributions should (logically) also be a conservative approximation of a protective criterion. As a more reliably determined statistic, the 10^{th} percentile of water quality parameters will also derive reasonably protective criteria, especially for small sample sizes where there may be greater uncertainty at lower percentile estimates. The 10^{th} percentile estimates presented in this document were initially developed to implement the copper BLM published by EPA in 2007 and will apply to other metals as well.

2.2 Geostatistical Analysis of National Data for Geochemical Ions

The ESRI ArcGIS Geostatistical Analyst tool was used to create statistically valid two-dimensional surface models for conductivity and for each of the BLM GI parameters. Using the 10^{th} percentile daily average concentrations at each sampling location from the NWIS data, Geostatistical Analyst was used to create predictions for unmeasured locations throughout the continental U.S. For each parameter, the surface models were fit by minimizing the statistical error of the predicted surface. Surface fitting involved three steps: exploratory spatial data analysis, structural analysis (modeling the semivariogram to analyze surface properties of data from nearby locations), and surface prediction and assessment of the results. The semivariogram represents autocorrelation of measured data points spatially.

Modeling of the semivariogram was based on cross-validation, which calculates error statistics that serve as diagnostics to indicate whether the model is reasonable for map production. Cross-validation was used to select the models that provided the most accurate predictions. The following criteria were used to evaluate goodness of fit for the semivariogram model:

- Mean Standardized Error: close to 0;
- Root Mean Square Error (RMSE): as small as possible;

- Root-Mean-Square Standardized Error: close to 1; and,
- RMSE close to Average Standard Error ASE).

The difference between the prediction and the measured data value is the prediction error. For a model that provides accurate predictions, the mean prediction error should be close to 0 if the predictions are unbiased. The root-mean-square standardized prediction error should be close to 1 if the standard errors are accurate, and RMSE should be small if the predictions are close to the measured values (ESRI, 2003).

A tabulation of the geostatistical model selected for each water quality parameter, the number of data points interpolated, and the resulting error statistics are presented in Table 2. We used the optimal parameters for a spherical semivariogram as calculated by the Geostatistical Analyst. No transformations were applied to the data. Anisotropy (directional influence on the semivariogram) was not incorporated in the semivariogram models.

Table 2. Model selection and cross validation statistics for geostatistical fitting of 10th percentiles of BLM GI parameters

Parameter	Geostatistical model	Number of samples	Mean standardized error	Root mean square error	RMS standardized error	Average standard error
Conductivity	Universal kriging	4833	-0.01038	1361	1.081	1259
Alkalinity	Universal cokriging with conductivity	1372	-0.001115	36.62	1.09	33.23
Calcium	Universal cokriging with conductivity	2590	0.0001694	26.81	1.186	22.02
Magnesium	Universal cokriging with conductivity	2578	-0.002258	15.92	1.16	13.58
Sodium	Universal cokriging with conductivity	2439	-0.002929	156.3	1.583	95.78
Potassium	Universal cokriging with conductivity	2379	-0.001184	3.488	1.429	2.381
Sulfate	Universal cokriging with conductivity	2650	-0.0000225	114.5	1.29	87.04
Chloride	Universal cokriging with conductivity	2792	0.001653	375.2	1.51	247

2.2.1 Kriging of Conductivity Data

Universal kriging with a constant trend was used to map the surface of 10th percentile conductivity values. Kriging weights the surrounding measured values to derive a prediction for each location. The weights are based on the distances between the measured points and the prediction location, as well as the overall spatial arrangement among the measured points. The kriged prediction surface of 10th percentiles of conductivity is mapped in Figure 2. As the kriging results show, conductivities are highest in the south-central and southwestern regions, as well as along the Gulf and southern Atlantic coasts. Regions of lower conductivity are found in a number of parts of the country.

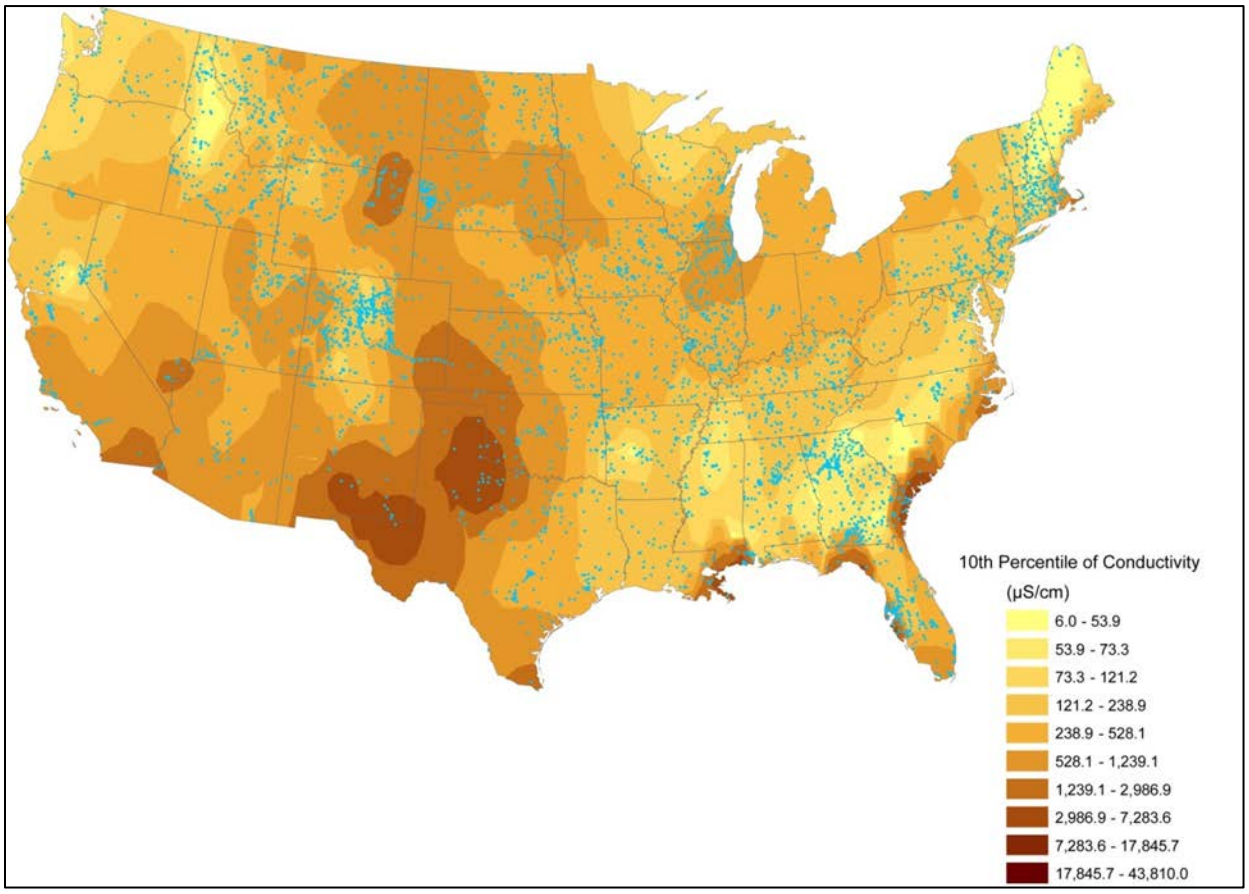

Figure 2. Kriged prediction surface for 10th percentile of conductivity in the continental U.S. (sample locations in blue)

2.2.2 Co-kriging of GI Data

Co-kriging was used to improve surface predictions of the BLM GI parameters by taking into account secondary variables, in this case conductivity. As demonstrated above, conductivity is significantly correlated to all of the BLM GIs. Universal co-kriging with conductivity, assuming a constant trend, was used to map the surface of 10th percentile BLM GIs concentrations. For each of these parameters, co-kriging produced cross-validation errors that were superior in terms of the goodness-of-fit criteria to errors produced by universal kriging. Prediction surfaces for calcium and alkalinity are mapped in Figures 3 and 4. The spatial distribution of calcium (Figure 3) shares a number of similarities with the mapping of conductivity (Figure 2). The co-kriged alkalinity surface (Figure 4) is rather different, with high alkalinity values reflecting geographic features (such as the carbonate geology of the prairie states) and low alkalinity values that reflect the granitic geology of the northeast. Prediction surfaces for the other BLM GI's are generally similar to those for conductivity and calcium.

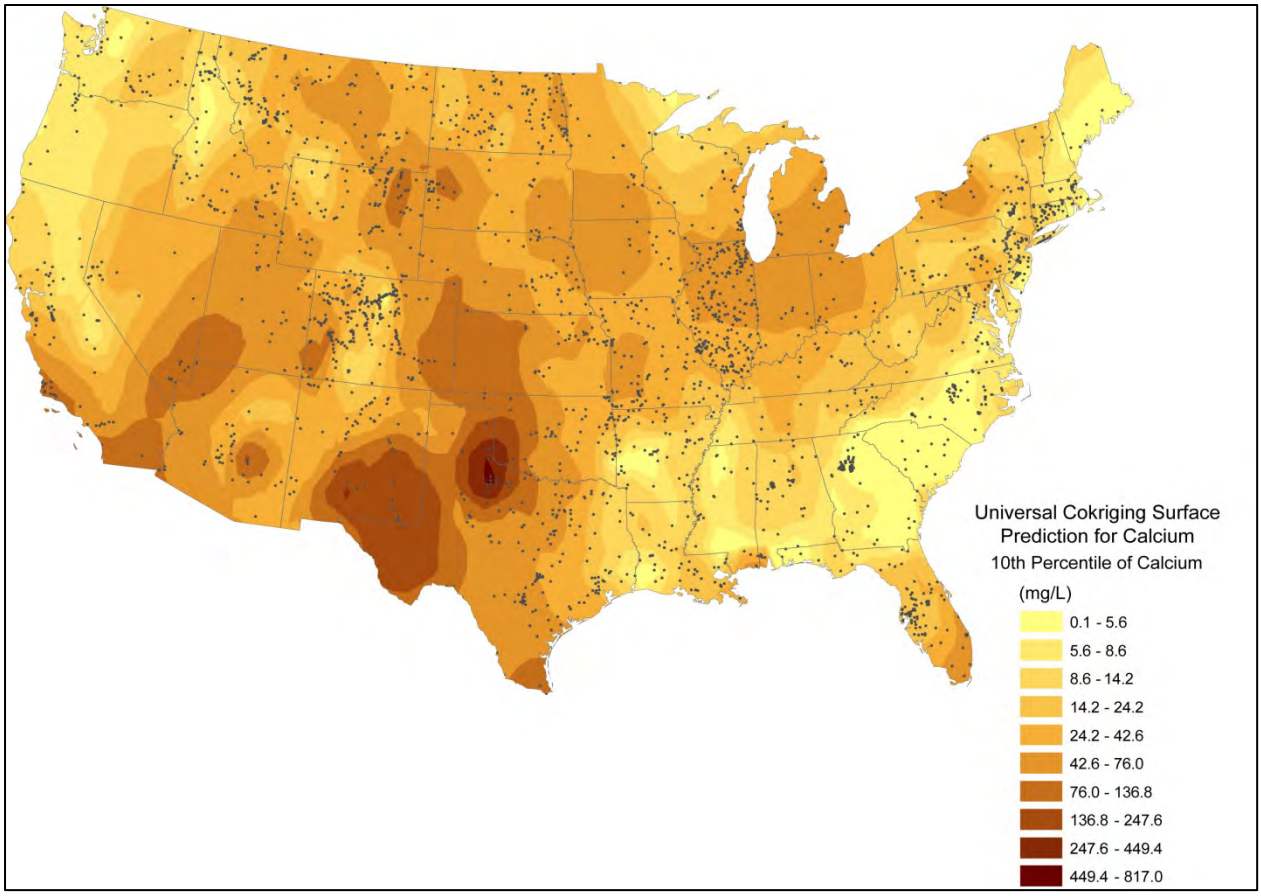

Figure 3. Co-kriged prediction surface for 10th percentile of calcium in the continental U.S.

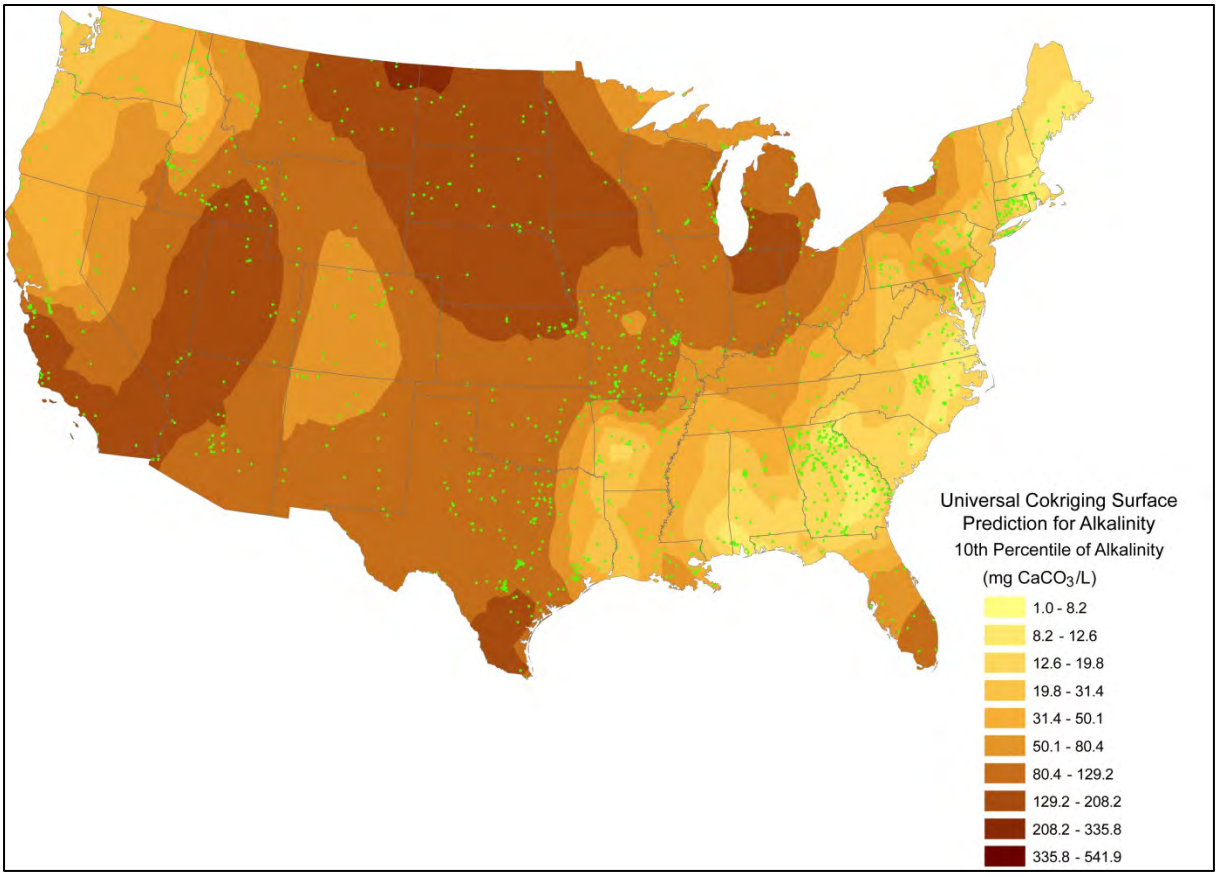

Figure 4. Co-kriged prediction surface for 10th percentile of alkalinity in the continental U.S.

2.2.3 Projection of Geostatistical Predictions onto Level III Ecoregions

Although maps of the geostatistical predictions are informative, a tabulation of the results is preferable for the purpose of providing guidance to BLM users. We chose to spatially average the geostatistical predictions of BLM water quality parameters according to the Level III ecoregions of the continental U.S. (Table 3), as these ecoregions provide a sound basis for spatial averaging of the water quality predictions. Ecoregions are designed to serve as a spatial framework for environmental resource management and denote areas within which ecosystems (and the type, quality, and quantity of environmental resources) are generally similar. They typically provide a logical and useful spatial (geographical) framework for organizing the results of environmental measurements (Omernik and Griffith, 2014). Ecoregions can be distinguished by landscape-level characteristics that cause ecosystem components to reflect different patterns in different regions (Omernik, 1987). "Level III Ecoregions of the Continental U.S." map layer shows ecoregion delineation based on common patterns of geology, physiography, vegetation, climate, soils, land use, wildlife, water quality, and hydrology. The map layer in Figure 5 was compiled by EPA (USEPA, 2013a) (http://www.epa.gov/wed/pages/ecoregions/level_iii_iv.htm).

Table 3. Level III ecoregions of the U.S. organized according to broader Level I ecoregions	
Level I Ecological Regions	
Level III Ecoregion	**Name of Ecoregion**
Marine West Coast Forest	
1	Coast Range
2	Puget Lowland
3	Willamette Valley
111	Ahklun and Kilbuck Mountains
113	Alaska Peninsula Mountains
115	Cook Inlet
119	Pacific Coastal Mountains
120	Coastal Western Hemlock-Sitka Spruce Forests
Northwestern Forested Mountains	
4	Cascades
5	Sierra Nevada
9	Eastern Cascades Slopes and Foothills
11	Blue Mountains
15	Northern Rockies
16	Idaho Batholith
17	Middle Rockies
19	Wasatch and Uinta Mountains
21	Southern Rockies
41	Canadian Rockies
77	North Cascades
78	Klamath Mountains
105	Interior Highlands
116	Alaska Range
117	Copper Plateau
118	Wrangell Mountains
Mediterranean California	
6	Southern and Central California Chaparral and Oak Woodlands
7	Central California Valley
8	Southern California Mountains
North American Deserts	
10	Columbia Plateau
12	Snake River Plain
13	Central Basin and Range
14	Mojave Basin and Range
18	Wyoming Basin
20	Colorado Plateaus
22	Arizona/New Mexico Plateau

Table 3. Level III ecoregions of the U.S. organized according to broader Level I ecoregions	
Level I Ecological Regions	
Level III Ecoregion	**Name of Ecoregion**
24	Chihuahuan Deserts
80	Northern Basin and Range
81	Sonoran Basin and Range
Temperate Sierras	
23	Arizona/New Mexico Mountains
Great Plains	
25	Western High Plains
26	Southwestern Tablelands
27	Central Great Plains
28	Flint Hills
29	Central Oklahoma/Texas Plains
30	Edwards Plateau
31	Southern Texas Plains
34	Western Gulf Coastal Plain
40	Central Irregular Plains
42	Northwestern Glaciated Plains
43	Northwestern Great Plains
44	Nebraska Sand Hills
45	Piedmont
46	Northern Glaciated Plains
47	Western Corn Belt Plains
48	Lake Agassiz Plain
Eastern Temperate Forest	
32	Texas Blackland Prairies
33	East Central Texas Plains
35	South Central Plains
36	Ouachita Mountains
37	Arkansas Valley
38	Boston Mountains
39	Ozark Highlands
51	North Central Hardwood Forests
52	Driftless Area
53	Southeastern Wisconsin Till Plains
54	Central Corn Belt Plains
55	Eastern Corn Belt Plains
56	Southern Michigan/Northern Indiana Drift Plains
57	Huron/Erie Lake Plains
59	Northeastern Coastal Zone
60	Northern Appalachian Plateau and Uplands

Table 3. Level III ecoregions of the U.S. organized according to broader Level I ecoregions	
Level I Ecological Regions	
Level III Ecoregion	**Name of Ecoregion**
61	Erie Drift Plain
63	Middle Atlantic Coastal Plain
64	Northern Piedmont
65	Southeastern Plains
66	Blue Ridge
67	Ridge and Valley
68	Southwestern Appalachians
69	Central Appalachians
70	Western Allegheny Plateau
71	Interior Plateau
72	Interior River Valleys and Hills
73	Mississippi Alluvial Plain
74	Mississippi Valley Loess Plains
75	Southern Coastal Plain
82	Laurentian Plains and Hills
83	Eastern Great Lakes and Hudson Lowlands
84	Atlantic Coastal Pine Barrens
Northern Forests	
49	Northern Minnesota Wetlands
50	Northern Lakes and Forests
58	Northeastern Highlands
62	North Central Appalachians
Tropical Wet Forests	
76	Southern Florida Coastal Plain
Southern Semi-Arid Highlands	
79	Madrean Archipelago
Taiga	
101	Arctic Coastal Plain
102	Arctic Foothills
103	Brooks Range
104	Interior Forested Lowlands and Uplands
106	Interior Bottomlands
107	Yukon Flats
108	Ogilvie Mountains
Tundra	
109	Subarctic Coastal Plains
110	Seward Peninsula
112	Bristol Bay-Nushagak Lowlands
114	Aleutian Islands

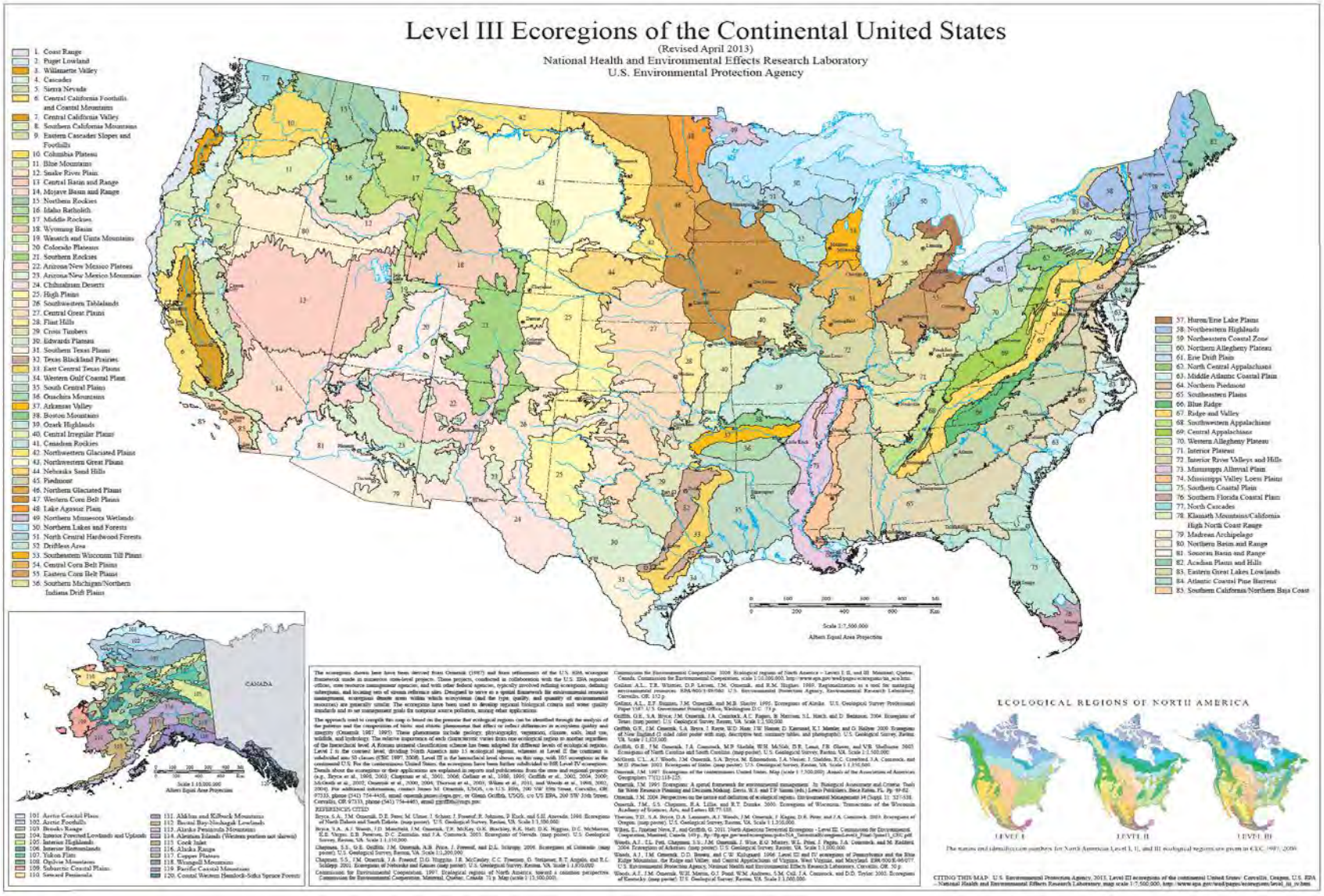

Figure 5. Map of Level III ecoregions in the U.S.

(Image taken from ftp://ftp.epa.gov/wed/ecoregions/us/Eco_Level_III_US.pdf)

Using the differences in land and water interactions, regional variations in attainable water quality, distinct biogeographical patterns (MacArthur, 1972), and similarities and differences in ecosystems to delineate ecoregions makes the application of ecoregions in environmental analyses a powerful tool with which to organize environmental information. The approach can take into account regional factors related to attainable water quality, and thus can be used to designate lakes for protection and to establish lake-restoration goals that are appropriate for each ecoregion (National Research Council (NRC), 1992). The NRC of the National Academy of Sciences has similarly endorsed the use of the concept in restoring and managing streams, rivers, and wetlands (NRC, 1992).

The theory of ecoregion delineation states that natural water quality characteristics of lakes and streams within a single ecoregion will be more similar than the characteristics between ecoregions (Perry and Vanderklein, 1996). Water quality characteristics exist in a landscape framework; neither normal nor impacted conditions of water resources can be separated from controlling influences of the surrounding landscape. The ecoregion concept has been applied and tested rather extensively in streams, rivers, and lakes. Testing and validation has been conducted in many diverse areas of the U.S., including several streams in Arkansas, Colorado, Kansas, Minnesota, Ohio, and Oregon, and in lakes of Michigan, Minnesota, Ohio, and Wisconsin (NRC, 1992).

Carleton (2006) used HUCs instead of ecoregions as the basis for averaging geostatistical results. HUCs are spatial delineations used for river basin management. Although a river basin may offer a logical framework for water supply management, for water quality management river basins are less applicable. The assumption that basins share similar properties is not always borne out, because river basins are often linked only by the water that flows through them. As Carleton (2006) noted, the use of HUCs for spatial averaging of surface water concentrations presents other conceptual difficulties. Only about 45% of HUCs are actual watersheds (Omernik, 2003); the rest receive drainage from additional upgradient areas. Concentrations measured in flowing waters reflect the soil, vegetation, and land use properties of the aggregate upstream drainage areas rather than of the sampling locations themselves (Smith et al., 1997).

2.2.3.1 Averaging Methods

To average the geostatistical predictions, a uniform grid was laid over the predicted surfaces and the predictions were sampled at the grid points falling within the polygons representing each ecoregion. The grid spacing was sized so that at least 30 points were sampled within each ecoregion. Unbiased log means were then calculated from the sampled concentration predictions in each ecoregion. The logarithmic transformation was applied because this normalized the concentration distributions in almost all of the ecoregions.

2.2.3.2 Tabulations of Ecoregional Estimated BLM Water Quality Parameters

The average predicted 10^{th} percentile concentrations for conductivity and the GI parameters for each of the Level III ecoregions in the continental U.S. are presented in Table 4. For each of these parameters, there is considerable variation between the ecoregional averages nationally. Chloride concentrations exhibit the greatest variation, with ecoregional 10^{th} percentile averages that range from 0.7 to 573 milligrams per liter (mg/L). Alkalinity was the least variable, but the ecoregional 10^{th} percentile average concentrations still ranged from 12 to 163 mg/L.

Table 4. Predicted 10th percentile concentrations for conductivity (µS/cm), BLM GI water quality parameters (mg/L) and hardness in each Level III ecoregion in the continental U.S.										
		Unbiased log mean of 10th percentile concentrations								
Level III Ecoregion	Ecoregion Name	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness[1]
1	Coast Range	102	8.4	3.2	4.1	0.64	33	3.2	4.8	34.12
2	Puget Lowland	80	7.1	1.9	2.8	0.64	22	2.3	5.6	25.54
3	Willamette Valley	91	8.2	2.9	4.4	0.90	30	4.7	3.8	32.39
4	Cascades	107	6.6	2.9	3.5	0.74	35	2.2	3.2	28.39
5	Sierra Nevada	195	8.3	4.7	8.8	1.3	58	5.8	11	40.02
6	Southern and Central California Chaparral and Oak Woodlands	600	42	24	48	2.5	124	56	136	203.4
7	Central California Valley	378	21	16	25	1.7	91	21	58	118.1
8	Southern California Mountains	772	63	25	63	3.8	150	54	171	260
9	Eastern Cascades Slopes and Foothills	212	8.2	3.8	6.0	1.0	44	3.2	5.0	36.08
10	Columbia Plateau	166	15	5.2	9.3	1.8	40	3.3	10	58.82
11	Blue Mountains	142	11	3.9	7.7	1.4	49	3.3	7.1	43.49
12	Snake River Plain	273	33	10	13	2.3	109	10	22	123.5
13	Central Basin and Range	426	43	16	45	3.6	120	45	83	173.1
14	Mojave Basin and Range	976	69	27	81	6.3	138	85	258	283.2
15	Northern Rockies	90	11	3.1	2.3	0.67	44	0.72	4.4	40.21
16	Idaho Batholith	91	13	3.8	3.6	0.88	62	1.9	5.9	48.08
17	Middle Rockies	300	30	10	14	1.9	105	7.6	55	116
18	Wyoming Basin	446	35	13	33	1.7	96	7.2	104	140.8
19	Wasatch and Uinta Mountains	426	61	27	61	3.3	155	55	155	263.2
20	Colorado Plateaus	639	65	26	57	2.6	117	28	197	269.1
21	Southern Rockies	259	26	8.0	12	1.4	55	3.8	56	97.8
22	Arizona/New Mexico Plateau	697	50	15	65	3.0	96	65	143	186.5
23	Arizona/New Mexico Mountains	879	66	18	85	3.4	102	100	189	238.8
24	Chihuahuan Deserts	2712	176	50	379	8.6	106	573	608	645
25	High Plains	1770	104	35	191	6.0	112	281	353	403.5
26	Southwestern	2147	114	34	316	4.9	94	512	374	424.4

Table 4. Predicted 10th percentile concentrations for conductivity (µS/cm), BLM GI water quality parameters (mg/L) and hardness in each Level III ecoregion in the continental U.S.										
		Unbiased log mean of 10th percentile concentrations								
Level III Ecoregion	Ecoregion Name	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness[1]
	Tablelands									
27	Central Great Plains	1228	84	24	176	6.9	121	245	204	308.4
28	Flint Hills	406	42	8.5	30	4.3	121	42	45	139.85
29	Central Oklahoma/Texas Plains	925	60	16	107	4.1	95	164	108	215.6
30	Edwards Plateau	596	48	14	38	2.7	98	62	52	177.4
31	Southern Texas Plains	798	56	14	58	3.8	129	73	91	197.4
32	Texas Blackland Prairies	364	39	5.8	21	3.2	92	26	33	121.28
33	East Central Texas Plains	367	36	6.3	23	3.8	98	29	29	115.83
34	Western Gulf Coastal Plain	565	43	10	62	3.9	87	86	78	148.5
35	South Central Plains	160	12	2.8	11	2.3	34	15	15	41.48
36	Ouachita Mountains	116	7.9	2.8	8.4	1.3	34	10	11	31.23
37	Arkansas Valley	192	16	4.7	15	1.8	51	20	16	59.27
38	Boston Mountains	152	18	3.3	4.3	1.3	53	6.7	8.2	58.53
39	Ozark Highlands	258	31	10	4.5	1.6	96	6.0	20	118.5
40	Central Irregular Plains	310	39	8.5	11	3.0	100	13	50	132.35
41	Canadian Rockies	164	22	8.7	15	0.57	80	1.7	38	90.67
42	Northwestern Glaciated Plains	545	37	20	61	5.9	163	8.1	147	174.5
43	Northwestern Great Plains	828	49	24	84	5.3	151	10	247	220.9
44	Nebraska Sand Hills	486	47	13	35	6.9	151	10	96	170.8
45	Piedmont	75	5.8	1.9	4.0	1.5	19	3.9	4.1	22.29
46	Northern Glaciated Plains	524	40	20	38	9.1	163	13	106	182
47	Western Corn Belt Plains	464	48	16	16	3.4	136	16	45	185.6
48	Lake Agassiz Plain	441	42	18	16	5.1	140	8.6	62	178.8
49	Northern Minnesota	229	24	10	3.2	1.4	95	2.6	8.4	101

Table 4. Predicted 10th percentile concentrations for conductivity (µS/cm), BLM GI water quality parameters (mg/L) and hardness in each Level III ecoregion in the continental U.S.

Level III Ecoregion	Ecoregion Name	Unbiased log mean of 10th percentile concentrations Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness[1]
	Wetlands									
50	Northern Lakes and Forests	166	19	6.5	2.5	0.78	83	3.4	6.1	74.15
51	North Central Hardwood Forests	295	31	12	5.6	1.7	115	8.9	15	126.7
52	Driftless Area	348	34	15	5.1	1.6	107	10	17	146.5
53	Southeastern Wisconsin Till Plains	510	39	20	16	2.1	112	31	25	179.5
54	Central Corn Belt Plains	546	53	24	14	1.7	124	30	46	230.9
55	Eastern Corn Belt Plains	463	49	15	11	2.0	116	23	32	184
56	Southern Michigan/Northern Indiana Drift Plains	463	52	15	14	1.9	134	28	29	191.5
57	Huron/Erie Lake Plains	467	52	15	13	2.2	125	27	32	191.5
58	Northeastern Highlands	97	11	1.9	5.7	0.69	24	10	7.4	35.29
59	Northeastern Coastal Zone	176	8.3	2.0	14	1.3	15	22	8.4	28.95
60	Northern Appalachian Plateau and Uplands	271	33	7.3	37	1.3	53	64	22	112.43
61	Erie Drift Plain	364	31	8.1	19	2.3	64	29	38	110.71
62	North Central Appalachians	184	13	3.9	7.1	1.0	41	11	15	48.49
63	Middle Atlantic Coastal Plain	793	6.7	2.3	6.8	1.8	15	83	22	26.18
64	Northern Piedmont	208	21	5.8	10	1.9	39	17	15	76.28
65	Southeastern Plains	121	7.4	2.2	5.5	1.5	19	15	7.2	27.52
66	Blue Ridge	121	11	3.2	3.0	1.3	23	3.4	6.0	40.62
67	Ridge and Valley	163	17	4.5	4.6	1.4	33	6.3	15	60.95
68	Southwestern Appalachians	151	13	3.2	2.5	1.3	42	3.2	11	45.62
69	Central Appalachians	193	16	5.6	4.6	1.3	34	4.5	33	62.96

Table 4. Predicted 10th percentile concentrations for conductivity (µS/cm), BLM GI water quality parameters (mg/L) and hardness in each Level III ecoregion in the continental U.S.										
		Unbiased log mean of 10th percentile concentrations								
Level III Ecoregion	Ecoregion Name	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness[1]
70	Western Allegheny Plateau	276	23	7.3	10	1.7	46	14	47	87.43
71	Interior Plateau	237	27	5.9	4.4	1.5	65	7.0	20	91.69
72	Interior River Valleys and Hills	326	34	12	11	2.4	87	17	46	134.2
73	Mississippi Alluvial Plain	255	14	5.5	17	2.6	44	22	13	57.55
74	Mississippi Valley Loess Plains	102	10	2.9	3.9	1.6	38	4.5	8.4	36.89
75	Southern Coastal Plain	726	18	5.0	20	2.0	41	390	38	65.5
76	Southern Florida Coastal Plain	682	49	6.6	25	2.4	103	43	15	149.56
77	North Cascades	93	6.5	2.3	2.3	0.64	27	1.2	4.9	25.68
78	Klamath Mountains	156	8.7	4.6	4.0	0.66	44	2.1	3.5	40.61
79	Madrean Archipelago	625	42	11	45	2.8	92	39	78	150.1
80	Northern Basin and Range	298	26	8.2	20	2.7	89	15	24	98.62
81	Sonoran Basin and Range	991	64	24	115	4.4	121	131	192	258.4
82	Laurentian Plains and Hills	104	4.8	0.78	2.5	0.48	12	2.7	4.4	15.198
83	Eastern Great Lakes and Hudson Lowlands	294	34	6.8	21	1.3	61	40	26	112.88
84	Atlantic Coastal Pine Barrens	261	7.4	2.7	10	1.7	23	16	11	29.57

[1] Water Hardness calculated as equivalents $CaCO3 = 2.5 (Ca^{2+}) + 4.1 (Mg^{2+})$

2.2.3.3 Confirmation of Results

To confirm the results of the geostatistical predictions, a number of comparisons were made between the ecoregional average predictions and averages based directly on the data. For each GI parameter, we compared the ecoregional average predictions against the corresponding averages calculated from the data for each ecoregion.

Scatter plot matrices provide a visual presentation of the correlations between different parameters. Scatter plot matrices were developed for the ecoregional averages of conductivity and GI parameters. Figure 6 is the scatter plot matrix for ecoregional averages based on the data, and Figure 7 is the scatter plot matrix for ecoregional averages based on the geostatistical predictions. Comparison of

these figures shows that the predicted averages capture the same trends in terms of distributions and parameter correlations as those that are found for the ecoregional data. The similarities in the distributions and correlation structures between the ecoregional averages in Figures 6 and 7 demonstrate that the geostatistical ecoregion predictions are reasonable.

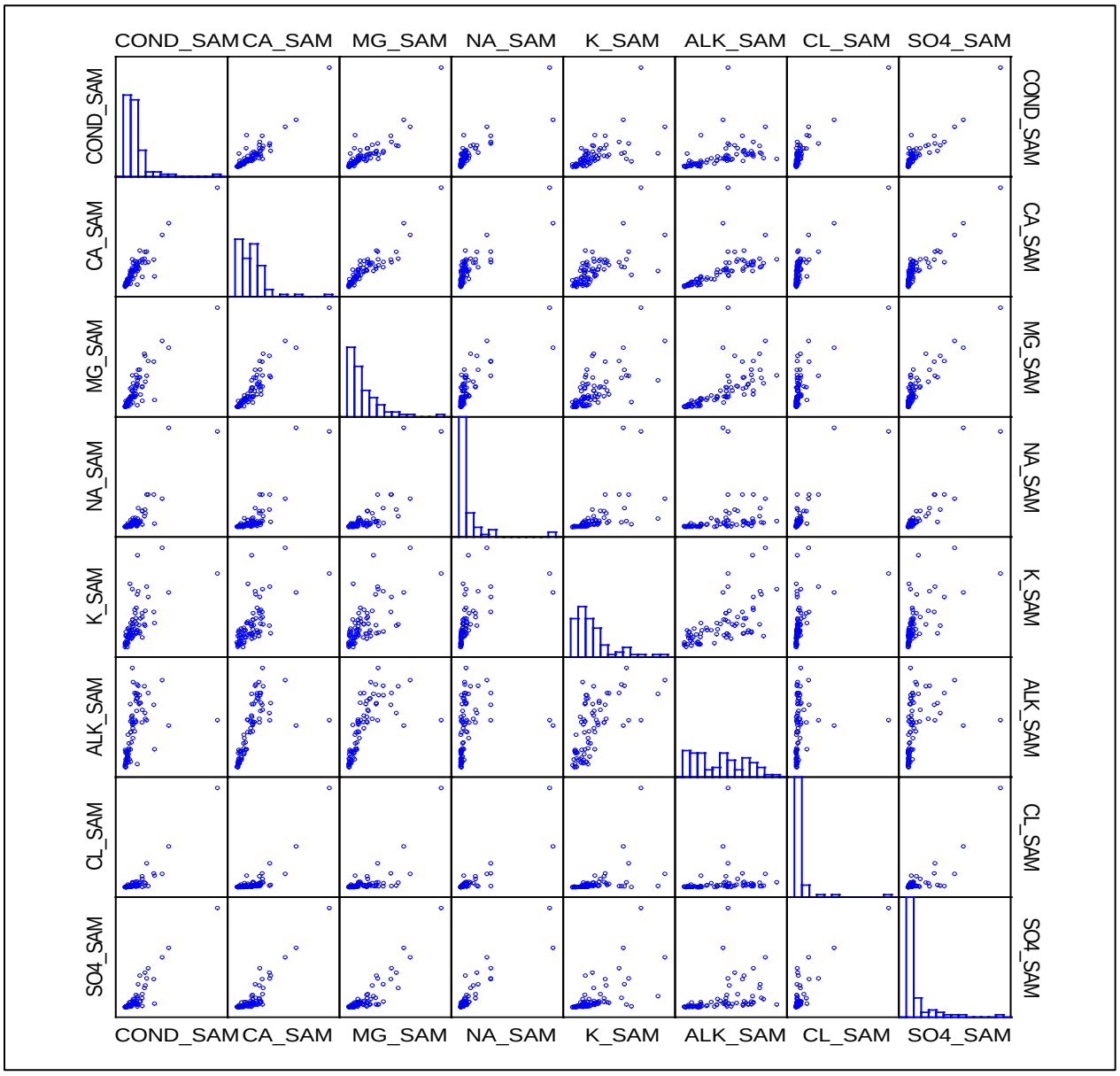

Figure 6. Scatter plot matrix of ecoregional average 10th percentiles of data for conductivity (COND_SAM) and GI parameters (calcium=CA_SAM, magnesium=MG_SAM, sodium=NA_SAM, potassium=K_SAM, alkalinity=ALK_SAM, chloride=CL_SAM, sulfate=SO4_SAM)

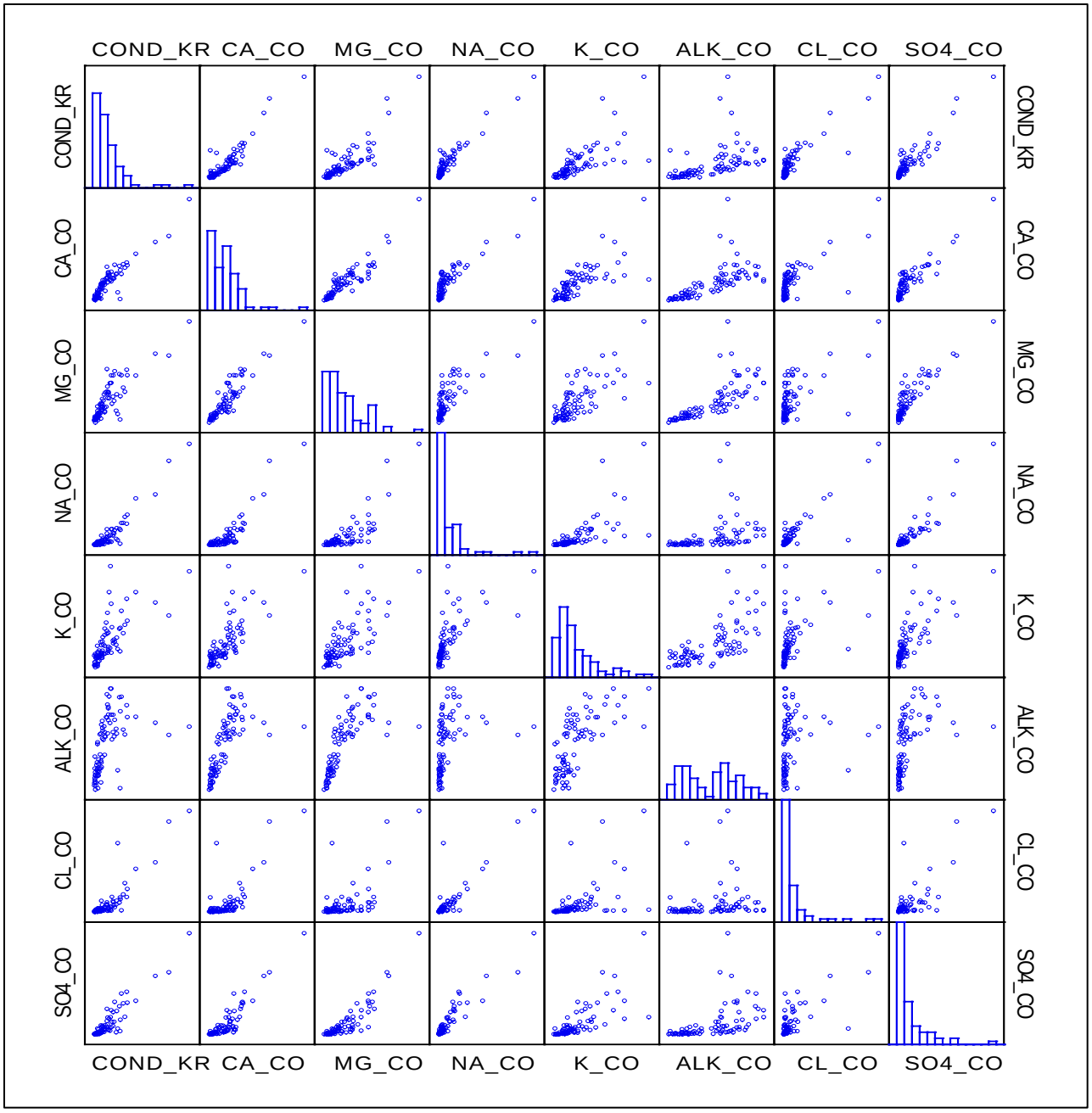

Figure 7. Scatter plot matrix of ecoregional average 10th percentiles of geostatistical predictions of conductivity (COND_KR) and BLM GI parameters (calcium=CA_CO, magnesium=MG_SCO, sodium=NA_CO, potassium=K_CO, alkalinity=ALK_CO, chloride=CL_CO, sulfate=SO4_CO)

In addition to scatter plots, correlation coefficient matrices between the parameters in each of the two data sets were generated. The Spearman (rank order) correlation coefficients for data-based ecoregional averages are presented in Table 5; correlation coefficients for ecoregional average geostatistical predictions are presented in Table 6. Although not identical, the correlation coefficients are similar between the two datasets, again demonstrating that the geostatistical predictions are reasonable.

Table 5. Spearman rank correlation matrix for unbiased log means of 10th percentile concentrations measured in Level III ecoregions								
	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate
Conductivity	1							
Calcium	0.895	1						
Magnesium	0.877	0.927	1					
Sodium	0.865	0.819	0.813	1				
Potassium	0.823	0.758	0.73	0.859	1			
Alkalinity	0.769	0.881	0.898	0.698	0.679	1		
Chloride	0.815	0.774	0.702	0.855	0.788	0.595	1	
Sulfate	0.894	0.864	0.85	0.883	0.786	0.725	0.744	1

Table 6. Spearman rank correlation matrix for unbiased log means of 10th percentile predicted (kriged/cokriged) concentrations in Level III ecoregions								
	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate
Conductivity	1							
Calcium	0.872	1						
Magnesium	0.843	0.924	1					
Sodium	0.87	0.82	0.778	1				
Potassium	0.842	0.803	0.753	0.836	1			
Alkalinity	0.747	0.868	0.901	0.672	0.745	1		
Chloride	0.829	0.715	0.592	0.826	0.725	0.461	1	
Sulfate	0.889	0.873	0.875	0.893	0.831	0.751	0.715	1

As a final test of the accuracy of the geostatistical predictions, we regressed the ecoregional averages based on the geostatistical predictions against the ecoregional averages based on the data. Scatter plots and fitted regression lines are shown for each of the parameters: conductivity (Figure 8), alkalinity (Figure 9), calcium (Figure 10), magnesium (Figure 11), sodium (Figure 12), potassium (Figure 13), sulfate (Figure 14), and chloride (Figure 15). Statistics for the linear regressions are provided in Table 7. For each of the parameters, the predicted and data-based ecoregional averages are significantly correlated. In each case, the linear regression coefficient is nearly 1.0, with a highly significant P value. As with the previous comparisons, the linear regression results demonstrate that the accuracy of the geostatistical predictions is high.

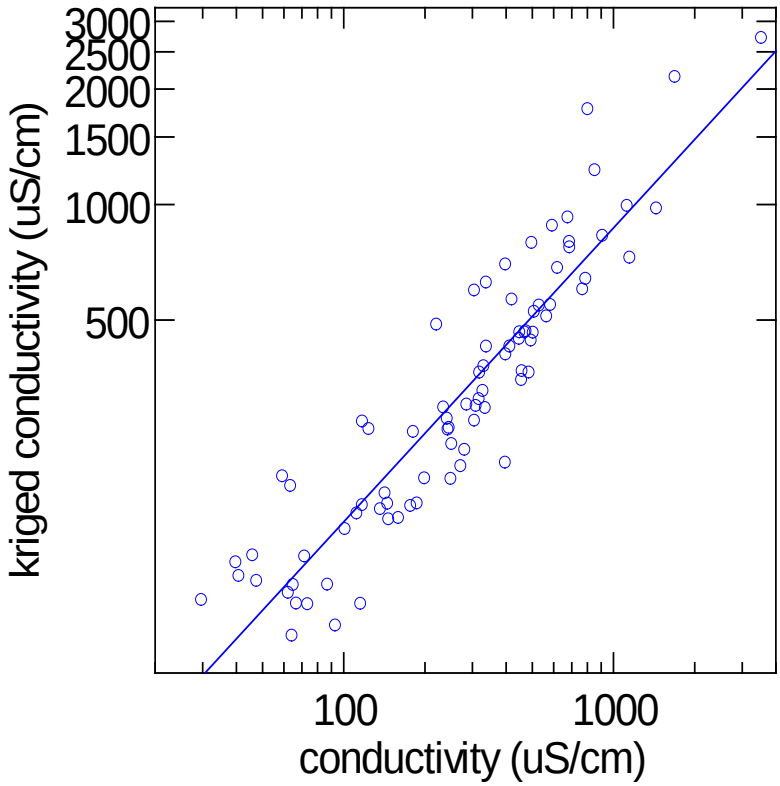

Figure 8. Ecoregional averages of kriged 10th percentiles of conductivity versus data

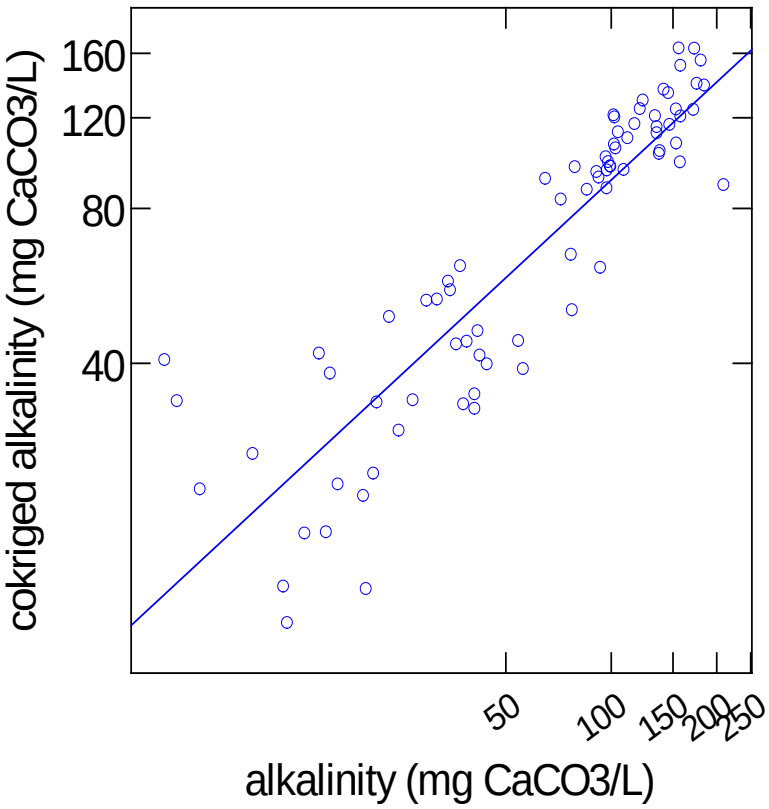

Figure 9. Ecoregional averages of cokriged 10th percentiles of alkalinity versus data

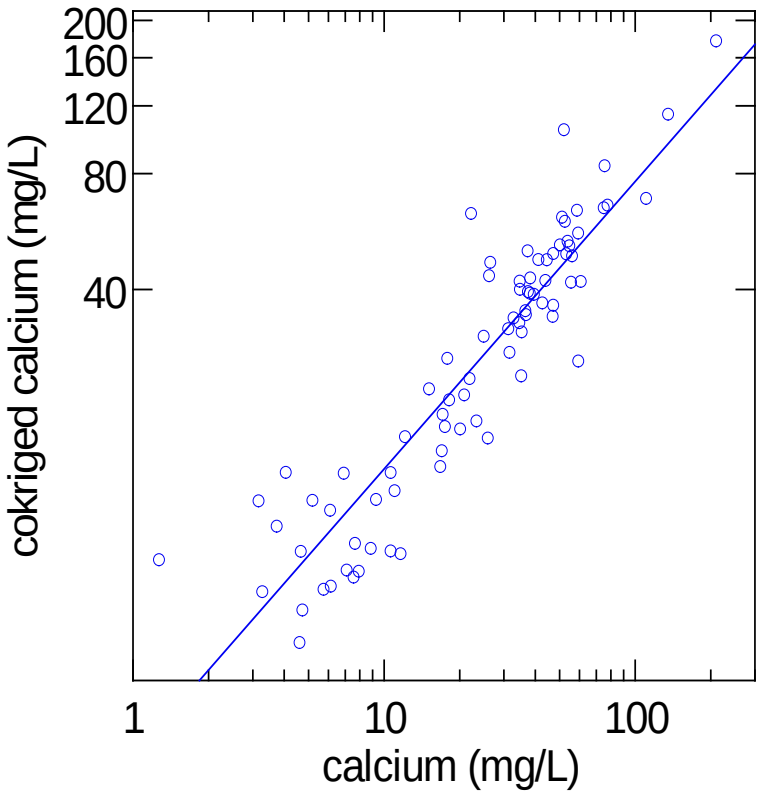

Figure 10. Ecoregional averages of cokriged 10th percentiles of calcium versus data

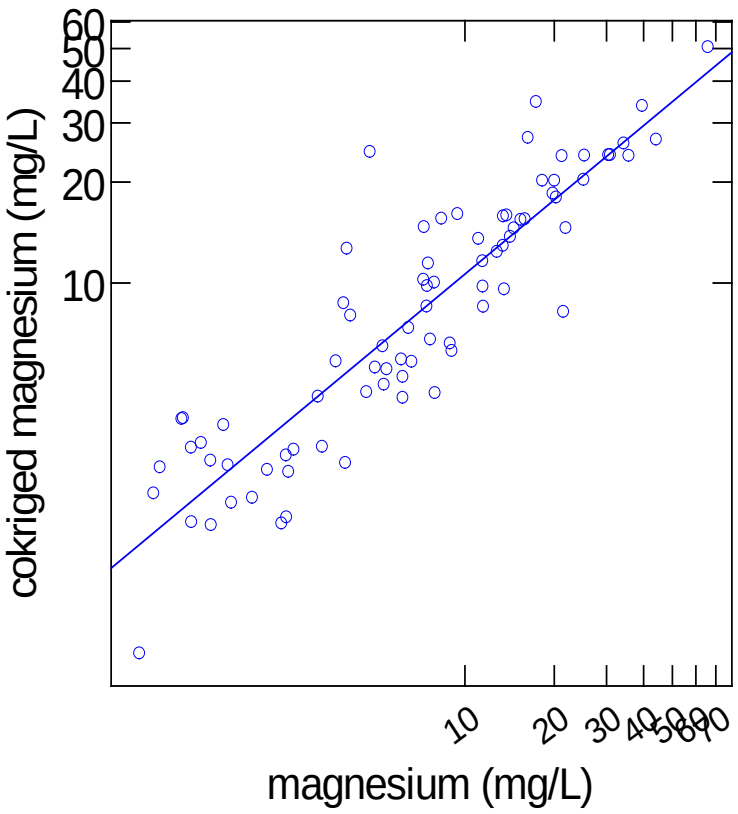

Figure 11. Ecoregional averages of cokriged 10th percentiles of magnesium versus data

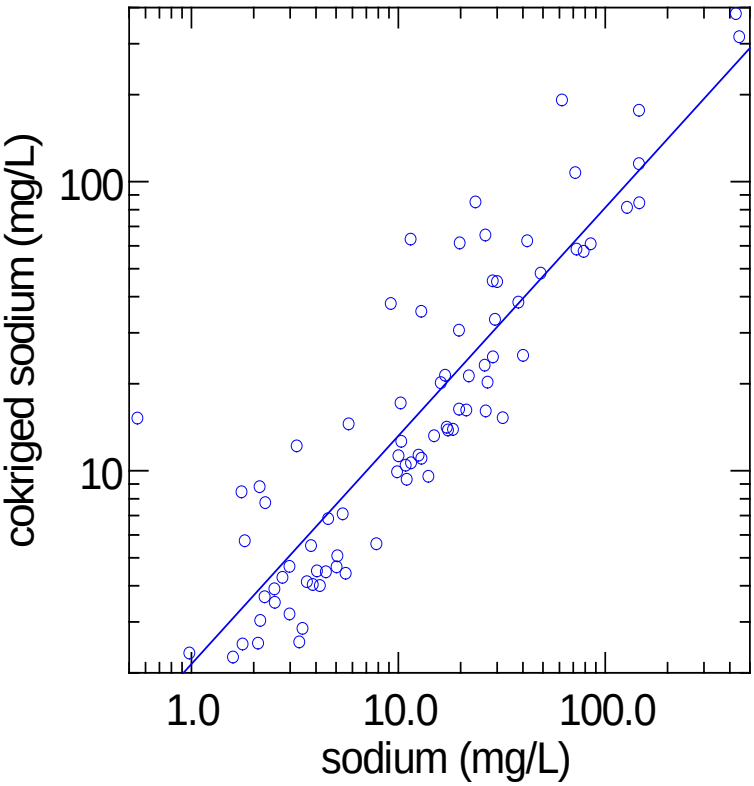

Figure 12. Ecoregional averages of cokriged 10th percentiles of sodium versus data

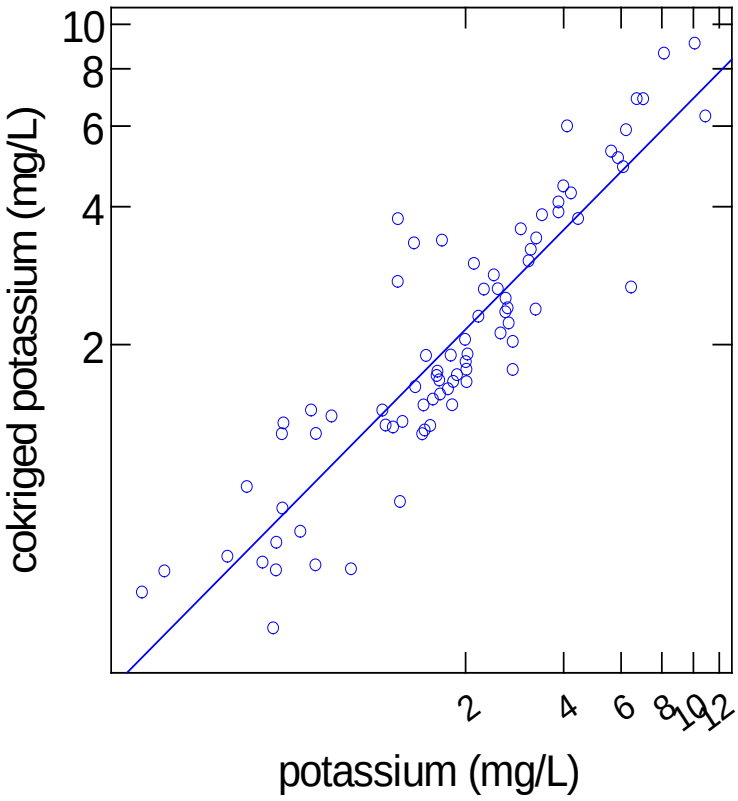

Figure 13. Ecoregional averages of cokriged 10th percentiles of potassium versus data

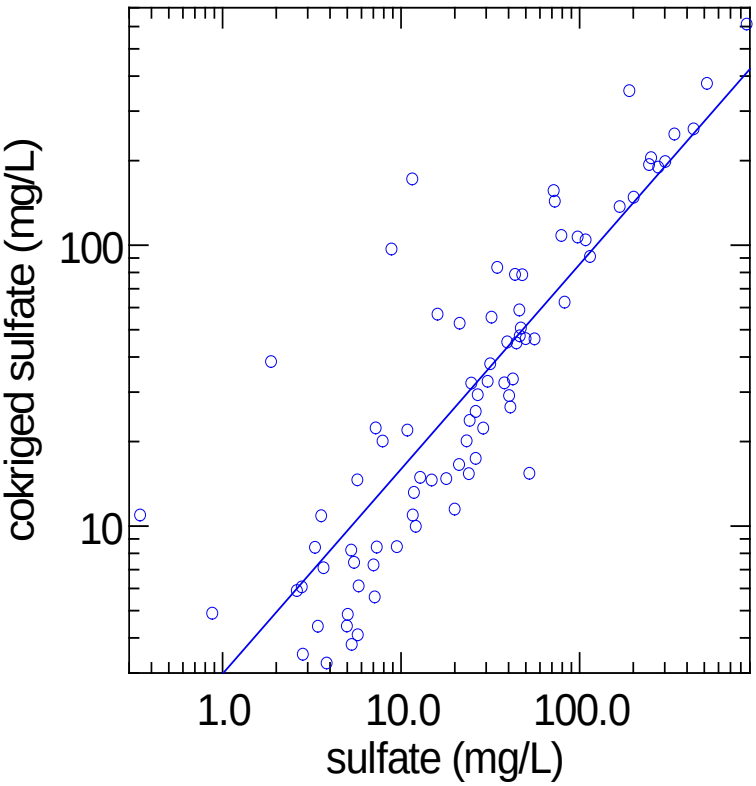

Figure 14. Ecoregional averages of cokriged 10th percentiles of sulfate versus data

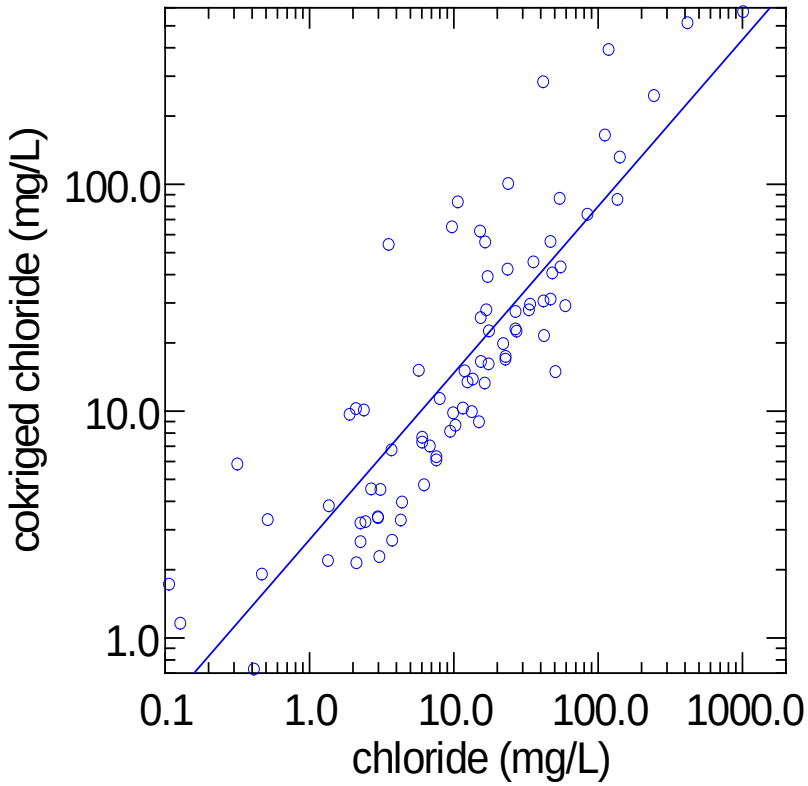

Figure 15. Ecoregional averages of cokriged 10th percentiles of chloride versus data

Table 7. Correlation coefficients and linear regression (LR) statistics between ecoregional average 10th percentiles of data and geostatistical predictions

Parameter	Correlation coefficient, r	LR coefficient	P (2 Tail)	LR constant	P (2 Tail)
Conductivity	0.908	0.98	<0.001	-20.992	0.495
Alkalinity	0.915	1.209	<0.001	-12.161	0.028
Calcium	0.922	1.061	<0.001	-1.994	0.365
Magnesium	0.885	1.097	<0.001	-1.169	0.22
Sodium	0.93	1.101	<0.001	-4.78	0.169
Potassium	0.906	1.073	<0.001	-0.184	0.297
Sulfate	0.865	1.264	<0.001	-13.599	0.041
Chloride	0.839	1.039	<0.001	-7.271	0.386

2.2.3.4 Conclusions for Selection of Water Quality Parameters

In this section we used geostatistics to estimate an intermediate step in generating missing GI parameter values based on geography. We supplemented the geostatistical approach by adding conductivity as an additional explanatory variable to generate a more robust spatial estimate of the GI water quality inputs for the BLM because conductivity is one of the most widely monitored water quality indicators in the U.S. and correlates well with GIs. In section 3, these estimates are further refined by stream order. We present here the average predicted 10^{th} percentile concentrations for the BLM GI water quality parameters, as presented in Table 4 by ecoregions. Because they are based on the 10^{th} percentiles of the daily average data from each USGS monitoring station, they are expected to yield copper criteria that are reasonably protective of aquatic life when applied as missing data for parameters in the BLM model. These data could also be used to fill in missing water chemistry parameters in the application of other metal BLM models. The most appropriate parameter selection however would include consideration of stream order in GIs estimates. Section 3 presents further refinements of estimates of the GI parameters by stream order and EPA's recommendations for default GI parameters for the BLM when data are lacking.

As with any estimate or prediction, it is appropriate to seek alternative estimates for the purpose of comparison or confirmation. If conductivity data are available for the site, either site-specific measurement data or data of opportunity from a database such as the NWIS, the regressions in EPA (2008; Appendix C) can be used to make independent estimates of the missing BLM water quality parameters. If the regression projections differ from the geostatistical average predictions, the lower (more conservative) estimate is recommended for application to ensure protection of aquatic life. As always, users of the BLM should be also encouraged to sample the water body of interest and to analyze for the constituent (parameter) concentrations as a basis for determining reliable BLM inputs.

2.2.3.5 Guidance Regarding Selection of Water Quality Parameters: pH and DOC

Although the geostatistical and regression-based approaches can be used to reliably estimate GI parameters used as BLM inputs, the same approaches do not produce accurate site-specific estimates for the two most important BLM inputs: pH and DOC. The BLM is less sensitive to the GI parameters than to pH and DOC predicting site-specific criteria for copper. Since our analysis indicates that there is little or no trend in relationships between conductivity and pH, and direct kriging produced similarly ambiguous predictions, site-specific data for pH must be used for BLM application at a site.

For DOC, analysis of NWIS data indicated a weak relationship with conductivity, so the regression approach is not appropriate for this parameter. In 2008, EPA recommended use of the ecoregional DOC concentration percentiles tabulated by EPA for the *Development of National Bioaccumulation Factors Technical Support Document* (USEPA, 2003) because they appeared to offer reasonable estimates of lower percentile DOC concentrations, and were based on substantially more DOC data than were available in the NWIS. In Section 4 of this report we further tested these ecoregional DOC concentrations for use in the BLM where site-specific data are not available.

3 USING STREAM ORDER TO REFINE PREDICTION OF GI PARAMETERS

The following section discusses how stream order was used to address anthropogenic impacts. The goal is to provide BLM users with tables of appropriately protective estimates of GI parameters, building on the ecoregional work described in Section 2.

Estimations of values for the GI parameters (alkalinity, calcium, magnesium, sodium, potassium, sulfate, and chloride) tend to vary regionally. As demonstrated in Section 2, the spatial variation of these factors is generally known or at least predictable, and therefore spatial or geographic analysis of data can be used to estimate GI input parameter values. However, these values also vary due to anthropogenic impact. In the case of conductivity and GI parameters, a positive correlation between ion concentrations and measures of human activity, such as population density, urban and agricultural land use, road density, point and nonpoint pollutant sources, among other activities, is expected and may confound the pattern of geographic variability both within and between ecoregions.

One way to account for surface water quality variability within ecoregions is to distinguish water bodies according to the Strahler stream order (SO). The SO is used to define stream size based on a hierarchy of tributaries (Strahler, 1952; 1957) and may range from 1st order (a stream with no tributaries) to 12th order (the Amazon, at its mouth). First through 3rd order streams are called headwater streams (source waters of a stream). Over 80% of Earth's waterways are headwater streams (Strahler, 1957). A stream that is 7th order or larger constitutes a river. For example, the Ohio River is 8th order and the Mississippi River is 10th order. According to the River Continuum Concept, changes in water quality are commonly observed between the upper, middle, and lower reaches of a stream (FISRWG, 1998; Ward, 1992; USEPA, 2015).

In this section we consider variability in GIs by determining the SO of each surface water sampling location in the USGS NWIS[2] database, and explore methods of incorporating SO variation in the parameter estimates. Tables are provided in this section showing tabulations of parameter estimates based upon both ecoregion and SO to maximize the accuracy of estimated input parameters.

3.1 Determining SO of NWIS Surface Water Sampling Locations

GIS was used to determine the SO of each NWIS surface water sampling location. Flowlines and catchments with SO were obtained from the NHD-Plus V2 geospatial hydrologic framework (McKay et al., 2012).[3] The point locations corresponding to the latitude-longitude coordinates of the NWIS sampling stations were snapped to the NHD-Plus flowlines using ArcGIS. A spatial join was then performed between these shapefiles and the NHD-Plus flowlines to link stream order to the sampling locations. Some of the NHD-Plus flowlines did not have SO data associated with the record. When a sampling location occurred on a flowline that didn't have a SO, the SO from the catchment was used. When the catchment also did not have a SO, the SO of the nearest stream was applied. SO was added as an attribute to the information for each station in the database.

[2] http://waterdata.usgs.gov/nwis
[3] http://www.horizon-systems.com/NHDPlus/NHDPlusV2_home.php

3.2 Estimating BLM Parameters for Ecoregions and SO

Estimated (10th percentile) BLM water quality parameters were presented in Section 2 for 84 Level III ecoregions of the continental U.S. In the work presented here, the parameter estimates were recalculated for individual SOs or ranges (groups) of SOs within each ecoregion.

3.3 Results

The distribution of NWIS sampling locations by SO is presented in Figure 16. The largest proportion of sampled locations (78%) was found to be in SO 1 through 4.

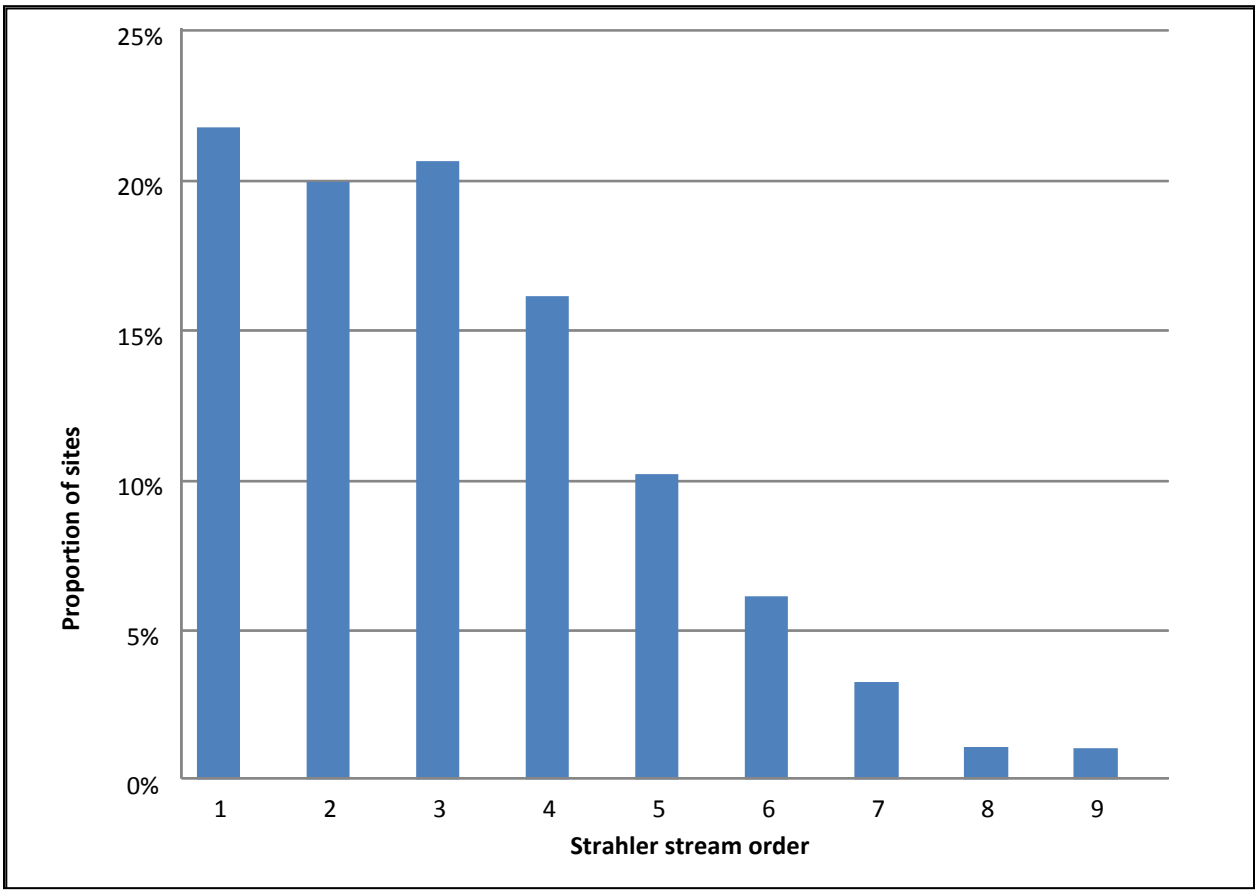

Figure 16. Distribution of NWIS surface water sampling locations by SO

3.3.1 Dependence of Ecoregional Parameter Estimates on SO

Box plots were constructed to examine how the GIs estimates varied with SO. Box plots of conductivity (Figure 17), alkalinity (Figure 18), calcium (Figure 19), magnesium (Figure 20), sodium (Figure 21), potassium (Figure 22), sulfate (Figure 23) and chloride (Figure 24) all show a general increase in the magnitude of the estimate with SO. This trend was most apparent and consistent when comparing medium stream orders (SO 4-6) to higher stream orders (SO ≥7). In addition, the upper quartile parameter estimates were generally higher in SOs 4 through 6 than in lower order streams (SO ≤3). Based upon these trends, we grouped the estimates for each parameter by SO: 1 through 3 (headwater

streams), 4 through 6 (mid-reaches) and 7 through 9 (rivers). There were no data for rivers with SO>9. Grouping simplified the presentation of results and improved the robustness of the parameter estimates, without losing significance of the SO trends. Parameter estimates for these three SO groups are included in the box plots in Figures 18 through 24, labeled as "13," "46," and "79." The classes depicted as 13, 46, and 79 reflect groupings according to SO (i.e., 1 through 3, 4 through 6, and 7 through 9).

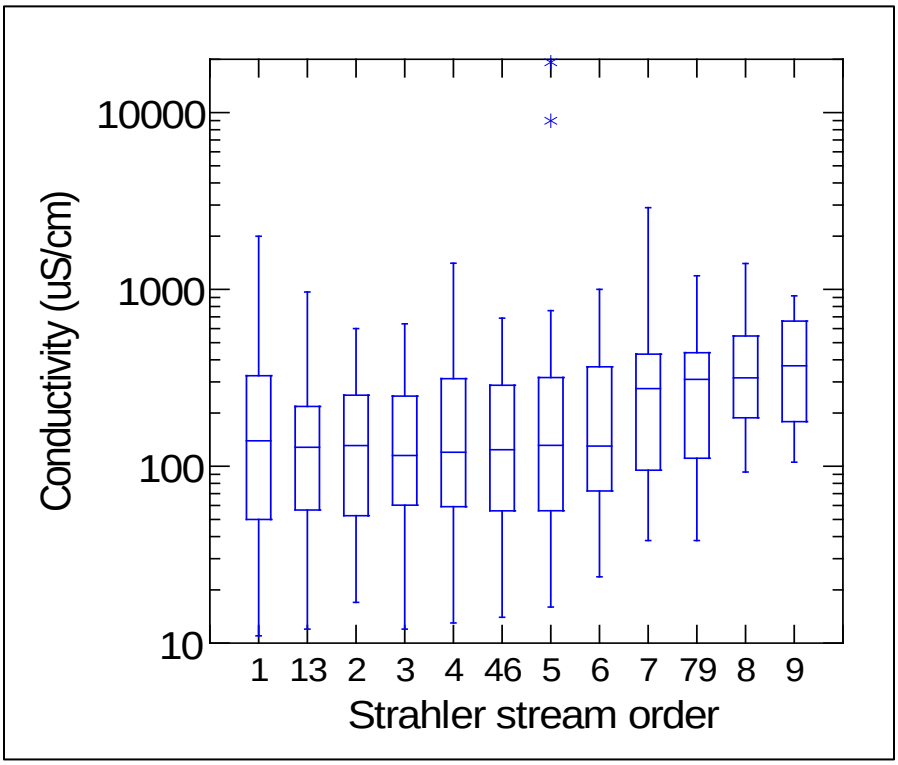

Figure 17. Box plot of estimated ecoregional conductivities as a function of SO

Note: Classifications depicted as 13, 46, and 79 reflect groupings according to stream order (i.e., 1 through 3, 4 through 6, and 7 through 9) as described in the text. For box plots, the bottom and top of each "box" displays the 25th and 75th percentile concentrations defined as the interquartile range (IQR) (i.e., the box contains 50% of the data values), respectively. The median is displayed as the horizontal line within the box. The "whiskers" show the relative distribution of data points outside of the IQR and represent 1.5 times the IQR. Data not included between the whiskers are plotted as outliers with a star/asterisk.

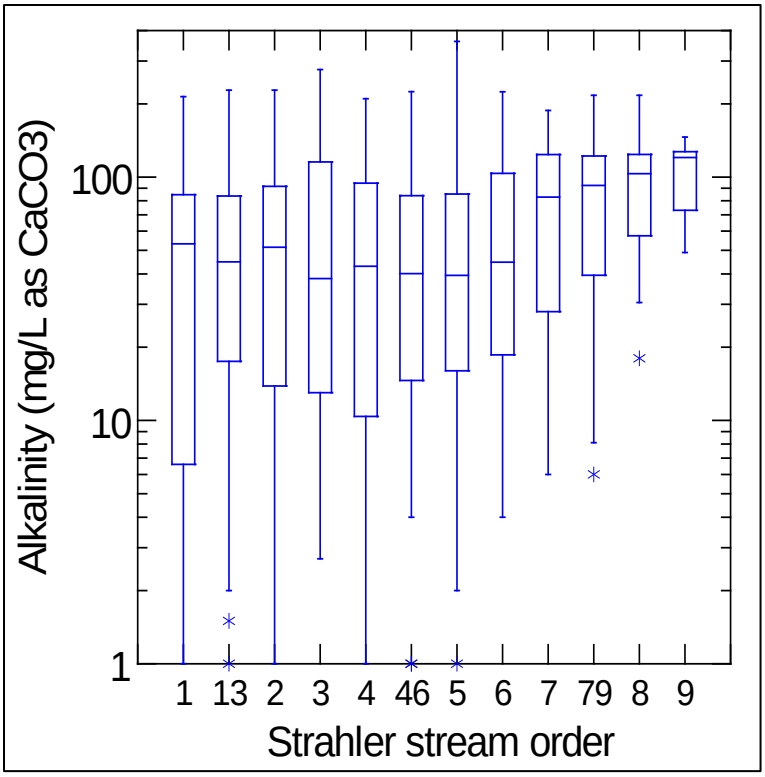

Figure 18. Box plot of estimated ecoregional alkalinity concentrations as a function of SO

(Refer to note in Figure 17.)

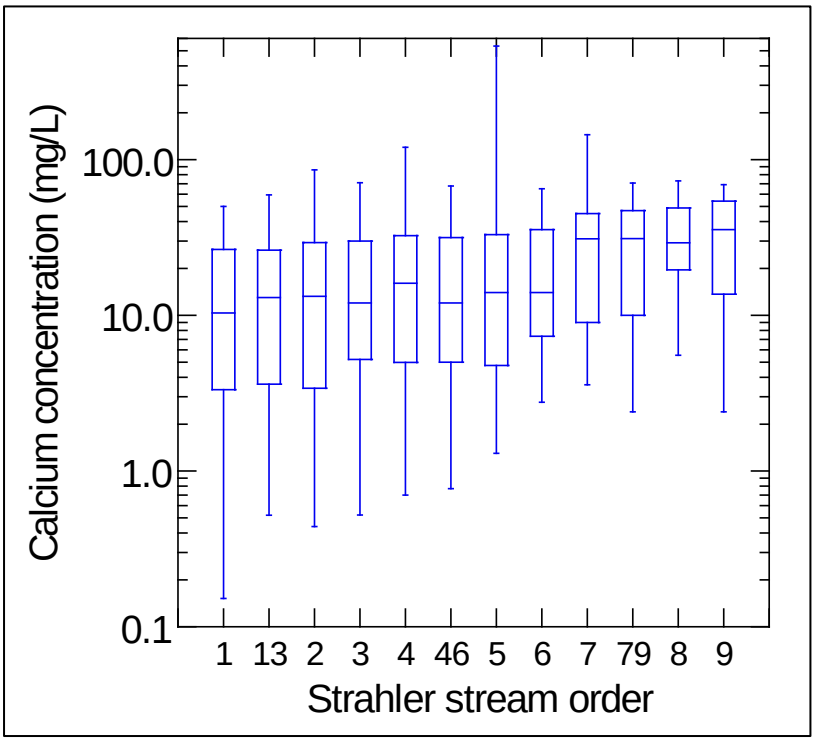

Figure 19. Box plot of estimated ecoregional calcium concentrations as a function of SO

(Refer to note in Figure 17.)

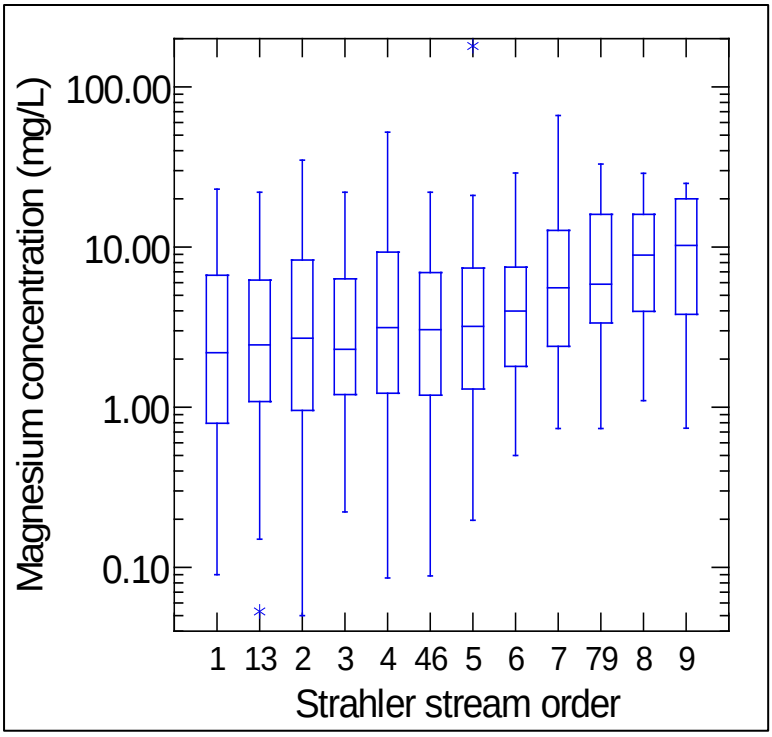

Figure 20. Box plot of estimated ecoregional magnesium concentrations as a function of SO

(Refer to note in Figure 17.)

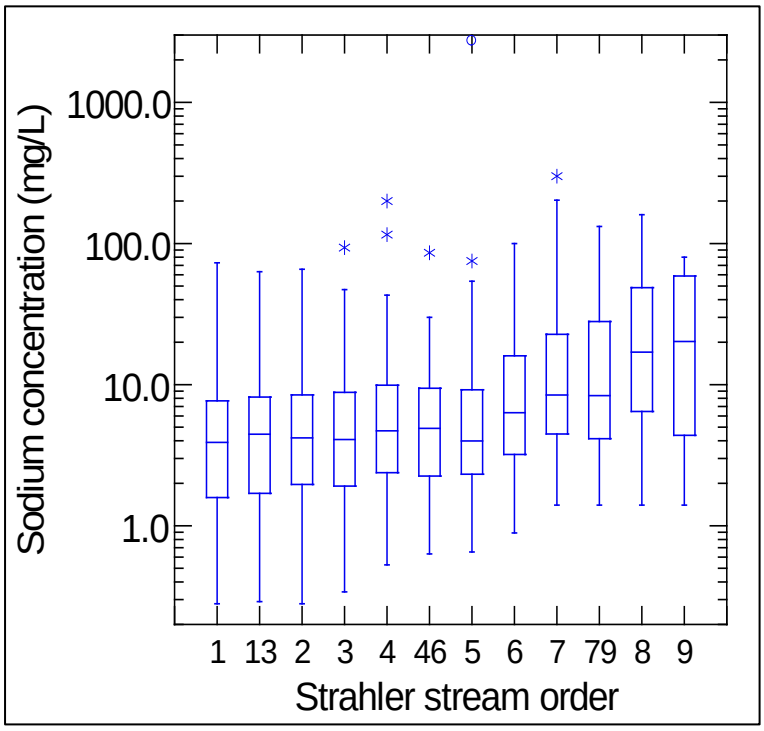

Figure 21. Box plot of estimated ecoregional sodium concentrations as a function of SO

(Refer to note in Figure 17.)

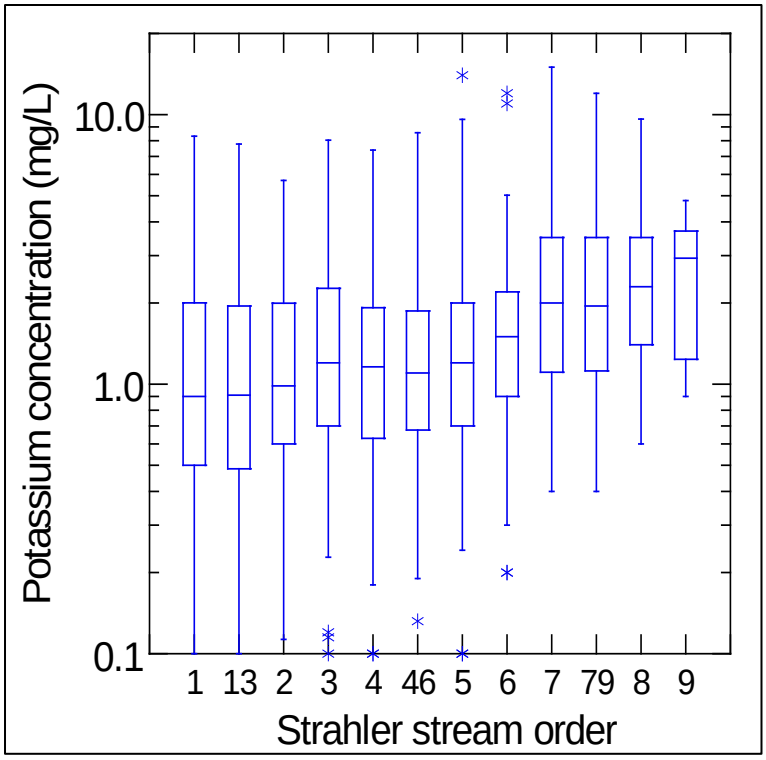

Figure 22. Box plot of estimated ecoregional potassium concentrations as a function of SO

(Refer to note in Figure 17.)

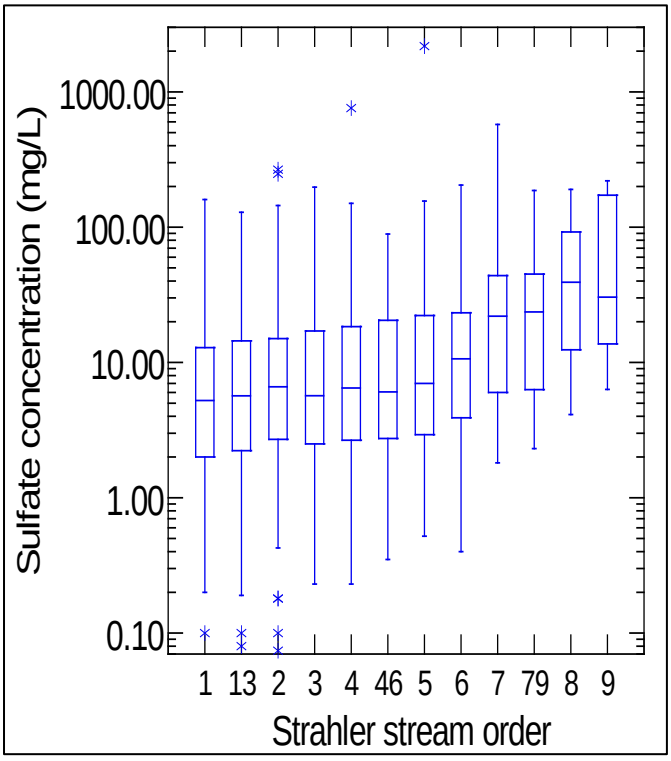

Figure 23. Box plot of estimated ecoregional sulfate concentrations as a function of SO

(Refer to note in Figure 17.)

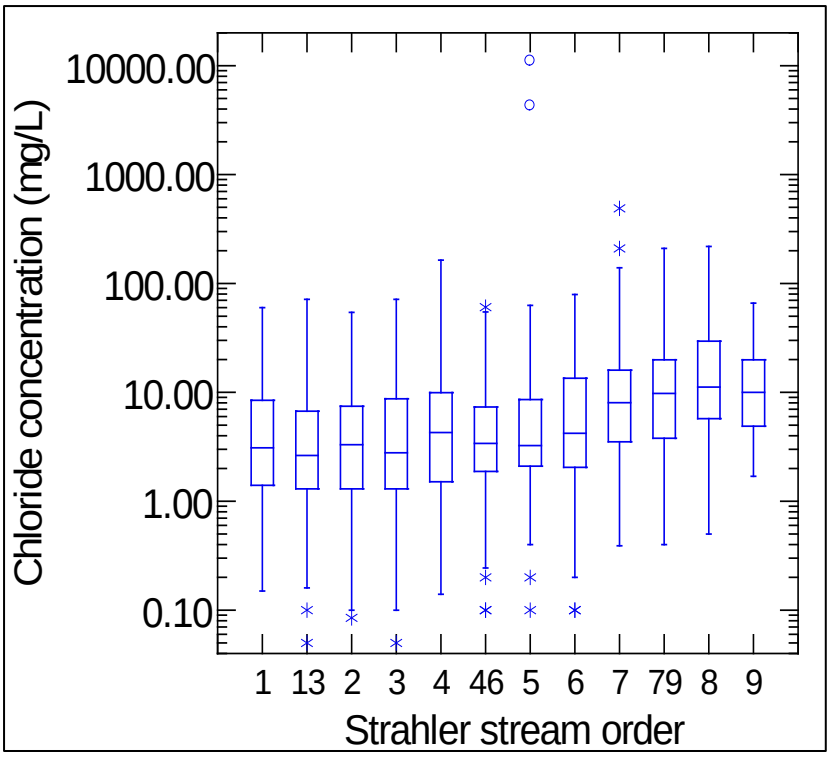

Figure 24. Box plot of estimated ecoregional chloride concentrations as a function of SO

(Refer to note in Figure 17.)

The range of values across SOs overlap greatly due to the inclusion of data across ecoregions. Tenth percentile estimates of conductivity increase with SO group in 58% of ecoregions when comparing low to medium SO groups, and 84% of ecoregions when comparing medium to high SO groups. The same trend was evident for the GIs. For example, 10th percentiles of calcium increased with SO group in 68% of ecoregions for low versus medium SO and 83% of ecoregions for medium versus high SO. In general, parameter estimates (10th percentiles of conductivity and ion concentrations) increased with SO, and the increase was most apparent and consistent for higher SOs (SO ≥7).

3.3.2 SO-Based Parameter Estimates

Tenth percentile parameter estimates for conductivity, GIs and hardness are grouped by SO and Level III ecoregions in Tables 8 through10. Tenth percentile parameter estimates for SOs 1 through 3 are presented in Table 8, for SOs 4 through 6 are presented in Table 9, and SOs 7 through 9 are presented in Table 10. The tables include the sample size for instances in which parameter estimates are highly uncertain due to limited data, i.e., cases where sample size is <10. Water quality data were limited in four ecoregions (11, 16, 49, and 78) for SO group 1 through 3, in Ecoregion 76 for SO group 4 through 6, and in 28 ecoregions for SO group 7 through 9. With the exception of the specific ecoregions and SO classes where data are limited, the parameter estimates in Tables 8 through 10 are recommended as improved default values for use in the BLM when data are not available for a location in a specific Level III ecoregion and SO group.

Table 8. Recommended 10th percentile conductivity, GIs, and hardness estimates for SO Group 1 through 3 (number of stations shown in parentheses if n<10)

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
1	58	6.0	0.8	1.3	0.1	44	0.6	1.1	18.28
2	74	8.8	2.8	3.9	0.5		2.8	3.3	33.48
3	68	9.9	3.8	5.6	1.5		2.3	1.5	40.33
4	16	1.0	0.2	1.8	0.2		0.5	0.2	3.32
5	28	0.6	0.1	0.3	0.1	38	0.1	0.1	1.91
6	279	3.6	8.2	8.4	0.9	73	8.9	7.2	42.62
7	164	19	6.0	14	1.8	120	8.6	6.6	72.1
8	157	29	4.3	10	1.5	70 (7)	2.6	0.4	90.13
9	55	4.4 (8)	0.9 (8)	2.3 (9)	0.4 (9)	35 (2)	0.2	0.2	14.69
10	137	24.0	9.4	10.2	1.4	127	4.6	11	98.54
11	88	8.6 (2)	3.2 (2)			169 (2)			34.62
12	133	13	2.0	6.1	0.8	35	1.4	3.7	40.7
13	109	9.4	1.6	2.7	0.6	45	0.5	3.7	30.06
14	967	15	2.8	6.0	3.0	90 (7)	2.7	6.3	48.98
15	24	3.1	0.8	0.9	0.4	9.0	0.2	1.3	11.03
16	21								
17	93	6.9	1.6	1.5	0.5	31	0.3	3.0	23.81
18	92	22	6.3	4.7	0.9		3.3	7.3	80.83
19	76	59	11	5.1	0.6	96	2.5	44	192.6
20	189	59	12	19	1.4	157	6.7	129	196.7
21	37	3.5	0.7	0.8	0.3	18	0.2	1.9	11.62
22	115	13	1.1	2.3	0.8	55	1.2	7.2	37.01
23	62	6.3	1.8	3.7	0.7	20	0.8	4.3	23.13
24	453	43	7.9	35	3.4	32	20	74	139.89
25	194	43	11	31	3.7	228	7.3	35	152.6
26	199	18	3.0	63	3.4	53	3.6	11	57.3
27	293	21	5.0	9.9	1.5	122	5.1	13	73
28	346	50	8.2	4.4	0.8	125	1.5	22	158.62
29	217	30	4.0	17	2.9	74	8.2	9.9	91.4
30	189	25	1.8	1.6	0.9	99	2.6	4.9	69.88
31	639	48	5.5	47	2.9		71	51	142.55
32	183	26	1.7	5.9	1.9	52	5.1	15	71.97
33	132	24	2.1	5.8	2.3	29	8.1	6.0	68.61
34	141	13	2.3	9.8	2.6	44	13	4.7	41.93
35	25	0.9	0.5	1.9	0.2	5.0	3.0	1.7	4.3
36	19	0.9	0.7	0.9	0.3	4.0	1.3	2.0	5.12
37	107	23	3.5	23	3.3	36	2.5	3.5	71.85
38	51	0.9 (7)	0.6 (7)	0.63 (7)	0.6 (7)	35	1.1	1.8	4.71
39	172	26	1.9	1.3	0.7	62	2.0	4.2	72.79

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
40	223	20	4.6	8.4	2.8	46	5.2	23	68.86
41	93	13	3.8	0.4	0.1	60 (9)	0.1	1.6	48.08
42	256	23	7.3	7.4	1.6	91	1.3	14	87.43
43	327	17	16	26	3.7	144	2.7	119	108.1
44	156	21	3.2	6.5	4.7	80 (2)	0.7	5.5	65.62
45	44	2.7	0.8	2.5	1.2	12	2.4	1.9	10.03
46	400	32	13	15	7.8	94	5.4	60	133.3
47	380	41	11	4.8	1.3	83	12	15	147.6
48	295	31	15	4.9	2.5		4.3	13	139
49	402 (4)					227 (1)			
50	69	5.1	1.0	1.2	0.3	32	0.4	1.2	16.85
51	137	28	11	2.0	0.9	53 (4)	3.3	4.5	115.1
52	432	42	12	2.9	0.7	75	4.8	12	154.2
53	502	27	8.0	11	2.0	42	25	22	100.3
54	574	51	22	8.4	1.2	202	28	44	217.7
55	420	40	11	7.8	1.5	130	19	16	145.1
56	219	28	7.4	3.6	0.9	79	6.3	16	100.34
57	446	35	10	6.8	2.3	84	19	33	128.5
58	25	1.2	0.4	0.3	0.2	1.5	0.4	4.2	4.64
59	69	3.6	1.1	6.2	0.9	3.0	10	5.8	13.51
60	61	4.7	1.2	1.5	0.4	17	1.5	6.6	16.67
61	131	13.0	2.5	8.0	1.1	33	7.8	8.3	42.75
62	32	1.8	0.7	0.8	0.3	2.0	0.8	4.4	7.37
63	108	1.1	0.8	2.8	0.8	5.0	5.5	3.2	6.03
64	134	12.0	4.6	7.0	1.3	19	10	11	48.86
65	25	0.7	0.5	1.5	0.3	3.0	2.6	0.7	3.8
66	12	0.5	0.3	0.7	0.3	3.0	0.7	1.1	2.48
67	81	4.7	2.0	2.0	0.7	3.0	1.8	7.1	19.95
68	32	1.9	1.2	1.3	0.7	3.0	1.4	3.6	9.67
69	36	2.8	0.8	0.4	0.4	4.0	1.0	8.1	10.28
70	125	5.6	4.0	2.7	1.4	12	2.4	16	30.4
71	179	10	1.8	1.1	0.7	54	2.2	2.8	32.38
72	180	17	5.3	5.2	1.7	69	3.5	21	64.23
73	102	6.9	2.7	3.4	2.0	31	2.0	2.9	28.32
74	52	3.2	1.4	2.5	1.5	9.3	2.6	3.9	13.74
75	75	6.6	1.5	4.5	0.5	9.0	9.0	1.5	22.65
76	430	41	1.8	7.8	0.3	116	30	0.1	109.88
77	14	1.6	0.4	0.4	0.2	7.0	0.2	0.7	5.64
78	19						2.1		
79	340	30	6.2	25	2.8	92 (2)	13	23	100.42
80	78	6.3	1.1	4.3	2.2	24	0.2	2.5	20.26
81	203	19	2.4	10	2.8	52	2.6	6.1	57.34

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
82	37	1.5	0.5	4.3	0.2		6.6	1.8	5.8
83	198	16	3.9	5.0	1.0	51	21	29	55.99
84	50	0.8	0.6	2.8	0.6	1.0	5.0	4.4	4.46

Table 9. Recommended 10th percentile conductivity, GIs, and hardness estimates for SO group 4 through 6 (number of stations shown in parentheses if n<10)

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
1	52	3.6	1.0	2.0	0.2	15	1.6	2.2	13.1
2	49	5.3	1.2	1.3	0.1	16	0.8	1.8	18.17
3	62	7.1	2.5	4.3	0.8	29	4.6	2.8	28
4	35	3.5	1.0	2.8	0.4	16	0.8	0.8	12.85
5	18	0.9	0.1	0.6	0.2	5.0	0.4	0.4	2.66
6	316	9.1	4.8	5.4	1.0	32	2.3	4.1	42.43
7	67	6.5	2.5	2.9	0.9	33	1.7	3.2	26.5
8	93	9.0	1.5	8.4	1.0	17	3.2	6.0	28.65
9	52	5.5	0.8	2.4	0.5	22	0.9	2.2	17.03
10	83	8.6	3.2	4.0	0.9	33	1.4	3.1	34.62
11	52	3.7	0.8	1.6	0.7	16	0.3	0.7	12.53
12	95	13	2.5	4.9	1.2	40	2.2	3.8	42.75
13	124	12	3.4	9.6	1.8	68	3.9	8.1	43.94
14	688	58	13	86	7.9	225	55	86	198.3
15	34	3.5	1.1	0.9	0.4	16	0.2	1.2	13.26
16	22	2.4	0.4	1.3	0.4	10	0.2	0.6	7.64
17	123	10	2.5	1.4	0.6	44	0.5	2.2	35.25
18	145	15	4.0	7.1	0.9	57	1.4	16	53.9
19	135	34	9.1	6.7	1.3		7.3	9.5	122.31
20	260	38	9.6	16	1.5	107	4.0	55	134.36
21	74	6.5	1.3	1.9	0.6	18	0.4	3.2	21.58
22	215	25	4.3	9.5	1.3	62	2.4	18	80.13
23	289	31	9.5	5.6	1.1	101	3.5	2.4	116.45
24	240	24	5.1	18	1.5	80	7.0	21	80.91
25	220	14	3.4	8.7	1.6	84	4.1	29	48.94
26	367	39	9.1	29	2.8	79	11	56	134.81
27	351	39	7.1	11	5.1	79	5.6	16	126.61
28	298	68	14	9.6	2.1	150	7.4	39	227.4
29	351	39	6.8	20	2.7	80	20	19	125.38
30	377	44	12	6.5	0.8	140	10	13	159.2
31	447	54	9.4	10	1.8	142	12	29	173.54
32	311	35	2.8	12	2.8	94	12	22	98.98
33	334	33	5.0	16	2.1	45	23	28	103
34	125	10	3.5	13	2.7	40	12.2	3.1	39.35

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
35	67	4.2	1.1	6.5	1.5	10	3.5	5.0	15.01
36	24	1.0	0.8	1.1	0.5	4.0	1.5	2.0	5.78
37	55	3.8	2.3	5.2	1.3	9.6	2.0	2.0	18.93
38	41	8.4	1.0	1.2	0.8	13	1.5	3.2	25.1
39	160	19	1.8	1.6	0.9	73	2.3	3.6	54.88
40	258	33	5.7	6.8	2.7	64	6.7	20	105.87
41	133	19	4.5	0.7	0.2		0.1	1.9	65.95
42	285	22	9.2	12	2.3	141	2.3	34	92.72
43	342	28	8.5	13	2.7	145	2.2	45	104.85
44	232	27	3.8	9.4	7.0	184 (1)	2.5	2.7	83.08
45	44	3.0	1.2	3.0	1.3	13	2.9	2.9	12.42
46	480	32	17	30	8.6	153	9.1	89	149.7
47	390	43	10	7.0	1.7	109	8.5	19	148.5
48	422	40	18	8.5	4.1	170	7.0	32	173.8
49	75	7.9	2.4	2.3	0.7	25	2.3	4.5	29.59
50	78	7.9	2.4	1.2	0.4	37	0.9	2.7	29.59
51	125	19	6.5	2.3	0.9	102	3.0	3.6	74.15
52	221	15	6.3	4.2	1.5	50	6.4	10	63.33
53	389	36	19	9.1	2.2	138	18	20	167.9
54	520	49	22	7.4	1.0	148	22	37	212.7
55	413	43	12	5.6	1.9	162	19	22	156.7
56	389	44	14	10	1.5	133	18	21	167.4
57	489	56	15	10	2.6	108	22	31	201.5
58	38	4.9	0.9	2.9	0.5	5.0	4.3	6.1	15.94
59	81	5.1	1.4	7.8	1.1	10	11	7.3	18.49
60	101	12	2.5	5.3	1.0	20	3.9	8.4	40.25
61	178	20	4.7	8.4	1.8	47	61	11	69.27
62	50	3.9	0.9	4.0	0.6	5.0	2.6	6.1	13.44
63	65	3.6	0.9	3.9	1.2	8.0	6.6	3.2	12.69
64	175	13	4.7	9.9	1.2	48	15	13	51.77
65	43	2.8	0.9	2.2	0.8	6.0	3.4	1.7	10.69
66	14	1.0	0.3	0.8	0.5	4.0	0.4	1.1	3.73
67	89	7.9	2.0	2.9	1.0	14	3.4	8.8	27.95
68	42	4.0	0.9	1.0	0.8	16	1.4	4.4	13.69
69	115	6.8	1.5	1.7	0.6	9.0	1.8	8.8	23.15
70	108	11	3.1	4.2	1.2	11	4.5	22	40.21
71	145	18	2.6	1.4	1.0	53	2.8	3.7	55.66
72	251	25	7.7	8.2	2.1	61	10	30	94.07
73	99	6.4	2.4	3.9	2.2	34	3.0	5.0	25.84
74	46	2.3	1.0	2.2	1.1	11	2.0	1.0	9.85
75	57	2.2	1.0	3.8	0.5	1.0	6.1	1.7	9.6
76									

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
77	44	4.7	1.5	1.1	0.2	34 (3)	0.1	0.9	17.9
78	92	7.9	3.2	4.0	0.6	36	2.1	2.4	32.87
79	371	33	7.1	25	2.0	89	6.5	18	111.61
80	204	15	5.7	4.1	0.8	54	2.0	9.3	60.87
81	146	30	5.7	11	2.0	54	4.5	25	98.37
82	29	3.5	0.7	2.1	0.4	9.2	2.3	5.4	11.62
83	97	11	1.9	3.2	0.7	54	22	25	35.29
84	41	0.8	0.5	2.4	0.7	1.0	4.4	4.5	4.05

Table 10. Recommended 10th percentile conductivity, GIs, and hardness estimates for SO group 7 through 9 (number of stations shown in parentheses if n<10)

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
1	111	12	3.4	4.3	0.8	56	2.3	6.3	43.94
2									
3	58	5.0	1.6	3.4	0.6	20	2.7	2.3	19.06
4	118	13	3.6	3.7	0.9	52	1.7	6.9	47.26
5									
6	101	9.0	4.5	4.9	0.9		1.7	3.1	40.95
7	122	10	5.0	6.2	1.0	56	3.5	4.8	45.5
8									
9									
10	71	5.7	1.5	2.0	0.7	16	0.8	4.2	20.4
11	72	8.5	1.5	3.3	0.7	32	0.8	5.0	27.4
12	310	37	10	13	2.5	122	11	30	133.5
13	430	38	10	32	5.6	175	15	27	136
14	810	64	23	69	3.2	121	55	181	254.3
15	51	5.2	1.5	1.4	0.5	20	0.4	3.1	19.15
16									
17	189	20	5.6	3.7	1.1	69	1.3	13	72.96
18	342	35	11	14	1.3	119	2.5	45	132.6
19	608	55	20	44	2.2	145	13.1	149	219.5
20	373	39	12	25	1.7	102	9.7	85	146.7
21									
22	279	28	4.9	15	1.9	80	4.5	37	90.09
23									
24	554	60	11	76	4.3	107	49	145	195.1
25	830	64	20	60	4.6	127	16	184	242
26	876	56	20	61	3.5	128	24	187	222
27	648	61	16	43	6.2	96	25	112	218.1
28	395	41	8.9	15	6.4	119	10	36	138.99
29	1194	71	19	132	4.7	89	210	130	255.4

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
30									
31	817	59	16	76	4.1	102	88	141	213.1
32	438	45	6.2	28	3.9	107	37	38	137.92
33	428	46	5.3	31	4.7	128	42	28	136.73
34	477	47	6.9	36	4.3	103	49	35	145.79
35	85	6.9	1.6	5.8	1.4	46	5.0	8.0	23.81
36	179	2.4	0.7 (3)	1.4 (3)	1.7 (3)		20	21	8.87
37	355	28	7.0	29	2.9	73	30	28	98.7
38									
39	215	28	7.7	2.8	1.1	96	3.4	6.0	101.57
40	310	34	5.2	6.4	3.0	96	7.3	24	106.32
41									
42	338	49	18	26	3.4	144	8.7	87	196.3
43	394	36	12	24	2.4	122	6.2	74	139.2
44									
45	53	4.1	1.7	6.0	1.5	13	5.0	7.3	17.22
46	642	52	25	49	12	176	22	149	232.5
47	570	48	12	15	3.7	159	11	44	169.2
48	425	44	19	14	5.3	188	9.9	61	187.9
49									
50									
51	353	44	16	7.2	2.1	217	10	13	175.6
52	115	12	4.4	2.9	1.1	40	4.4	5.0	48.04
53	544	53	33	7.9	1.8		19	22	267.8
54	388	41	18	9.7	2.1	131	16	25	176.3
55	502	48	18	20	3.0	182 (4)	32	33	193.8
56									
57	405	43	12	9.5	2.8	104	20	30	156.7
58									
59	65	3.9	0.7	8.5	0.8	6.0	13	6.0	12.62
60									
61									
62									
63	80	3.6	1.4	5.1	2.0	8.5	6.5	7.4	14.74
64	148	14	3.5	4.6	1.3	28	8.0	20	49.35
65	69	4.7	1.2	3.7	1.2	15	4.1	5.7	16.67
66									
67	96	15	3.4	4.7	1.2	28	5.7	12	51.44
68	138	17	3.5	4.1	1.1	57 (8)	5.7	12	56.85
69									
70	225	21	5.4	9.8	1.4	29	10	44	74.64
71	183	23	4.3	3.2	1.4	56	3.8	13	75.13

Ecoregion	Conductivity	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Chloride	Sulfate	Hardness
72	310	34	10	8.3	2.3	88	12	23	126
73	146	14	3.7	3.4	1.7	54	3.5	6.0	50.17
74									
75	70	4.8	1.2	3.2	1.0	15	4.6	3.9	16.92
76									
77									
78									
79									
80	71	8.9 (3)	2.4 (3)	7.7 (3)	2.1 (3)		2.1 (3)	5.1	32.09
81	898	64	23	80	3.8	123	69	160	254.3
82	38	4.0	0.8	1.9	0.4	8.1	2.4	4.5	13.28
83	174	18	3.2	6.1	0.8	41	10	12	58.12
84									

At the level of individual ecoregions, the trends in parameter estimates as a function of SO group often reflect the assessment presented in the previous section. In the majority of ecoregions, most of the parameter estimates increase with SO group, as illustrated in Figure 25 for Ecoregion 46, the Northern Glaciated Plains. However, other trends were observed as well. In Ecoregion 83 (Eastern Great Lakes Lowland), conductivity and cation concentrations were approximately equal in the low and high SO groups and lower in the medium SO group, as shown in Figure 26. Figure 27 illustrates the trends in Ecoregion 54, the Central Corn Belt Plains. In this ecoregion (and several others), most of the parameter estimates decreased with SO group. The explanation for different trends within ecoregions, which may reflect different causes, is beyond the scope of this effort.

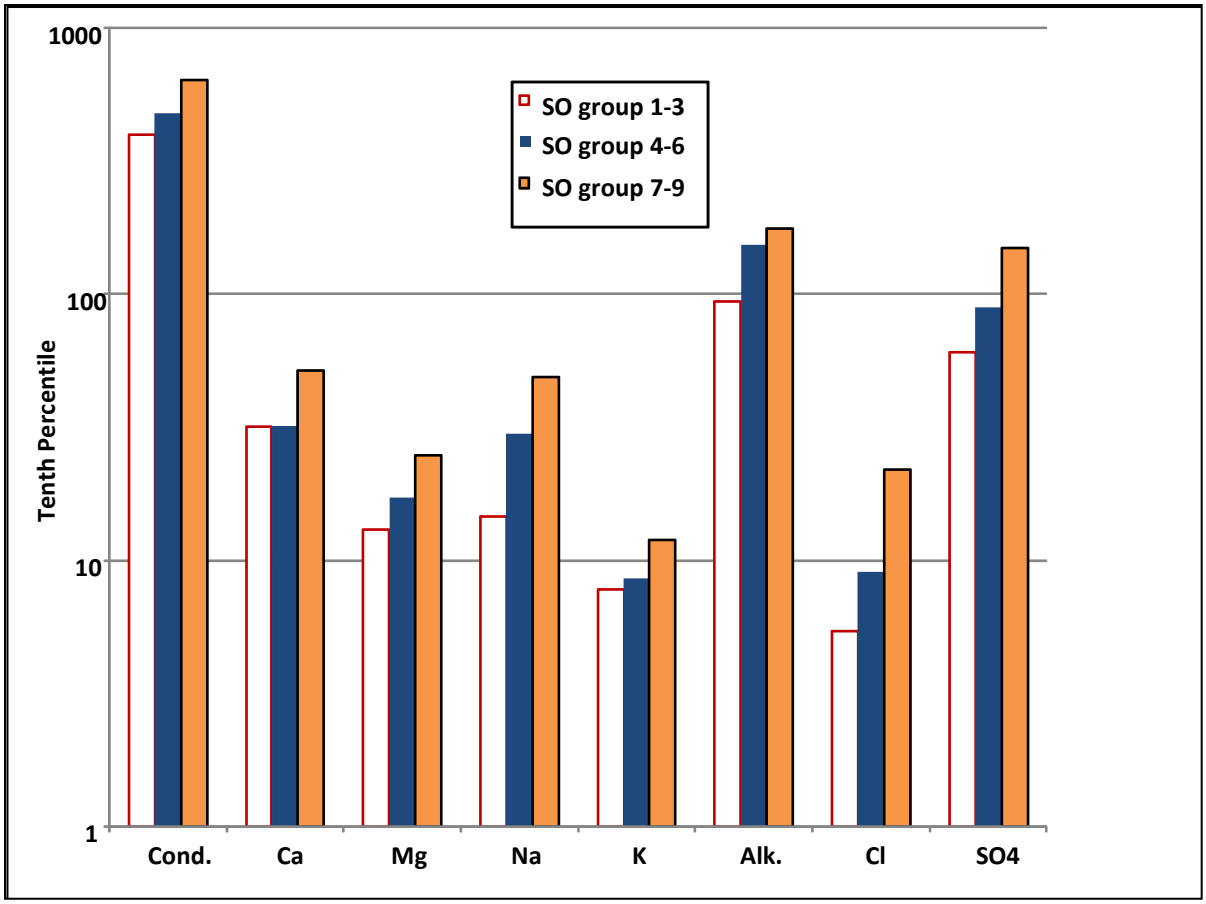

Figure 25. BLM parameter estimates (10th percentile values) for each SO group in Ecoregion 46 (Northern Glaciated Plains)

Key: Stream order: 1-3 are headwater streams, 4-6 are mid-reaches, and 7-9 are rivers.

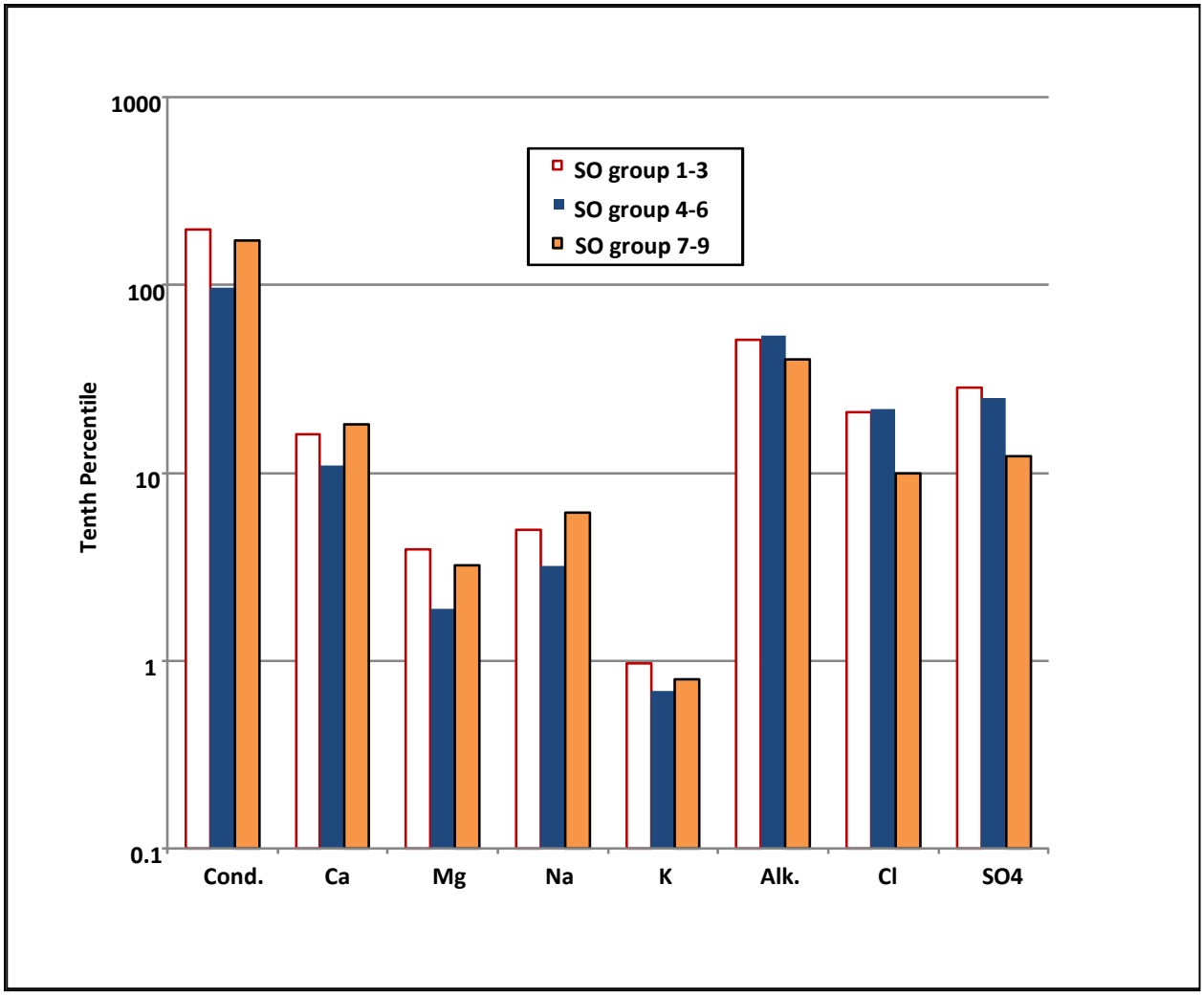

Figure 26. BLM parameter estimates for each SO group in Ecoregion 83 (Eastern Great Lakes Lowland)

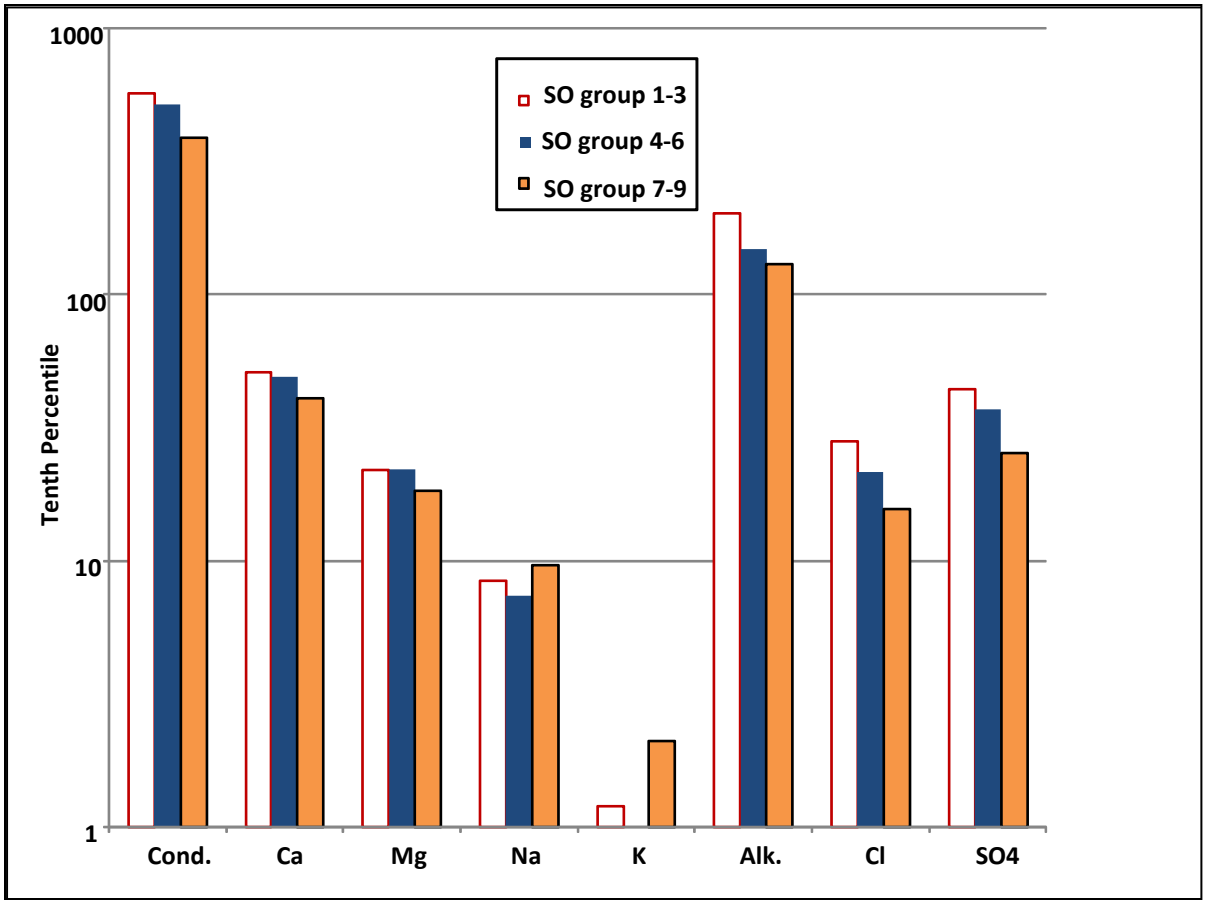

Figure 27. BLM parameter estimates for each SO group in Ecoregion 54 (Central Corn Belt Plains)

3.3.3 Comparison of Parameter Estimates to Results of Probability-Based Surface Water Sampling

There have been relatively few efforts to estimate GI concentrations in surface water at the national scale. Carleton (2006) developed a prototype geostatistical approach to estimate BLM parameters averaged over 8-digit HUC polygons. Carleton examined data from the NWIS and noted several limitations of this dataset in terms of uneven spatial sampling intensity. Carleton's prototype did not incorporate SO in the analysis, nor did it generate BLM parameter estimates.

Griffith (2014) compiled data from probability-based surface water quality sampling surveys conducted by EPA between 1985 and 2009, mostly from wadeable streams (SO group 1 through 4). These surveys included the National Acid Precipitation Assessment Program surveys, EMAP and regional EMAP surveys, WSA, and NRSA. The probability-based sample designs ensured that the results of these surveys represented the character of streams across the continental U.S. The water quality parameters included the same GIs as discussed above, and the results were presented on the same Level III ecoregion-specific basis.

We compared current results to those of Griffith (2014) because the lack of a probability-based sample design is a potential source of bias in the NWIS dataset. Parameter estimates based on the NWIS data for SO group 1 through 3 were compared to the corresponding estimates calculated by Griffith (2014). While Griffith did not tabulate 10th percentiles, he did tabulate first quartile (i.e., 25th percentile)

statistics for each ecoregion in supplemental material published with his article. Accordingly, we calculated 25th percentiles of the ecoregional NWIS data in SO class 1 through 3 (in addition to the 10th percentiles) to facilitate this comparison. The 25th percentiles from the two datasets are compared for conductivity in Figure 28 and calcium in Figure 29. The scatter plots reveal significant log-linear relationships between 25th percentiles for the two datasets; the coefficient of determination (R^2) was 0.668 for conductivity and 0.551 for calcium. For conductivity, the 25th percentiles differed by more than a factor of 2 in 17% of the Level III ecoregions; for calcium, 26% of the ecoregional results differed by more than a factor of 2.

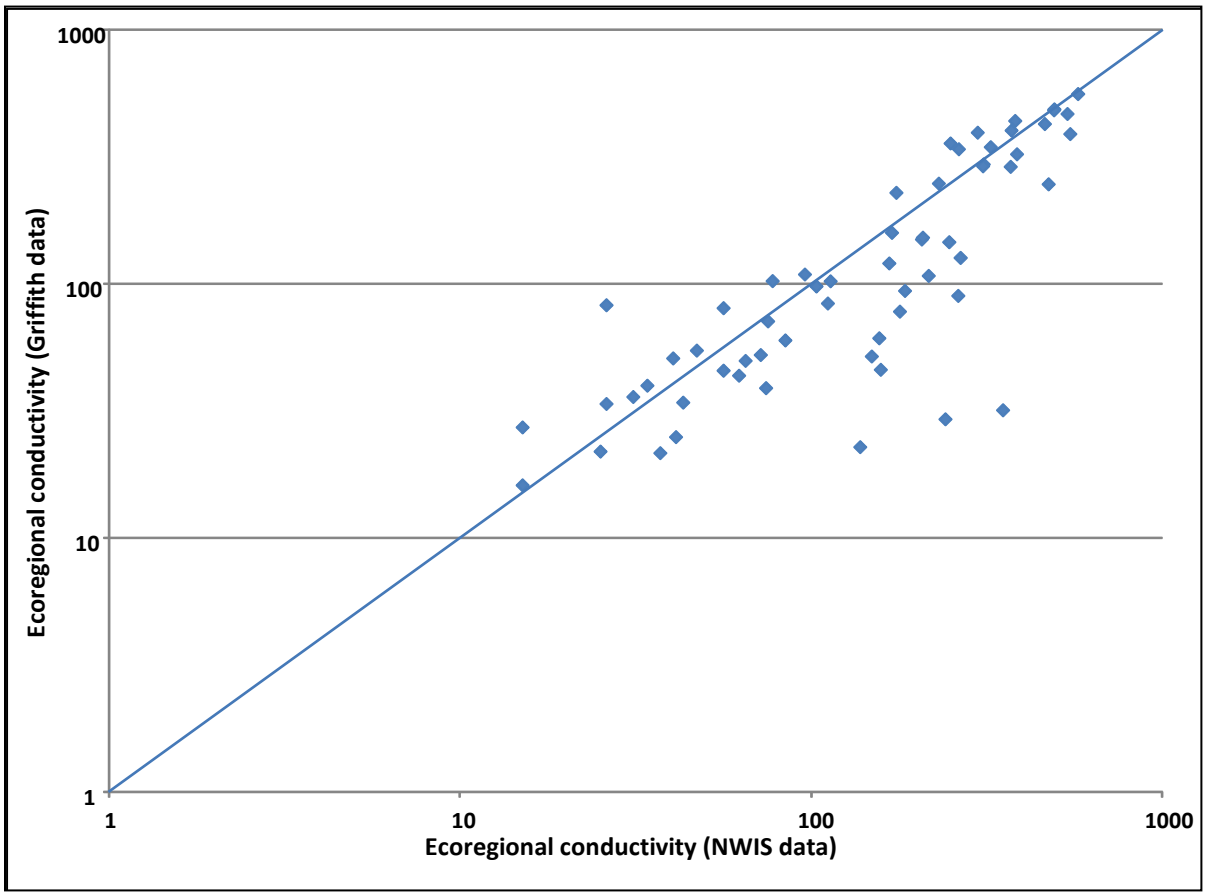

Figure 28. Scatter plot of ecoregional 25th percentile conductivity for NWIS Data (SO Class 1-3) versus ecoregional 25th percentile conductivity for Griffith data (mostly SO 1-4)

Solid diagonal line represents 1:1 agreement.

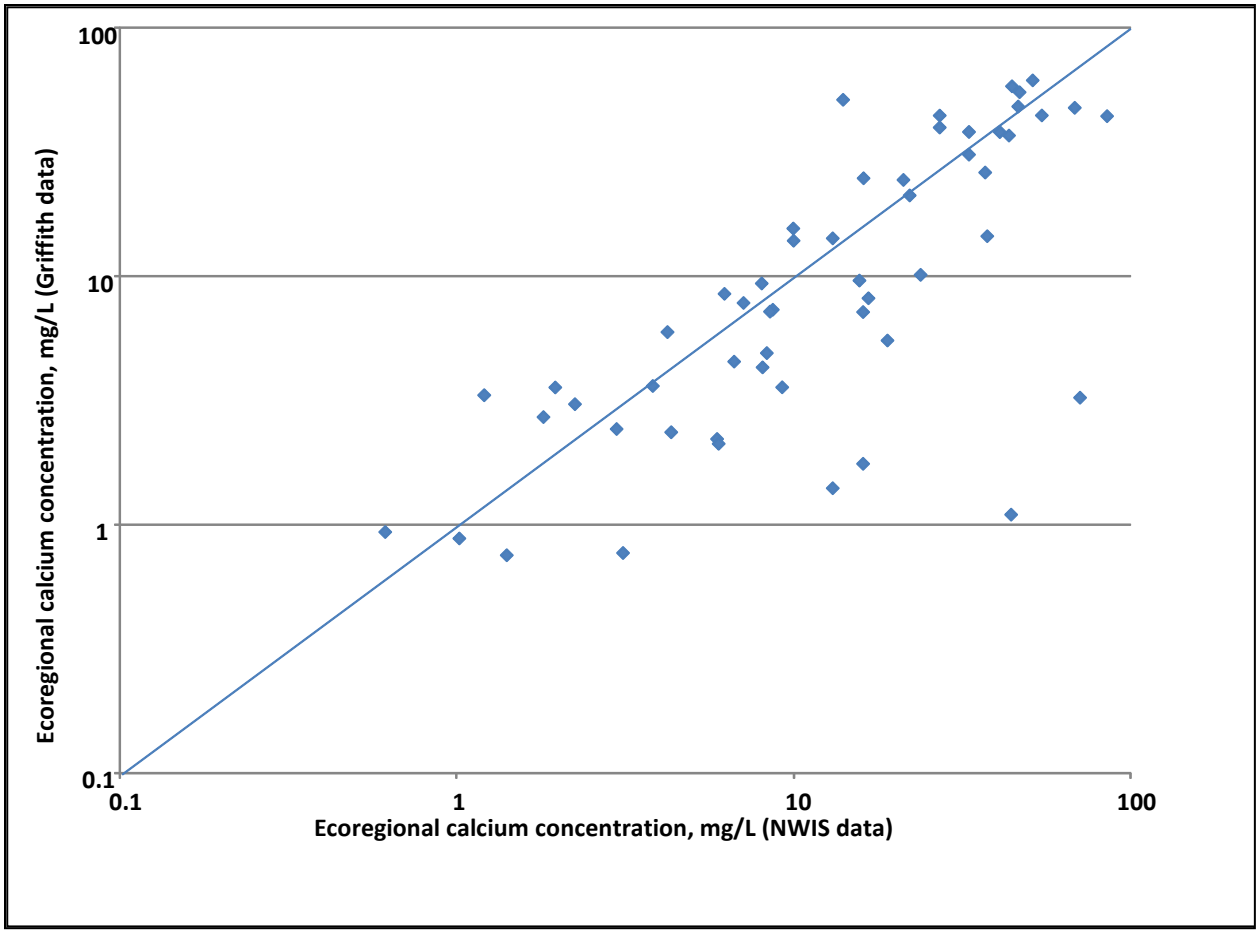

Figure 29. Scatter plot of ecoregional 25th percentile calcium concentration for NWIS data (SO class 1-3) versus ecoregional 25th percentile calcium concentration for Griffith data (mostly SO 1-4)

Solid diagonal line represents 1:1 agreement.

These results suggest reasonable overall consistency between datasets, as well as significant disparity in specific ecoregions. For example, agreement was especially poor in Ecoregions 19 (Wasatch and Uinta Mountains), 37 (Arkansas Valley), 38 (Boston Mountains), 39 (Ozark Highlands), 75 (Southern Coastal Plain), and 78 (Klamath Mountains). NWIS data were examined at the station-specific level to understand why these ecoregional 25th percentiles of conductivity and calcium in the low SO group were so inconsistent with corresponding percentiles presented by Griffith. Table 11 presents salient characteristics of the conductivity data for Ecoregion 19, including the number of stations, samples per station, 25th percentile conductivity, the lowest station-specific 25th percentile conductivity in the NWIS data, and other remarks. In that ecoregion, conductivity data were reported for 62 stations in the NWIS database; the number of observations per station ranged from 1 to 189, with a median of 22 observations per station. In comparison, the EPA data analyzed by Griffith reported conductivity data for 32 stations, with a single observation per station. The 25th percentile conductivity based on NWIS data was 240 versus 22.9 microsiemens per centimeter (µS/cm) based on Griffith's analysis of EPA data. When recalculated for individual stations in Ecoregion 19 low SO group, the 25th percentile conductivities varied widely, from 18.25 to 1590 µS/cm. Griffith reported a median conductivity of 213

µS/cm (nine times larger than the 25th percentile), indicating considerable variability in that data as well.

Table 11. Characteristics of the conductivity data for Ecoregion 19 in the low SO group.		
	NWIS data	Griffith data
Number of stations	62	32
Samples per station	1 – 189	1
Median samples per station	22	1
25th percentile conductivity (µS/cm)	240	22.9
Range of station-specific 25th percentile conductivity (µS/cm)	18.25-1590	
Other remarks	25th percentiles of conductivity reported by Griffith were marginally higher than the *minimum* station-specific 25th percentiles calculated from the NWIS data	Median conductivity = 213 µS/cm (19x the 25th percentile value) indicates high variability

The same information is tabulated for Ecoregions 37, 38, 75, and 78 in Tables 12 through 15. Although the details regarding the data vary in each of these ecoregions, a number of commonalities emerge from these examinations:

- In four ecoregions, the number of NWIS stations in SO group 1 through 3 was small relative to other ecoregions (the median number of ecoregional stations in SO group 1 through 3 was 68).
- In three ecoregions, 10th and 25th percentiles of conductivity and/or calcium decreased with SO group, contrary to the general trend.
- In three ecoregions, 25th percentiles of conductivity reported by Griffith were marginally higher than the minimum station-specific 25th percentiles calculated from the NWIS data for the corresponding ecoregion and low SO group.

Table 12. Characteristics of the conductivity data for Ecoregion 37 in the low SO group.		
	NWIS data	Griffith data
Number of stations	34	45
Samples per station	1 – 129	1
Median samples per station	2	1
25th percentile conductivity (µS/cm)	350	32
Range of station-specific 25th percentile conductivity (µS/cm)	29-846.5	
Other remarks	10th and 25th conductivity percentiles *decrease* with SO group; Small number of NWIS stations; 25th percentiles of conductivity reported by Griffith were marginally higher than the *minimum* station-specific 25th percentiles calculated from the NWIS data	

Table 13. Characteristics of the conductivity data for Ecoregion 38 in the low SO group		
	NWIS data	Griffith data
Number of stations	31	38
Samples per station	1 – 8	1
Median samples per station	3	1
25th percentile conductivity (µS/cm)	137	22.9
Range of station-specific 25th percentile conductivity (µS/cm)	22-384	
Other remarks	10th and 25th conductivity percentiles *decrease* with SO group; Small number of NWIS stations; 25th percentiles of conductivity reported by Griffith were marginally higher than the *minimum* station-specific 25th percentiles calculated from the NWIS data	

Table 14. Characteristics of the calcium data for Ecoregion 75 in the low SO group.		
	NWIS data	Griffith data
Number of stations	360	42
Samples per station	1 – 177	1
Median samples per station	17	1
25th percentile calcium (mg/L)	13	1.41
Range of station-specific 25th percentile calcium (mg/L)	0.02-91	
Other remarks	10th and 25th calcium percentiles *decrease* with SO group	

Table 15. Characteristics of the conductivity data for Ecoregion 78 in the low SO group.		
	NWIS data	Griffith data
Number of stations	15	45
Samples per station	1 – 18	1
Median samples per station	8	1
25th percentile conductivity (µS/cm)	26	98.4
Range of station-specific 25th percentile conductivity (µS/cm)	19-326.5	
Other remarks	Small number of NWIS stations; 6 of 15 stations were Ashland Creek (OR) or tributaries	

It is possible that the disparities noted above arise in part from non-representative sampling in the NWIS data. For example, representativeness of NWIS data is questionable in Ecoregion 78 because 40% of the stations were sampled in a single water body. There was also a difference in the way data for repeated sampling at individual stations were processed, due to differences between the NWIS data and data compiled by Griffith. In the NWIS data, water quality was sampled repeatedly at a significant

number of stations, and we included daily averages of all of these measurements in the calculation of 10th percentile estimates. In the probabilistic EPA surveys analyzed by Griffith, individual stations were usually sampled once. In the case of repeated sampling at a station, Griffith used data from only the first sample reported for the station in the statistics he calculated. This difference implies that our estimates of BLM parameters incorporate temporal as well as spatial variability in water quality, while Griffith's results do not. Thus, it would be unrealistic to expect complete agreement between these results. It should also be reiterated that sampling bias in SO is probably not a factor in these disparities because the estimates from the NWIS data were based on measurements from stream orders 1 through 3, which is generally consistent with the data compiled by Griffith (2014).

It is particularly of concern when percentiles based on NWIS data are higher than those calculated based on Griffith data because this suggests the parameter estimates may result in non-conservative BLM predictions of copper, or others metals, criteria. To evaluate this concern, we ran the BLM (version 2.1.2) using the 25th percentile GI estimates of Griffith and those from the current analysis for NWIS SO group 1 through 3 for each ecoregion in which parameters were available. If the 25th percentile of a GI was not available, the value was projected from the 25th percentile of conductivity using regressions based on NWIS data. If the 25th percentile of conductivity was not available (this occurred in 24 ecoregions), no BLM prediction was made. The inputs for pH and DOC were ecoregional values. There were 60 ecoregions where BLM predictions of copper criteria using the 25th percentile GI estimates from NWIS and Griffith could be compared. The criteria estimated in these ecoregions using GI input parameters from the two sources agree very well, as shown in Figure 30. The R^2 was 0.9897, and relative percent differences (RPDs) ranged from -21 to 39% with an average RPD of 3% and a median of 0.1%.

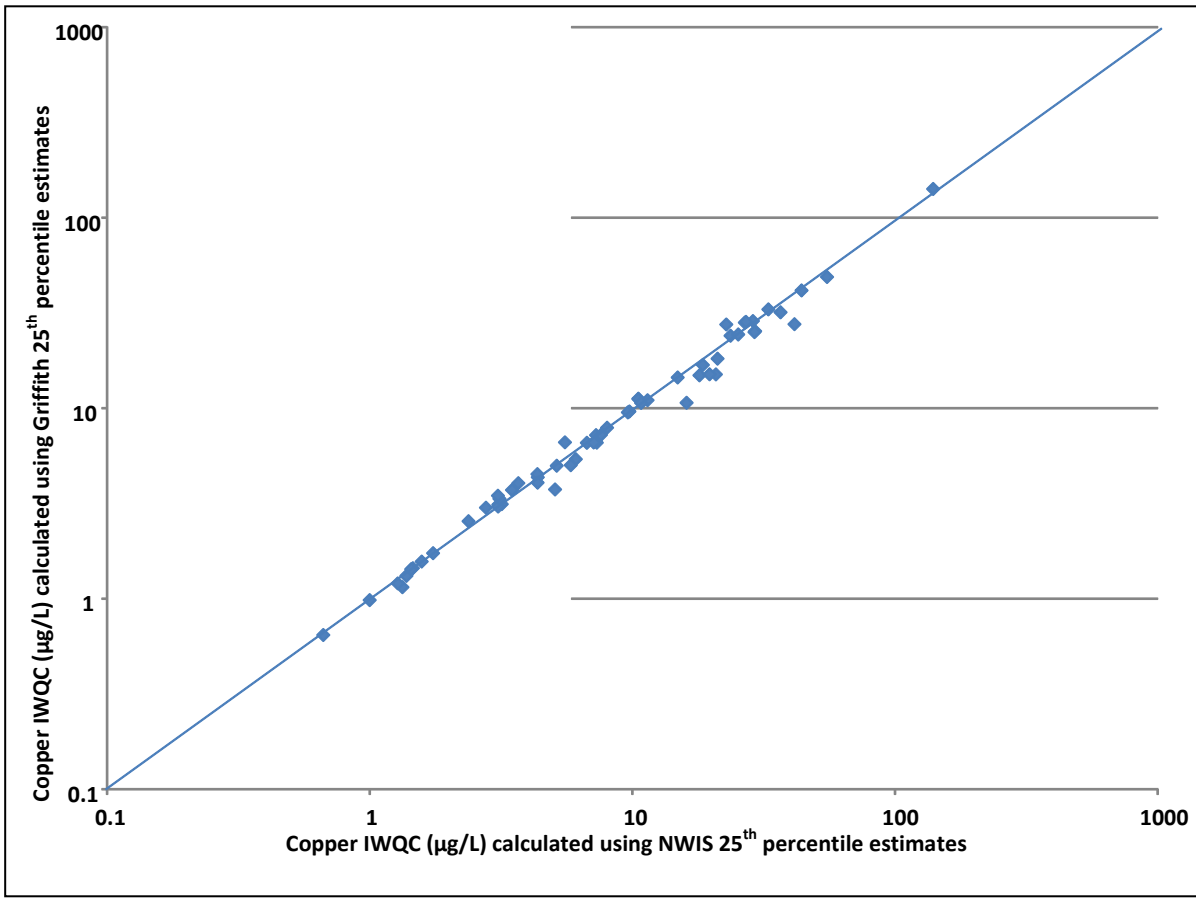

Figure 30. BLM predictions of copper criteria made with GI water quality parameters based on ecoregional 25th percentile from NWIS data (SO class 1-3) versus ecoregional 25th percentile calcium Concentration for Griffith data (mostly SO 1-4).

R^2 = 0.99. Solid diagonal line represents 1:1 agreement

While it is possible that the GI estimates presented here as recommended input values for use in the BLM for copper could be improved, it is not clear that such improvement would result in substantially better predictions of protective copper criteria based on the BLM. This is because BLM predictions for copper are much more sensitive to pH and DOC.

3.4 Summary

In this section we have incorporated SO variation in the GI water quality parameter estimates to refine the default parameters estimates for use in the BLM when data are not available.

EPA found that values of the GI parameter estimates generally increased with SO. This trend was most apparent and consistent for higher order streams (SO≥7). Tenth percentile estimates of conductivity increase with stream order group in 58% of ecoregions (comparing low to medium SO groups) and 84% of ecoregions (comparing medium to high SO groups). The same trend was evident for the GIs that are input parameters to the BLM.

We compared the parameter estimates for SO group 1 through 3 to those calculated by Griffith (2014). This comparison revealed significant log-linear relationships between 25th percentiles for the two datasets; the coefficient of determination (R^2) was 0.668 for conductivity and 0.551 for calcium. For conductivity, the 25th percentiles differed by more than a factor of 2 in 17% of the Level III ecoregions; for calcium, 26% of ecoregional results differed by more than a factor of 2. There is also considerable variability in the relationship between ecoregional statistics based on NWIS versus Griffith's data. Possible causes of these disparities may be due to sampling bias in the NWIS database, limited numbers of samples in some ecoregions, and differences in the degree and treatment of repeated sampling at individual locations.

NWIS percentiles higher than Griffith (2014) suggest that the recommended parameter estimates may result in non-conservative BLM predictions of copper criteria. The BLM was run to predict copper criteria for 60 ecoregions using 25th percentile GI estimates as parameter inputs from NWIS and Griffith's data. The criteria predicted using the two sets of GI parameter inputs agreed favorably. The R^2 was 0.990, and RPDs ranged from -21 to 39% with an average RPD of 3% and a median of 0.1%. These results demonstrated that the recommended default GI parameter estimates are reasonably conservative.

EPA incorporated SO variation in the parameter estimates to refine recommended input values for use in the BLM. EPA found the 10th percentile ecoregion, SO group specific estimates to be reasonably protective inputs and recommends their use where site-specific parameters are not readily available.

4 DOC ESTIMATION USING THE NATIONAL ORGANIC CARBON DATABASE

The following section summarizes our investigation into whether ecoregion and water body-type-specific DOC concentration percentiles tabulated by EPA for the *Development of National Bioaccumulation Factors Technical Support Document* (hereafter referred to as the National Organic Carbon Database (NOCD)) (USEPA, 2003)) offer reasonable estimates of lower-percentile DOC concentrations. A summary of the NOCD's data sources, analysis, and uncertainty associated with ecoregional statistics derived from the NOCD is provided below. This section also discusses how we recalculated ecoregional DOC percentiles for rivers and streams, and then tested for bias in the NOCD. Finally, we compared results based on the NOCD and data from the Wadeable Stream Assessment (WSA) (USEPA, 2006b)) and the National River and Stream Assessment (NRSA) (USEPA, 2013)).

4.1 Description of the NOCD

The NOCD is a compilation of pre-2003 organic carbon data derived from two sources: EPA's Storage and Retrieval Data Warehouse (STORET), recently renamed the STORET Legacy Data Center (LDC), and USGS's National Water Data Storage and Retrieval System (WATSTORE), the predecessor of NWIS. A complete background on the NOCD is available in USEPA 2003.

EPA's LDC database contains water quality monitoring data collected by academia, volunteer groups, and tribes, as well as federal, state, and local agencies. Geographically, the LDC data represent all 50 states and all U.S. territories and jurisdictions, along with portions of Canada and Mexico. The database queried for this investigation is often referred to as the "historical" or "old" STORET database because it contains water quality data dating back to the early part of the 20th century through the end of 1998. Data from 1999 to the present are stored in the "modernized" STORET Data Warehouse.[4] The LDC contains raw biological, chemical, and physical data for both surface water and groundwater. Each sampling result is accompanied by: information on sample collection location (latitude, longitude, state, county, HUC, and a brief site description), date the sample was gathered, the medium sampled, and the name of the organization that sponsored the monitoring.

We retrieved data from LDC and WATSTORE in January 2000. Approximately 800,000 records containing data on particulate organic carbon (POC), dissolved organic carbon (DOC), or total organic carbon (TOC) were obtained for the period beginning in 1970 through the latest year that data were available (1999 for WATSTORE and 1998 for LDC). This initial retrieval was limited to samples taken from ambient surface waters (i.e., samples from wells, springs, effluents, and other non-ambient sources were excluded). Additionally, this retrieval included multiple types of organic carbon measurements to ensure that the data would be sufficiently comprehensive.

WATSTORE was established in 1972 to provide an effective and efficient means for processing and maintaining water data collected through USGS activities and to facilitate release of that data to the public. The WATSTORE database resides on the central computer facilities of the USGS and contains results of approximately two million analyses of both surface and groundwater that provide data on

[4] Refer to http://www.epa.gov/storet/dbtop.html for more information on the STORET Data Warehouse and the LDC.

chemical, physical, biological, and radiological characteristics. EPA queried WATSTORE, the Water Quality File, to retrieve DOC data.

After retrieval, the data from LDC and WATSTORE were combined into a single database. The data were then processed and screened to ensure that only the most appropriate data would be retained. This screening process is outlined below:

- Values that were coded in such a way as to suggest uncertainty in the measurement were deleted from the database.
- The database was restricted to the following water body types: estuaries, lakes, reservoirs, and streams (including rivers).
- "Pseudo-ecoregions" were added for the five Great Lakes.
- The time period for the data was restricted to 1980 through 1999.
- Some values for DOC were reported to be below analytical detection levels. In this situation, the value was assumed to be half of the reported detection level. Values with "high" detection levels (i.e., >1.0 mg/L for DOC) were deleted from the database because of the greater uncertainty involved in estimating definitive values of DOC in these situations.
- A small fraction of the DOC and POC concentrations obtained from the LDC database exceeded concentrations considered to represent upper limits of DOC concentrations reported in U.S. water bodies (i.e., 0.2% exceeded 60 mg/L for DOC). These extreme values were based on a review of organic carbon data by Thurman (1985), who reported extreme values of DOC concentrations as high as 50 mg/L in dystrophic lakes and 60 mg/L in tributaries draining wetland systems. Therefore, values for DOC above 60 mg/L were removed from the database.

The NOCD that resulted from processing and screening data retrieved from the LDC and WATSTORE databases has some limitations, which are described below:

- The WATSTORE and LDC databases do not reflect a random sampling of U.S. surface waters. They contain datasets with a diversity of sampling design and thus data may be biased towards locations and water bodies with known water quality impairments.
- These data also reflect spatial bias due to unequal sampling efforts in different areas. For example, about half of the DOC and POC values in the databases were from samples collected in Maryland, New York, Ohio, Florida, and Delaware. Therefore some states are disproportionally represented, even when one considers the relative surface water area likely to be contained within each state.
- WATSTORE and LDC generally contain more data from sampling sites in larger river and stream systems, and areas subjected to proportionately greater human influence compared with random statistical sampling.

4.2 Recalculation of Ecoregional DOC Percentiles for Rivers and Streams

Lower percentile (1^{st}, 5^{th}, 10^{th}, and 25^{th} percentiles) DOC concentrations were calculated from all data for rivers and streams in each Level III ecoregion (Table 16). Nonparametric (i.e., rank) percentiles were calculated following the recommendations of Dierickx (2008) and Hyndman and Fan (1996). We also calculated confidence limits for the percentiles using the method presented in Berthouex and Brown

(1994). Upper and lower 95% confidence limits (UCLs and LCLs) were calculated if 20 or more DOC concentrations were available for an ecoregion (Berthouex and Brown, 1994).

Table 16. Lower percentile values of DOC in U.S. streams and rivers by ecoregion, including 95% confidence limits for percentile concentrations if n>20.

Level III Ecoregion	Ecoregion Name	n (count)	1%	1% (LCL)	1% (UCL)	5%	5% (LCL)	5% (UCL)	10%	10% (LCL)	10% (UCL)	25%	25% (LCL)	25% (UCL)
1	Coast Range	91	≤0.2	≤0.2	0.6	0.9	≤0.2	1.1	1.12	0.78	1.4	1.8	1.4	2
2	Puget Lowland	835	0.84	0.47	1	1.9	1.7	2	2.5	2.3	2.7	3.9	3.5	4.1
3	Willamette Valley	66	≤0.5	≤0.5	0.73	0.8	≤0.5	1.08	1.07	0.68	1.2	1.48	1.2	2
4	Cascades	101	0	≤0	0.2	0.3	≤0	0.44	0.5	0.3	0.5	0.7	0.5	0.9
5	Sierra Nevada	32	≤1.9	≤1.9	1.9	1.9	≤1.9	2.3	2.09	≤1.9	2.3	2.55	2.13	3.11
6	Southern and Central California Chaparral and Oak Woodlands	480	1.1	≤0	1.5	1.8	1.7	1.9	2.1	1.9	2.2	3	2.67	3.2
7	Central California Valley	180	1.21	≤0.8	1.8	2.11	1.66	2.48	2.71	2.3	3.5	5.3	4.4	6.26
8	Southern California Mountains	6	≤4.4			≤4.4			≤4.4			5.45		
9	Eastern Cascades Slopes and Foothills	13	≤1.3			≤1.3			1.42			1.75		
10	Columbia Plateau	73	≤0.7	≤0.7	1.3	1.67	≤0.7	2.03	2.04	1.29	2.34	2.9	2.3	3.2
11	Blue Mountains	26	≤1	≤1	1.05	1.07	≤1	1.45	1.34	≤1	1.81	1.9	1.28	2.61
12	Snake River Plain	50	≤2	≤2	2	2	≤2	2.2	2.2	≤2	2.43	3.08	2.27	3.68
13	Central Basin and Range	1553	0.8	0.69	0.9	1.2	1.2	1.3	1.5	1.4	1.6	2	2	2.1
14	Mojave Basin and Range	35	≤2.5	≤2.5	2.55	2.58	≤2.5	3	2.84	≤2.5	3.32	3.6	2.99	3.8
15	Northern Rockies	778	0.7	0.6	0.72	0.9	0.9	1	1	1	1.1	1.3	1.3	1.4
16	Idaho Batholith	29	≤1.2	≤1.2	1.2	1.2	≤1.2	1.4	1.4	≤1.2	1.8	1.9	1.39	2.31
17	Middle Rockies	87	≤0	≤0	0	0	≤0	0.14	0.18	0	0.46	1.1	0.42	1.4
18	Wyoming Basin	150	2.05	≤1.9	3.03	3.56	2.26	4.18	4.31	3.59	4.6	5.48	5.1	5.8
19	Wasatch and Uinta Mountains	46	≤1.5	≤1.5	1.5	1.57	≤1.5	1.82	1.8	≤1.5	2.51	2.88	1.9	3.6
20	Colorado Plateaus	798	1.5	0.36	1.6	2.4	2	2.6	3	2.7	3.25	4.3	4.1	4.6
21	Southern Rockies	1129	0.2	0.1	0.3	0.5	0.4	0.5	0.6	0.6	0.6	0.8	0.8	0.9
22	Arizona/New Mexico Plateau	281	1.65	≤1.2	2.01	2.2	2.09	2.4	2.62	2.3	2.91	3.7	3.3	3.9
23	Arizona/New Mexico Mountains	37	≤1.3	≤1.3	1.41	1.48	≤1.3	2.35	2.16	≤1.3	2.64	2.8	2.33	3.57
24	Chihuahuan Deserts	116	0.5	≤0.5	0.64	1	0.5	2.15	2.34	1	3	3.68	3	4.44
25	High Plains	439	0.3	≤0.1	0.4	0.8	0.5	3	4.4	3.2	4.9	7	6.42	7.78
26	Southwestern Tablelands	167	1.94	≤1.8	2.04	2.5	2	2.98	3.28	2.52	3.7	4.4	4	4.9
27	Central Great Plains	228	0.59	≤0.4	2.01	3	1.8	3.2	3.8	3	4	5	4.8	5.71
28	Flint Hills	10	≤4.9			≤4.9			4.9			5.15		
29	Central Oklahoma/Texas Plains	289	1.9	≤1	2.09	3	2.5	3.38	3.8	3	4	4.8	4.21	5
30	Edwards Plateau	200	0.5	≤0.5	0.5	0.5	0.5	0.82	1	0.5	1	1	1	1.6
31	Southern Texas Plains	58	≤0.5	≤0.5	0.5	0.5	≤0.5	1	1	0.5	1.34	2	1	2
32	Texas Blackland Prairies	829	0.5	0.5	0.5	1	1	1	2	2	2	3	3	3.99

Level III Ecoregion	Ecoregion Name	n (count)	1%	1% (LCL)	1% (UCL)	5%	5% (LCL)	5% (UCL)	10%	10% (LCL)	10% (UCL)	25%	25% (LCL)	25% (UCL)
33	East Central Texas Plains	268	1	≤1	1.95	2.6	2	3	3	3	3	3.83	3.1	4
34	Western Gulf Coastal Plain	399	2.8	≤1.7	3	3.7	3	4	4	4	4	5	5	5.9
35	South Central Plains	523	1.45	≤0.4	2.07	3.92	3.2	4	4.6	4.1	4.98	5.7	5.4	6
36	Ouachita Mountains	198	0	≤0	0.47	0.8	0.4	1.1	1.1	0.86	1.6	2.38	2.08	2.8
37	Arkansas Valley	184	0.49	≤0.4	0.65	0.8	0.55	1.4	1.9	0.85	2.39	3.73	3	4.4
38	Boston Mountains	21	≤0.4	≤0.4	0.4	0.4	≤0.4	0.5	0.42	≤0.4	0.5	0.5	0.4	0.74
39	Ozark Highlands	233	1.67	≤1.5	2	2	2	2.12	2.3	2	2.5	3	2.7	3.15
40	Central Irregular Plains	434	2.71	≤1.4	3	3.5	3.1	3.6	4	3.6	4	4.9	4.6	5
41	Canadian Rockies	36	≤0.3	≤0.3	0.35	0.39	≤0.3	0.6	0.57	≤0.3	0.9	0.93	0.6	1.03
42	Northwestern Glaciated Plains	36	≤3	≤3	3.05	3.09	≤3	6.63	5.05	≤3	11.23	13.25	6.12	16.34
43	Northwestern Great Plains	679	2.22	0.72	3.19	4.4	3.8	5.2	6.2	5.6	6.73	9.9	9.19	10
44	Nebraska Sand Hills	4	≤1.4			≤1.4			≤1.4			1.4		
45	Piedmont	309	0.4	≤0	0.5	0.7	0.6	0.9	1	0.8	1.1	1.7	1.4	2.1
46	Northern Glaciated Plains	142	4.03	≤3.3	8.01	9.13	5.11	9.82	9.9	9.16	11	12	11	12.86
47	Western Corn Belt Plains	193	0.44	≤0.2	2.43	2.77	2.28	2.96	3.14	2.8	3.5	3.9	3.6	4.3
48	Lake Agassiz Plain	261	3.1	≤0.8	4.52	6.41	4.72	7.1	7.6	6.87	7.97	9	8.7	9.22
49	Northern Minnesota Wetlands	44	≤7.8	≤7.8	8.25	9.05	≤7.8	11	11	≤7.8	12	13	11	14.88
50	Northern Lakes and Forests	403	2	≤0.1	2.2	2.72	2.36	3.08	3.7	3.06	4.5	5.8	5.5	6.2
51	North Central Hardwood Forests	153	0.54	≤0	2.19	2.67	1.42	2.9	3.1	2.7	3.77	4.95	3.9	5.6
52	Driftless Area	49	≤1.7	≤1.7	2.31	2.4	≤1.7	3.25	3.1	≤1.7	4.31	4.5	3.4	5.8
53	Southeastern Wisconsin Till Plains	439	2	≤0.25	2.1	3.4	2.7	4.3	5.3	4.3	5.6	6.8	6.5	7
54	Central Corn Belt Plains	202	1	≤0.7	1.68	2.12	1.61	2.5	2.73	2.2	3	3.9	3.57	4.8
55	Eastern Corn Belt Plains	1398	0	0	0	0	0	0.2	3.8	3	4	5.2	5	5.3
56	S. Michigan/N. Indiana Drift Plains	287	1.4	≤1.4	1.92	2.7	2.05	3.43	3.8	3.17	4.2	5.4	4.9	5.7
57	Huron/Erie Lake Plains	3825	0	0	0	3.9	3.7	4.07	4.7	4.52	4.8	5.8	5.7	5.9
58	Northeastern Highlands	14044	0.25	0.25	0.25	0.25	0.25	0.5	0.64	0.62	0.65	0.95	0.94	0.97
59	Northeastern Coastal Zone	102	0.05	≤0	1.6	2.02	≤0	2.6	2.63	1.74	2.92	3.28	3	3.63
60	Northern Appalachian Plateau and Uplands	354	0.6	≤0.5	0.8	1	0.87	1.1	1.3	1.1	1.4	2	1.9	2.3
61	Erie Drift Plains	919	0	0	0	4.8	4.6	4.9	5.1	5	5.1	5.5	5.4	5.6
62	North Central Appalachians	106	0.41	≤0.4	0.5	0.5	≤0.4	0.7	0.7	0.5	0.78	0.98	0.8	1.2
63	Middle Atlantic Coastal Plain	16730	1.1	1.01	1.2	1.8	1.8	1.8	2.2	2.2	2.2	2.7	2.7	2.7
64	Northern Piedmont	1525	0.43	0.3	0.5	1	0.99	1	1.3	1.2	1.4	2.2	2.1	2.4
65	Southeastern Plains	3813	1	0.51	1.1	2	1.9	2.03	2.4	2.38	2.5	3.3	3.2	3.3
66	Blue Ridge	699	0	0	0	0.4	0.3	0.4	0.5	0.4	0.5	0.6	0.6	0.7
67	Ridge and Valley	733	0.1	0.1	0.1	0.4	0.3	0.5	0.6	0.5	0.7	1.05	1	1.1
68	Southwestern Appalachians	47	≤0.5	≤0.5	0.5	0.58	≤0.5	0.9	0.9	≤0.5	1	1	0.9	1.38
69	Central Appalachians	864	0.3	0.15	0.4	0.6	0.5	0.6	0.7	0.7	0.8	1.1	1.1	1.2
70	Western Allegheny Plateau	1795	0	0	0	0.78	0.2	1	1.6	1.4	1.7	2.7	2.5	2.9
71	Interior Plateau	559	0.1	≤0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.3	0.2	0.3

Level III Ecoregion	Ecoregion Name	n (count)	1%	1% (LCL)	1% (UCL)	5%	5% (LCL)	5% (UCL)	10%	10% (LCL)	10% (UCL)	25%	25% (LCL)	25% (UCL)
72	Interior River Valleys and Hills	328	1.32	≤0.15	1.8	2.1	1.9	2.4	2.69	2.4	3.05	4.2	3.6	4.6
73	Mississippi Alluvial Plain	503	1.7	≤1	2.14	2.82	2.4	3.1	3.4	3.2	3.6	4.3	4.1	4.5
74	Mississippi Valley Loess Plains	21	≤1.4	≤1.4	1.41	1.41	≤1.4	2.61	1.72	≤1.4	2.7	2.7	1.46	4.72
75	Southern Coastal Plain	4223	0.9	0.9	1.09	5.3	4.7	6	8	7.6	8.3	12.1	11.9	12.3
76	Southern Florida Coastal Plain	-	-			-			-			-		
77	North Cascades	50	≤0.2	≤0.2	0.2	0.26	≤0.2	0.4	0.4	≤0.2	0.4	0.48	0.4	0.5
78	Klamath Mountains	8	≤1.7			≤1.7			≤1.7			2.3		
79	Madrean Archipelago	9	≤2.6			≤2.6			2.6			4.05		
80	Northern Basin and Range	16	≤1.6			≤1.6			1.81			2.5		
81	Sonoran Basin and Range	133	1.33	≤1.3	1.7	1.8	1.38	2.1	2.2	1.8	2.6	3.85	2.94	4.4
82	Laurentian Plains and Hills	21	≤4.6	≤4.6	4.7	4.69	≤4.6	5.68	5.52	≤4.6	7.98	8.45	5.15	9.3
83	Eastern Great Lakes and Hudson Lowlands	1430	0	0	0	0	0	0.2	1.9	1	2	5.1	5	5.5
84	Atlantic Coastal Pine Barrens	243	1	≤0.9	1.1	1.22	1.1	1.5	1.6	1.32	2	2.6	2.4	3
	Lake Erie	9	≤1.1			≤1.1			1.18			1.4		
	Lake Michigan	5	≤2.6			≤2.6			≤2.6			2.6		
	Lake Ontario	14	≤0.4			≤0.4			0.4			2.2		
	Lake Superior	7	≤1.2			≤1.2			≤1.2			1.4		

As was the case for the BLM GI input parameters, we consider low percentiles of the ecoregional DOC concentration distributions to be reasonably protective inputs to the BLM for sites where DOC measurements are not available.

4.3 Testing for Bias in the NOCD

EPA conducted an evaluation of bias in the NOCD (USEPA, 2003) using data from EPA's Environmental Monitoring and Assessment Program's (EMAP) 1997 to 1998, which sampled mid-Atlantic streams and rivers (USEPA, 2006b). This effort is described below in Section 4.3.1.

We also evaluated the bias in the NOCD using independent data from EPA's Wadeable Streams Assessment (WSA), which included DOC measurements from a statistically based random sample of approximately 2,000 wadeable, perennial 1st through 5th order streams (USEPA, 2006c). In Section 4.3.2, we compare the WSA data to the ecoregion-specific DOC concentration percentiles calculated from the NOCD.

4.3.1 Previous Efforts Using EMAP Data

Ideally, the data used to generate the distribution of national organic carbon concentration values should originate from a random sampling of U.S. surface waters, and should be appropriately stratified and weighted by spatial and temporal factors that would be expected to influence organic carbon concentrations in aquatic ecosystems (e.g., water body type, hydrologic and watershed characteristics, ecoregion, season). However, these data are not available on a national scale. The strength of this analysis is that the data from USGS's WATSTORE and EPA's LDC databases include a large number of

records (e.g., >110,000 DOC values), a representation of DOC values for all 50 states, and reasonably long period over which data were collected (1980 through 1999 for these analyses).

Data generated by EMAP are based on a stratified, random sampling strategy that was specifically designed to minimize the influence of sampling bias on the data and to enable statistically based extrapolations across geographic regions (Herlihy et al., 2000). At the time the NOCD was developed, the EMAP databases containing DOC measurements were limited to smaller geographic scales and specific water body types.

Previously, to address the question of sampling bias and its impact on the representativeness of the NOCD values, EPA made quantitative comparisons that involved contrasting geographically distinct subsets of the WATSTORE/LDC databases with geographically similar subsets of data produced by EMAP. DOC data from EMAP's 1997 to 1998 sampling of mid-Atlantic streams and rivers were compared with similar geographic subsets from the WATSTORE/LDC databases. The mid-Atlantic EMAP database was chosen because sufficient DOC data were available for rivers and streams to make meaningful comparisons at the state and ecoregion levels. Similar comparisons are made for four mid-Atlantic ecoregions (Piedmont, Ridge and Valley, Central Appalachians, Western Allegheny Plateau) which is well represented in the WATSTORE/LDC databases (USEPA, 2003).

Based on both sets of comparisons, it is apparent that the agreement between the WATSTORE/LDC and EMAP data was best at the middle to lower tails of the distributions, and poorest at the higher end of the distributions. At the lower tails of the distributions (e.g., 10^{th}, 25^{th} percentiles) the WATSTORE/LDC DOC data are generally within 30% of the EMAP data (Ecoregion 70 being the only exception). The median DOC values of the WATSTORE/LDC data show a slightly higher bias compared with median values from the EMAP data, but are usually within a factor of 1.5 (Ecoregions 47 and 70 are about a factor of 2 greater). This result is expected, given the greater focus of the WATSTORE/LDC sampling sites on larger river and stream systems, and on areas subjected to proportionately greater human influence compared with the EMAP sampling sites. Since EPA is interested in supporting the generation of BLM values that are protective of aquatic life, the lack of bias noted for the lower tails of the DOC concentration distributions is noteworthy.

4.3.2 Testing for Bias Using Data from the WSA

A more comprehensive evaluation of the effects of sampling bias on the NOCD can now be made using the results of national statistically-designed water quality sampling surveys. We assembled a database of organic carbon data from 1,313 randomly selected sites throughout the continental U.S. collected for the WSA. GIS procedures were used to associate each site with the Level III ecoregion corresponding to its location.

The 1,392 sites sampled for the WSA were identified using a probability-based sample design, a technique in which every element in the population has a known probability of being selected for sampling (USEPA, 2006). This ensured that the results of the WSA reflect the full range of variation among wadeable streams across the U.S. The target population for the WSA was wadeable, perennial streams in the conterminous U.S. (lower 48 states). The WSA used the National Hydrography Dataset (NHD), a comprehensive set of digital spatial data on surface waters (USGS, 2012), to identify the location of wadeable perennial streams. Rules for site selection included weighting to provide balance in the number of stream sites from each of the 1^{st} through 5^{th} SO size classes (Strahler, 1952, 1957),

and controlled spatial distribution to ensure that sample sites were distributed across the U.S.). The basic sampling design drew 50 sampling sites randomly distributed in each of the EPA Regions and WSA ecoregions. The unbiased site selection of the survey design ensures that assessment results represent the condition of the streams throughout the nation.

4.3.2.1 Selection of Statistical Test to Assess Potential Bias in DOC Data

The most appropriate statistical test for determining bias in the NOCD is the comparison of WSA and organic carbon database DOC data within each ecoregion as independent groups of data to determine if one group tends to contain larger values (Helsel and Hirsch, 2002; USGS, 2002). The WSA and organic carbon database DOC data are independent because there is no natural structure in the order of observations across groups. A nonparametric statistical test is most appropriate since no assumptions regarding normality of the data are required. As noted by Helsel and Hirsch (2002), nonparametric tests are, in general, never worse than their parametric counterparts in their ability to detect departures from the null hypothesis, and may be better. These considerations led us to select the rank-sum test, a nonparametric procedure for determining whether data are significantly different between two independent groups. This test is also known as the Wilcoxon Rank-Sum Test or, alternatively, the Mann-Whitney U-Test.

In its most general form, the rank-sum test is a test to determine whether one group tends to produce larger observations than another group. It has as its null hypothesis:

H_0: Prob [x > y] = 0.5

where x are data from one group and y are from another group (the probability of an x value being higher than any given y value is one-half). The test is typically used to determine whether two groups come from the same population (same median and other percentiles), or alternatively whether they differ only in location (central value or median). If both groups of data are from the same population, about half of the time an observation from either group could be expected to be higher than that from the other, so the above null hypothesis applies. If the groups belong to different populations the null hypothesis does not apply.

In practice, the rank-sum test takes several forms, depending upon the size of the smaller sample (*n* observations) and the larger sample (*m* observations). Walpole and Myers (1978; Section 13.2) present the details of four alternative forms of the rank-sum test, depending on the sizes of *n* and *m*. The exact form of the rank-sum test is the only form appropriate for comparing groups of sample sizes of 10 or smaller per group. When both groups have samples sizes greater than 10 (*n*, *m* > 10), the large-sample approximation may be used.

4.3.2.2 Rank-Sum Test Comparing WSA DOC Data to NOCD

Table 17 presents the results of the rank-sum test comparing Level III ecoregional DOC data from the WSA (USEPA, 2006b) and the NOCD. The left-hand columns present statistics (sample size, median, and Mann-Whitney U_x, and U_y) for the ecoregion-specific DOC data from the two datasets. The next six columns to the right present the test statistics for the appropriate form of the rank-sum test. The right-hand column provides a summary interpretation of the test for each ecoregion indicating whether the null hypothesis (H_0: DOC concentrations from both datasets are not different) should be rejected at the 5% level of significance, in favor of the alternative hypothesis (H_1: DOC concentrations are higher in the

national organic carbon database). In other words, rejection of the null hypothesis implies that DOC concentrations from the National Organic Carbon Database are biased high in that ecoregion relative to the WSA data.

Table 17. Results of rank-sum test comparing Level III ecoregional DOC data from WSA and NOCD

Level III Ecoregion	Ecoregion Name	WSA dataset			NOC database			M-W test (n1>10 and n2>10)		One large sample (n2>20)		Exact test (n2<20)	Exact test (n2<9)	Interpretation of test 1-sided @ 0.05
		n	median DOC	Ux	n	median DOC	Uy	Z	P	Z	P	critical U (0.05 level of signif.)	P	(H0: same mean of distributions)
1	Coast Range	38	1.41	2535.5	91	2.2	922.5	4.17	1.54E-05					reject H0
2	Puget Lowland	3	0.96	2289	835	6.6	216			-2.48	6.63E-03			reject H0
3	Willamette Valley	2	2.47	84	66	2.9	48			-0.65	2.57E-01			do not reject H0
4	Cascades	23	0.75	1673	100	1.4	627	3.39	3.46E-04					reject H0
5	Sierra Nevada	14	0.91	448	32	3.6	0	5.35	5.96E-08					reject H0
6	Southern and Central California Chaparral and Oak Woodlands	32	2.5	12372	479	4.6	2957	5.82	0					reject H0
7	Central California Valley	2	4.2	302	180	13	58			-1.65	4.98E-02			reject H0 (P~5%)
8	Southern California Mountains	41	1.66	245	6	8.9	1			-3.89	5.03E-05			reject H0
9	Eastern Cascades Slopes and Foothills	19	0.97	209	13	2.3	38	3.28	5.18E-04					reject H0
10	Columbia Plateau	8	2.59	477	73	3.6	107			-2.93	1.70E-03			reject H0
11	Blue Mountains	77	1.59	1625	26	3.1	376.5	4.74	1.07E-06					reject H0
12	Snake River Plain													no test

Level III Ecoregion	Ecoregion Name	WSA dataset			NOC database			M-W test (n1>10 and n2>10)		One large sample (n2>20)		Exact test (n2<20)	Exact test (n2<9)	Interpretation of test 1-sided @ 0.05
		n	median DOC	Ux	n	median DOC	Uy	Z	P	Z	P	critical U (0.05 level of signif.)	P	(H0: same mean of distributions)
13	Central Basin and Range	42	1.93	47300	1553	3	17926	4.986	2.98E-07					reject H0
14	Mojave Basin and Range	2	2.65	67	35	5.6	3			-2.15	1.58E-02			reject H0 (P~1.6%)
15	Northern Rockies	19	1.54	9149	778	1.8	5634	1.77	3.81E-02					reject H0 (P~3.8%)
16	Idaho Batholith	19	1.21	495	29	2.4	56	4.63	1.85E-06					reject H0
17	Middle Rockies	70	1.43	3142	81	1.9	2529	1.14	0.126					do not reject H0
18	Wyoming Basin	29	2.29	4006	150	7.3	344	7.17	0					reject H0
19	Wasatch and Uinta Mountains	25	2.11	978	46	4.8	172	4.85	5.96E-07					reject H0
20	Colorado Plateaus	24	2.22	15697	798	6.3	3455	5.34	5.96E-08					reject H0
21	Southern Rockies	43	2.05	18637	1129	1.3	29911	-2.59	4.83E-03					reject H0
22	Arizona/New Mexico Plateau	7	1.83	1723	281	5.6	244			-3.40	3.40E-04			reject H0
23	Arizona/New Mexico Mountains	31	1.94	984	37	5.3	163	5.02	2.38E-07					reject H0
24	Chihuahuan Deserts	1	1.48	110	116	6	6			-1.54	6.18E-02			do not reject H0
25	High Plains	6	3.5	2450	439	11	184.5			-3.62	1.48E-04			reject H0
26	Southwestern Tablelands	17	4.21	2246	167	6.3	593	3.95	3.90E-05					reject H0
27	Central Great Plains	12	4.71	2189	228	7	547	3.50	2.31E-04					reject H0
28	Flint Hills	2	4.91	19	10	8.7	1					1		reject H0

Level III Ecoregion	Ecoregion Name	WSA dataset			NOC database			M-W test (n1>10 and n2>10)		One large sample (n2>20)		Exact test (n2<20)	Exact test (n2<9)	Interpretation of test 1-sided @ 0.05
		n	median DOC	Ux	n	median DOC	Uy	Z	P	Z	P	critical U (0.05 level of signif.)	P	(H0: same mean of distributions)
29	Central Oklahoma/ Texas Plains	6	5.58	1098	289	6.7	636			-1.12	1.32E-01			do not reject H0
30	Edwards Plateau													no test
31	Southern Texas Plains													no test
32	Texas Blackland Prairies	2	7.37	880	829	6	778			-0.15	4.40E-01			do not reject H0
33	East Central Texas Plains	3	15.03	108.5	268	5	695.5			2.17	9.85E-01			do not reject H0
34	Western Gulf Coastal Plain	9	7.47	1860	399	7	1732			-0.183	0.427			do not reject H0
35	South Central Plains	24	7.7	6744	523	7.7	5808	0.62	2.68E-01					do not reject H0
36	Ouachita Mountains	6	2.2	963.5	196	3.7	212.5			-2.66	3.88E-03			reject H0
37	Arkansas Valley	3	4.61	327	184	7	225			-0.55	2.92E-01			do not reject H0
38	Boston Mountains	3	2.18	19	21	0.8	44			1.09	8.62E-01			do not reject H0
39	Ozark Highlands	10	1.8	2248	233	4.1	82			-4.98	3.26E-07			reject H0
40	Central Irregular Plains	8	6.7	1772	434	6.3	1700			-0.10	4.60E-01			do not reject H0
41	Canadian Rockies	5	0.8	127.5	36	1.1	52.5			-1.49	6.76E-02			do not reject H0
42	Northwestern Glaciated Plains	13	9.27	362	36	18	106	2.90	1.87E-03					reject H0
43	Northwestern Great Plains	81	7.45	43205	679	14	11794	8.41	0					reject H0

Level III Ecoregion	Ecoregion Name	WSA dataset			NOC database			M-W test (n1>10 and n2>10)		One large sample (n2>20)		Exact test (n2<20)	Exact test (n2<9)	Interpretation of test 1-sided @ 0.05
		n	median DOC	Ux	n	median DOC	Uy	Z	P	Z	P	critical U (0.05 level of signif.)	P	(H0: same mean of distributions)
44	Nebraska Sand Hills	1	6.46	0	4	3.2	4						0.2	do not reject H0
45	Piedmont	28	2.11	5379	308	3.4	3245	2.17	1.51E-02					reject H0 (P~1.5%)
46	Northern Glaciated Plains	17	14.62	1443	142	15	971	1.315	9.42E-02					do not reject H0
47	Western Corn Belt Plains	42	2.84	6200	193	5.1	1907	5.38	5.96E-08					reject H0
48	Lake Agassiz Plain	13	10.38	1962	261	10	1431	0.95	1.71E-01					reject H0 (P~1.7%)
49	Northern Minnesota Wetlands	1	11.71	37	44	17	7			-1.15	1.24E-01			do not reject H0
50	Northern Lakes and Forests	20	12.28	3099	403	8.1	4961	-1.74	4.05E-02					reject H0 (P~4%)
51	North Central Hardwood Forests	7	8.08	573.5	152	9.2	490.5			-0.35	3.64E-01			do not reject H0
52	Driftless Area	11	2.4	509	49	7.6	30	4.58	2.38E-06					reject H0
53	Southeastern Wisconsin Till Plains	5	3	2067	439	8	128.5			-3.40	3.41E-04			reject H0
54	Central Corn Belt Plains	9	2.69	1498	202	6.1	320			-3.29	5.07E-04			reject H0
55	Eastern Corn Belt Plains	6	2.83	7199	1325	6.8	751			-3.43	3.00E-04			reject H0

Level III Ecoregion	Ecoregion Name	WSA dataset			NOC database			M-W test (n1>10 and n2>10)		One large sample (n2>20)		Exact test (n2<20)	Exact test (n2<9)	Interpretation of test 1-sided @ 0.05
		n	median DOC	Ux	n	median DOC	Uy	Z	P	Z	P	critical U (0.05 level of signif.)	P	(H0: same mean of distributions)
56	Southern Michigan/Northern Indiana Drift Plains	9	4.62	1967	287	7	616			-2.67	3.77E-03			reject H0
57	Huron/Erie Lake Plains	3	5.05	8246	3762	7.1	3040			-1.38	8.33E-02			do not reject H0
58	Northeastern Highlands	23	3.54	79049	14044	1.4854	2E+05	-4.24	1.13E-05					reject H0
59	Northeastern Coastal Zone	10	4.01	561	101	4.2	449			-0.58	2.82E-01			do not reject H0
60	Northern Appalachian Plateau and Uplands	5	3.74	831	354	3.2	939			0.23	5.93E-01			do not reject H0
61	Erie Drift Plain	9	2.99	7834	901	6.2	275.5			-4.82	7.32E-07			reject H0
62	North Central Appalachians	4	3.34	102	106	1.7	322			1.76	9.60E-01			do not reject H0
63	Middle Atlantic Coastal Plain	2	18.54	16060	16726	3.4	17392			0.10	5.39E-01			do not reject H0
64	Northern Piedmont	15	2.18	18010	1524	4	4850	3.84	6.11E-05					reject H0
65	Southeastern Plains	18	2.55	49987	3801	4.3	18432	3.38	3.61E-04					reject H0
66	Blue Ridge	16	1.09	4915	686	0.9	6061	0.71	2.37E-01					do not reject H0
67	Ridge and Valley	27	1.56	10612	733	1.7	9180	0.64	2.61E-01					do not reject H0
68	Southwestern Appalachians	9	1.91	212.5	47	1.7	210.5			-0.02	4.91E-01			do not reject H0

Level III Ecoregion	Ecoregion Name	WSA dataset			NOC database			M-W test (n1>10 and n2>10)		One large sample (n2>20)		Exact test (n2<20)	Exact test (n2<9)	Interpretation of test 1-sided @ 0.05
		n	median DOC	Ux	n	median DOC	Uy	Z	P	Z	P	critical U (0.05 level of signif.)	P	(H0: same mean of distributions)
69	Central Appalachians	10	1.84	3651	864	1.6	4990			0.84	8.01E-01			do not reject H0
70	Western Allegheny Plateau	19	2.17	25890	1735	4	7075	4.28	9.18E-06					reject H0
71	Interior Plateau	14	2.24	2153	559	0.4	5674	-2.88	2.00E-03					reject H0
72	Interior River Valleys and Hills	14	3.86	3143	328	6.2	1449	2.34	9.70E-03					reject H0 (P~0.97%)
73	Mississippi Alluvial Plain	10	8.31	1646	503	5.5	3384			1.87	9.69E-01			do not reject H0
74	Mississippi Valley Loess Plains	1	1.26	21	21	5.4	0			-1.66	4.90E-02			reject H0 (P~4.9%)
75	Southern Coastal Plain	6	6.7	20141	4222	15.5	5191			-2.50	6.18E-03			reject H0
76	Southern Florida Coastal Plain													no test
77	North Cascades	54	0.82	1169	50	0.7	1531	1.18	1.19E-01					do not reject H0
78	Klamath Mountains	43	0.77	343	8	2.6	1			-4.43	4.74E-06			reject H0
79	Madrean Archipelago	3	1	27	9	7.9	0					3		reject H0
80	Northern Basin and Range	26	1.51	372.5	16	3.2	43.5	4.26	1.02E-05					reject H0
81	Sonoran Basin and Range	3	1.72	364	133	5.1	35			-2.44	7.40E-03			reject H0
82	Laurentian Plains and Hills	5	5.68	94	21	9.9	11			-2.70	3.47E-03			reject H0

Level III Ecoregion	Ecoregion Name	WSA dataset			NOC database			M-W test (n1>10 and n2>10)		One large sample (n2>20)		Exact test (n2<20)	Exact test (n2<9)	Interpretation of test 1-sided @ 0.05
		n	median DOC	Ux	n	median DOC	Uy	Z	P	Z	P	critical U (0.05 level of signif.)	P	(H0: same mean of distributions)
83	Eastern Great Lakes and Hudson Lowlands	12	6.72	8228	1346	7	7925	0.11	4.55E-01					do not reject H0
84	Atlantic Coastal Pine Barrens	2	12.62	150	243	5.7	336			0.93	8.24E-01			do not reject H0

4.3.3 Results and Implications of Bias Testing

The results of the rank-sum test indicate that DOC concentrations from the NOCD are biased high (i.e., the null hypothesis was not rejected) in 52 of 81 (64%) of Level III ecoregions in which comparable data were available (no comparison was possible in four ecoregions). For those ecoregions where the null hypothesis was not rejected, BLM users can be confident that the lower percentile DOC concentrations listed in Table 16 are representative for that ecoregion.

For ecoregions where the null hypothesis was rejected, the result suggests that the DOC data from the national organic carbon database are from biased samples. Recall discussion of both database in Sections 4.3, 4.3.1, and 4.3.2 that WSA is a random design sampling that ensures unbiased site selection. Whereas the NOCD is more influenced by locations with known water quality impairments and reflect unequal sampling efforts potentially creating a bias. It is likely that the percentile DOC concentrations tabulated for those ecoregions in Table 16 also reflect this bias towards high concentrations. This was confirmed by comparing the probability distributions of DOC concentrations in the ecoregions where n and m were large (n, m > 30).

In large-sample ecoregions where the null hypothesis was rejected by the rank-sum test (Ecoregions 1, 6, 11, 13, 23, 43, and 47), the probability distributions also show that the DOC concentration percentiles are substantially different, with the NOCD showing higher values. An example of such a comparison of DOC probability distribution is shown in Figure 31. On the other hand, in such large-sample ecoregions where the null hypothesis was not rejected (Ecoregions 17, 21, and 77), the probability distributions show that the DOC concentration percentiles are comparable. An example of such a probability distribution is shown in Figure 32. In all cases where it was possible to compare the DOC probability distributions, the results of the rank-sum test were confirmed.

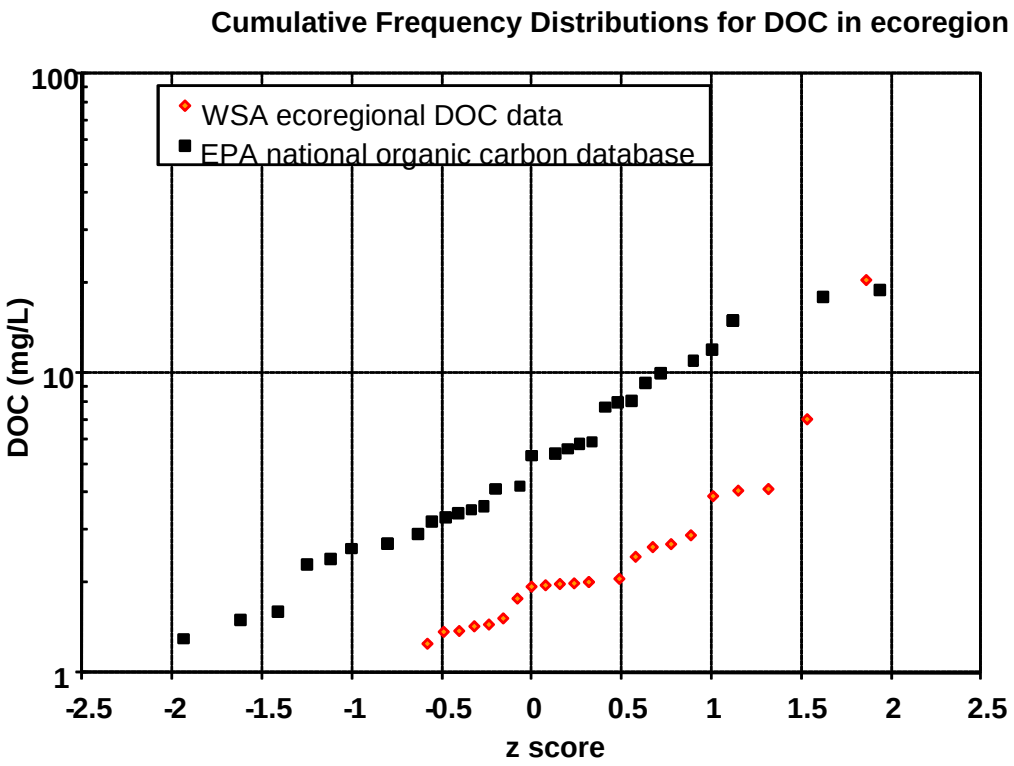

Figure 31. Comparison of probability distributions of DOC concentrations in Ecoregion 23

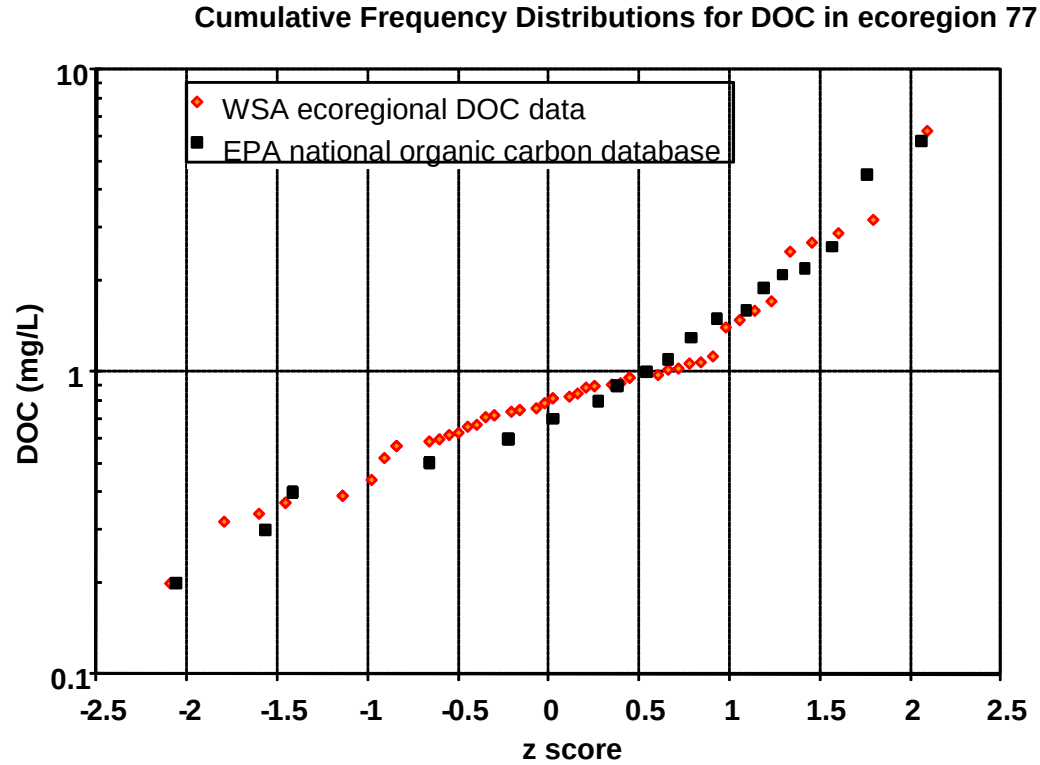

Figure 32. Comparison of probability distributions of DOC concentrations in Ecoregion 77

Because using a DOC concentration that is biased high as input to the BLM may lead to a non-conservative (high) site-specific copper criterion, it would be inappropriate to use the 10th percentile DOC concentrations in Table 16 for ecoregions in which the data from the NOCD come from biased samples.

We have not addressed the issue of whether the streams sampled for the WSA are representative in terms of DOC concentrations for all lotic (flowing) waters. It is possible that larger rivers may have DOC concentrations that are different from streams. For this reason, we recommend that the estimated ecoregional DOC values be compared to data from EPA's NRSA (NRSA, USEPA, 2013b; EPA 841-D-13-001). If necessary, adjustment of the estimated DOC values can be made at that time.

4.4 Comparing NOCD to WSA/NRSA DOC Data

The representativeness of the DOC data in the NOCD was evaluated by statistically comparing the data, at the ecoregional level, with the combined DOC data from two smaller random statistical surveys of rivers and streams. The two smaller surveys were:

(1) the 2004-05 WSA (1,313 sites), and

(2) the 2008-09 NRSA (2,113 sites).

The NRSA was the first nationally consistent survey assessing the ecological condition of the full range of flowing waters in the conterminous U.S. The target population includes the Great Rivers (such as the Mississippi and the Missouri), small perennial streams, and urban and non-urban rivers. Run-of-the-river ponds and pools are included, along with tidally influenced streams and rivers up to the leading edge of dilute sea water.

NRSA sampling locations were selected by random selection. The locations of perennial streams were identified using the EPA-USGS National Hydrography Dataset Plus (NHD-Plus), a comprehensive set of digital spatial data on surface waters at the 1:100,000 scale. Information about stream order was also obtained from the NHD-Plus. The 1,924 sites sampled for the NRSA were identified using a probability-based sample design. Details about the NRSA probabilistic sampling design are described in Section 1.1 of the NRSA: Field Operations Manual (USEPA, 2007; EPA-841-B-07-009). Site selection rules included weighting to provide balance in the number of river and stream sites from each of the size classes. Site selection was also controlled for spatial distribution to make sure sample sites were distributed across the U.S. Among these randomly selected sample sites were 359 of the original 2004 WSA sites. These were revisited as part of the NRSA to examine whether conditions have changed. When sites were selected for sampling, research teams conducted office evaluations and field reconnaissance to determine if the sites were accessible or if a river or stream labeled as perennial in NHD-Plus was, in fact, flowing during the sampling season. If a river or stream was not flowing or was determined to be inaccessible, it was dropped from the sampling effort and replaced with a perennial river or stream from a list of replacement sites within the random design.

The DOC data from these two smaller datasets were combined and described, hereafter, as WSA/NRSA data. GIS was used to determine which sampling sites were in each Level III ecoregion. The statistical test used was the non-parametric Wilcoxon 2-sample test, with a null hypothesis that DOC concentrations in the two different DOC sample datasets were equal. The alternative hypothesis was that DOC concentrations in the NCOD were significantly greater than those in the WSA/NRSA data,

indicating positive bias possibly due to over-representation of impacted sites. The test was applied for each of 84 Level III ecoregions at alpha=0.05.

Table 18 below includes the number of data (n) in each ecoregion, and the 10th percentiles of DOC based upon data from the EPA DOC database (NOCD) and the combined WSA/NRSA data. For each ecoregion, the table also provides the result of the Wilcoxon 2-sample test, in terms of whether the null hypothesis (the two samples are equal) should be accepted or rejected. The null hypothesis was rejected in 59 of 83 ecoregions, indicating bias in DOC concentrations higher in the national organic carbon dataset for the majority of ecoregions. In these 59 ecoregions, low-end percentiles based on DOC concentrations in the WSA/NRSA data were selected as reasonably protective estimates of ecoregional DOC concentrations.

Table 18. DOC concentrations (mg/L) in each Level III ecoregion based upon data from the NOCD and the combined WSA/NRSA data: number of data (n); 10th percentiles; and results of the Wilcoxon 2-sample test

Ecoregion	NOC database		WSA/NRSA database		H_0 (equal means)
	n	10%	n	10%	
1	91	1.1	60	0.7	reject
2	835	2.5	8	0.36	reject
3	66	1.1	12	0.4	reject
4	100	0.5	37	0.3	reject
5	32	2.1	21	0.5	reject
6	479	2.1	42	0.8	reject
7	180	2.7	7	1.1	reject
8	6	4.4	43	0.7	reject
9	13	1.4	25	0.5	reject
10	73	2.0	22	1.0	reject
11	26	1.3	91	0.8	reject
12	50	2.2	6	1.2	reject
13	1553	1.5	82	0.7	reject
14	35	2.8	8	0.8	reject
15	778	1.0	39	0.8	reject
16	29	1.4	34	0.8	reject
17	81	0.2	94	0.7	accept
18	150	4.3	52	1.1	reject
19	46	1.8	41	0.9	reject
20	798	3.0	61	1.2	reject
21	1129	0.6	76	0.8	accept
22	281	2.6	27	0.7	reject
23	37	2.2	48	0.7	reject
24	116	2.3	10	1.4	reject
25	439	4.4	29	1.3	reject

Ecoregion	NOC database		WSA/NRSA database		H_0 (equal means)
	n	10%	n	10%	
26	167	3.3	47	1.9	reject
27	228	3.8	92	2.2	reject
28	10	4.9	8	1.2	reject
29	289	3.8	30	1.7	reject
30	200	1.0	4	1.0	accept
31	58	1.0	4	0.3	accept
32	829	2.0	9	3.1	accept
33	268	3.0	5	5.6	accept
34	399	4.0	18	2.8	accept
35	523	4.6	66	3.3	reject
36	196	1.1	18	0.7	reject
37	184	1.9	18	1.4	reject
38	21	0.4	7	0.5	accept
39	233	2.3	39	0.8	reject
40	434	4.0	32	3.3	reject
41	36	0.6	7	0.6	reject
42	36	5.1	43	2.5	reject
43	679	6.2	234	2.3	reject
44	4	1.4	4	1.0	accept
45	308	1.0	93	1.1	reject
46	142	9.9	28	6.1	reject
47	193	3.1	103	1.7	reject
48	261	7.6	26	5.4	reject
49	44	11.0	9	6.0	accept
50	403	3.7	77	2.7	accept
51	152	3.1	44	3.2	reject
52	49	3.1	49	1.1	reject
53	439	5.3	12	1.9	reject
54	202	2.7	21	1.8	reject
55	1325	3.6	30	2.1	reject
56	287	3.8	38	2.9	reject
57	3762	4.7	14	1.5	accept
58	14044	0.6	92	1.2	accept
59	101	2.6	81	2.7	accept
60	354	1.3	29	1.4	reject
61	901	5.1	26	1.8	reject
62	106	0.7	22	0.9	accept
63	16726	2.2	45	1.7	accept
64	1524	1.3	47	1.0	reject
65	3801	2.4	108	1.4	reject
66	686	0.5	40	0.6	accept

Ecoregion	NOC database		WSA/NRSA database		H_0 (equal means)
	n	10%	n	10%	
67	733	0.6	88	0.9	accept
68	47	0.9	17	0.8	accept
69	864	0.7	31	1.1	accept
70	1735	1.5	67	1.5	reject
71	559	0.1	54	1.1	accept
72	328	2.7	65	2.2	reject
73	503	3.4	107	2.8	reject
74	21	1.7	18	1.2	accept
75	4222	8.0	41	3.6	reject
76	1	na	0	na	na
77	50	0.4	61	0.4	accept
78	8	1.7	56	0.6	reject
79	9	2.6	9	0.8	reject
80	16	1.8	49	1.0	reject
81	133	2.2	13	1.0	reject
82	21	5.5	18	2.8	reject
83	1346	1.0	32	2.6	reject
84	243	1.6	4	3.3	accept

In the 24 ecoregions where the null hypothesis was not rejected (i.e., no significant difference in DOC concentrations was found between datasets), the data were combined and the percentiles of the combined dataset were recalculated (Table 19). In these 24 ecoregions, low-end percentiles based on DOC concentrations in the combined data (NOCD and WSA/NRSA) were selected as reasonably protective estimates of ecoregional DOC concentrations.

Recommended DOC estimated values for 83 of the 84 ecoregions are summarized in Table 20. In the remaining ecoregion (76; Southern Florida Coastal Plain), there were insufficient data in either dataset (NOC database or WSA/NRSA) to calculate DOC concentration percentiles.

Table 19. DOC concentrations (mg/L) in 24 ecoregions where no significant difference in DOC concentrations was found between national organic carbon database (NOCD) and the WSA/NRSA datasets: number of data (n); 10th percentiles from combined NOCD & WSA/NRSA data

Ecoregion	n	DOC (mg/L) 10%
17	175	0.6
21	1205	0.6
30	204	1.0
31	62	1.0
32	838	2.0

Ecoregion	n	DOC (mg/L) 10%
33	273	3.0
34	417	4.0
38	28	0.5
44	8	1.0
49	53	10.4
50	480	3.5
57	3776	4.6
58	14136	0.6
59	182	2.7
62	128	0.7
63	16771	2.2
66	726	0.5
67	821	0.6
68	64	0.9
69	895	0.7
71	613	0.1
74	39	1.5
77	111	0.4
84	247	1.6

Table 20. Recommended ecoregional DOC concentrations (mg/L) based upon combined data from the NOCD and the WSA/NRSA data in 83 Level III ecoregions: number of observations (n); 10th percentiles; and source of data for each ecoregion

Ecoregion	n	DOC (mg/L) 10%	Data Source
1	60	0.7	WSA/NRSA
2	8	0.3	WSA/NRSA
3	12	0.4	WSA/NRSA
4	37	0.3	WSA/NRSA
5	21	0.5	WSA/NRSA
6	42	0.8	WSA/NRSA
7	7	1.1	WSA/NRSA
8	43	0.7	WSA/NRSA
9	25	0.5	WSA/NRSA
10	22	1.0	WSA/NRSA
11	91	0.8	WSA/NRSA
12	6	1.2	WSA/NRSA
13	82	0.7	WSA/NRSA
14	8	0.8	WSA/NRSA
15	39	0.8	WSA/NRSA

Ecoregion	n	DOC (mg/L) 10%	Data Source
16	34	0.8	WSA/NRSA
17	175	0.6	NOCD & WSA/NRSA
18	52	1.1	WSA/NRSA
19	41	0.9	WSA/NRSA
20	61	1.2	WSA/NRSA
21	1205	0.6	NOCD & WSA/NRSA
22	27	0.7	WSA/NRSA
23	48	0.7	WSA/NRSA
24	10	1.4	WSA/NRSA
25	29	1.3	WSA/NRSA
26	47	1.9	WSA/NRSA
27	92	2.2	WSA/NRSA
28	8	1.2	WSA/NRSA
29	30	1.7	WSA/NRSA
30	204	1.0	NOCD & WSA/NRSA
31	62	1.0	NOCD & WSA/NRSA
32	838	2.0	NOCD & WSA/NRSA
33	273	3.0	NOCD & WSA/NRSA
34	417	4.0	NOCD & WSA/NRSA
35	66	3.3	WSA/NRSA
36	18	0.7	WSA/NRSA
37	18	1.4	WSA/NRSA
38	28	0.5	NOCD & WSA/NRSA
39	39	0.8	WSA/NRSA
40	32	3.3	WSA/NRSA
41	7	0.6	WSA/NRSA
42	43	2.5	WSA/NRSA
43	234	2.3	WSA/NRSA
44	8	1.0	NOCD & WSA/NRSA
45	93	1.1	WSA/NRSA
46	28	6.1	WSA/NRSA
47	103	1.7	WSA/NRSA
48	26	5.4	WSA/NRSA
49	53	10.4	NOCD & WSA/NRSA
50	480	3.5	NOCD & WSA/NRSA
51	44	3.2	WSA/NRSA
52	49	1.1	WSA/NRSA
53	12	1.9	WSA/NRSA
54	21	1.8	WSA/NRSA
55	30	2.1	WSA/NRSA
56	38	2.9	WSA/NRSA
57	3776	4.6	NOCD & WSA/NRSA

Ecoregion	n	DOC (mg/L) 10%	Data Source
58	14136	0.6	NOCD & WSA/NRSA
59	182	2.7	NOCD & WSA/NRSA
60	29	1.4	WSA/NRSA
61	26	1.8	WSA/NRSA
62	128	0.7	NOCD & WSA/NRSA
63	16771	2.2	NOCD & WSA/NRSA
64	47	1.0	WSA/NRSA
65	108	1.4	WSA/NRSA
66	726	0.5	NOCD & WSA/NRSA
67	821	0.6	NOCD & WSA/NRSA
68	64	0.9	NOCD & WSA/NRSA
69	895	0.7	NOCD & WSA/NRSA
70	67	1.5	WSA/NRSA
71	613	0.1	NOCD & WSA/NRSA
72	65	2.2	WSA/NRSA
73	107	2.8	WSA/NRSA
74	39	1.5	NOCD & WSA/NRSA
75	41	3.6	WSA/NRSA
77	111	0.4	NOCD & WSA/NRSA
78	56	0.6	WSA/NRSA
79	9	0.8	WSA/NRSA
80	49	1.0	WSA/NRSA
81	13	1.0	WSA/NRSA
82	18	2.8	WSA/NRSA
83	32	2.6	WSA/NRSA
84	247	1.6	NOCD & WSA/NRSA

4.5 Conclusions

EPA tested the 10th percentiles of ecoregional DOC concentrations against data from the Southern Rocky Mountains (Level III Ecoregion 21) as input to the copper BLM. Broad ranges of errors (including some that were larger than an order-of magnitude) were observed in BLM predictions made with the DOC estimates, in comparison to predictions made with actual measured site data. Although the copper criteria values predicted using the parameter estimates for DOC were found to be protective in 90% of the cases, in many of these cases these predictions were overly-protective (e.g., IWQC lower by a factor of 4 to 5). For this reason, BLM users should be cautious when considering lower percentiles of the distribution of DOC as estimates for missing input parameters to the BLM. In general, it is preferable to use site-specific measurements of DOC as BLM input because: (1) copper toxicities (and BLM model predictions) are highly sensitive to DOC concentrations and (2) reasonably protective DOC concentrations can be difficult to estimate at the ecoregional level, when data are limited.

For many ecoregions, the EPA recommended percentiles in Table 20 are based upon a relatively small number of DOC data, which can be a cause for concern in terms of the reliability of these values. For

example, in 47 ecoregions the DOC percentiles were calculated from 50 or fewer concentration values, and in seven ecoregions the DOC percentiles were calculated from fewer than 10 values. In the former case (n≤50), the lower 95% confidence limit of the 10th percentile cannot be calculated (Berthouex and Brown, 1994), while in the latter (n<9) the 10th percentile itself is below the lowest concentration value. Because of these and other limitations on the DOC database and the importance of this parameter in criteria calculation, users are encouraged to sample for DOC as a basis for determining BLM input rather than using default parameters where possible.

5 SUMMARY AND RECOMMENDATIONS

The BLM predicts acute copper toxicity based on site-specific water quality parameters, and calculates aquatic life criteria based on the predicted copper toxicity. The BLM requires 10 input parameters to calculate copper criteria: temperature, pH, DOC, alkalinity, calcium, magnesium, sodium, potassium, sulfate, and chloride, the last seven of which are also referred to as GIs. Given the broad geographical range over which the BLM is likely to be applied, and the limited availability of data for input parameters in many areas, a practical method to estimate missing water quality parameters was developed to support the use of the copper BLM for copper aquatic life criteria.

In this report we described three approaches EPA used to estimate default input parameters for GI and DOC for BLM that could be used where site-specific data are not available. EPA's goal was to provide estimates for these missing input parameters that are reasonably protective. EPA used geostatistics to predict ecoregional input parameters from national water quality databases, and developed correlations between GI parameters and conductivity. These estimates were further refined using stream order.

Our analysis of national data indicates that there is no relationship between conductivity and pH, and geostatistical methods were found to produce similarly ambiguous results. Because pH is one of the most important BLM inputs for predicting criteria for copper, we conclude that site-specific data for pH are needed for successful BLM application. Temperature is a commonly measured parameter and should be easy to obtain by users for input in the BLM.

5.1 Recommendations for BLM inputs for geochemical ions where site-specific data are not available

In Section 2 we used geostatistics to estimate missing GI parameter values based on geography. We supplemented the geostatistical approach by adding conductivity as an additional explanatory variable to generate a more robust spatial estimate of the GI water quality inputs for the BLM because conductivity is one of the most widely monitored water quality indicators in the U.S. and correlates well with GIs. We presented average predicted 10th percentile concentrations for the BLM GI water quality parameters Level III ecoregions. We further refined these estimates by considering the effect of stream order (size) in Section 3. We found that values of the GI estimates generally increased with stream order, a trend that was most apparent and consistent for higher order streams. Tables 8, 9, and 10 present best estimates of GI input parameters for the BLM. Estimated inputs are provided for each GI in each ecoregion categorized by stream order for low, medium, and high order streams, respectively. EPA recommends these 10th percentile Level III ecoregion, stream order group-specific values be used in the BLM where site-specific data are not available.

5.2 Recommendations for BLM inputs for DOC where site-specific data are not available

In Section 4 we determined that the geostatistical and regression-based approaches used to estimate GI input parameters for the BLM do not produce accurate site-specific estimates for DOC. Because previous analyses indicate that DOC is the most important BLM input for estimating criteria for copper, we further refined our approach in Section 4 based on analyses using the NOCD to estimate lower-percentile DOC concentrations. Based on statistical comparisons to an independent probabilistic dataset, we found that DOC concentrations from the NOCD are reasonably protective estimates of DOC for use as input parameters for the BLM for some ecoregions. For other ecoregions, EPA recommends using estimates based on the WSA dataset. Recommended 10^{th} percentile DOC estimated values for 83 of the 84 ecoregions are summarized in Table 20. In the remaining ecoregion (76; Southern Florida Coastal Plain), there were insufficient data in either dataset (NOC database or WSA/NRSA) to calculate DOC concentration percentiles. Because limitations in the DOC database and the importance of this parameter in criteria calculation, users are encouraged to sample for DOC as a basis for determining BLM input rather than using default parameters wherever possible.

5.3 Recommendations for BLM inputs for pH where site-specific data are not available

In Section 2 we determined that geostatistical and regression-based approaches used to estimate GI input parameters for the BLM did not produce accurate site-specific estimates for pH. Our analysis of national data indicates that there is no relationship between conductivity and pH, and geostatistical methods were found to produce similarly ambiguous results. Because pH is one of the most important BLM inputs for predicting criteria for copper, we conclude that site-specific data for pH are needed for successful BLM application. Temperature along with pH is similarly recommended to acquire site-specific data for BLM application with the advantage of both of these been easy parameters to measure.

5.4 Conclusions

The approaches described in this TSD can be used to provide reasonable default values for input parameters in the BLM to derive protective freshwater aquatic life criteria for copper when data are lacking. These data could also be used to provide reasonable default values to fill in missing water quality input parameters in the application of other metal BLM models as well when data are lacking. Default recommended values for GI parameters are 10^{th} percentile ecoregional, stream-order specific values. Default recommended values for DOC are 10^{th} percentile ecoregional values. Both pH and temperature should be measured values when using the BLM. It should be noted that site-specific data are always preferable for use in the BLM and should be used to develop copper criteria via the BLM when possible. Users of the BLM are encouraged to sample their water body of interest, and to analyze the samples for the constituent (parameter) concentrations as a basis for determining BLM inputs where possible.

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Appendix A: An Examination of Spatial Trends in Surface Water Chemistry in the Continental United States: Implications for the Use of Default Values as Inputs to the Biotic Ligand Model for Prediction of Acute Metal Toxicity to Aquatic Organisms

Internal EPA Report (2006)
James N. Carleton
EPA, Office of Water, Office of Science & Technology.

A.1 Abstract

A large database of surface water chemistry monitoring data was examined to look for spatial trends in five chemical constituents that are key inputs to a model for predicting metal toxicity to aquatic organisms. Continuous prediction maps of concentrations were generated using various kriging techniques to interpolate between site-median values measured at several thousand separate locations throughout the continental United States (U.S.). Continuous concentration surfaces were then averaged over 8-digit Hydrologic Unit Code (HUC) polygons to produce block-averaged mean estimates of site-median concentrations. Pairwise comparisons indicated distinct trends between various HUC-averaged predicted constituents. The same analyses performed on data from 772 locations where all five constituents had been measured revealed similar relationships between monitored constituents. Principal components analyses performed on these data sets showed that 80 to 90% of the variance in both cases could be explained by a single component with loadings on three of the five constituents. The use of kriging to produce appropriate quantile maps for block-averaging is suggested as a possible approach for developing regional values to use as default model inputs, when site-specific monitoring data are lacking.

A.2 Background

The U.S. Environmental Protection Agency is planning in the near future to release proposed water quality criteria for copper (*Note: EPA's BLM-based Freshwater Copper Aquatic Life Ambient Water Quality Criteria document was released in 2007, EPA-822-R-07-001*). These criteria are unlike most water quality criteria in that acceptable (safe) concentrations for aquatic life support, rather than being defined as simple numerical values that apply everywhere, will be addressed through the use of a chemical speciation model – the Biotic Ligand Model (BLM) (EPA, 2003). The BLM calculates metal toxicity to aquatic organisms as a function of simultaneous concentrations of additional chemical constituents of water, for example other ions that can either complex with copper and render it biologically unavailable, or compete with copper for binding sites at the point of entry into a vulnerable organism (i.e. at the fish gill). While the BLM has the potential to improve the accuracy of metal ecotoxicity predictions, its use requires input concentrations of nine separate chemical constituents and water temperature. Of these nine chemical constituents (Alkalinity (alk), calcium (Ca), magnesium (Mg), sodium (Na), sulfate (SO42-), potassium (K), chloride (Cl), dissolved organic carbon (DOC), and pH), model-predicted toxicity is most sensitive to five: Ca, alk, pH, Na, and DOC. States or other entities wishing to use the BLM to assess compliance with the proposed criteria in specific waters, or to develop effluent permit limits, will therefore require monitoring information on a suite of chemical

constituents – information that is not always available. One possible way to deal with such missing information is to develop reasonably protective default values for these various model inputs, especially the five to which the BLM is most sensitive. Given that ambient surface water chemistry reflects, among other things, the influences of local soil types and land uses, it may make sense that any such defaults be developed on some kind of regional or local basis.

The exercise described in this report comprises a geospatial examination of a large amount of water chemistry monitoring data collected in recent years by the U.S. Geological Survey, and recorded in their National Water Information System (NWIS) database. The data includes monitoring information from several thousand separate surface water sampling locations throughout the U.S. (Figure A-1). The latitudes and longitudes of each sampling location are part of the data record. The primary objective of this analysis is to look for any obvious spatial trends in typical concentrations of the five most sensitive constituents, and to suggest procedures for making use of these trends to define regional default values for use as inputs to the BLM. For purposes of expediency, the geographic extent of this analysis is limited to the continental U.S.

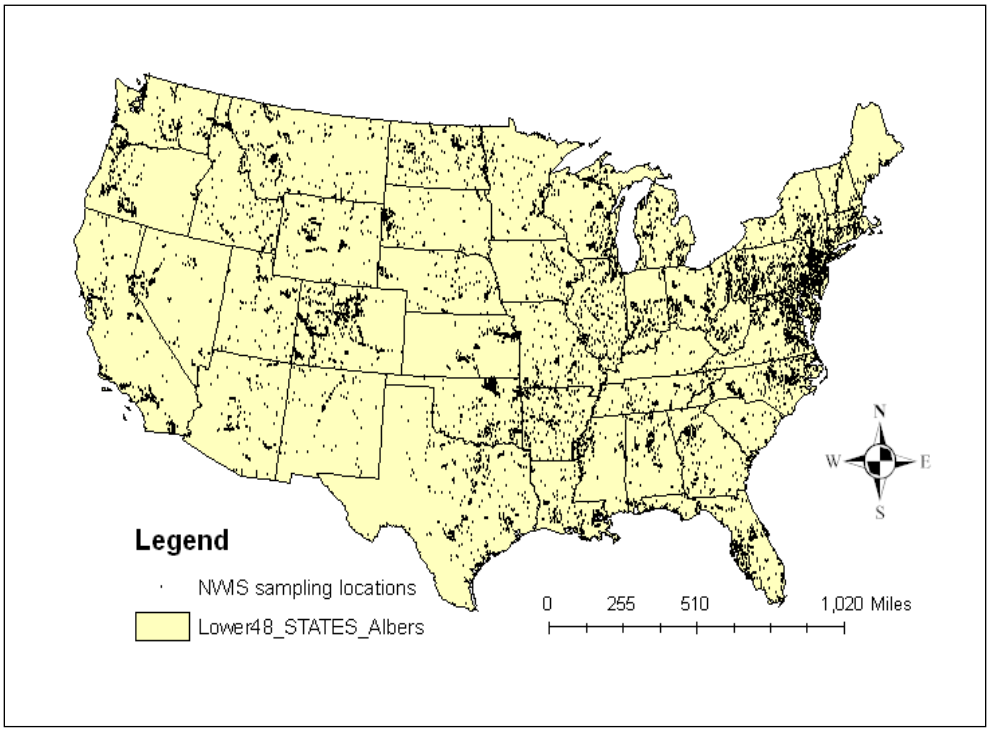

Figure A-1. NWIS sample collection locations in the continental U.S.

A.3 Description of Data

Although NWIS contained data from 207,153 sampling events at 13,824 individual sampling locations in the continental U.S. (Figure A-1), all 10 constituents of relevance to the BLM were not monitored at each location. For the five constituents of interest, the numbers of discrete sampling locations were as follows: alk, 5,900; Ca, 10,940; DOC, 3,726; Na, 10,424; pH, 11,780. Numbers of sampling events at individual locations ranged from 1 to 2,605, with a mean of 15, and a mode of one (i.e. most sites were

only sampled once). Examination of the spatial distribution of numbers of sampling events per site reveals that the most intensive sampling tended to occur in Midwestern and western states (Figure A-2). Because environmental sampling data tend to be lognormally distributed, disparities in numbers of samples may tend to produce higher mean and median values at more-frequently-sampled locations. As spatial distributions of representative (e.g., median) concentrations are examined, it should be kept in mind that apparent geographic trends in concentration may be in part simply the result of uneven sampling intensity.

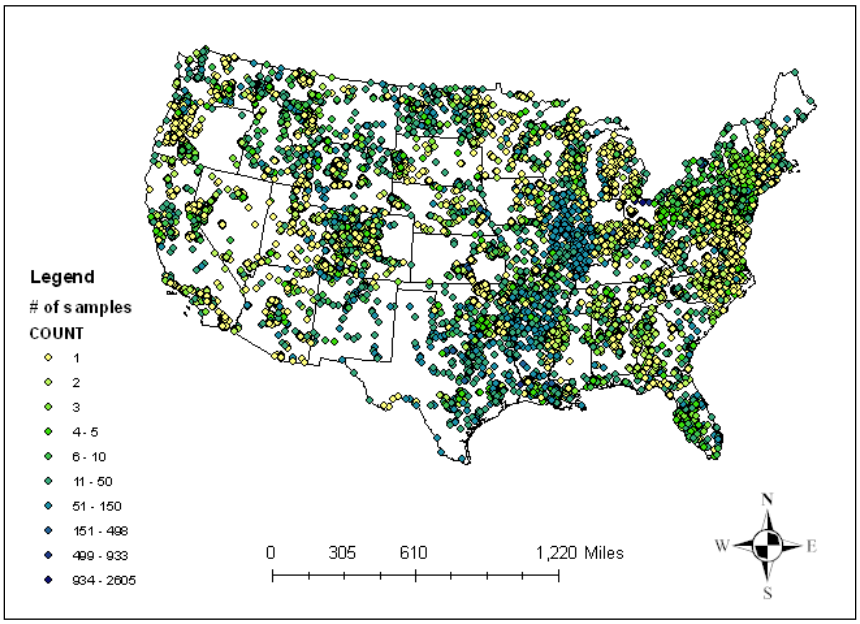

Figure A-2. Intensity of sampling (number of separate sampling dates) at each NWIS site

A.4 Data Analysis

Because environmental data tend to be positively skewed, the median statistic was chosen as providing the best central-tendency representation of each location's concentration. For the purpose of looking for general spatial trends in the five constituents, the first step involved simply mapping the sampling locations as points, color-graded by median concentration. Figure A-3, for example, shows some apparent trends in alkalinity across the country, with lower concentrations along the eastern seaboard, and higher concentrations in parts of the Midwest. Similar kinds of trends at the national scale were also seen with the other constituents.

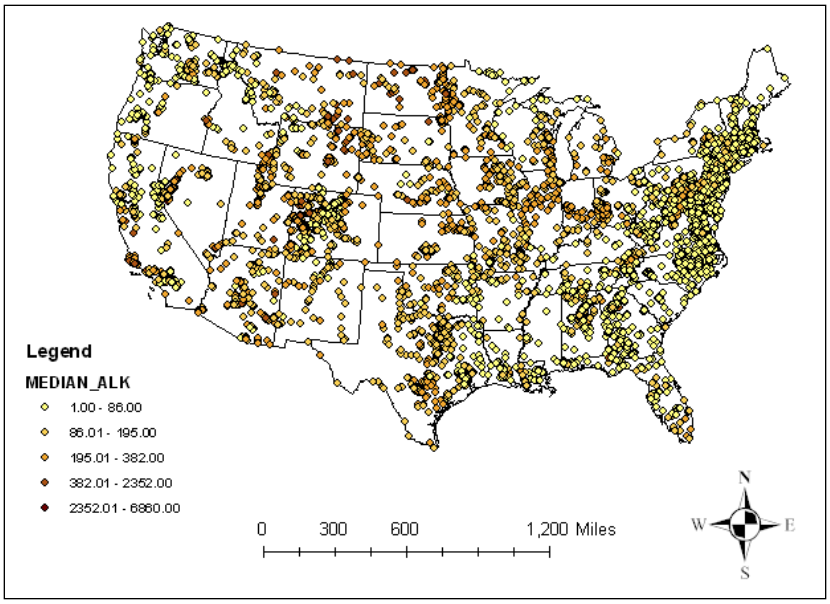

Figure A-3. Median measured alkalinity (mg/L as CaCO3) at NWIS locations

The next step in data visualization involved the calculation of median concentrations averaged over each 8-digit HUC containing sampling locations. These display essentially the same information as the point displays (Figure A-3), but with a degree of smoothing and summarization provided by the spatial averaging process, to make visual interpretation of general trends easier (Figure A-4).

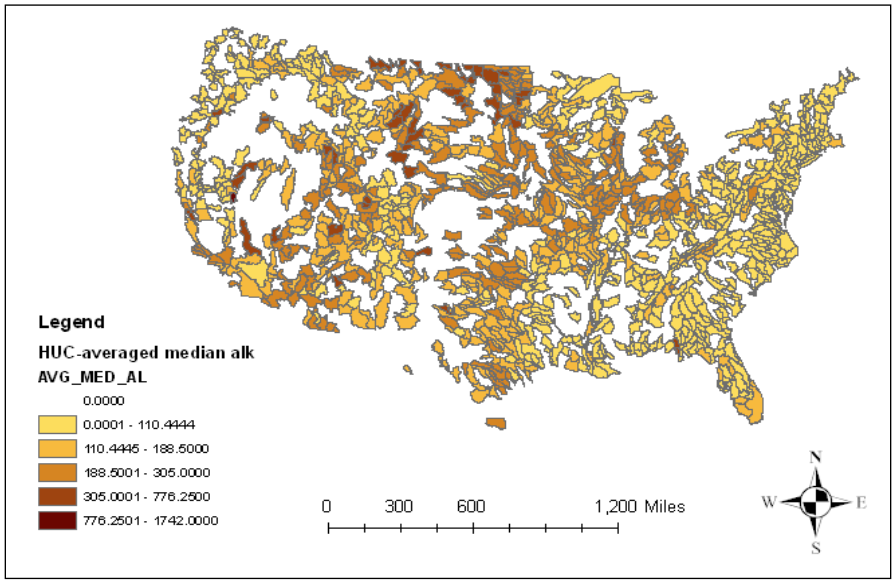

Figure A-4. HUC-averaged mean median observed alkalinity in the continental U.S.

The use of 8-digit HUCs as the areal units over which to calculate representative concentrations for default BLM inputs makes some physical sense: HUCs are areas that are defined by some degree of

interconnection between associated surface water features. HUCs may be either watersheds in their own right or downstream sections of larger watersheds (Omernik, 2003). In either case, all flowing surface water that passes through a HUC eventually (in theory) passes through the same downstream "pour point". One advantage of using HUCs is that they divide the land area into roughly equally sized areas at a level of resolution roughly consistent with gross variations in median concentration (Figure A-3). One problem with using HUCs for spatial aggregation is that not all HUCs contain NWIS sampling locations, as the blank areas in Figure A-4 make clear. The third step in this analysis therefore involved the use of kriging to create continuous surfaces of interpolated concentrations that cover the entire area of interest. Spatial averaging of the results over each HUC was then used to provide estimates of expected concentrations for all HUCs, including those lacking NWIS samples.

For each of the five key constituents, the Geostatistical Analyst extension in ArcGIS was used to explore the data, and to look for sets of kriging model options that provided the best fit to the data. The criteria used to evaluate goodness of fit were as follows:

1. Mean Standardized Error as small as possible
2. RMSE as small as possible
3. Root-Mean-Square Standardized Error close to 1.0
4. RMSE and Average Standard Error close together

Trial and error parameter selection was used to search for a set of model options that best attained each of these four goals simultaneously. For each constituent, 10 to 20 combinations were tried, until a best option for each emerged, as determined by judgment of the author. The results are as follows:

Alk: Universal kriging, log transformation, constant trend, 50% global, 50% local, spherical semivariogram, no anisotropy.

Ca: Ordinary kriging, log transformation, constant trend, 50% global, 50% local, exponential semivariogram, anisotropy.

DOC: Universal kriging, log transformation, constant trend, 50% global, 50% local, hole-effect semivariogram, anisotropy.

Na: Universal kriging, log transformation, constant trend, 50% global, 50% local, hole-effect semivariogram, anisotropy.

pH: Ordinary kriging, no transformation, constant trend, 50% global, 50% local, spherical semivariogram, no anisotropy.

Prediction surface maps were generated for each constituent using the above sets of kriging options. Figure A-5, which displays the results for alkalinity, shows patterns that are generally consistent with those in the data (Figures A-3 and A-4). Figure A-6 shows the predicted alkalinities projected into three dimensions using ArcScene.

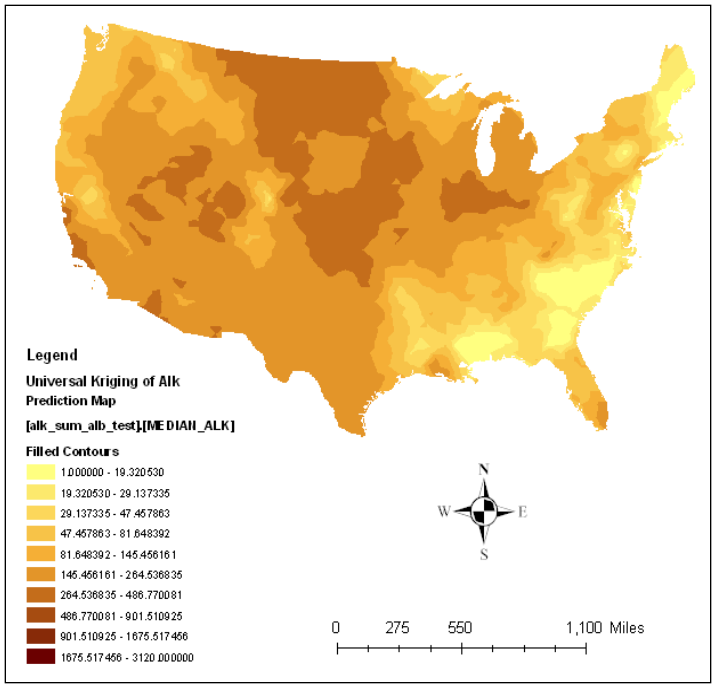

Figure A-5. Kriging prediction map of median alkalinity

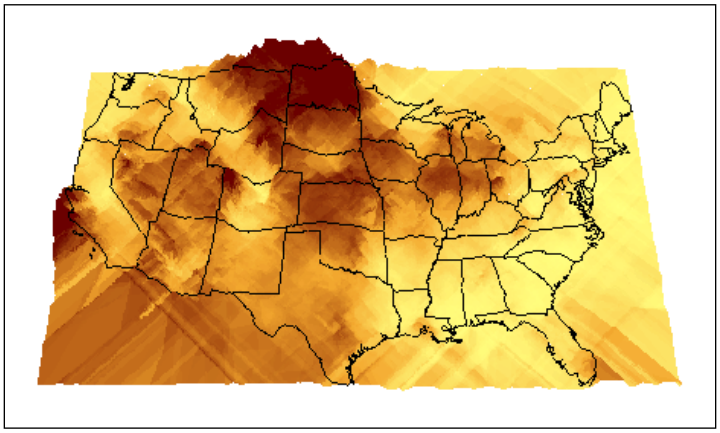

Figure A-6. Kriging map of alkalinity, projected into vertical dimension

This technique demonstrates broad geographic trends most dramatically, for example emphasizing the fact that the highest alkalinities are apparently found in northern North Dakota and Montana. Figure A-7 shows the predicted values averaged over HUC polygons by using the Zonal Statistics function of ArcGIS Spatial Analyst.

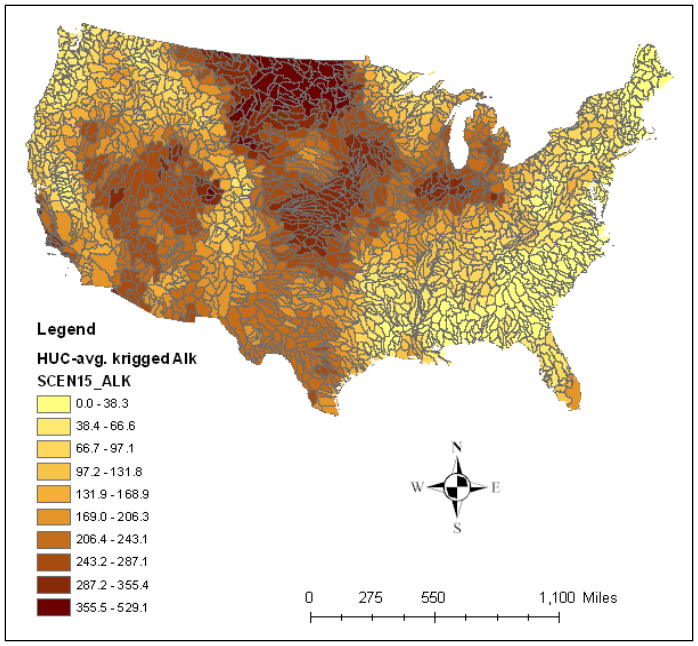

Figure A-7. Kriging-based alkalinity predictions, averaged over 8-digit HUC polygons

For HUCs containing NWIS sampling locations, linear regression plots of predicted versus measured concentrations (Figure A-8) provided a check on the accuracy of the kriging predictions. R-squared values for the five constituents were: 0.537 (alk), 0.238 (Ca), 0.686 (DOC), 0.351 (Na), and 0.139 (pH). In most cases, a handful of outliers appeared to be responsible for smaller-than-expected correlation coefficients.

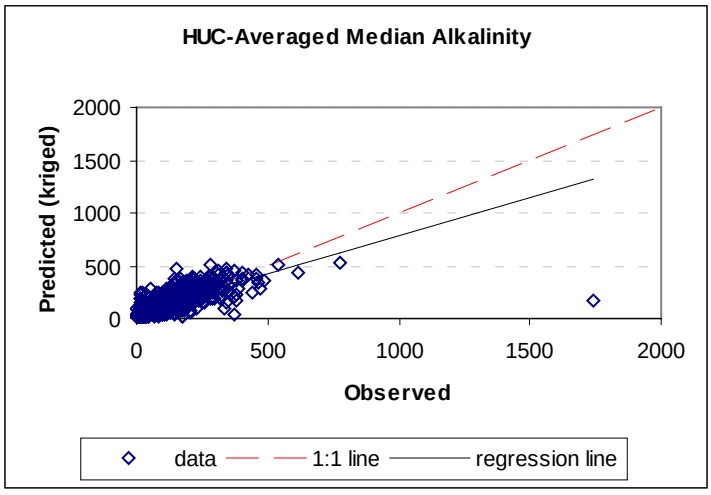

Figure A-8. Kriging-predicted vs. calculated HUC-averaged alkalinity; r^2=0.537

A scatter plot matrix of cross-constituent comparisons revealed some interesting, non-random relationships between HUC-averaged concentrations (Figure A-9). For comparative purposes, a subset of 772 sampling locations was also identified, at which sampling for all five of the constituents had taken place. Coincident concentrations of all constituents allowed a scatter plot matrix of this data (Figure A-10) to also be constructed. Similarities between the kinds of relationships in Figure A-9 and A-10 suggest that the predicted HUC mean median values are reasonable.

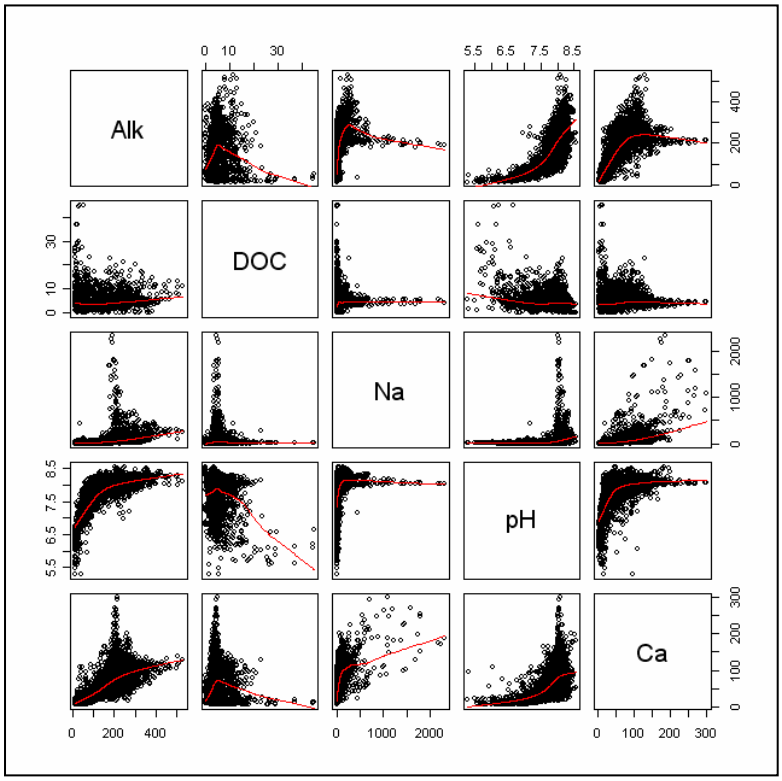

Figure A-9. Scatter plot matrix of median concentration kriged predictions, averaged over 8-digit HUCs regions covering the continental U.S.

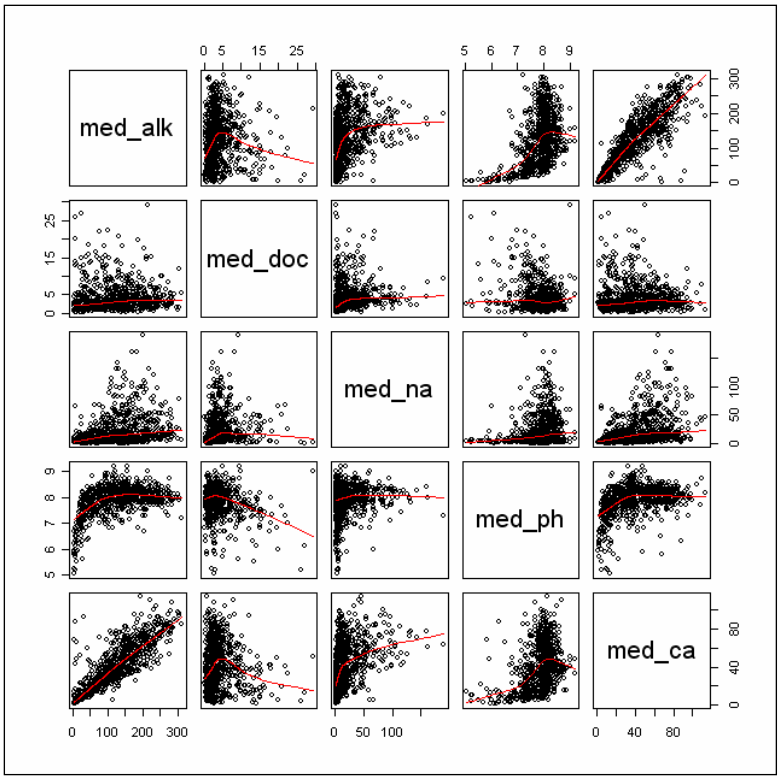

Figure A-10. Scatter plot matrix of median concentrations from 772 monitoring locations in the continental U.S.

In addition to scatter plots, correlation coefficient matrices between constituents in each of the two data sets (HUC-mean kriged median values and site median values for 772 locations) were generated (Table A-1). Although not identical, the coefficients were generally similar between the two datasets, again suggesting that the kriging predictions are reasonable.

Table A-1. Matrices of correlation coefficients between constituent concentrations

2096 HUC-averaged predicted median values					
	Alk	DOC	Na	pH	Ca
Alk	1				
DOC	-0.01456	1			
Na	0.327599	-0.02661	1		
pH	0.761675	-0.27746	0.286512	1	
Ca	0.698379	-0.02585	0.531727	0.58514	1
772 site-median values					
	Alk	DOC	Na	pH	Ca
Alk	1				
DOC	0.019145	1			
Na	0.327028	0.165445	1		
pH	0.453161	-0.24067	0.169238	1	
Ca	0.842484	-0.05097	0.387617	0.374592	1

Principal components analyses (PCA) were also run on both the HUC-averaged predictions and the 772 sets of monitored constituent concentrations to look for linear combinations of variables that might explain most of the observed variation. Figures A-11 and A-12 show the resulting plots of the variance explained by each component, and Table A-2 lists the loadings of the components onto the original variables. The first component comprised 80 and 88% of the variance in the HUC-based and site-based analyses, respectively. As Table A-2 indicates, this component loaded entirely onto alk, Na, and Ca in both cases. For the HUCs, component 1 was primarily loaded on Na, while for the sites, it primarily loaded on alk.

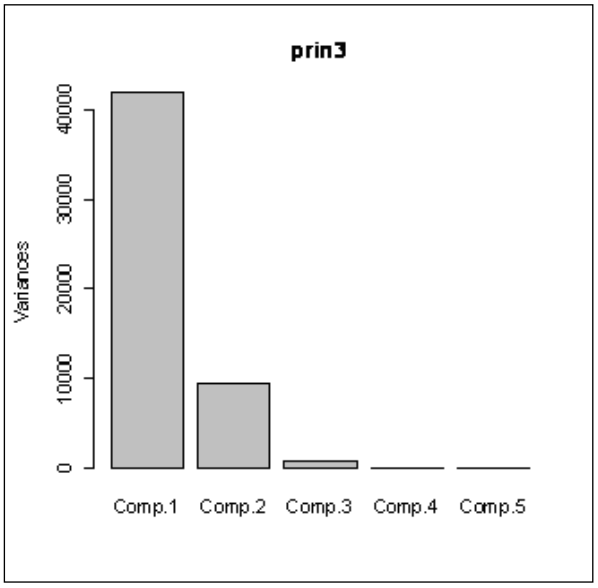

Figure A-11. Variance plot from PCA of HUC-average kriging-predicted concentrations

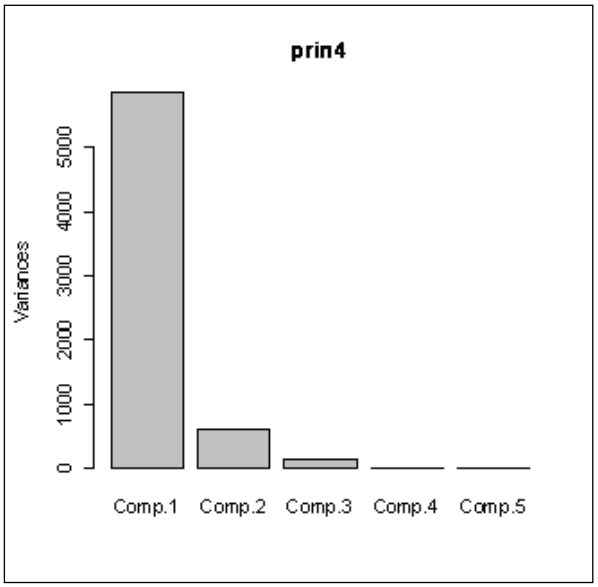

Figure A-12. Variance plot from PCA of site-median measured concentrations

Table A-2. Loadings onto original variables from PCA on HUC-averaged predictions and site-median concentrations

HUCs:					
	Comp.1	Comp.2	Comp.3	Comp.4	Comp.5
Alk	0.219	0.932	-0.289		
DOC				0.999	
Na	0.965	-0.25			
pH					-0.999
Ca	0.142	0.263	0.954		
Sites:					
	Comp.1	Comp.2	Comp.3	Comp.4	Comp.5
Alk	0.952	0.162	0.259		
DOC				-0.997	
Na	0.13	-0.982	0.133		
pH					0.999
Ca	0.277		-0.955		

A.5 Developing Regional Defaults

Besides prediction maps of best-estimate median concentrations, the Geostatistical Analyst can be used, with the same sets of kriging parameters listed previously, to generate quantile surface maps that represent reasonably protective inputs to the BLM than standard kriging predicted values. The five key inputs examined in this paper are all positively associated with BLM-predicted LC50s. Thus, lower values of all of them tend to result in lower (i.e., more protective) site-specific criteria. Lower quantile predictions can be used to produce protective regional default inputs. As an example, Figure A-13 displays the 25th percentile prediction map for alkalinity. When these values are block-averaged over the HUC polygons, the resulting alkalinities are lower than 67% of the site-minimum alkalinities (Figure A-14) measured inside the same areas.

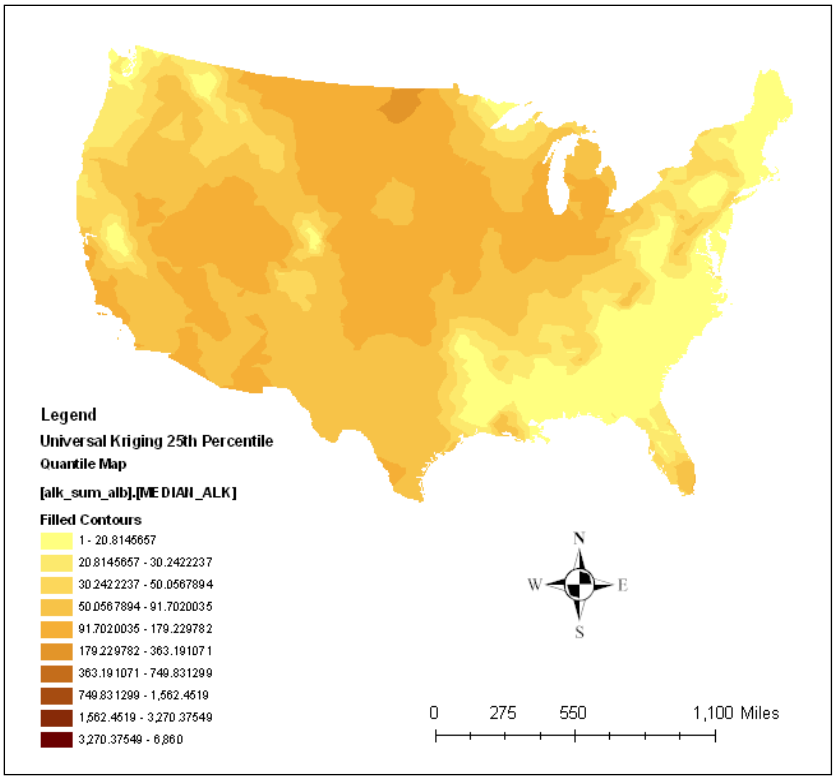

Figure A-13. Kriging 25th percentile map of median alkalinity

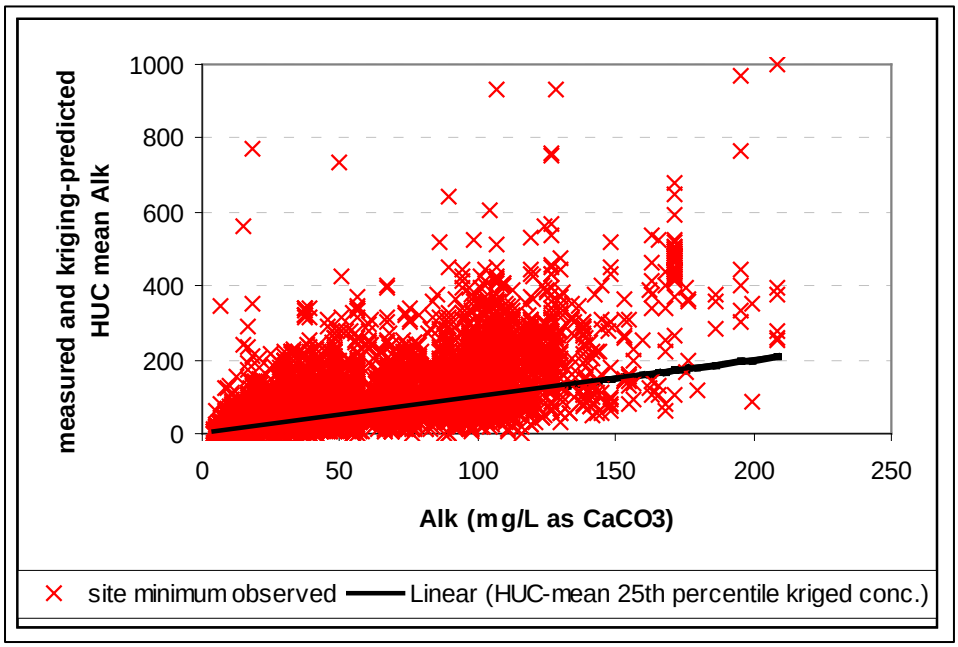

Figure A-14. Comparison of observed site-minimum alkalinities with HUC-mean 25th percentile kriging-predicted values

A.6 Discussion

The use of HUCs for spatial averaging of surface water concentrations is not without conceptual difficulties. First, only about 45% of HUCs are actual watersheds (Omernik, 2003); the rest receive drainage from additional upgradient areas. Concentrations measured in flowing waters reflect the soil, vegetation and land use properties of the aggregate upstream drainage areas, rather than of the sampling locations themselves (Smith et al., 1997). Assignment of measured concentrations to a HUC through block averaging may understate the spatial relevance of the samples for HUCs that are only parts of watersheds. One way to address this concern might be to use, as the aggregation polygons, only samples from watersheds that are entirely contained within single ecoregions (Omernik, 1987). However, this would have the unacceptable consequence of excluding large areas, and perhaps much of the data, from analysis. Another critical problem with this idea is that watershed boundaries for all of the NWIS sampling locations are not readily available, so there is currently no basis for deciding which points should be included or excluded. One advantage provided by the use of HUCs is that they divide the entire land mass of interest in this case into roughly equal sized polygons, at a level of resolution that appears to be roughly compatible with that of observed concentration trends. Block averaging using other sets of similarly sized polygons, such as counties, might serve equally well for empirically capturing broad spatial variability in concentrations. However the resulting concentrations would be less useful because they would lack even the incomplete degree of organization by connected hydrology that HUCs provide.

A.7 Conclusions

Kriging-predicted median concentrations of five water quality constituents, averaged over 8-digit HUCs, showed similar inter-constituent relationships as median concentrations from 772 specific sampling locations. PCA analyses revealed that in both cases, most of the observed variability was related to variations in three of the five constituents: alk, Na, and Ca. Results suggest that block averaging of kriging predictions over irregularly spaced sampling points can provide estimates that preserve much of the interrelationships between different measured entities. The use of suitable low-quantile kriging predictions is suggested as a way to estimate reasonably protective concentrations to serve as regional default inputs to the BLM.

A.8 References

Omernik, J., 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77:118-125.

Omernik, J., 2003. The misuse of hydrologic unit maps for extrapolation, reporting, and ecosystem management. Journal of the American Water Resources Association 39(3):563-573.

Smith, R.A., Schwartz, G.E., and R.B. Alexander, 1997. Regional interpretation of water-quality monitoring data. Water Resources Research 33912):2781-2798.

U.S. Environmental Protection Agency, 2003. The Biotic Ligand Model: Technical Support Document for its Application to the Evaluation of Water Quality Criteria for Copper, EPA 822-R-03-027.

Appendix B: Approaches for Estimating Missing Biotic Ligand Model Input Parameters. Correlation approaches to estimate Biotic Ligand Model input parameters using conductivity and discharge as explanatory variables

B.1 Introduction

Derivation of water quality criteria for copper and other metals from predictions of bioavailability generated by the Biotic Ligand Model (BLM) introduces a number of issues. For example, obtaining the data needed to apply the BLM may be problematic for many dischargers and receiving waters. The BLM requires 10 input parameters to characterize water quality at a particular site; the most important ones for predicting copper bioavailability and toxicity include pH, dissolved organic carbon (DOC), calcium, magnesium, sodium, alkalinity, and temperature. In stream segments with only small dischargers, or possibly no dischargers at all, the data needed to apply the BLM may not be available. Water quality criteria that rely upon BLM predictions would be greatly facilitated by the development of practical approaches to estimate values for BLM water quality parameters, which could be applied when data for one or more of these parameters are missing at a site.

Given the broad geographical range over which the BLM is likely to be applied, potentially over the entire Nation, and the limited information that is available for many areas, a practical method to estimate missing water quality parameters is needed. The geostatistical methods employed by the U.S. Environmental Protection Agency (EPA) (Carleton, 2006) presented a viable system to estimate missing water quality parameters required by the BLM. The prototype work developed by Carleton applied kriging to predict average concentrations of alkalinity, DOC, sodium, pH and calcium over hydrologic units (8-digit Hydologgic Unit Codes [HUCs]), using the U.S. Geological Service (USGS) National Water Information Service (NWIS) as the source of spatial data. Comparison of measured concentrations with kriging predictions were encouraging for several of the BLM water quality parameters, although the errors and uncertainties associated with these predictions were not fully explored.

The geostatistical approach utilizes knowledge of spatial correlation to project values of a water quality parameter at sites where it has not been measured. The accuracy of these projections depends upon the availability of sufficient and spatially-proximate data for the specific parameter of interest. In addition, the seasonal and annual temporal variation in water quality must also be addressed in order to apply the BLM at a site. Water quality parameters often experience large changes during periods of snowmelt or intense rainfall. In many rivers and streams, the chemical composition and physical properties of water are following trends associated with increased land use in watersheds, water diversion for irrigation, regulation of river flow by dams, and other anthropogenic disturbances.

The acute BLM predicts an *instantaneous* acute copper criterion (i.e., a maximum short-term, non-toxic concentration of copper), which will vary according to changes in the water quality parameters. An appropriately protective copper criterion must therefore reflect the variability of water quality parameters at the site. In previous analyses we found that protective water quality criteria for copper

generally corresponded to approximately the 2.5th percentile of the distribution of instantaneous water quality criteria (IWQC) predicted by the BLM.[5] BLM criteria predictions made for a site using the corresponding percentiles (i.e., 2.5%) of the water quality parameter distributions will be a conservative approximation of this protective criterion. The sensitivity of criteria predictions to the most important BLM water quality inputs is proportional (sensitivity to DOC is ~100%[6], [H^+] is ~50%, calcium, magnesium and sodium is ~20%). Relevant site-specific water quality parameters will be values from the lower "tail" of the measured or estimated distributions.

There may be great value in supplementing the geostatistical approach with classical estimation methods, such as regression and correlation. Examination of the NWIS data used to develop the geostatistical approach suggests that two variables, discharge (flow rate) and conductivity, may be useful for estimating BLM input water quality parameters. The USGS maintains the most comprehensive routine water flow and water quality data for streams and rivers in the Nation. Discharge may be a relevant explanatory variable because the USGS measures or estimates flow on a daily basis for a large number of stream and river segments. Among water quality parameters, the data for conductivity are the most complete and cover the longest time period (Wang and Yin, 1997). The literature also indicates that conductivity is one of the most widely monitored water quality indicators in the U.S. In part, this is because conductivity measurements are usually included in automated multiparameter systems for monitoring changes in the quality of surface waters (Allen and Mancy, 1972).

Conductivity is useful as a general measure of stream water quality. Each stream tends to have a relatively constant range of conductivity that, once established, can be used as a baseline for comparison with regular conductivity measurements (USEPA, 1997). Conductivity in streams and rivers is affected primarily by the geology of the area through which the water flows. Streams that run through areas with granite bedrock tend to have lower conductivity because granite is composed of more inert materials that do not ionize (dissolve into ionic components) when washed into the water. On the other hand, streams that run through areas with clay soils tend to have higher conductivity because of the presence of materials that ionize when washed into the water. Ground water inflows can have the same effects depending on the bedrock they flow through.

Conductivity reflects the strength of major ions in water and is a good estimator of total dissolved solids (TDS). Linear relationships between conductivity and TDS have been developed for many USGS monitoring sites. Conductivity is also linearly related to the sum of cations (McCutcheon et al., 1993). In addition, conductivity measurements provide information about the total concentration of ionic species in a water sample (Tyson, 1988). Figure B-1 illustrates how conductivity relates to hardness and anion concentrations in a river that has a rather saline base flow maintained by irrigation drainage and groundwater inflows. The chemical characteristics of the base flow are generally constant but they are subject to seasonal dilution by runoff. Relationships between conductivity and chloride and sulfate concentrations are well defined. A similarly good association with hardness (calcium+magnesium) is

[5] This was the median for 17 sites; the range was 1 to 36%.

[6] 100% sensitivity implies that a model prediction (in this case, the criteria predicted by the BLM) varies in direct proportion to the change in the value of a specified input parameter.

indicated. Lines drawn by eye through the points for chloride and sulfate show slight curvature, but the departure from linearity is insignificant. It seems evident that a record of conductivity at this station could be used to compute the other chemical characteristics of the water with a good level of accuracy for major ions, except at high flow when the relationships would not be as well defined (Hem, 1985).

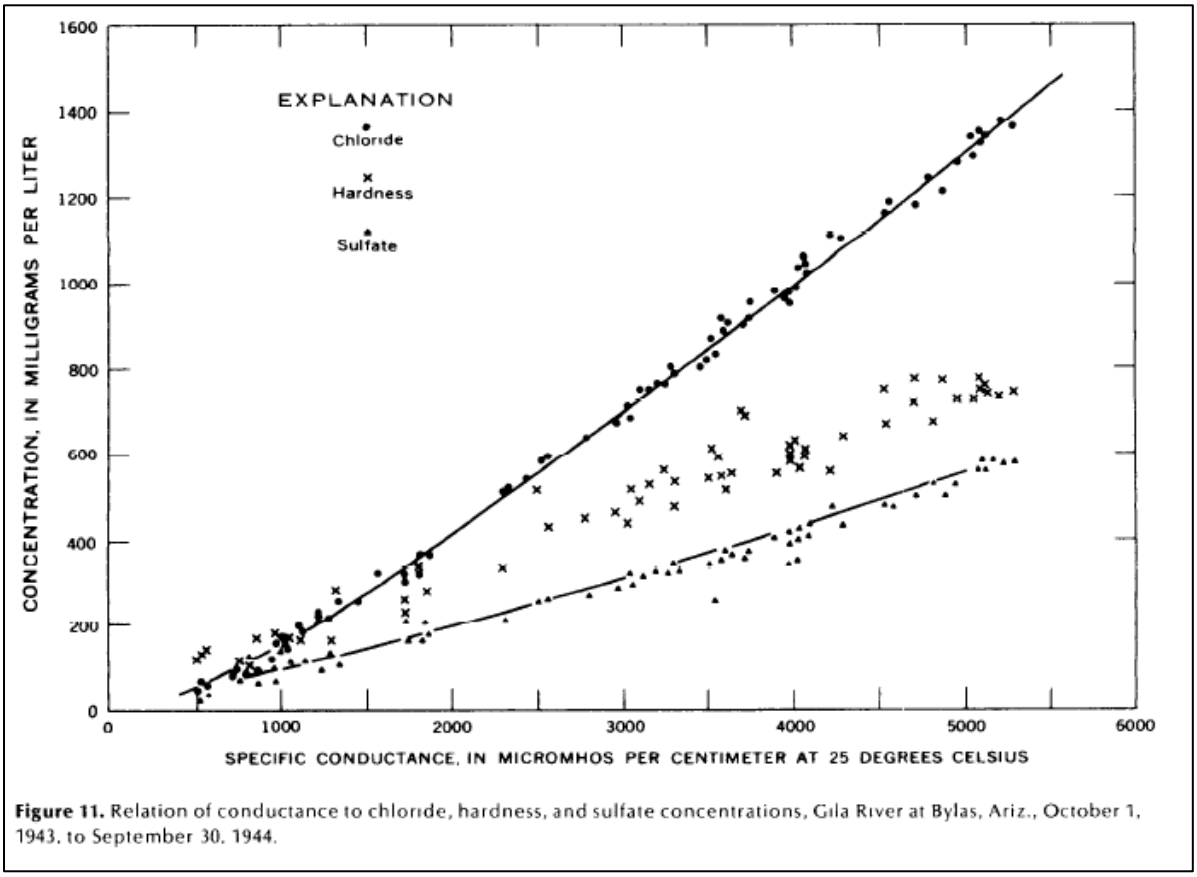

Figure 11. Relation of conductance to chloride, hardness, and sulfate concentrations, Gila River at Bylas, Ariz., October 1, 1943, to September 30, 1944.

Figure B-1. Relation of conductivity to chloride, hardness and sulfate concentrations in the Gila River at Bylas, Arizona

(reprinted from Hem, 1985)

Wang and Yin (1997) established conductivity as a general water quality indicator based on spatial data. The concentration of major base metal cations in water explained the positive correlation between conductivity and hardness. This also explained a rather weak correlation between conductivity and the pH value. Its relationship with other materials, however, most likely resulted from the dilution effect of stream flow. Conductivity was negatively correlated with discharge (ρ=-0.729), and the same was found for most water quality variables that were positively correlated with conductivity. With increasing stream flow, the concentration of the dissolved material decreased, as did the conductivity. Wang and Yin's analysis suggests that conductivity could be used as a general indicator of water quality, which is positively related to dissolved materials and soluble metals. As it is

widely monitored and has relatively long records, conductivity has the potential to be a very useful variable for estimating missing water quality input parameters for the BLM.

We explored this possibility by assessing the degree of correlation between conductivity and each of the BLM water quality parameters. We used NWIS data from three contiguous states in the Western U.S. (Colorado, Utah and Wyoming) for this analysis. These states were selected because of the large spatial and temporal variability observed in BLM water quality parameters, and because they provided us a tractable dataset for analysis.

Discharge was included as one of the parameters in this correlation analysis. However, discharge is most often used to explain water quality variation at a particular site (Hem, 1985). The concentration of dissolved solids in the water of a stream is related to many factors, but it seems obvious that one of the more direct and important factors is the volume of water from rainfall available for dilution and transport of weathering products. Presumably, therefore, the concentrations of dissolved solids should be an inverse function of the rate of discharge of water over all or at least most of the recorded range (Hem, 1985). Regressing water quality parameter measurements against discharge is a common practice in environmental engineering, and many references on this subject are available (McDiffett et al.,1989; Chanat and Hornberger, 2002; Christensen et al., 2005; Godsey and Kirchner, 2005). We should also point out that correlating the variation in water quality parameters to streamflow is also necessary for effluent dilution calculations associated with use of the BLM (for example, the probabilistic dilution framework incorporated in the BLM-Monte software [HydroQual, 2001]).

B.2 Data

Data for discharge, conductivity, and BLM water quality parameters (temperature, pH, DOC, alkalinity calcium, magnesium, sodium, potassium, sulfate, and chloride,) were retrieved from the USGS NWIS web interface (http://nwis.waterdata.usgs.gov/usa/nwis/qwdata). Data were selected for 790 stream and river stations in Colorado, Utah, and Wyoming reporting 100 or more water quality observations. This latter constraint was imposed to eliminate the large number of stations reporting very few (often one) water quality observations. Even when the analysis was restricted to sites with more than 100 water quality observations, there were frequently a marginal number of data for the multiple parameters needed to measure between-parameter correlations. We also restricted the analysis to observations made since 1975 to avoid the possible influence of pre-Clean Water Act discharges on water quality.

Natural logarithms of the discharge data were used in the analysis, because discharge was clearly lognormally distributed at the majority of sites. In cases where a parameter was measured simultaneously by more than one method (field pH versus laboratory pH, for example), the reported results were averaged for analysis. We did not consider other approaches for selecting data based on preference for a particular analytic method (Roberson et al., 1963).

B.3 Results

Table B-1 provides an inventory of the number of observations, and number of sites with data, for several of the parameters in the state of Colorado (these numbers reflect the full NWIS dataset, uncensored for minimum number of observations or date). From this table, it is apparent that a vast amount of conductivity data exists, both in terms of the total number of observations and the number of sites reporting this parameter in comparison to the BLM water quality parameters. For example,

there are almost four times as many observations of conductivity as there are for calcium, and they are measured at more than twice the number of sites. Discharge data is similarly abundant.

Table B-1. Number of observations and sites reported in NWIS for streams and rivers in Colorado

Parameter	Number of Observations	Number of Sites
pH	62,005	3668
alkalinity	8136	839
calcium	45,490	2708
conductivity	168,110	6101
discharge	127,275	3340

To quantify the relationship between conductivity levels and values of water quality parameters required by the BLM, we performed correlation analyses on the NWIS water quality data for the three states. We estimated correlations for several statistics that summarized the distribution of conductivity and water quality values at each station. These included median levels, as well as the first quartile and fifth quantile. The last two statistics represent the lower end of the distribution of parameter values at a site, and are appropriate statistics for calculation of BLM instantaneous criteria. A non-parametric correlation (Spearman's rank correlation) was employed to avoid the problems of unknown data distributions and possibly non-linear relationships. To determine the statistical significance of the rank correlation coefficient (ρ), the significance level (P) was also calculated. The Spearman's rank correlation was also used to examine the relationship between stream discharge and the water quality variables to reveal the effect of dilution.

For the median site concentrations, we found that six BLM water quality parameters, two-thirds of the nine variables examined in this study, had non-zero rank correlation coefficients at the 0.001 significance level (Table B-2). As expected, strong positive correlations between conductivity and salt concentrations were found. For example, the correlation coefficients between conductivity and the concentration of salt cations and anions (sodium, potassium, magnesium, calcium, sulfate and chloride) were all higher than 0.80. However, median site conductivity was not significantly correlated to several other important BLM parameters including pH, DOC, and alkalinity. In terms of the site medians, there appears to be limited correlation between conductivity and the BLM water quality parameters. Furthermore, for the median site concentrations neither conductivity nor any of the BLM water quality parameters were significantly correlated to discharge.

Table B-2. Results of Spearman rank tests for correlation (ρ) between median values of variables at each site.

Probability values (P) are not exact due to the presence of ties in the data

	Conductivity	Discharge
Conductivity		ρ: 0.012 P: 0.892
pH	ρ: 0.175 P: 0.019	ρ: 0.441 P: 0.008
DOC	ρ: 0.866 P: 0.333	ρ: P:
Ca	ρ: 0.867 P: <0.001	ρ: -0.371 P: 0.068
Mg	ρ: 0.882 P: <0.001	ρ: -0.516 P: 0.008
Na	ρ: 0.921 P: <0.001	ρ: 0.139 P: 0.695
K	ρ: 0.846 P: <0.001	ρ: -0.128 P: 0.551
SO_4	ρ: 0.905 P: <0.001	ρ: -0.514 P: 0.010
Alkalinity	ρ: -0.600 P: 0.350	ρ: P:
Cl	ρ: 0.827 P: <0.001	ρ: -0.866 P: 0.333

We then repeated the correlation analysis for the site first quartiles (Table B-3) and fifth quantiles (Table B-4). For both of these low-end distribution statistics, *all* of the BLM water quality parameters were significantly correlated to conductivity, having non-zero rank correlation coefficients at the 0.001 significance level, as listed in Tables B-3 and B-4. The correlation coefficients are lower for pH and DOC than for the salts and alkalinity, but are nevertheless significant. Apparently, the correlation structure between conductivity and the BLM water quality parameters is much stronger at the lower end of the site distributions. Ambiguity in correlations between conductivity and BLM water quality parameters disappears when low-end distribution statistics are analyzed. As was the case for the median site concentrations; neither conductivity nor any of the BLM water quality parameters were correlated with discharge for the low-end distribution statistics.

Table B-3. Results of Spearman rank tests for correlation (ρ) between the first quartile of values at each site.

Probability values (P) are not exact due to the presence of ties in the data

	Conductivity	Discharge
Conductivity		ρ: 0.057 P: 0.144
pH	ρ: 0.287 P: <0.001	ρ: 0.070 P: 0.168
DOC	ρ: 0.618 P: <0.001	ρ: -0.149 P: 0.031
Ca	ρ: 0.920 P: <0.001	ρ: -0.060 P: 0.305
Mg	ρ: 0.935 P: <0.001	ρ: -0.107 P: 0.066
Na	ρ: 0.910 P: <0.001	ρ: -0.075 P: 0.129
K	ρ: 0.773 P: <0.001	ρ: -0.109 P: 0.075
SO_4	ρ: 0.941 P: <0.001	ρ: -0.068 P: 0.247
Alkalinity	ρ: 0.829 P: <0.001	ρ: 0.099 P: 0.381
Cl	ρ: 0.752 P: <0.001	ρ: 0.004 P: 0.958

Table B-4. Results of Spearman rank tests for correlation (ρ) between the fifth quantile of values at each site.

Probability values (P) are not exact due to the presence of ties in the data

	Conductivity	Discharge
Conductivity		ρ: 0.056 P: 0.213
pH	ρ: 0.382 P: <0.001	ρ: 0.032 P: 0.579
DOC	ρ: 0.558 P: <0.001	ρ: -0.107 P: 0.134
Ca	ρ: 0.920 P: <0.001	ρ: 0.017 P: 0.791
Mg	ρ: 0.929 P: <0.001	ρ: -0.056 P: 0.383
Na	ρ: 0.845 P: <0.001	ρ: -0.089 P: 0.078

K	ρ: 0.694 P: <0.001	ρ: -0.065 P: 0.353
SO_4	ρ: 0.908 P: <0.001	ρ: 0.017 P: 0.790
Alkalinity	ρ: 0.784 P: <0.001	ρ: 0.184 P: 0.102
Cl	ρ: 0.706 P: <0.001	ρ: 0.034 P: 0.671

To further illustrate these correlations, scatter plot matrices (or SPLOMs) were prepared for the first quartiles (Figure B-2) and fifth quantiles (Figure B-3). SPLOMs show scatter plots for each combination of parameters, arrayed as a matrix, with parameters labeled along the borders of the plot. Histograms for each parameter are plotted on the main diagonal. The correlations between conductivity and each of the BLM water quality parameters are apparent by examining the second row (from the top) of scatter plots in Figures B-2 and B-3. Likewise, the lack of correlation between these parameters and discharge is apparent in the top row of the scatter plots in these same figures.

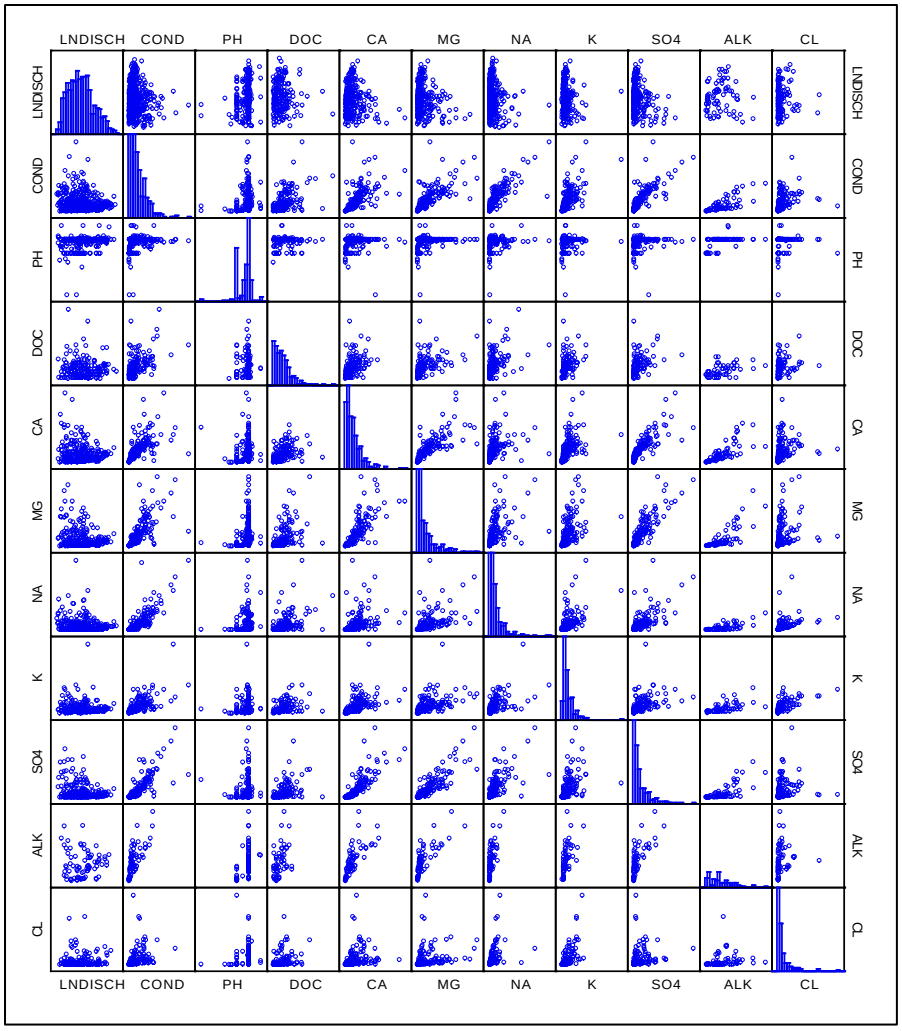

Figure B-2. Scatter plot matrix for first quartile of site-specific data for discharge (LNDISCH), conductivity (COND), and BLM water quality parameters

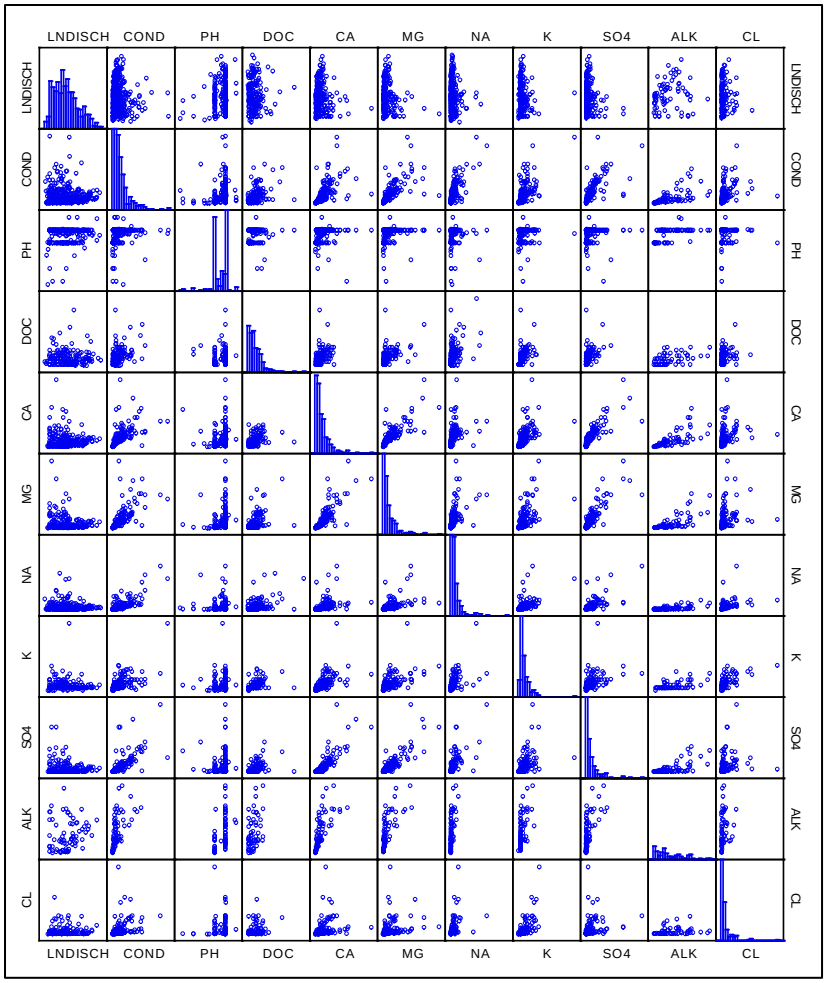

Figure B-3. Scatter plot matrix for fifth quantile of site-specific data for discharge (LNDISCH), conductivity (COND), and BLM water quality parameters

To understand why the correlations between conductivity and the other BLM water quality parameters are so much stronger for the low-end distribution statistics than for the medians, it is necessary to examine the site-specific data itself. Figure B-4 is a SPLOM of the conductivity, discharge, and BLM water quality parameter data for a representative USGS station in Colorado. The histograms for conductivity, salts, and alkalinity are remarkable in that the distribution of each is clearly bimodal (i.e., two separate peaks are evident in the histograms). This was observed for many of the sites in this dataset (not shown). In Figure B-5, the conductivity and discharge data for this site are plotted as a time series, which reveals why the water quality data are bimodal: high values of conductivity (> 5,000 micromhos/cm) occur when streamflow discharge is low, and low values of conductivity (< 2,000 micromhos/cm) occur when the discharge is high. At this station (and many others in this region), streamflow discharge is high in the May-June period coinciding with snowmelt at higher elevations. Feth and others (1964) reported conductivities of melted snow in the Western US ranging from about 2 to 42 picomhos/cm. Thus, the low values of conductivity (as well as concentrations of the salts and alkalinity) are the result of annual dilution from snow melt. At most other times, conductivity and salt and alkalinity concentration values are much higher. Depending upon how the water quality samples

are allocated at a site, the median concentration of these parameters may fall in either mode of the bimodal distribution, resulting in quite different values that appear almost random. Fortunately, the low-end distribution statistics avoid this seeming randomness because they consistently reflect sampling from the lower mode of the concentration distribution.

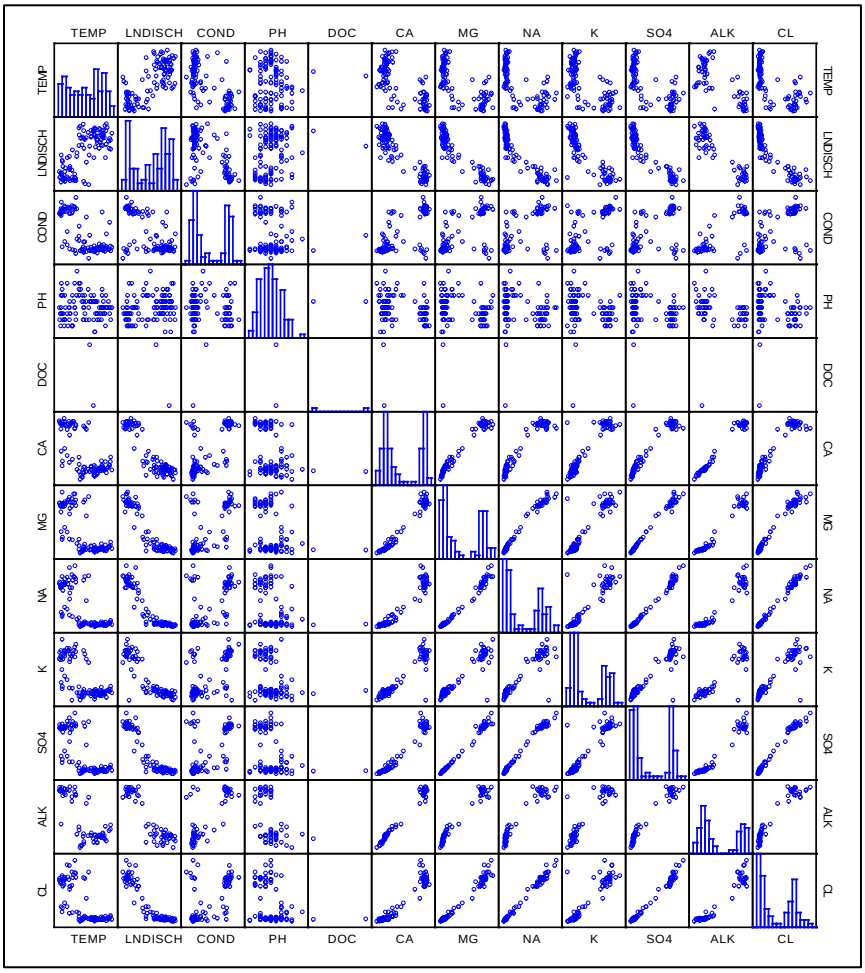

Figure B-4. Scatter plot matrix of BLM water quality parameter data from NWIS Station 384551107591901 (Sunflower Drain at Highway 92, near Read, Delta County, Colorado)

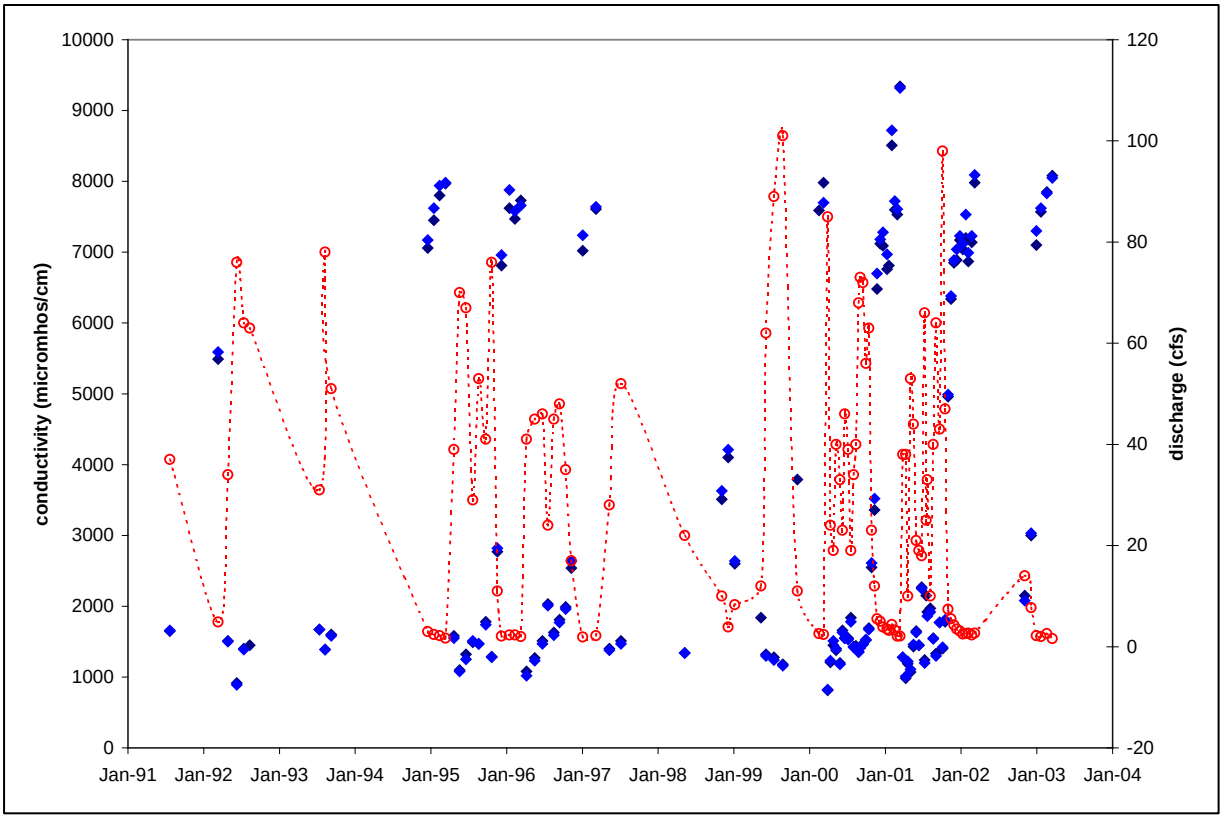

Figure B-5. Time series plot of conductivity (diamond symbols) and discharge (open circles connected by dashed line) at Station 384551107591901 (Sunflower Drain at Highway 92, near Read, Delta County, Colorado)

Figure B-4 also illustrates that, in terms of explaining site-specific variability, discharge is a much better predictive variable for a number of the BLM water quality parameters than conductivity. Each of these parameters (alkalinity calcium, magnesium, sodium, potassium, sulfate, and chloride) is clearly correlated to discharge, but not to conductivity. Discharge correlations are observed at many locations, and are commonly used to project water quality for various applications (Hem, 1985).

B.4 Discussion

Incorporating classical water quality correlation approaches, using conductivity and discharge as explanatory variables, within the geostatistical approach prototyped by EPA, appears promising. Conductivity, but not discharge, is significantly correlated to BLM water quality parameters between sites, especially for the low-end distribution statistics of interest for criteria calculations. Since conductivity data is abundant and it correlates well to BLM water quality parameters, it is reasonable to incorporate conductivity in spatial projections of BLM parameters. This may simplify the geostatistical approach and allow more robust spatial extrapolation of BLM water quality parameters.

Conversely, discharge is correlated to concentrations of a number of BLM parameters (salts and alkalinity) within many sites. Streamflow is a good explanatory variable for a number of the BLM water quality parameters (the salts and alkalinity) because their variabilities largely reflect dilution at high flow rates. Discharge data are also plentiful, so we believe that incorporating classical methods of

correlating concentration to discharge may be a useful means to address within-station variability for BLM water quality parameters.

It should also be recognized that geostatistical and/or correlation approaches appear to most often fail for those water quality parameters which are the most sensitive and important to the BLM, namely DOC and pH. Additional sampling effort will likely be required to address these deficiencies. In the case of pH, it is worth noting that many surface water sampling crews carry electronic multiparameter instruments which measure pH, conductivity, and temperature simultaneously in the field. Therefore, data collection strategies which incorporate these three measurements may be especially effective.

Measurement of DOC is considerably more difficult and expensive. It may be worthwhile to investigate whether ultraviolet (UV) absorption spectroscopy could be used as a surrogate measurement technique for DOC. The organic ligands that bind metals are humic and fulvic compounds (HydroQual, 2005). At least some of these compounds can be measured by UV absorption spectroscopy or related methods (Kalbitz et al., 2000; Wang and Hsieh, 2001), which may be easier and less expensive than DOC analysis.

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Appendix C: Development of Tools to Estimate Biotic Ligand Model Parameters

C.1 Introduction

The U.S. Environmental Protection Agency (EPA) explored using regression models that project BLM water quality parameters from conductivity data for sites where there may be few or no data available to characterize water. We demonstrated previously (USEPA, 2007) that conductivity (specific conductance) is significantly correlated to Biotic Ligand Model (BLM) water quality parameters between a large number of monitoring sites in three western states (Colorado, Utah, and Wyoming), especially for the low-end distribution statistics of interest for site-specific fixed water quality criteria calculations. Since conductivity data are also abundant, it is reasonable to incorporate conductivity in spatial projections of BLM parameters.

C.2 Regression Analysis

Water quality data were retrieved from the U.S. Geological Survey (USGS) National Water Information System (NWIS; http://waterdata.usgs.gov/nwis/qw). We focused our efforts on data collected from rivers and streams in the western states of Colorado, Utah, and Wyoming between 1984 and 2005. Data from these three states were selected because conductivity was known to vary substantially, and the legacy of past mining in the region made the contamination of waterbodies by trace metals a possibility. Data collected prior to 1984 was excluded because a number of the analytical methods used by USGS prior to that date have been replaced by methods with improved precision and lower detection limits. Furthermore, only sites with 40 or more samples were included in the analysis. Data were retrieved for all BLM water quality input parameters including pH, dissolved organic carbon (DOC) (or total organic carbon (TOC), if no DOC data were available) and the geochemical ions (GIs). We also retrieved discharge measurements and filtered (dissolved) copper concentration data, although these data were not included in the regression analysis.

In work described in Appendix B, we found that the correlation structure between conductivity and the BLM water quality parameters was much stronger at the lower end of the concentration distributions. For various low-end distribution statistics, all of the BLM water quality parameters were significantly correlated to conductivity, having non-zero rank correlation coefficients at the 0.001 significance level. The correlation coefficients for pH and DOC were lower than for the GIs, but were nevertheless significant. We exploited this feature of the data in our current work. For each site, we estimated the 10^{th} percentile (i.e., the value exceeded by 90% of the data) of conductivities and the 10^{th} percentile of BLM water quality parameter values. We then fit regression models to project 10^{th} percentiles of BLM parameter values as a function of 10^{th} percentiles of conductivities.

We also fit regression models to the full NWIS dataset (data for all rivers and streams sampled in Colorado, Utah, and Wyoming between 1984 and 2005). This was done out of concern that the lower percentile data might be skewed due to sampling bias, censoring, fewer sites, etc. The results of both approaches are presented below.

C.2.1 pH

The following regression model appeared to be optimum for projecting the 10^{th} percentile of pH from the 10^{th} percentile of conductivity at the sites for which appropriate data were available:

$$\ln(pH) = 1.85 + 0.0352 \cdot \ln(EC)$$

We did not fit a regression model to the full NWIS dataset, because no trend was evident between conductivity and pH.

C.2.2 DOC

The following regression model appeared to be optimum for projecting the 10th percentile of DOC concentrations from the 10th percentile of conductivity at the sites for which appropriate data were available:

$$\ln(DOC) = 0.671 \cdot \ln(EC) - 1.60$$

As with pH, we did not fit a regression model to the full NWIS dataset, since no trend was evident between conductivity and DOC.

C.2.3 Alkalinity

The following regression model appeared to be optimum for projecting the 10th percentile of alkalinity concentrations from the 10th percentile of conductivity at the sites for which appropriate data were available:

$$\ln(alkalinity) = 1.14 \cdot \ln(EC) - 4.68$$

For the full NWIS dataset, the following regression model was developed to project alkalinity concentrations from conductivity:

$$\ln(alkalinity) = 0.652 \cdot \ln(EC) + 0.530$$

C.2.4 Calcium

The following regression model appeared to be optimum for projecting the 10th percentile of calcium concentrations from the 10th percentile of conductivity at the sites for which appropriate data were available:

$$\ln(Ca) = 1.14 \cdot \ln(EC) - 4.35$$

For the full NWIS dataset, the following regression model was developed to project calcium concentrations from conductivity:

$$\ln(Ca) = 0.866 \cdot \ln(EC) - 1.51$$

C.2.5 Magnesium

The following regression model appeared to be optimum for projecting the 10th percentile of magnesium concentrations from the 10th percentile of conductivity at the sites for which appropriate data were available:

$$\ln(Mg) = 1.27 \cdot \ln(EC) - 4.81$$

For the full NWIS dataset, the following regression model was developed to project magnesium concentrations from conductivity:

$$\ln(Mg) = 0.986 \cdot \ln(EC) - 3.48$$

C.2.6 Sodium

The following regression model appeared to be optimum for projecting the 10th percentile of sodium concentrations from the 10th percentile of conductivity at the sites for which appropriate data were available:

$$\ln(Na) = 0.578 \cdot \ln(EC) - 2.62$$

For the full NWIS dataset, the following regression model was developed to project sodium concentrations from conductivity:

$$\ln(Na) = 1.32 \cdot \ln(EC) - 4.96$$

C.2.7 Potassium

The following regression model appeared to be optimum for projecting the 10th percentile of potassium concentrations from the 10th percentile of conductivity at the sites for which appropriate data were available:

$$\ln(K) = 0.882 \cdot \ln(EC) - 3.29$$

For the full NWIS dataset, the following regression model was developed to project potassium concentrations from conductivity:

$$\ln(K) = 0.647 \cdot \ln(EC) - 3.04$$

C.2.8 Sulfate

The following regression model appeared to be optimum for projecting the 10th percentile of sulfate concentrations from the 10th percentile of conductivity at the sites for which appropriate data were available:

$$\ln(SO_4) = 1.16 \cdot \ln(EC) - 4.85$$

For the full NWIS dataset, the following regression model was developed to project sulfate concentrations from conductivity:

$$\ln(SO_4) = 1.43 \cdot \ln(EC) - 4.47$$

C.2.9 Chloride

For the full NWIS dataset, the following regression model was developed to project chloride concentrations from conductivity:

$$\ln(\text{chloride}) = 1.39 \cdot \ln(EC) - 6.15$$

Unfortunately, there were an insufficient number of sites reporting chloride data for a regression model to be developed for the 10th percentile of chloride.

C.3 Application of Conductivity Regressions

There are a number of ways in which the conductivity regressions could be used to project BLM water quality inputs. However, the most important situation may be when a fixed copper criteria value must be calculated for a site where there may be little data available to characterize water quality. In such cases, the regressions allow some or all of the BLM water quality inputs to be projected from either (1)

a limited number of conductivity measurements or (2) a low-end conductivity value estimated by geostatistical or other methods. The first approach, projecting BLM water quality inputs from conductivity measurements, will be demonstrated in this section for a limited number of test sites. The second approach, projecting the BLM water quality inputs based on conductivities estimated by geostatistical methods, is demonstrated in the following section (Section C4).

The regression models presented above for projecting BLM water quality inputs from conductivity were tested using data and BLM predictions from a number of sites. For each site, a fixed copper criteria value was calculated using the Monte Carlo method described in EPA 2002. The BLM version 2.2.3 was used for all BLM calculations. Fixed copper criteria values were determined by the Monte Carlo method, utilizing site-specific data for parameter distributions and variance-covariance structure of all BLM water quality parameter inputs as well as filtered copper concentrations. The test sites (below) were selected on the basis of convenience, number of water quality observations, and geographic location.

The BLM water quality inputs projected from the conductivity regressions are low-end percentiles appropriate for predicting the instantaneous criterion (IC) predicted by the BLM to estimate the fixed site criteria (FSC) value. We suggested this approximation previously, based on the observation that protective FSC for copper generally corresponded to approximately the 2.5^{th} percentile of the distribution of IC predicted by the BLM. BLM estimates made for a site using the corresponding percentiles of the water quality parameter distributions will be a conservative approximation of this protective criteria values. For the present work, we are using this approach to test the 10^{th} percentile water quality parameter values projected from the conductivity regressions.

Previously we noted that filtered copper concentrations were correlated to BLM input water quality parameters at many sites. Furthermore, we found that the degree of correlation between copper concentrations and BLM input parameters appeared to be an important site-specific factor in determining the relationship between the FSC and the IC. Copper concentrations are not required to run the BLM in its toxicity prediction mode, but they are used in the Monte Carlo method to determine the FSC. Because of this, we calculated the FSC both with and without (neglecting) the correlation between copper concentrations and BLM input parameters at each test site.

C.3.1 Naugatuck River, Connecticut

The USGS has sampled the Naugatuck River near Waterville, Connecticut (Station 01208049) since 1967. Ninety-one water samples collected since 1984 provided near-concurrent measurements of all BLM water quality inputs and filtered copper concentrations. The water is low in hardness and alkalinity, slightly acidic (mean pH = 7.32), and fairly low in conductivity (10^{th} percentile = 134 µS/cm at 25º C). Organic carbon concentrations are representative for rivers and streams in this region and nationwide (logmean TOC = 4.02 mg/L), and the filtered copper concentrations are low (logmean filtered copper = 3.62 µg/L). The FSC for copper at this site was calculated to be 11.4 µg/L when the correlation between copper concentrations and BLM parameters was considered, and 7.0 µg/L when this correlation was neglected. Test results at this site are show in Table C-1.

Table C-1. Copper Fixed Site Criterion predictions for the Naugatuck River, Connecticut using various calculation methods

Calculation Method	DOC (mg/L)	pH	Geochemical Ions	Copper Fixed Site Criterion (µg/L)
Monte Carlo FSC with [copper] correlated to inputs (r =0.7)	Data	Data	Data	11.4
Monte Carlo FSC with no [copper] correlation	Data	Data	Data	7.0
IC calculated with 10th % of input data	2.8 (10th % of data)	7.1 (10th % of data)	(10th % of data)	6.4
IC calculated with input from 10th % of conductivity and correlations	5.49 (projected from correlations)	7.55 (projected from correlations)	(projected from correlations)	21.5
IC calculated with input from 10th % of conductivity and correlations except DOC	2.7 (10th % from L3 ecoregion)	7.55 (projected from correlations)	(projected from correlations)	10.5
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	2.7 (10th % from L3 ecoregion)	7.1 (10th % of data)	(projected from correlations)	5.7
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	2.8 (10th % of data)	7.1 (10th % of data)	(projected from correlations)	5.9

The BLM was then applied to predict IC for copper, using low-end percentiles of the measured BLM water quality inputs. Using the 10th percentile values of all measured input data, an IC of 6.4 µg/L was predicted. This IC is 43% smaller than the FSC calculated considering the copper correlation, but only 21% smaller than the FSC neglecting this correlation. When the 10th percentiles of all of the BLM water quality inputs were instead projected from conductivity using the regression models, the predicted ICs were 21.7 µg/L (using the regressions based on 10th percentiles of the three-state data) and 21.5 µg/L (using the regressions based on all of the data). These results illustrate two important points. First, the BLM predictions based on water quality inputs all projected from conductivity correlations are quite different from BLM predictions based on site data; this will be further considered below. Secondly, however, the BLM predictions based on projected water quality inputs do not really depend on which correlations are used.

Clearly, this result shows that the regression models are unable to accurately project all BLM water quality inputs at this site. However, this was almost entirely due to inaccuracy in the pH and organic carbon projections. To demonstrate this, we recalculated the IC several times, using better estimates of the organic carbon and/or pH data, but all other BLM water quality inputs projected from conductivity using the regression models. The first recalculation was made using the 10th percentile of DOC from rivers and streams in the Northeastern Coastal Zone, the Level III ecoregion where the

Naugatuck River is located.[7] In this case the predicted IC was 10.5 µg/L, a value much closer to the FSCs as well as the IC calculated using the 10th percentile values of all the measured input data. A second recalculation was made using the 10th percentile of the pH data, together with the ecoregional 10th percentile of DOC and all other BLM water quality inputs projected from conductivity using the regression models. Finally, a third recalculation was made in which the 10th percentiles of both pH and DOC data were input, with the remaining BLM water quality inputs projected from conductivity using the regression models. For both of these cases, the ICs were within about 10% of the prediction made using the 10th percentile values of all the measured input data. In summary, if BLM predictions are made for copper IC using measured values of pH and organic carbon, minimal error results from projecting the other BLM water quality inputs using conductivity and the regression models. As will be shown in the following sections, the same result was found for the other test sites.

The correlation between filtered copper concentrations and BLM parameters and output was quite strong at this location (r = 0.70 between filtered copper and IC predictions). As a result, the FSC corresponds to an elevated percentile (40%) of the IC predictions. If this correlation is neglected in the Monte Carlo method, the FSC corresponds to only the 14th percentile of the IC predictions. This suggests that the relationship between FSC and IC (in terms of the percentile of the IC distribution corresponding to the FSC) may be somewhat site-specific. Regardless of this complication, the conductivity regressions appear to project reliable low-end percentile estimates of the BLM water quality inputs other than pH and organic carbon. This was demonstrated by repeating the analysis described above using 5th, 2.5th, and 1st percentile input values and projections, each of which produced comparable results (not shown).

C.3.2 San Joaquin River, California

The USGS has sampled the San Joaquin River near Vernalis, California (Station 1130500) since 1950. Water samples collected since 1984 provided 283 near-concurrent measurements of all BLM water quality inputs and 77 filtered copper concentrations. The water has moderate values of hardness and alkalinity, neutral pH, and moderately high conductivity (10th percentile = 307 µS/cm at 25º C). DOC concentrations are representative for rivers and streams in this region and nationwide (logmean DOC = 5.35 mg/L), and the filtered copper concentrations are low (logmean filtered copper = 1.75 µg/L). The FSC for copper at this site was calculated to be 39.1 µg/L, and the correlation between copper concentrations and BLM parameters was strong (r = 0.624 between filtered copper and IC predictions). This FSC value corresponds to the 46th percentile of the distribution of IC. When the FSC for copper was recalculated assuming no correlation between copper concentrations and BLM parameters, the value decreased to 11.1 µg/L (corresponding to the 4.5th percentile of the IC distribution). Test results at this site are tabulated Table C-2.

[7] Ecoregion and water body-type specific DOC concentration percentiles were tabulated for the *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000), Technical Support Document Volume 2: Development of National Bioaccumulation Factors* (EPA-822-R-03-030).

Table C-2. Copper Fixed Site Criterion predictions for the San Joaquín River, California using various calculation methods

Calculation Method	DOC (mg/L)	pH	Geochemical Ions	Fixed Site Criterion (µg/L)
Monte Carlo FSC with [copper] correlated to inputs (r =0.6)	Data	Data	Data	39.1
Monte Carlo FSC with no [copper] correlation	Data	Data	Data	11.1
IC calculated with 10th % of input data	2.7 (10th % of data)	7.5 (10th % of data)	(10th % of data)	11.9
IC calculated with input from 10th % of conductivity and correlations	9.38 (projected from correlations)	7.77 (projected from correlations)	(projected from correlations)	54.0
IC calculated with input from 10th % of conductivity and correlations except DOC	2.79 (10th % from L3 ecoregion)	7.77 (projected from correlations)	(projected from correlations)	16.0
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	2.79 (10th % from L3 ecoregion)	7.5 (10th % of data)	(projected from correlations)	11.6
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	2.7 (10th % of data)	7.5 (10th % of data)	(projected from correlations)	11.2

The BLM was then applied to predict IC for copper, using low-end percentiles of the BLM water quality inputs. Using the 10th percentile values of all measured input data, an IC of 11.9 µg/L was predicted, which is 70% smaller than the FSC calculated considering the copper concentration correlation but 7% higher than the FSC neglecting this correlation. When the 10th percentiles of all of the BLM water quality inputs were instead projected from conductivity using the regression models, the predicted IC were 50.0 µg/L (using the regressions based on 10th percentiles of the three-state data) and 54.0 µg/L (using the regressions based on all of the data). Again, the BLM predictions based on projected water quality inputs do not really depend on which correlations are used. And, as was the case at the Naugatuck River site, the regression models were unable to accurately project all BLM water quality inputs at this site, although the error is again almost entirely due to inaccuracy in the pH and organic carbon projections. As in the previous case, we demonstrated this by recalculating the IC several times, using better estimates of the organic carbon and/or pH data, but all other BLM water quality inputs projected from conductivity using the regression models. The first recalculation was made using the 10th percentile of DOC from rivers and streams in the Central California Valley, the Level III ecoregion where the San Joaquin River is located. In this case the predicted IC was 16.0 µg/L, a value much closer to the uncorrelated FSCs as well as the IC calculated using the 10th percentile values of all the measured input data. A second recalculation was made using the 10th percentile of the pH data, together with the ecoregional 10th percentile of DOC and all other BLM water quality inputs projected from conductivity using the regression models. Finally, a third recalculation was made in which the 10th percentiles of both pH and DOC data were input, with the remaining BLM water quality inputs projected from conductivity using the regression models. For both of these cases, the ICs were within

about 5% of the prediction made using the 10th percentile values of all the measured input data. BLM predictions made for copper IC at this site using measured values of pH and organic carbon, but all other BLM water quality inputs projected using conductivity regressions, were found to be accurate in comparison to model predictions made using all measured input data.

C.3.3 South Platte River, Colorado

The South Platte River has been sampled by the USGS at Denver, Colorado (Station 06714000) since 1972. Water samples collected since 1984 provided 93 near-concurrent measurements of all BLM water quality inputs and 10 filtered copper concentrations. The water is moderately high in hardness and alkalinity, neutral pH, and moderate conductivity (10th percentile = 229 µS/cm at 25° C). Organic carbon concentrations are representative for rivers and streams in this region (logmean DOC = 5.50 mg/L), and the filtered copper concentrations are low (logmean filtered copper = 3.27 µg/L). The FSC for copper at this site was calculated to be 35.4 µg/L. This FSC value corresponds to the 32nd percentile of the distribution of IC. Moderate correlation between copper concentrations and BLM parameters was observed at this site (r = 0.50 between filtered copper and IC predictions). When the FSC for copper was recalculated assuming no correlation between copper concentrations and BLM parameters, the value decreased to 20 µg/L (corresponding to the 4.3rd percentile of the IC distribution). Test results at this site are shown in Table C-3.

Table C-3. Copper Fixed Site Criterion predictions for the South Platte River, Colorado using various calculation methods

Calculation Method	DOC (mg/L)	pH	Geochemical Ions	Copper Fixed Site Criterion (µg/L)
Monte Carlo FSC with [copper] correlated to inputs (r =0.5)	Data	Data	Data	35.4
Monte Carlo FSC with no [copper] correlation	Data	Data	Data	20.0
IC calculated with 10th % of input data	4.1 (10th % of data)	7.5 (10th % of data)	(10th % of data)	17.3
IC calculated with input from 10th % of conductivity and correlations	7.7 (projected from correlations)	7.7 (projected from correlations)	(projected from correlations)	37.5
IC calculated with input from 10th % of conductivity and correlations except DOC	4.5 (10th % from L3 ecoregion)	7.7 (projected from correlations)	(projected from correlations)	21.6
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	4.5 (10th % from L3 ecoregion)	7.5 (10th % of data)	(projected from correlations)	17.3
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	4.1 (10th % of data)	7.5 (10th % of data)	(projected from correlations)	15.9

The BLM was applied to predict IC for copper, using low-end percentiles of the BLM water quality inputs. Using the 10th percentile values of all measured input data, an IC of 17.3 µg/L was predicted, which is 51% smaller than the FSC calculated considering the copper concentration correlation but only 14% smaller than the FSC neglecting this correlation. When the 10th percentiles of all of the BLM water quality inputs were instead projected from conductivity using the regression models, the predicted IC was 37.5 µg/L. As was the case at the previous sites, the regression models were again unable to accurately project pH and organic carbon concentrations for input to the BLM. We demonstrated this by recalculating the IC several times, using better estimates of the organic carbon and/or pH data, but all other BLM water quality inputs projected from conductivity using the regression models. The first recalculation was made using the 10th percentile of DOC from rivers and streams in the Western High Plains, the Level III ecoregion where the South Platte River is located. In this case the predicted IC was 21.6 µg/L, a value much closer to the uncorrelated FSCs as well as the IC calculated using the 10th percentile values of all the measured input data. A second recalculation was made using the 10th percentile of the pH data, together with the ecoregional 10th percentile of DOC and all other BLM water quality inputs projected from conductivity using the regression models. Finally, a third recalculation was made in which the 10th percentiles of both pH and DOC data were input, with the remaining BLM water quality inputs projected from conductivity using the regression models. For both of these cases, the ICs were within 10% of the prediction made using the 10th percentile values of all the measured input data. As with the previous cases, the BLM predictions made for copper IC at this site using measured values of pH and organic carbon, but where all other BLM water quality inputs were projected using conductivity regressions, were found to be accurate in comparison to model predictions made using all measured input data.

C.3.4 Halfmoon Creek, Colorado

The USGS has sampled Halfmoon Creek near Malta, Colorado (Station 07083000) since 1959. Seventy-three water samples collected since 1984 provided near-concurrent measurements of all BLM water quality inputs and 18 filtered copper concentrations. The water is very low in hardness and alkalinity, slightly acidic (mean pH = 7.76), and low in conductivity (10th percentile = 50.1 µS/cm at 25° C). Organic carbon concentrations are low (logmean DOC = 0.92 mg/L), as are the filtered copper concentrations (logmean filtered copper = 1.75 µg/L). The FSC for copper at this site was calculated to be 1.56 µg/L, corresponding to the 6th percentile of the distribution of IC. The correlation between copper concentrations and BLM parameters was negligible at this site, so the Monte Carlo FSC were not calculated twice (i.e., with and without the copper correlation) as was done at the other sites. Test results at this site are shown in Table C-4.

Table C-4. Copper Fixed Site Criterion predictions for the Halfmoon Creek, Colorado using various calculation methods

Calculation Method	DOC (mg/L)	pH	Geochemical Ions	Copper Fixed Site Criterion (µg/L)
Monte Carlo FSC with [copper] correlated to inputs (r =0.01)	Data	Data	Data	1.56
IC calculated with 10th % of input data	0.6 (10th % of data)	7.2 (10th % of data)	(10th % of data)	1.42
IC calculated with input from 10th % of conductivity and correlations	2.8 (projected from correlations)	7.3 (projected from correlations)	(projected from correlations)	7.43
IC calculated with input from 10th % of conductivity and correlations except DOC	0.6 (10th % from L3 ecoregion)	7.3 (projected from correlations)	(projected from correlations)	1.58
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	0.6 (10th % from L3 ecoregion)	7.2 (10th % of data)	(projected from correlations)	1.39
IC calculated with input from 10th % of conductivity and correlations except pH & DOC	0.6 (10th % of data)	7.2 (10th % of data)	(projected from correlations)	1.39

The BLM was then applied to predict IC for copper, using low-end percentiles of the BLM water quality inputs. Using the 10th percentile values of all measured input data, an IC of 1.42 µg/L was predicted, only 9% smaller than the FSC. When the 10th percentiles of all of the BLM water quality inputs were instead projected from conductivity using the regression models, the predicted IC was 7.43 µg/L. Again, this result clearly shows that the regression models are unable to accurately project all BLM water quality inputs at this site. As in the previous examples, this was almost entirely due to inaccuracy in the pH and organic carbon projections. As in the previous cases, we demonstrated this by recalculating the IC several times, using better estimates of the organic carbon and/or pH data, but all other BLM water quality inputs projected from conductivity using the regression models. The first recalculation was made using the 10th percentile of DOC from rivers and streams in the Southern Rockies, the Level III ecoregion where Halfmoon Creek is located. In this case the predicted IC was 1.58 µg/L, a value within about 10% of the FSC as well as the IC calculated using the 10th percentile values of all the measured input data. A second recalculation was made using the 10th percentile of the pH data, together with the ecoregional 10th percentile of DOC and all other BLM water quality inputs projected from conductivity using the regression models. Finally, a third recalculation was made in which the 10th percentiles of both pH and DOC data were input, with the remaining BLM water quality inputs projected from conductivity using the regression models. For both of these cases, the ICs were within about 2% of the prediction made using the 10th percentile values of all the measured input data. As in the previous examples, if BLM predictions are made for copper IC using measured values of pH and organic carbon, minimal error results from projecting the other BLM water quality inputs using conductivity and the regression models.

C.3.5 Summary of Site-Specific Test Results

The results of this work can be summarized as follows:

Regression models were developed to project 10th percentiles of BLM water quality parameters from the 10th percentile of conductivity distributions at sites in Colorado, Utah, and Wyoming. The regression models were tested using data and copper BLM predictions for four sites, and produced highly consistent results. The regression models for pH and DOC, the most sensitive of BLM water quality parameters, were not sufficiently accurate to make reliable BLM predictions. However, regression models for the GI parameters (alkalinity, calcium, magnesium, sodium, potassium, sulfate, and chloride,) were reasonably accurate, as judged by comparison of model predictions made using projected values of the GI BLM input parameters to model predictions made using all measured input data. The regression models used to project GI parameters from conductivity were calculated two different ways; however, the BLM predictions of IC were not sensitive to this difference.

We were unable to find an estimate for site-specific pH that was superior to the (admittedly poor) conductivity regression. To improve upon this estimate it was necessary to use actual site-specific pH data. This appears to be the general case for reliable site-specific BLM application.

For DOC, the ecoregion and water body-type specific DOC concentration percentiles tabulated by EPA for the National Bioaccumulation Factors Technical Support Document appear to be far better estimates of lower-percentile DOC concentrations than the projections made using the conductivity regression. These tabulations are based on an organic carbon database compiled prior to 2003 from a number of sources including EPA's STOrage and RETrieval Data Warehouse (STORET) and the USGS NWIS. The utility of these tabulations could be improved by updating them to incorporate newer information. For example, EPA recently released data from the Wadeable Stream Assessment, which included DOC measurements from a statistically based random sample of ~2,000 streams. Other statistically-based national water quality surveys, including national assessments of lakes and large rivers, will also be providing additional data in future years.

The Monte Carlo method developed to calculate FSC for copper was applied at each of the four sites, both with and without the correlation between filtered copper concentrations and the BLM water quality parameters that were found to be significant at three of the sites. We also approximated the FSC using the 10th percentile of the distribution of IC predicted by the BLM at each site. When copper concentration correlations were considered in the FSC calculations, the 10th percentile of the IC distributions was found to be highly conservative approximations of the FSC, underestimating the FSC by 44 to 70%. This is illustrated in Figure C-1, which also shows the good agreement between IC predicted with the BLM using site-specific data and IC predicted using measured pH and organic carbon but projected values of the GI BLM input parameters. Ecoregion and water body-type specific DOC concentration percentiles ("L3-DOC" in the figure below) were also an improvement over the projections based on conductivity regressions.

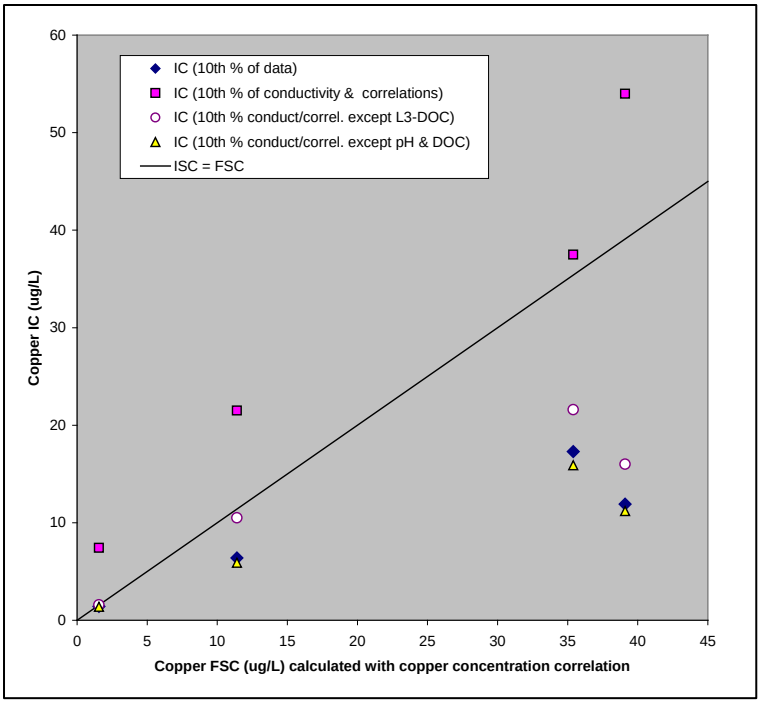

Figure C-1. Instantaneous Criteria (IC) predicted with the BLM using site-specific data and IC predicted using measured pH and organic carbon and projected values of the GI BLM input parameters

When copper concentration correlations were neglected in the FSC calculations, the 10th percentile of the IC distributions did a much better job approximating the FSC. This is shown in Figure C-2. In this case, the 10th percentile of the IC distributions was within 15% of the FSC. This figure also shows the good agreement between IC predicted with data and projected values of the GI BLM input parameters.

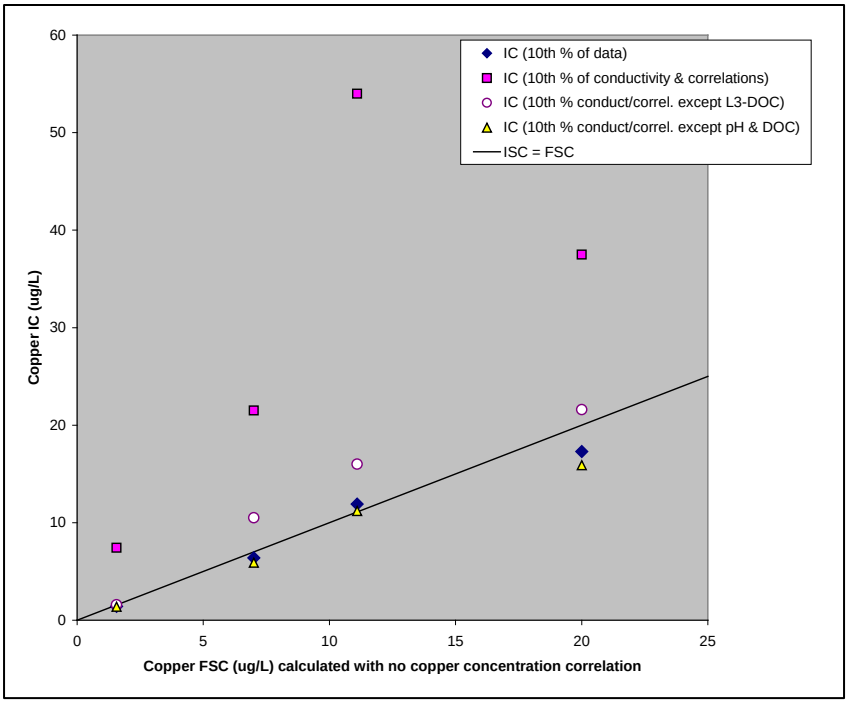

Figure C-2. 10th percentile of the IC distributions using data and projected (predicted) values of the GI BLM parameters

The degree of correlation between filtered copper concentrations and BLM input water quality parameters appears to be an important site-specific factor in determining the relationship between the FSC and the IC. Figure C-3 plots the percentile of the IC corresponding to the FSC for each site as a function of the correlation coefficient between the copper concentrations and the IC, for two cases: (1) FSC calculated by the Monte Carlo method including the observed correlations between concentrations of copper and BLM input water quality parameters, and (2) FSC calculated with no correlation between concentrations of copper and BLM input water quality parameters. In the first case (plotted with dark diamond symbols), the percentile of the IC corresponding to the FSC increases substantially (6th to 46th percentile) as the correlation coefficient between the copper concentrations and the IC increases. If the correlation between concentrations of copper and BLM input water quality parameters is neglected (the second case, plotted in lighter square symbols), the percentile of the IC corresponding to the FSC is considerably lower (4.3rd to 14th percentile). This suggests that correlations between copper concentrations and BLM input parameters should be given careful consideration when calculating FSC.

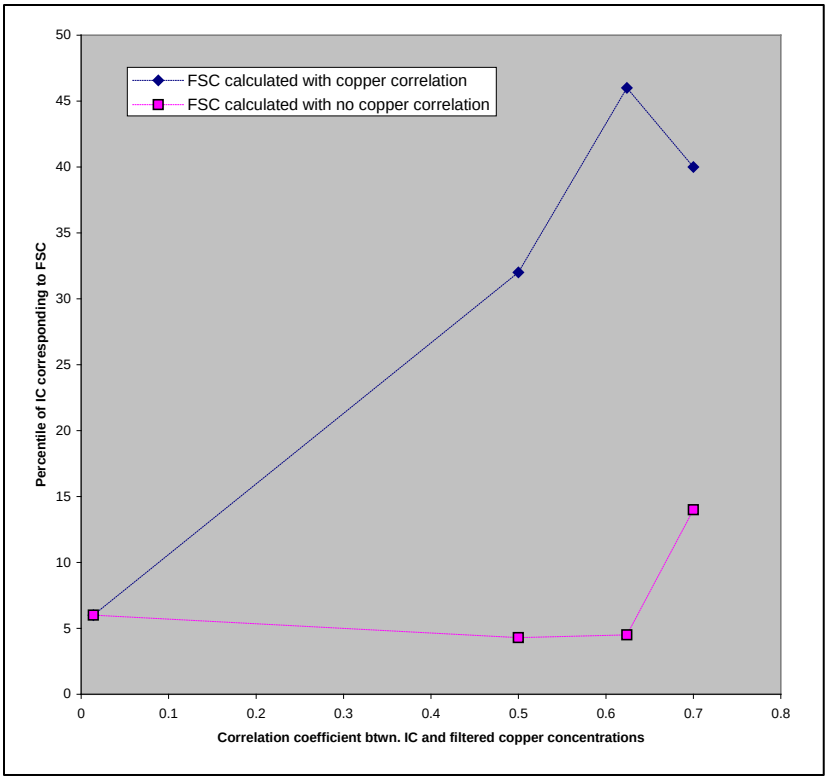

Figure C-3. Percentile of the IC corresponding to the FSC for each site as a function of the correlation coefficient between the copper concentrations and the IC when the FSC is calculated with Copper correlation and when FSC is calculated without Copper correlation

C.4 Combining GI-Conductivity Regressions with Geostatistical Techniques

Geostatistical techniques are attractive because they explain parameter variation arising from spatial correlations, which are otherwise ignored by (and may, in fact, violate the assumptions of) conventional statistics. BLM input water quality parameters (except for pH and DOC) are GIs, the concentrations of which vary in surface water due to dissolution, weathering, ground water-surface water interactions, and other geologic processes in the watershed. Consequently, the concentrations of GI parameters tend to vary according to the regional geology. For example, water hardness has noticeable geographic trends. Areas with limestone geology, such as in the prairie states, tend toward high hardness and alkalinity. Areas of with granite geology, such as parts of the Northeast, tend toward low hardness and alkalinity. The estimation of GI parameter values based on geography thus seems possible. EPA has provided a prototype of a geostatistical approach[8] that demonstrated this potential. That work applied kriging to predict median concentrations of five of the BLM water quality input parameters (pH, DOC, alkalinity, sodium, and calcium) averaged over 8-digit HUCs, using the USGS NWIS as the source of spatial data. Comparison of measured concentrations with kriging predictions were encouraging, especially for DOC and alkalinity. Geostatistical techniques to project BLM GI input

parameters might well be developed from the same nationwide monitoring data used to develop a correlation approach. By the same token, geostatistical techniques based on these data may suffer the same problems experienced when developing the correlation approach. Most significantly, the NWIS data are not randomly distributed in either time or space, and the measurements of the BLM GI parameters are generally uneven (considerable differences in terms of the number of observations for different parameters) and/or inconsistent (i.e., relatively few concurrent measurements of BLM GI parameters).

There may be great value in supplementing the geostatistical approach with classical estimation methods, such as regression and correlation. Examination of the NWIS data suggests that conductivity may be useful for estimating BLM input water quality parameters in conjunction with geostatistics. The literature indicates that conductivity is one of the most widely monitored water quality indicators in the US. Among water quality parameters, the data for conductivity are the most complete and cover the longest time period (Wang and Yin, 1997). In part, this is because conductivity measurements are usually included in automated multiparameter systems for monitoring changes in the quality of surface waters (Allen and Mancy, 1972). A vast amount of conductivity data exists, both in terms of the total number of observations and the number of sites reporting this parameter in comparison to the BLM GI quality parameters. For example, NWIS data for the state of Colorado have almost four times as many observations of conductivity as for calcium, and they are measured at more than twice the number of sites. There are 20 times as many observations of conductivity as for alkalinity, and they are measured at more than seven times the number of sites. Since conductivity data are abundant, and correlate well to the BLM GI parameters (GLEC, 2007), it is reasonable to incorporate conductivity in spatial projections of BLM parameters. This may simplify the geostatistical approach and allow more robust spatial projections of BLM water quality parameters.

Although combining GI-conductivity regressions with geostatistical techniques seems promising for the reasons mentioned above, this approach had never been demonstrated. We conducted a simple test using NWIS conductivity and hardness data from the state of Colorado. We used data from Colorado because many more stations were sampled in comparison to the surrounding states.

The data were processed in a manner similar to the methods used to develop the regressions in Section C.2. For each station, we calculated the 10^{th} percentiles of conductivity and hardness. A regression model was fit to the full dataset (data for all rivers and streams sampled in Colorado between 1984 to 2005). The following regression model was developed to project hardness from conductivity:

$$\ln(\text{hardness}) = 0.984 \cdot \ln(\text{EC}) - 0.870$$

We also kriged the 10^{th} percentiles of conductivity and hardness, using latitude and longitude coordinates reported by USGS for each sampling station. Figure C-4 shows the kriged surface of the 10^{th} percentile of conductivity at all stations in Colorado, Utah, and Wyoming. Data are far more abundant in Colorado, as shown by the density of the dots representing the locations of sampling stations. Figure C-5 shows the kriged surface of the 10^{th} percentile of hardness at all stations in Colorado. Kriging was done using the Vertical Mapper program, version 3.1; no attempts were made to optimize the kriging of conductivity or hardness by parameter adjustment.

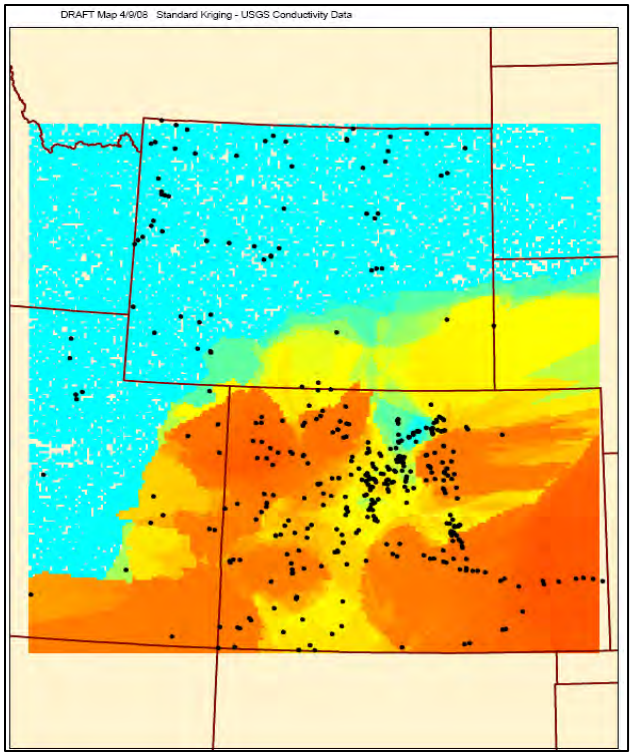

Figure C-4. Kriged surface of the 10th percentile of conductivity at all stations in Colorado, Utah and Wyoming

Dots represent sampling stations; notice that data are far more abundant in Colorado.

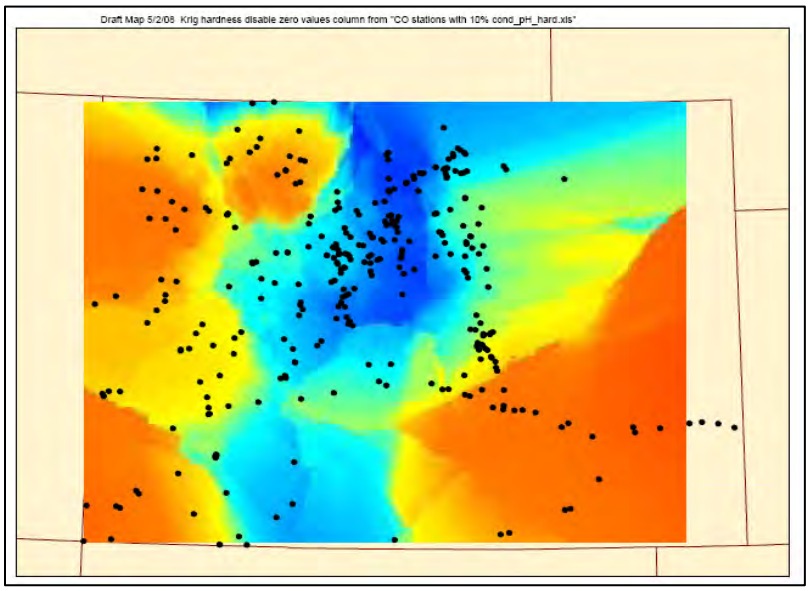

Figure C-5. Kriged surface of the 10th percentile of hardness at all stations in Colorado

Our goal was to see whether combining the kriged conductivities with the conductivity-hardness regression would project the 10th percentiles of hardness better than direct kriging of the hardness

data. For the combined kriging/regression approach, we determined the kriged conductivity values at all of the sampling locations and then projected the 10th percentiles of hardness at these locations using the regression equation. We also determined the directly-kriged 10th percentiles of hardness at all of the sampling locations.

The hardness estimates obtained by each approach were then compared to the 10th percentiles of hardness measured at each station. The results of this comparison are shown graphically in Figure C-6. Both approaches produce estimates of hardness that correlate significantly with the measured data (correlation coefficient r= 0.80 for direct kriging of hardness; r= 0.950 for conductivity kriging + regression projection). However, the kriging+regression approach fits the hardness data substantially better than direct kriging. To quantify this, we calculated the residual sum of squares (RSS), a composite measure of the discrepancy between the data and our alternative hardness estimates. The smaller this discrepancy is, the better the estimation will be. In natural log space, the RSS for the kriging+regression approach is 18.6 (135 degrees of freedom, or df) while the log-space RSS for the direct kriging approach is 73.4 (136 df). Thus, for this test case substantially better estimates of the 10th percentile of hardness were made by the kriging/regression approach compared to direct kriging.

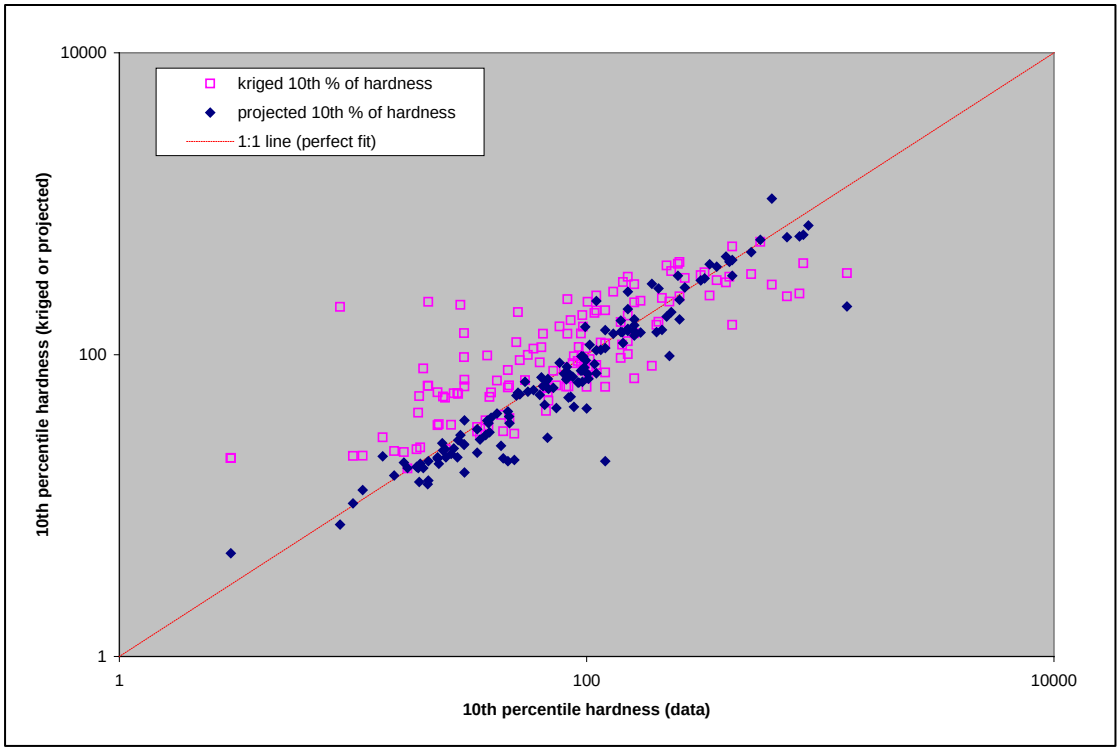

Figure C-6. Comparison of the 10th percentile of hardness at all stations in Colorado with estimates based on (a) direct kriging of hardness data and (b) kriging of conductivity to station locations and projecting conductivity to hardness via regression ("kriging/regression")

As this test demonstrated, combining kriging with regressions to project BLM GI inputs from conductivity appears to improve the accuracy of estimates of parameters used as BLM inputs. Applying the conductivity kriging/regression projection approach on a broader scale should be considered as a

"next step" in developing tools to estimate BLM water quality parameters for sites where there may be few or no data available to characterize water quality. Since direct kriging of most BLM GI parameters has already been done using data from NWIS, it will also be worthwhile to continue comparing the alternative estimates to the observed data in order to obtain the best estimates.

We should also note that although the kriging/regression approach can be used to improve the accuracy of estimates of GI parameters used as BLM inputs, this approach cannot be expected to produce accurate site-specific estimates for the two most important BLM inputs: pH and DOC. As shown in Section C.3, accurate estimates of the GI parameters are less important than pH and DOC in terms of predicting appropriate site-specific IC and FSC. Since our analysis of NWIS data indicates there to be either little no trend between conductivity and pH, and direct kriging produced similarly ambiguous predictions, we must conclude that site-specific data for pH must either be available or be collected for BLM application at a site. This may not be a significant obstacle, since pH data can be cheaply and readily acquired.

Lack of methods to accurately estimate DOC is a bigger problem, since measurements of this parameter are comparatively rare and DOC is a relatively expensive measurement to make. For DOC, analysis of NWIS data again indicates no trend with conductivity, so the kriging/regression approach is not appropriate for this parameter. However, other analyses conducted suggested that DOC could be kriged with some success. And, as was demonstrated for the test sites in Section C.3, the ecoregion and water body-type specific DOC concentration percentiles tabulated by EPA for the National Bioaccumulation Factors Technical Support Document appear to offer reasonable estimates of lower-percentile DOC concentrations. Further development of these approaches for estimating site-specific DOC appears worthwhile, for example by incorporating new data from the Wadeable Stream Assessment and other statistically-based national water quality surveys.

C.5 References

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USEPA. 2002. Development of Methodologies for Incorporating the Copper Biotic Ligand Model into Aquatic Life Criteria: Application of BLM to Calculate Site-Specific Fixed Criteria. GLEC Work Assignment 3-38, Contract No. 68-C-98-134. 63p plus figures.

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Wang, X. and Z.Y. Yin. 1997. Using GIS to assess the relationship between land use and water quality at a watershed level. Environment International. 23(1): 103-114.

Appendix D: Approaches for Estimating Missing Biotic Ligand Model Input Parameters: Projections of Total Organic Carbon as a Function of Biochemical Oxygen Demand

D.1 Introduction

The 2007 Update of the Ambient Water Quality Criteria for Copper (EPA-822-R-07-001) employs the Biotic Ligand Model (BLM) to estimate bioavailability of this metal in toxicity tests used in Criterion Maximum Concentration derivation, which requires data on the 10 input parameters for the BLM, including dissolved organic carbon (DOC). Data for DOC concentrations, in both effluents and receiving waters, are extremely limited. The BLM is very sensitive to DOC concentrations (HydroQual, 2005), which means that to ensure accurate predictions of copper bioavailability and toxicity reliable data on DOC concentrations in the water are needed. Effluent DOC concentrations, which are necessary for application of the BLM to predict copper toxicity associated with a wastewater discharge, are monitored by very few publicly-owned treatment works (POTWs).

Projections of DOC concentrations from biochemical oxygen demand (BOD) values may be a viable solution for surmounting the lack of data on DOC. Effluent BOD (most typically 5-day BOD) is monitored by most POTWs. We expect a positive correlation between BOD and DOC, because the two parameters are conceptually related. While DOC quantifies the concentration of many organic compounds dissolved in water, BOD is a routine surrogate test for estimating the load of organic carbon into the environment. Ideally, one might expect an almost stoichiometric relationship between organic carbon (i.e., DOC) and the oxygen consumed during its metabolization (i.e., BOD). For instance, Fadini et al. (2004) evaluated the possible replacement of BOD for DOC measurements in a number of different wastewater categories. A statistical relationship between effluent BOD and DOC would provide estimates of DOC concentrations, needed for application of the BLM, from routine BOD monitoring data. The effluent contribution to in-stream DOC could then be estimated, for example, by using a dilution model for a site.

Evidence, from analyses of effluent monitoring data from the New York State Department of Environmental Conservation (NYSDEC) Contaminant Assessment and Reduction Project (CARP), suggests that most of the total organic carbon (TOC) in POTW effluent is in the form of DOC. Therefore, a regression between BOD and TOC could be used as a surrogate for the relationship between BOD and DOC. The advantage of using TOC is the greater availability of data. TOC is reported for a significant number of major POTW dischargers.

D.2 Data

In 2006, monitoring data from all major POTWs reporting TOC and 5-day BOD in the United States were downloaded from the U.S. Environmental Protection Agency (EPA) Permit Compliance System (PCS) web site http://www3.epa.gov/enviro/facts/pcs-icis/search.html. Nine POTWs had 30 or more synchronous records of TOC and BOD, while 23 POTWs had at least 10 synchronous records. These numbers include both monthly average and maximum monthly values.

Review of the data indicated several extremely high (>1,000 milligrams per liter [mg/L]) effluent TOC values for discharger CA0079243. We assumed that they presented errors in the reported unit, and divided them by 1,000 to convert from units of microgram per liter (µg/L) to mg/L. TOC and BOD

records were matched by POTW, location (e.g., upstream, downstream, influent or effluent), year and month. Thus, "synchronous" measurements do not necessarily correspond to samples collected on the same day and time. The resulting table had 341 records.

D.3 Results

D.3.1 TOC and BOD at All Monitoring Locations

The first statistical evaluation involved data for all monitoring locations at the eight POTWs reporting 30 or more synchronous records of TOC and BOD. Table D-1 presents the results of least squares regression of the average monthly data: TOC*avg* = *a* + *b* BOD*avg*. A scatter plot of this data is shown in Figure D-1. Table D-2 presents the results of least squares regression of the maximum monthly data: TOC*max* = *a* + *b* BOD*max*. A scatter plot of this data is shown in Figure D-2. Bimodal distributions are observed for TOC and especially BOD in this data set. It should be noted that the BOD concentrations of 200 mg/L or higher were measured in samples of untreated (influent) wastewater; TOC concentrations were also quite high in these samples. Both scatter plots (Figures D-1 and D-2) show a fairly strong correlation between TOC and BOD in the combined data for all POTWs. The linear relationship between TOC and BOD is better defined in the average data (Figure D-1).

Table D-1. Least squares regression of average monthly TOC and BOD data for all monitoring locations

POTW	Location	Intercept (a)	Slope (b)	r^2	df
CA0054372	Effluent Gross Value				
CA0105295	Effluent Gross Value	7.551	-0.379	0.009	41
CA0105295	Raw Sew/Influent	19.935	0.142	0.104	59
CA8000326	Effluent Gross Value*				
CA8000383	Effluent Gross Value	4.952	0.725	0.344	31
CA8000383	Raw Sew/Influent	59.586	0.107	0.038	30
ID0020443	Upstream Monitoring*				
ID0020443	Downstream Monitoring	3.500	-0.400	0.190	2
ID0023981	Effluent Gross Value	6.268	0.281	0.196	26
ID0023981	Upstream Monitoring	3.200	-0.300	0.127	3
LA0073521	Effluent Gross Value*				
TN0023353	Effluent Gross Value	2.628	0.391	0.438	35
All POTWs	All locations	4.828	0.237	0.873	243

*note: POTW/location without regression results indicates less than 2 synchronous data records

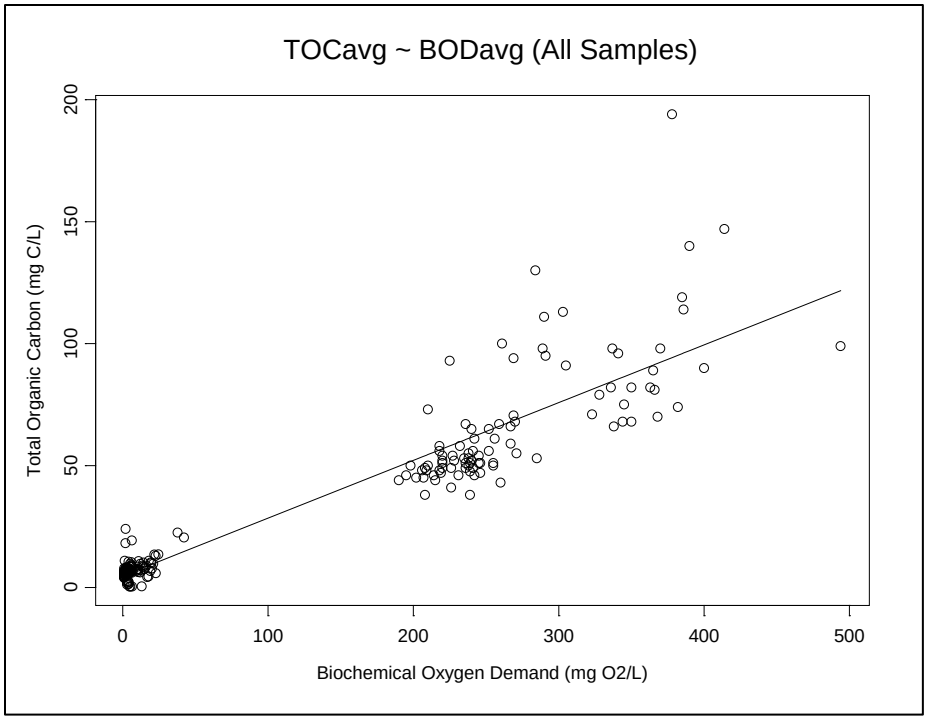

Figure D-1. Scatter Plot of Average Monthly Data (all Monitoring Locations)

Table D-2. Least squares regression of maximum monthly TOC and BOD data for all monitoring locations

POTW	Location	Intercept (a)	Slope (b)	r^2	df
CA0054372	Effluent Gross Value	9.420	0.499	0.154	29
CA0105295	Effluent Gross Value	8.567	-0.114	0.007	50
CA0105295	Raw Sew/Influent	56.439	0.062	0.046	59
CA8000326	Effluent Gross Value	7.307	0.006	0.000	28
CA8000383	Effluent Gross Value	6.674	0.507	0.235	31
CA8000383	Raw Sew/Influent	149.293	0.047	0.052	30
ID0020443	Upstream Monitoring	0.300	1.175	0.039	4
ID0020443	Downstream Monitoring	3.500	-0.400	0.190	2
ID0023981	Effluent Gross Value	6.210	0.208	0.202	28
ID0023981	Upstream Monitoring	3.200	-0.300	0.127	3
LA0073521	Effluent Gross Value	12.989	-0.818	0.110	18
TN0023353	Effluent Gross Value*				
All POTWs	All locations	11.183	0.196	0.700	302

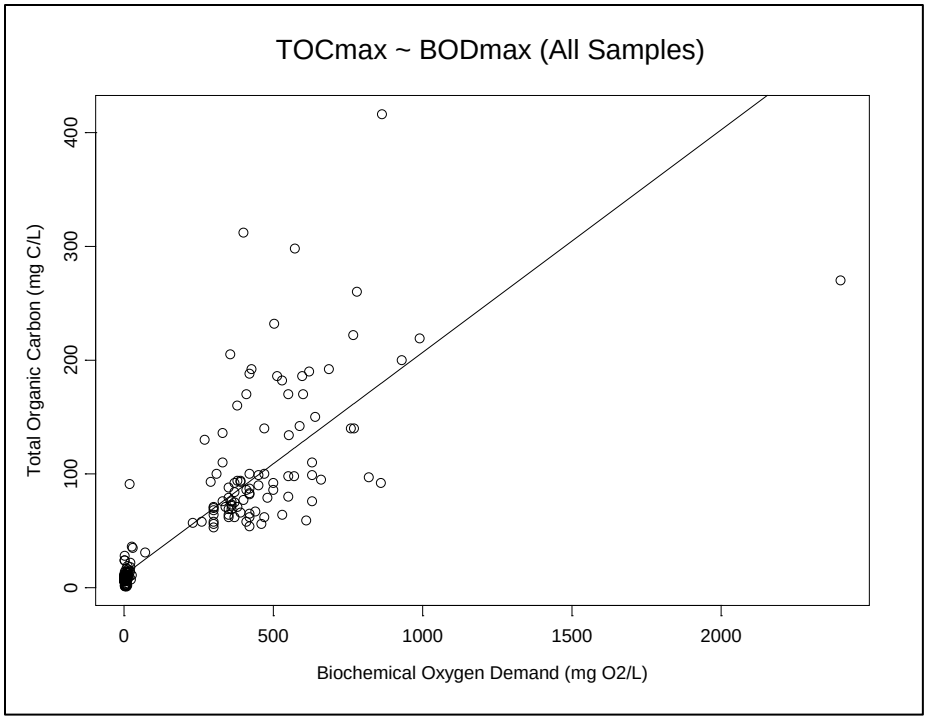

Figure D-2. Scatter Plot of Maximum Monthly Data (all Monitoring Locations)

Results of regression analyses revealed large differences in slopes of the linear model TOC = *a* + *b* BOD among locations and POTWs. Slopes for individual regressions ranged from –0.40 to 0.73 for average, and from -0.82 to 0.50 for maximum BOD and TOC values. Coefficients of determination (r^2) for the regressions were low; for most of them $r^2 < 0.2$. Pooling the data from all POTWs and locations increased the r^2 to 0.87 for average and 0.70 for maximum BOD and TOC values.

Diagnosis of regression analyses revealed that variance in both the average and maximum TOC rose with increasing values of biochemical oxygen demand. Such patterns were also evident from a simple inspection of the plots cited above. Homogeneity of variance, though, is a core assumption of ordinary least squares regression, and its violation compromises the quality of results generated by the analysis. The solution was to perform quantile regression analysis because it does not assume that variance of the response is homogeneous along the range of the independent variable. The fitted model for the 50^{th} quantile (median) was:

$$TOC_{avg} = 5.5647 + 0.2088\ BOD_{avg} \quad (243\ df,\ R^1 = 0.77)$$

D.3.2 TOC and BOD at Effluent Monitoring Locations

Although the quantile regression model above provided a reasonable fit of the data at all monitoring locations, we were specifically interested in the relationship between TOC and BOD measured in POTW effluents. Therefore, we conducted a separate statistical analysis of effluent monitoring data from the 17 POTWs with more than one synchronous record of TOC and BOD retrieved from PCS. TOC and BOD records were again matched by POTW, year, and month. The resulting data tabulation had 373 records.

The results of least squares regression of the average monthly effluent data: TOC*avg* = *a* + *b* BOD*avg* are presented in Table D-3. A scatter plot of this data is shown in Figure D-3. Table D-4 presents the results of least squares regression of the effluent maximum monthly data: TOC*max* = *a* + *b* BOD*max*. A scatter plot of this data is shown in Figure D-4.

Table D-3. Least squares regression of average monthly TOC and BOD data for effluent monitoring locations

POTW	Intercept (a)	Slope (b)	r^2	df
CA0054372				
CA0077691				
CA0079103				
CA0079243				
CA0102822				
CA0105295	7.5512	-0.3789	0.009	41
CA0107492				
CA0109991				
CA8000073				
CA8000326				
CA8000383	4.9522	0.7254	0.344	31
ID0023981	6.2679	0.2808	0.196	26
LA0069868				
LA0073521				
TN0023353	2.6284	0.3909	0.438	35
TN0023531	37.6422	-1.1348	0.076	3
TN0023574	7.4142	0.0882	0.016	25
All POTWs	5.8740	0.2859	0.245	174

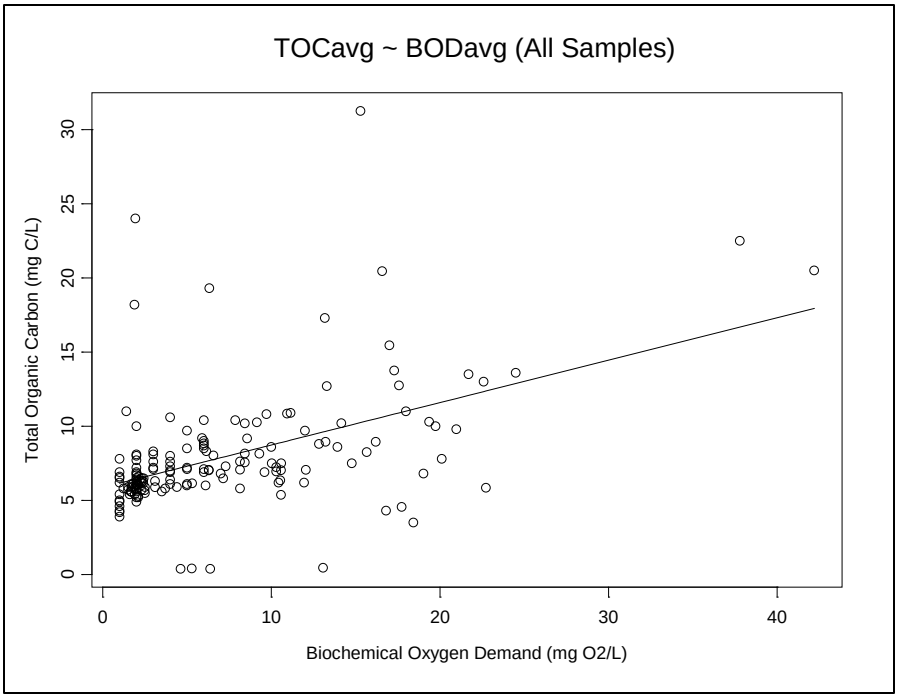

Figure D-3. Scatter Plot of Average Monthly Data (Effluent Monitoring Locations)

Table D-4. Least squares regression of maximum monthly TOC and BOD data for effluent monitoring locations

POTW	Intercept (a)	Slope (b)	r^2	df
CA0054372	9.4197	0.4993	0.154	29
CA0077691	4.2750	0.7008	0.422	6
CA0079103	23.3517	-0.1278	0.035	16
CA0079243	5.7979	-0.0367	0.008	8
CA0102822	6.3526	0.1780	0.195	28
CA0105295	8.5667	-0.1143	0.007	50
CA0107492	5.9043	1.0676	0.027	2
CA0109991	3.0331	0.8458	0.167	11
CA8000073	9.0000	0.0000	0.000	8
CA8000326	7.3073	0.0056	0.000	28
CA8000383	6.6743	0.5070	0.235	31
ID0023981	6.2101	0.2083	0.202	28
LA0069868	7.6475	0.1139	0.535	5
LA0073521	12.9886	-0.8184	0.110	18
TN0023353				
TN0023531				
TN0023574				
All POTWs	6.6930	0.4311	0.276	299

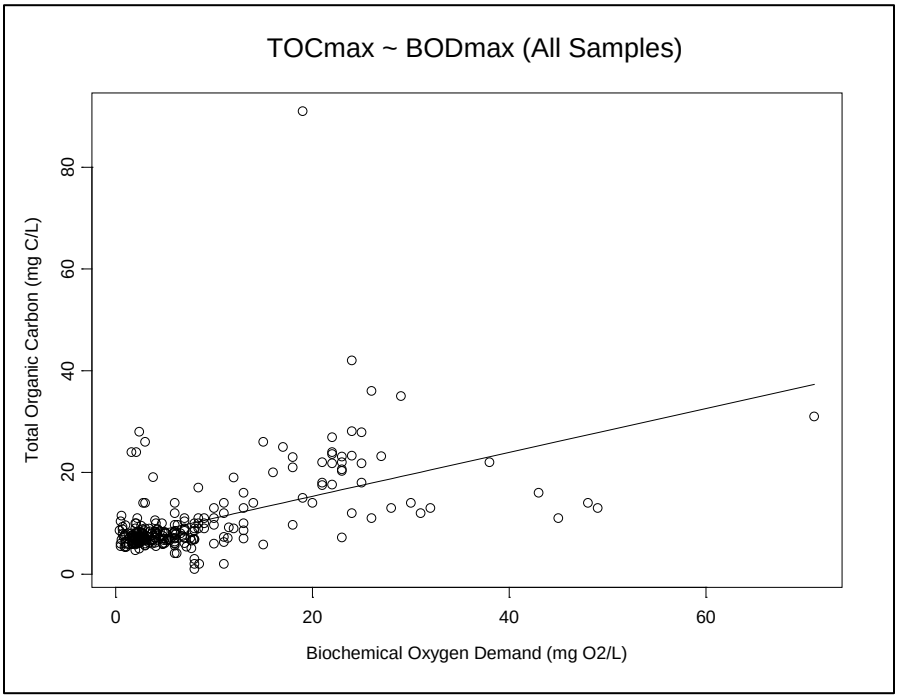

Figure D-4. Scatter Plot of Maximum Monthly Data (Effluent Monitoring Locations)

Results of the effluent regression analyses revealed large differences in slopes of the linear model TOC = *a* + *b* BOD among POTWs. Slopes for individual regressions ranged from –1.13 to 0.73 for average, and from -0.82 to 1.07 for maximum BOD and TOC values. Coefficients of determination (r^2) for the regressions were low; all $r^2 < 0.55$ and for most of them $r^2 < 0.24$. Low coefficients of determination were also recorded for regressions of TOC on BOD values from all POTWs ($r^2 = 0.245$ and 0.276, for average and maximum values, respectively). Further investigations of the effluent regression analyses were performed, because visual inspection of Figures D-3 and D-4 suggested the presence of outliers in the data.

We examined the fit of the linear model, TOC*avg* = *a* + *b* BOD*avg*, by inspecting its residuals (Figure D-5). Studentized residuals were plotted against projected (fitted) TOC values in the left pane, and against quantiles of the standard normal distribution in the right pane. Four suspiciously-low average TOC points in Figure D-4 are labeled '324', '308', plus the two points adjacent to the latter (left pane). This plot reveals that residuals for high-TOC points '489' and '64' are far larger in magnitude than residuals for the four suspiciously-low points. Residuals for these two points greatly deviate from the normal distribution (right pane). Furthermore, points '335' and '336' have much greater leverage than any other (leverages for '335': 0.164, '336': 0.127). Fitting the linear model without points '489', '64', '335', and '336' results in the following parameter values:

TOC*avg* = 6.0388 + 0.2171 BOD*avg* ($r^2 = 0.185$, 170 df) (Equation 1)

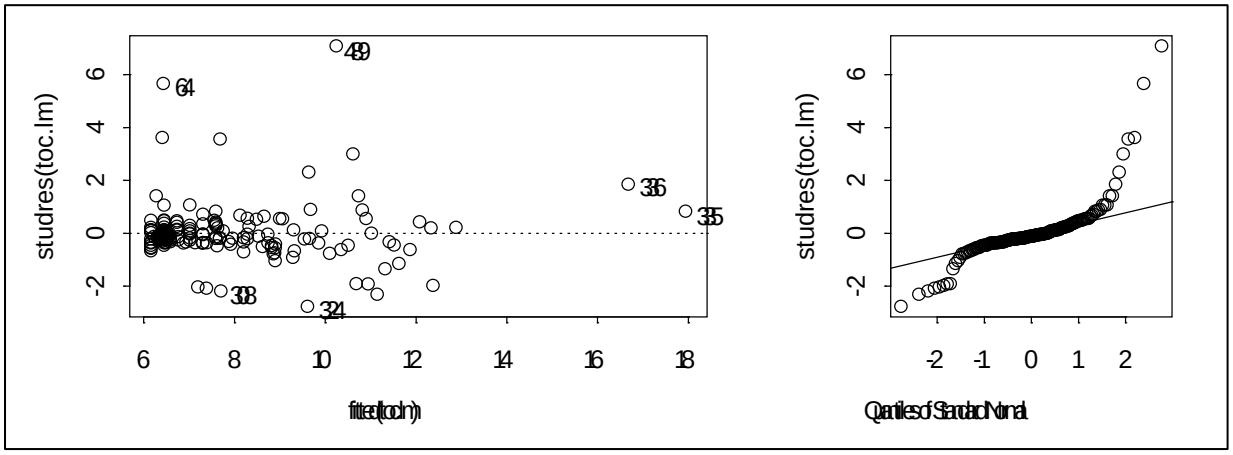

Figure D-5. Residuals of the linear model, TOCavg = a + b BODavg

(Left: plot of Studentized residuals (studres) against projected (fitted) TOC values; Right: plot of studies against quantiles of the standard normal distribution).

It should be noted that this model (Equation 1) projects average TOC concentrations very similar to the regression based upon the uncensored data (i.e., within ± 2 mg/L), for the range of BOD concentrations of interest (less than 30 mg/L).

Diagnosis of the regression analysis, TOC*max* = *a* + *b* BOD*max*, revealed an excessively high residue for the (19, 91) point and very high leverage for the (71, 31) point. The projected regression line without those two points was:

$$TOC_{max} = 6.6242 + 0.4095\ BOD_{max} \qquad (r^2 = 0.352,\ 297\ df)\ \text{(Equation 2)}$$

This model (Equation 2) fits a single slope for all data. Our results, though, revealed large differences in slopes of regression lines among POTWs (Table D-4). We tested the significance of such differences with an *F*-test, which required fitting two additional models, one with the same slope for all POTWs and the other with a distinct slope for each POTW. The *F*-test compares model fits while taking into account the loss in degrees of freedom associated with the computation of multiple slopes. The estimated *F*-value ($F = 4.92$, 13 df) was highly significant ($P < 0.001$), indicating that distinct slopes are necessary to accurately project maximum TOC from maximum BOD values.

D.3.3 TOC and DOC at CARP Effluent Monitoring Locations

Effluent discharge samples were collected from 11 New Jersey POTWs in 2000 and 2001 for the NYSDEC CARP project (www.dec.state.ny.us/website/dow/bwam/CARP). These samples were analyzed for DOC, particulate organic carbon (POC), and total suspended solids (TSS) by the U.S. Geological Survey. TOC was calculated by adding together DOC and POC concentrations. The results are shown in Table D-5. Effluent DOC concentrations are generally much higher than POC because most of the particulate organic matter is removed from wastewater during secondary treatment. A scatter plot of the TOC and DOC data, Figure D-6, shows the strong linear correlation between TOC and DOC that results from the predominance of DOC in effluent. These data are replotted in Figure D-7 for TOC concentrations less than 50 mg/L.

Table D-5. CARP organic carbon and total suspended solids (TSS) monitoring data for New Jersey discharger

DATE	SITE	DOC (mg/L)	POC (mg/L)	TOC (mg/L)	TSS (mg/L)
Oct. 2-4, 2000	PVSC	43.0	10.5	53.5	51.4
	MCMUA	0.10	6.75	6.85	36.3
	BCMUA	22.2	10.6	32.8	54.1
	JMEU	8.51	8.29	16.8	19.2
	RVMUA	12.2	3.81	16.0	22.1
	LRMUA	8.76	4.71	13.5	9.3
Dec. 11-15, 2000	PVSC	50.3	5.35	55.7	25.9
	MCMUA	260	9.22	269	62.6
	BCMUA	20.0	2.73	22.8	14.4
	JMEU	23.0	5.23	28.2	31.1
	RVMUA	23.4	8.73	32.1	42.0
	LRMUA	10.4	11.4	21.8	55.2
	NHH	14.0	3.07	17.1	22.5
	NBC	28.6	6.67	35.3	23.3
	NBW	21.8	3.38	25.2	7.8
	NHWNY	18.7	5.66	24.3	18.1
	SMUA	15.8	2.89	18.6	6.6
May 21-23, 2001	PVSC	34.5	14.2	48.7	41.1
	BCMUA	15.0	9.17	24.1	11.9

DATE	SITE	DOC (mg/L)	POC (mg/L)	TOC (mg/L)	TSS (mg/L)
	RVMUA	9.26	10.2	19.5	12.0
	LRMUA	14.7	9.34	24.1	10.9
	EMUA	0.25	0.15	0.40	19.9
August 6-9, 2001	PVSC	123	8.74	132	35.6
	MCMUA	20.6	5.34	25.9	22.6
	BCMUA	109	15.4	125	45.6
	JMEU	131	8.58	140	18.1
	RVMUA	8.78	3.39	12.2	6.7
	LRMUA	7.33	5.01	12.3	17.9
	NBC	191	8.39	199	17.4
	NBW	23.4	9.82	33.3	13.5
	EMUA	14.7	5.33	20.1	7.5
	NHWNY	17.7	12.1	29.8	13.8
	SMUA	10.7	2.67	13.4	3.8

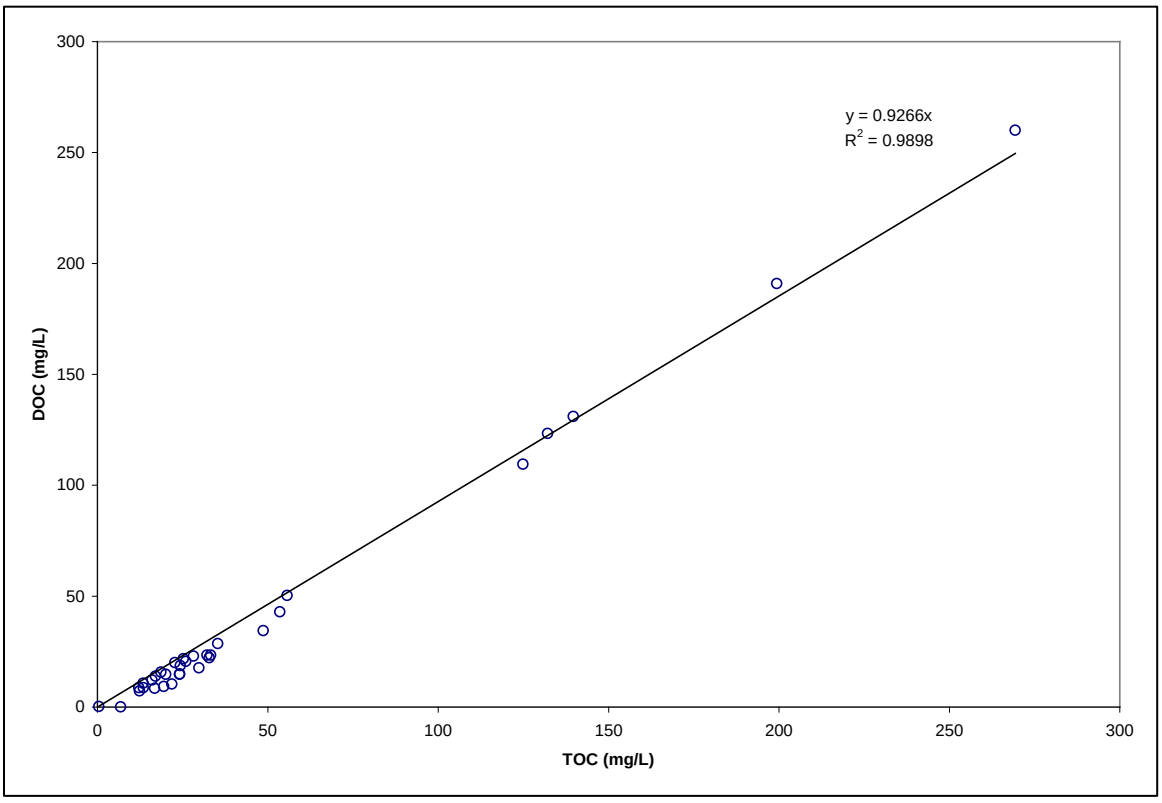

Figure D-6. Scatter plot of TOC versus DOC in CARP effluent monitoring data

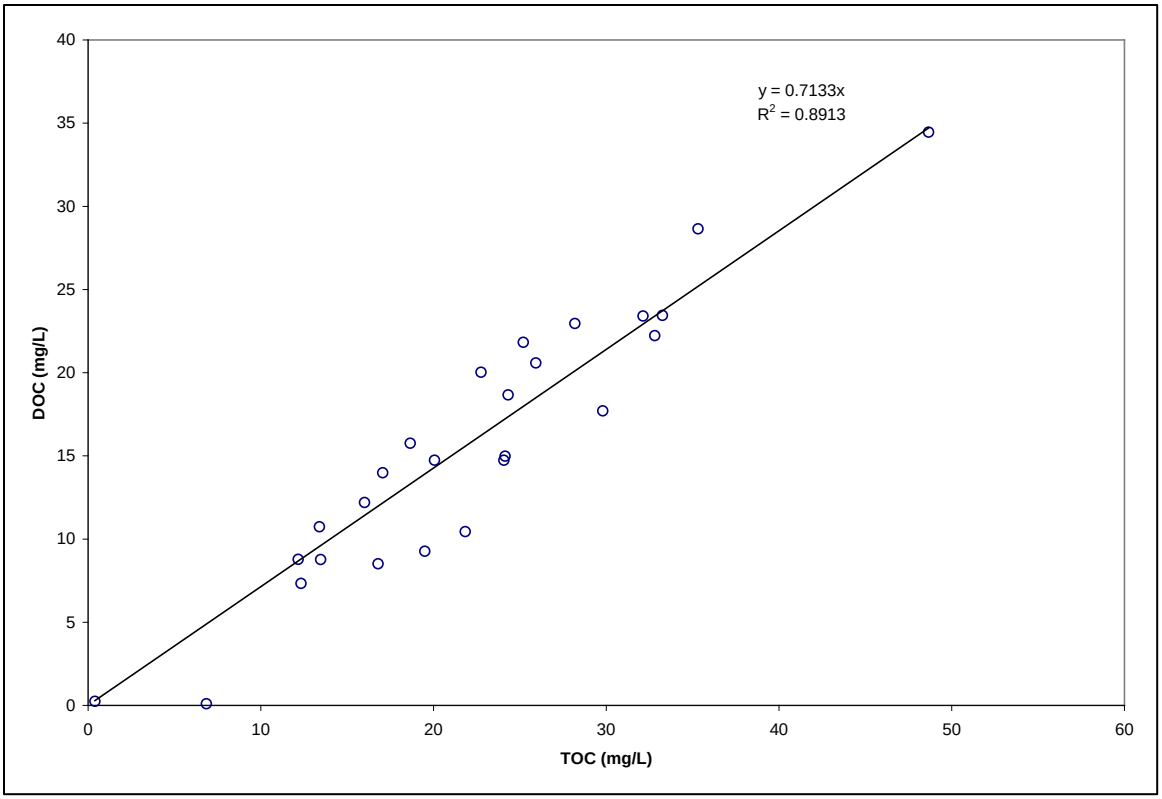

Figure D-7. Scatter plot of TOC versus DOC in CARP effluent monitoring data (TOC <50 mg/L)

Least squares regression of the CARP effluent data (Table D-5) produces the following model:

DOC = 0.9266 TOC (r^2 = 0.9898, 32 df)

This regression was forced through the origin by constraining the intercept to be zero. At the limit of removal efficiency (i.e., as effluent TOC approaches zero), any remaining TOC should be in the form of DOC, as mentioned above. This argument justifies forcing the regression through the origin. If only the data for which TOC falls in the expected range for effluent concentrations (TOC < 50 mg/L) are considered, the regression (again forced though the origin) is:

DOC = 0.7133 TOC (r^2 = 0.8913, 25 df)

For either case, the CARP effluent data show the strong linear relationship between TOC and DOC. Because TOC and DOC are linearly related in POTW effluent, the relationships between BOD and TOC reported above (Sections D.1 and D.2) also apply to DOC.

D.4 Discussion

Initially, we attempted to correlate BOD and TOC concentration measurements using data for all monitoring locations retrieved from PCS for major POTWs. We produced significant linear regression models for both average (Figure D-1) and maximum monthly (Figure D-2) data. Coefficients of determination were 0.87 and 0.70, respectively, for these data when combined for all locations. However, these correlations were substantially influenced by very high (i.e., greater than 50 mg/L) concentrations of BOD and TOC measured in untreated wastewater.

When we repeated the statistical analysis using effluent monitoring data, we found large differences in the slopes of the linear model TOC = *a* + *b* BOD among POTWs. Low coefficients of determination were also recorded for regressions of TOC on BOD values from all POTWs (r^2 = 0.245 and 0.276, for average and maximum values, respectively). In part, this may reflect random errors in the measurements of BOD and TOC, since data quality issues including loss of precision tend to be more frequent and significant at lower concentrations. The greater scatter in the plots of effluent BOD and TOC (Figures D-3 and D-4) may also reflect the limitations of working with the monthly average and maximum data reported by PCS.

Direct inspection of the TOC data in Figures D-3 and D-4 is nevertheless instructive. Aside from some extreme high and low values, the great majority of effluent TOC concentrations are in the range of 5 to 10 mg/L, especially for effluents with BOD concentrations below 10 mg/L. This is true for both average and maximum monthly TOC values. Table D-6 presents summary statistics for average monthly effluent TOC, for all data as well as data categorized according to the following BOD ranges: ≤5 mg/L, 5 to 10 mg/L, 10 to 20 mg/L and > 20 mg/L. As noted in Table D-6, four very low TOC values (≤ 0.5 mg/L) were judged to be anomalies and were therefore censored from the data for these statistics. The same summary statistics are presented for maximum monthly effluent TOC in Table D-7. In this context, the regressions of TOC on BOD values from effluent samples at all POTWs are quite reasonable, despite the low coefficients of determination. For average monthly effluent data, the regression of TOC on BOD is:

$$TOC_{avg} = 5.8740 + 0.2859\ BOD_{avg} \quad (r^2 = 0.245,\ 174\ df)$$

Table D-6. Summary statistics for POTW average monthly effluent TOC concentrations, categorized according to average monthly effluent BOD concentration

TOC_{avg}	BODavg level				
	≤5 mg/L	>5-10 mg/L	>10-20 mg/L	>20 mg/L	All levels
Mean	6.75	8.45	9.83	13.32	7.98
Median	6.17	8.15	8.70	13.25	6.90
Standard Deviation	2.47	2.41	5.26	5.79	3.76
5^{th} quantile	4.90	6.08	4.46	6.53	4.96
95^{th} quantile	10.00	10.58	18.40	21.8	14.52
n	98*	32*	34*	8	172*

*Four suspiciously-low TOC values were censored from the data for these statistics

Table D-7. Summary statistics for POTW maximum monthly effluent TOC concentrations, categorized according to maximum monthly effluent BOD concentration

TOCmax	BODmax level				
	≤5 mg/L	>5-10 mg/L	>10-20 mg/L	>20 mg/L	All levels
Mean	7.95	7.88	14.73	20.57	10.08
Median	7.30	7.85	11.00	20.60	7.90
Standard Deviation	3.28	2.74	15.97	7.75	7.60
5^{th} quantile	5.50	3.00	2.00	11.00	5.4
95^{th} quantile	10.94	12.00	25.55	35.3	23.6
n	164	72	30	35	301

Given the substantial limitations imposed by the data available from PCS, we believe that this regression gives reasonable estimates of TOC in POTW effluents. These are also probably the best available estimates of effluent TOC for dilution calculations to determine DOC concentrations for use in the BLM (for example, the probabilistic dilution framework incorporated in the BLM-Monte software [HydroQual, 2001]). As shown in Section D.3, effluent DOC concentrations can be reliably predicted from TOC values:

$$DOC = 0.7133\ TOC \qquad (r^2 = 0.8913,\ 25\ df)$$

It should be noted that the regressions presented here should not be applied to project water quality in natural receiving waters unimpacted by POTW effluent, because they are based solely on POTW effluent monitoring data. The characteristics of the constituents DOC, TOC, and BOD, as well as the relationships between them, may be quite dissimilar between natural waters and effluents.

D.5 References

Fadini, S.F., Jardim, W.F., J.R. Guimarães. 2004 Evaluation of organic load measurement techniques in a sewage and waste stabilisation pond. Journal of the Brazilian Chemical Society. 15(1): 131-135 (http://jbcs.sbq.org.br/jbcs/2004/vol15_n1/19-152-02.pdf).

HydroQual, Inc. 2001. BLM-Monte User's Guide, Version 2.0. HydroQual, Mahwah, NJ. October, 2001.

HydroQual, Inc. 2005. Biotic Ligand Model Windows Interface, Version 2.1.2, User's Guide and Reference Manual. HydroQual, Mahwah, NJ. June, 2005.

Comparison of Aquatic Life Use Criteria
for Aluminum
Prepared by New Mexico Environment Department Surface Water Quality Bureau
April 2021

Total Hardness (mg/L as *CaCO3)	* EPA's 2018 MLR Criteria (DOC=1 and pH=7)		EPA's 2018 MLR Criteria (DOC=5 and pH=7)		NMED's Hardness-Based Criteria (pH=7)	
	*CMC-Acute	*CCC-Chronic	CMC-Acute	CCC-Chronic	Acute	Chronic
10	430	240	970	430	146	59
20	570	290	1200	480	377	151
30	660	310	1400	510	658	263
40	730	330	1500	530	975	391
50	790	340	1600	550	1324	530
60	840	350	1700	570	1699	681
70	880	360	1800	580	2099	841
80	920	360	1900	590	2520	1010
90	950	370	2000	610	2961	1186
100	980	380	2100	620	3421	1370
110	1000	380	2200	630	3898	1562
120	1000	390	2200	640	4391	1759
130	1100	390	2300	650	4899	1963
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160	1100	400	2400	670	6511	2609
170	1200	410	2500	680	7075	2834
180	1200	410	2500	690	7651	3065
190	1200	410	2600	690	8239	3301
200	1200	420	2600	700	8838	3541
210	1200	420	2700	700	9449	3786
220	1300	420	2700	710	10071	4035
230	1300	430	2700	710	10071	4035
240	1300	430	2800	720	10071	4035
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260	1300	430	2800	730	10071	4035
270	1400	430	2900	730	10071	4035
280	1400	440	2900	740	10071	4035
290	1400	440	2900	740	10071	4035
300	1400	440	3000	750	10071	4035
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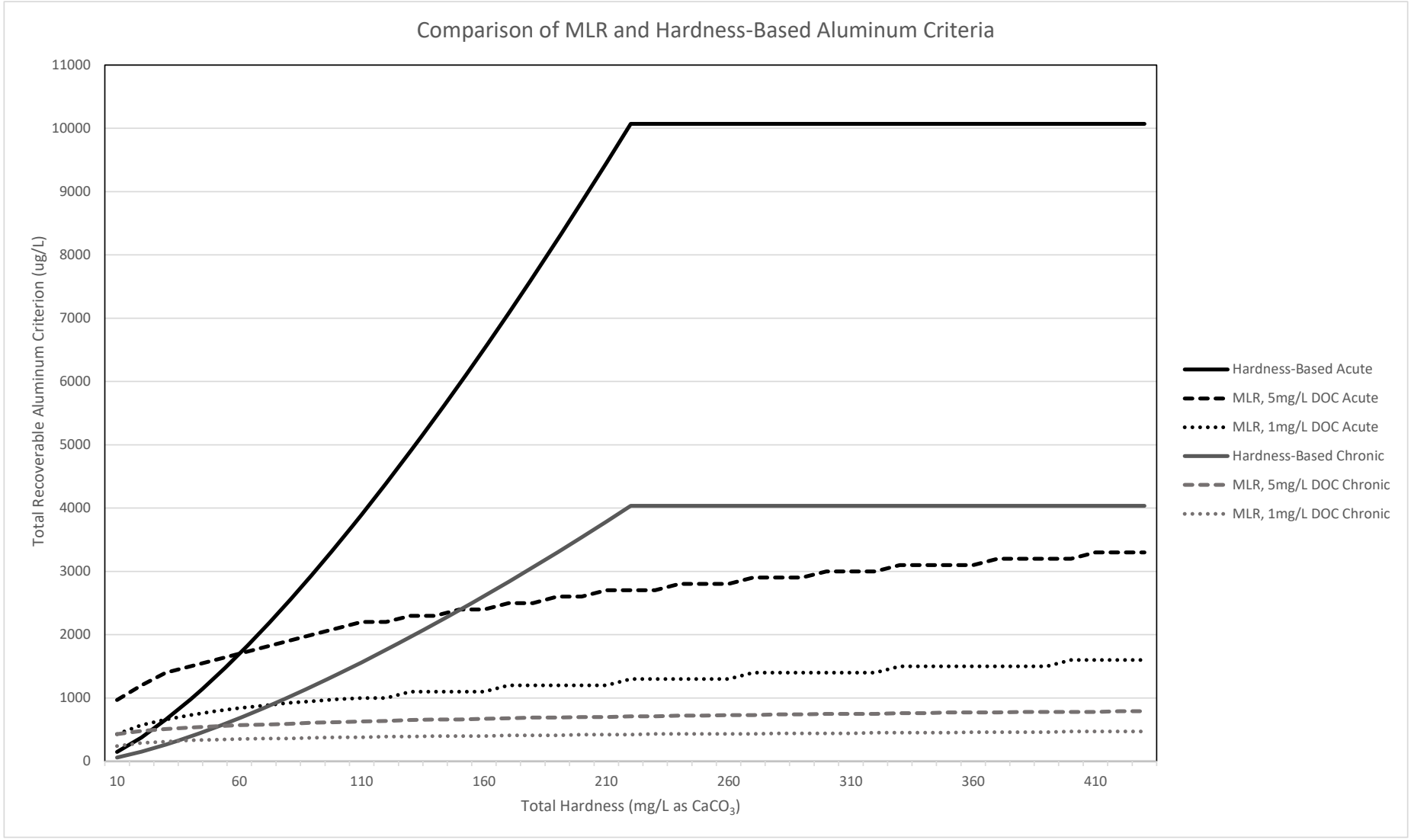
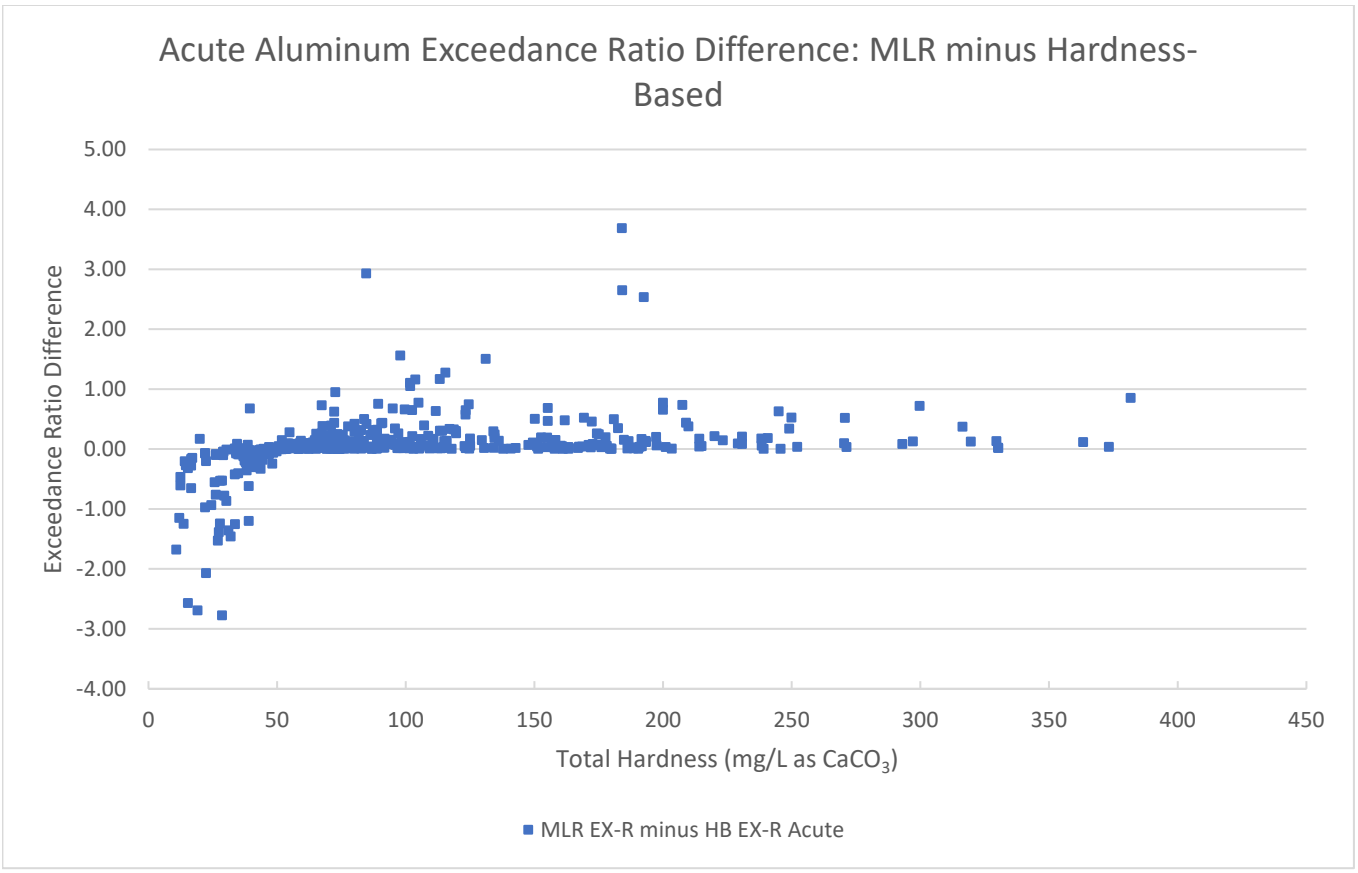
390	1500	460	3200	780	10071	4035
400	1600	470	3200	780	10071	4035
410	1600	470	3300	780	10071	4035
420	1600	470	3300	790	10071	4035
430	1600	470	3300	790	10071	4035

* Abbreviations		
	CaCO3	Total hardness of water expressed as calcium carbonate
	CMC	Criterion Maximum Concentration
	CCC	Criterion Continuous Concentration
	DOC	Dissolved Organic Carbon
	MLR	Multi-Linear Regresson



NMED Exhibit 45


Acute Aluminum Exceedance Ratio Difference: MLR minus Hardness-Based
Exceedance Ratio Difference
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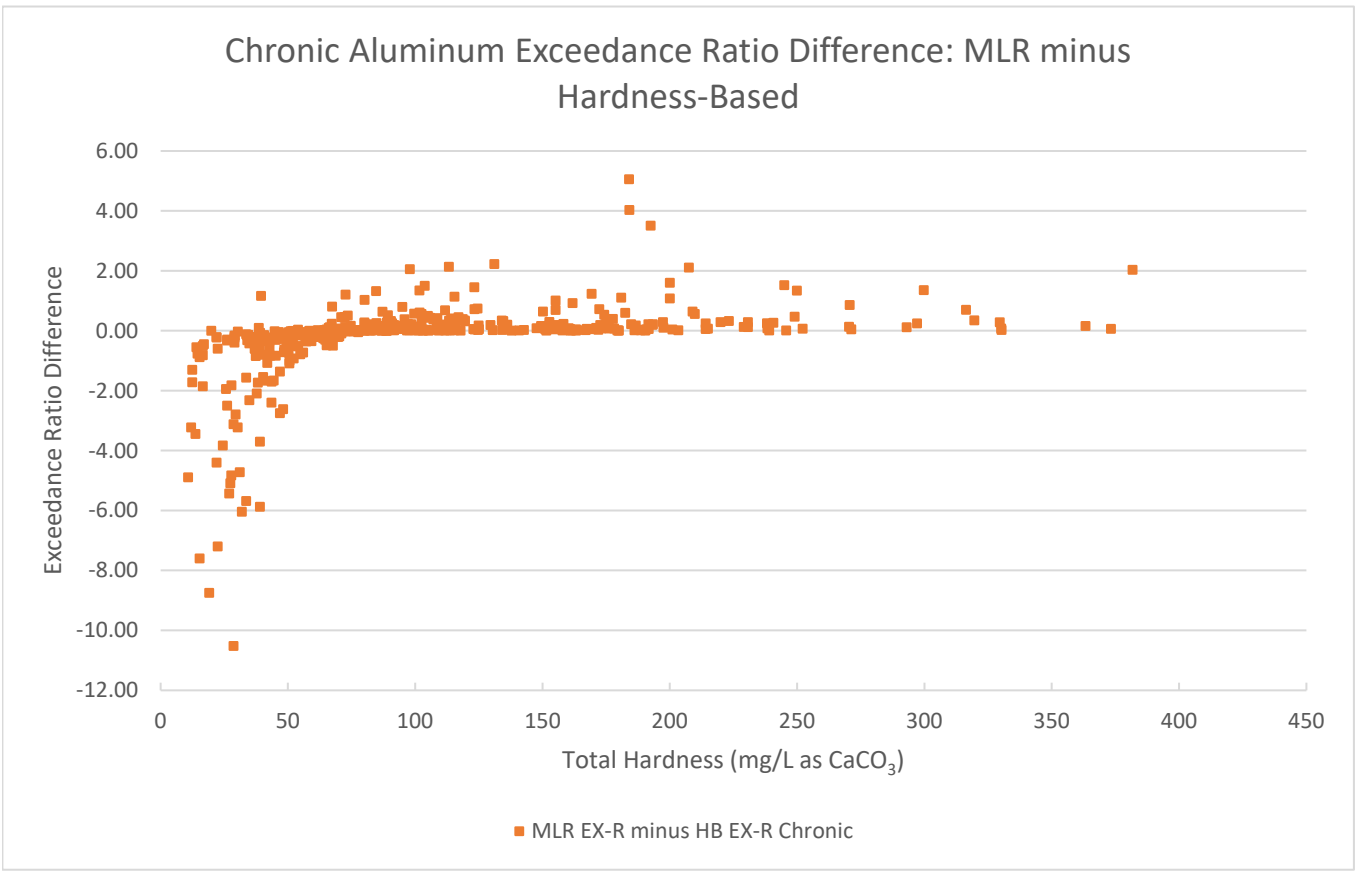
4.00
3.00
2.00
1.00
0.00
-1.00
-2.00
-3.00
-4.00
0
50
100
150
200
250
300
350
400
450
Total Hardness (mg/L as CaCO3)
MLR EX-R minus HB EX-R Acute


Chronic Aluminum Exceedance Ratio Difference: MLR minus Hardness-Based
Exceedance Ratio Difference
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-2.00
-4.00
-6.00
-8.00
-10.00
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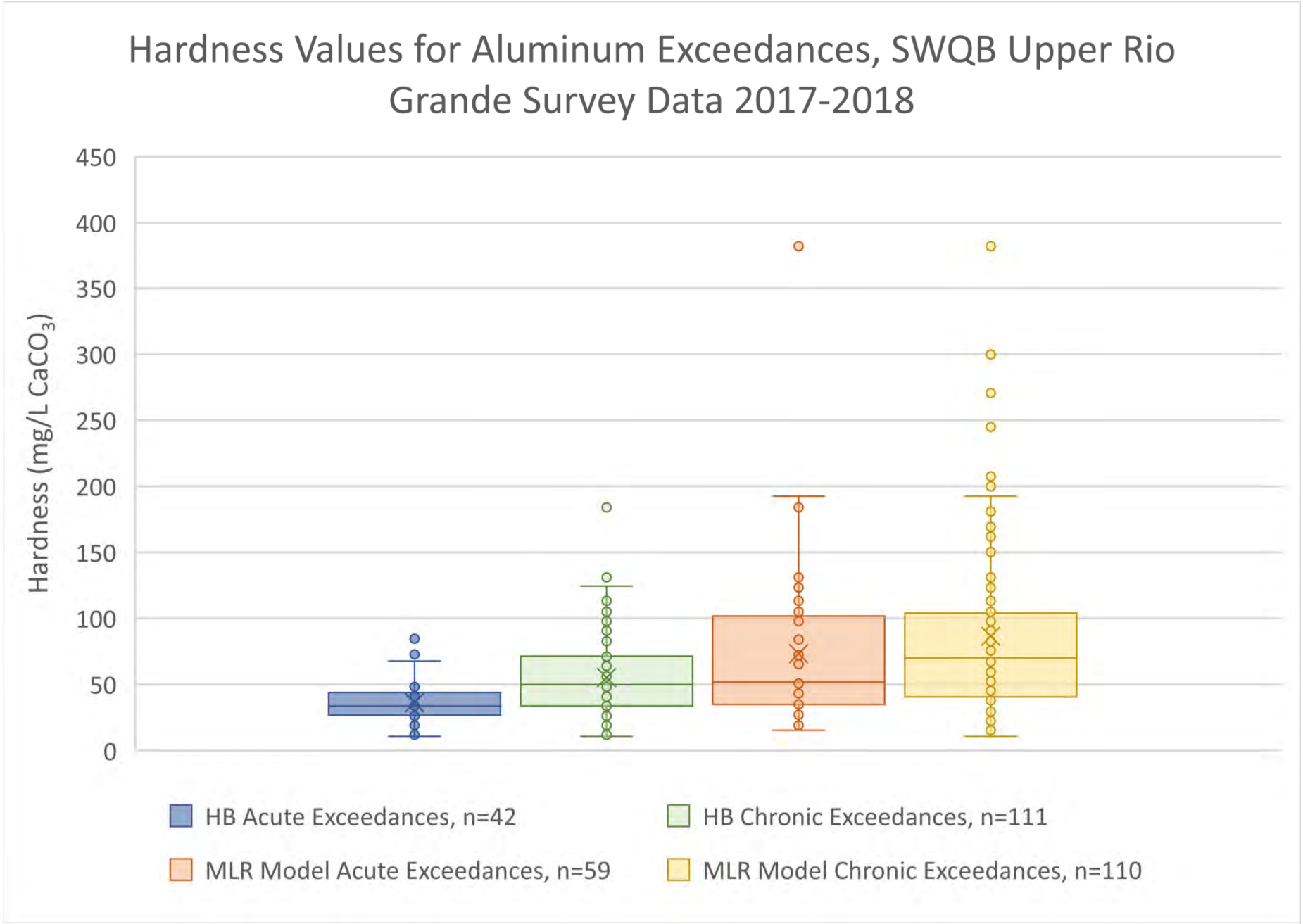
100
150
200
250
300
350
400
450
Total Hardness (mg/L as CaCO3)
MLR EX-R minus HB EX-R Chronic



NMED Exhibit 47

335. The Commission adopts the identical NMED and UC proposals to revise the chronic and acute criteria for mercury to be consistent with EPA's recommended criteria pursuant to 40 CFR §131.11.

336. The Commission adopts the identical NMED and UC proposals to add a methylmercury criterion of 0.3 mg/kg for protection of human health, as recommended by EPA in 2001.

337. The Commission adopts NMED's proposed revised nitrate criteria based on the research of Dr. Sam Fernald of New Mexico State University, who demonstrated that the revised criteria protect livestock watering.

338. Regarding the "Aquatic Life" criteria the Commission adopts the identical NMED and UC proposals to delete the beryllium criteria because EPA has withdrawn its recommended values. EPA no longer recommends beryllium aquatic life criteria.

339. Regarding the "Human Health" criteria the Commission adopts NMED's proposal to amend the criteria based upon the current EPA recommendations in National Recommended Water Quality Criteria: 2002, EPA-822-R-02-047. The recalculated criteria integrate an updated national default fish consumption rate (17.5 g/day) and, in some cases, relative source contribution values obtained from primary drinking water standards and new cancer potency information from EPA's Integrated Risk Information System.

340. Of these criteria, only the arsenic criterion is New Mexico-specific (e.g., the updated national default fish consumption rate applied to site-specific data collected during a 1997 joint agency study of arsenic in the middle Rio Grande in the vicinity of Albuquerque. The site specific data included: (1) geometric mean of dissolved arsenic concentrations from all river and drain stations of 2.88 mg/L; (2) geometric mean of total arsenic concentrations in eight composited fish-tissue samples from fish collected in the river and drains of 13.13 µg/kg; and (3) risk assumptions, including (a) risk level = 10^{-5}; (b) body weight = 70 kg; (c) cancer potency factor = 1.5; (d) bioaccumulation factor = 4.57 L/kg (geomean tissue 13.13/ geomean H20 2.88); and (e) inorganic As = 65 percent). The recalculation resulted in an arsenic criterion of 9.0 µg/L for consumption of organisms only, and 2.3 µg/L for consumption of water plus organisms. The Commission

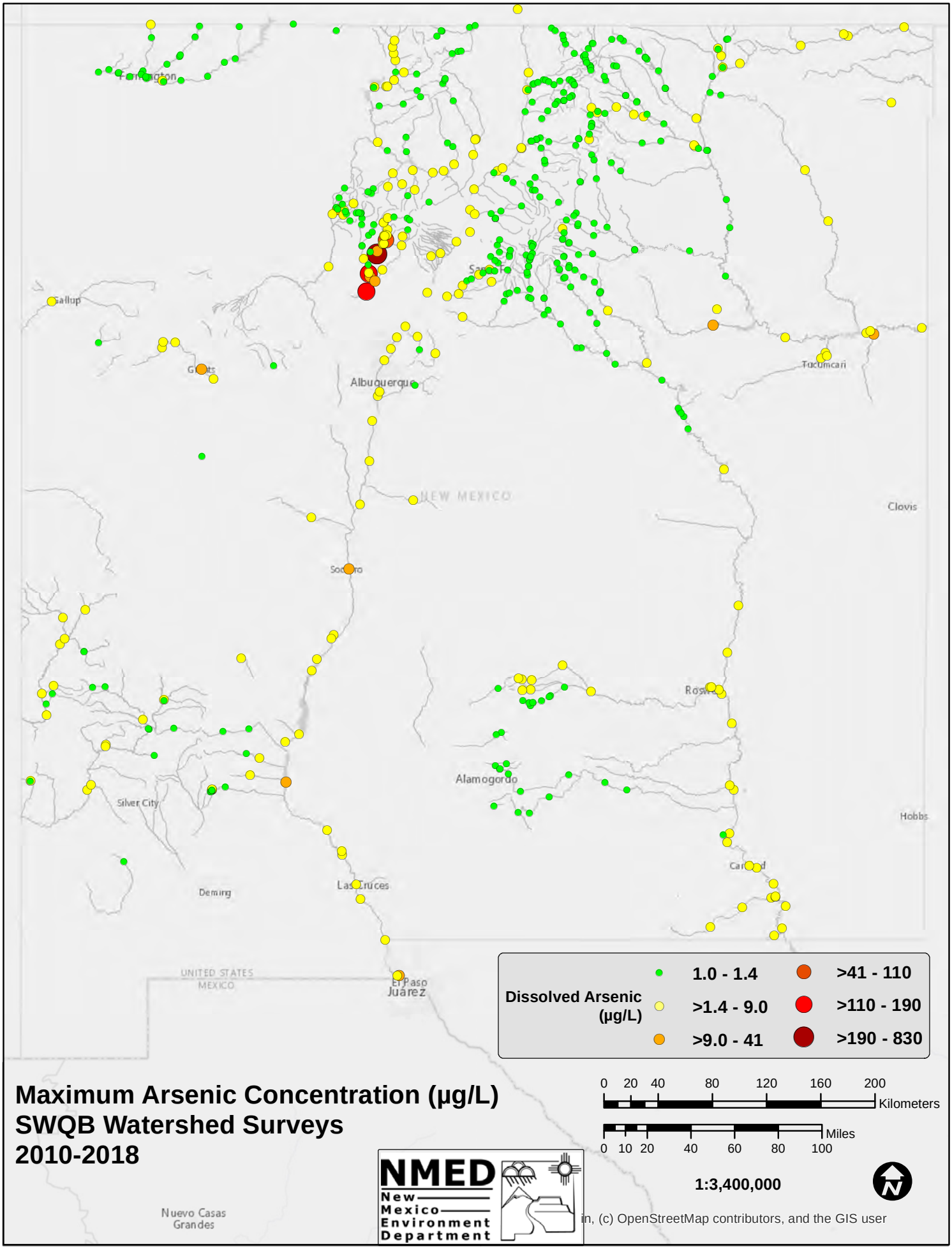
expresses its concern to the department that the assumptions of the level of fish consumption from the Rio Grande may be overstated.

341. The Commission rejects AB's proposal to include numeric criteria for perchlorate-domestic water supply- at 1 ug/L for lack of credible scientific data in the record, and because EPA has not recommended a criterion for ambient waters.

342. The Commission rejects AB's proposal to change the criteria for uranium, dissolved- Domestic Water Supply to 7 ug/L for lack of credible scientific data in the record, and because EPA has not recommended a criterion for ambient waters. The Commission is concerned about uranium, and mindful that it recently lowered the ground water standard, but the record in this triennial review simply does not support such a change at this time.

343. The Commission rejects AB's proposed wildlife habitat dissolved aluminum numeric criterion, proposed aquatic life cyanide numeric criteria, and proposed wildlife habitat and aquatic life selenium numeric criteria for lack of credible scientific data in the record.

[tables begin on next page]



Maximum Arsenic Concentration (µg/L)
SWQB Watershed Surveys
2010-2018

NMED Exhibit 49

United States
Environmental Protection
Agency

Office of Water
4304T

EPA-822-R-07-001
February 2007

AQUATIC LIFE AMBIENT FRESHWATER QUALITY CRITERIA - COPPER

2007 Revision

AQUATIC LIFE AMBIENT FRESHWATER QUALITY CRITERIA - COPPER

2007 Revision

(CAS Registry Number 7440-50-8)

February 2007

U.S. Environmental Protection Agency
Office of Water
Office of Science and Technology
Washington, DC

NOTICES

This document has been reviewed in accordance with U.S. EPA policy and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

This document can be downloaded from EPA's website at:
http://www.epa.gov/waterscience/criteria/aqlife.html

FOREWORD

Section 304(a)(l) of the Clean Water Act of 1977 (P.L. 95-217) requires the Administrator of the Environmental Protection Agency to publish water quality criteria that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that might be expected from the presence of pollutants in any body of water, including ground water. This document is a revision of criteria based upon consideration of comments received from independent peer reviewers and the public. Criteria contained in this document supplement any previously published EPA aquatic life criteria for the same pollutant(s).

The term "water quality criteria" is used in two sections of the Clean Water Act, section 304(a)(l) and section 303(c)(2). The term has a different program impact in each section. In section 304, the term represents a non-regulatory, scientific assessment of health or ecological effects. Criteria presented in this document are such scientific assessments. If water quality criteria associated with specific waterbody uses are adopted by a state or tribe as water quality standards under section 303, they become enforceable maximum acceptable pollutant concentrations in ambient waters within that state or tribe. Water quality criteria adopted in state or tribal water quality standards could have the same numerical values as criteria developed under section 304. However, in many situations states or tribes might want to adjust water quality criteria developed under section 304 to reflect local environmental conditions. Alternatively, states or tribes may use different data and assumptions than EPA in deriving numeric criteria that are scientifically defensible and protective of designated uses. It is not until their adoption as part of state or tribal water quality standards that criteria become regulatory. Guidelines to assist the states and tribes in modifying the criteria presented in this document are contained in the Water Quality Standards Handbook (U.S. EPA 1994). The handbook and additional guidance on the development of water quality standards and other water-related programs of this agency have been developed by the Office of Water.

This document is guidance only. It does not establish or affect legal rights or obligations. It does not establish a binding norm and cannot be finally determinative of the issues addressed. Agency decisions in any particular situation will be made by applying the Clean Water Act and EPA regulations on the basis of specific facts presented and scientific information then available.

Ephraim S. King, Director
Office of Science and Technology

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ACRONYMS

ACR	Acute-Chronic Ratio
BL	Biotic Ligand
BLM	Biotic Ligand Model
CCC	Criterion Continuous Concentration
CF	Conversion Factors
CMC	Criterion Maximum Concentration
CWA	Clean Water Act
DIC	Dissolved Inorganic Carbon
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
EC	Effect Concentration
EPA	Environmental Protection Agency
FACR	Final Acute-Chronic Ratio
FAV	Final Acute Value
FCV	Final Chronic Value
FIAM	Free Ion Activity Model
GMAV	Genus Mean Acute Value
GSIM	Gill Surface Interaction Model
LC50	Lethal Concentration at 50 Percent Effect Level
LOAEC	Lowest Observed Adverse Effect Concentration
NASQAN	National Stream Quality Accounting Network
NOAEC	No Observed Adverse Effect Concentration
pH	Negative logarithm of the concentration (mol/L) of the $H_3O^+[H^+]$ ion; scale range from 0 to 14
SMAV	Species Mean Acute Values
STORET	EPA STOrage and RETrieval Data System
WER	Water-Effect Ratio
WET	Whole Effluent Toxicity
WQC	Water Quality Criteria

1.0 INTRODUCTION

Copper is an abundant trace element found in the earth's crust and is a naturally occurring element that is generally present in surface waters (Nriagu, 1979). Copper is a micronutrient for both plants and animals at low concentrations and is recognized as essential to virtually all plants and animals (Kapustka et al., 2004). However, it may become toxic to some forms of aquatic life at elevated concentrations. Thus, copper concentrations in natural environments, and its biological availability, are important. Naturally occurring concentrations of copper have been reported from 0.03 to 0.23 µg/L in surface seawaters and from 0.20 to 30 µg/L in freshwater systems (Bowen, 1985). Copper concentrations in locations receiving anthropogenic inputs can vary anywhere from levels that approach natural background to 100 µg/L or more (e.g., Lopez and Lee, 1977; Nriagu, 1979; Hem, 1989) and have in some cases been reported in the 200,000 µg/L range in mining areas (Davis and Ashenberg, 1989; Robins et al., 1997). Mining, leather and leather products, fabricated metal products, and electric equipment are a few of the industries with copper-bearing discharges that contribute to anthropogenic inputs of copper to surface waters (Patterson et al., 1998).

Over the past 20 years, the U.S. Environmental Protection Agency (EPA) has published a number of guidance documents containing aquatic life criteria recommendations for copper (e.g., U.S. EPA 1980, 1985, 1986, 1996). The present document contains EPA's latest criteria recommendations for protection of aquatic life in ambient freshwater from acute and chronic toxic effects from copper. These criteria are based on the latest available scientific information, supplementing EPA's previously published recommendations for copper. This criteria revision incorporated new data on the toxicity of copper and used the biotic ligand model (BLM), a metal bioavailability model, to update the freshwater criteria. With these scientific and technical revisions, the criteria will provide improved guidance on the concentrations of copper that will be protective of aquatic life. The BLM is not used in the saltwater criteria derivation because further development is required before it will be suitable for use to evaluate saltwater data.

This document provides updated guidance to states and authorized tribes to establish water quality standards under the Clean Water Act (CWA) to protect aquatic life from elevated copper exposure. Under the CWA, states and authorized tribes are to establish water quality criteria to protect designated uses. Although this document constitutes EPA's scientific recommendations regarding ambient concentrations of copper, it does not substitute for the CWA or EPA's regulations, nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, states, tribes, or the regulated community, and might not apply to a particular situation based on the circumstances. State and tribal decision makers retain the discretion in adopting approaches, on a case-by-case basis, that differ from this guidance when appropriate. EPA may change this guidance in the future.

Although the BLM has been used in place of the formerly applied hardness-based approach, the updated freshwater criteria derivations in this document are still based on the principles set forth in the *Guidelines for Deriving Numerical Water Quality Criteria for the Protection of Aquatic Life and Their Uses* (Stephan et al. 1985, hereafter referred to as the Guidelines). Section 2 of this document provides an overview of copper bioavailability and the BLM. Additional information on the generalized BLM framework, theoretical background, model calibration, and application for the BLM can be found in the published literature. Section 3 of this document discusses general

procedures and requirements for applying the BLM to criteria. Section 4 provides the derivation of criteria Final Acute Value (FAV) and Final Chronic Value (FCV) for freshwater organisms. Section 5 discusses plant data and Section 6 discusses other data not included in the criteria derivation. Sections 7 and 8 provide the final criteria statements and information on implementation. Various supplementary information is provided in several appendices.

2.0 APPROACHES FOR EVALUATING COPPER BIOAVAILABILITY

2.1 *General Aspects of Copper Bioavailability*

The toxicity of a chemical to an aquatic organism requires the transfer of the chemical from the external environment to biochemical receptors on or in the organism at which the toxic effects are elicited. Often, this transfer is not simply proportional to the total chemical concentration in the environment, but varies according to attributes of the organism, chemical, and exposure environment so that the chemical is more or less "bioavailable". Definitions of bioavailability vary markedly (e.g., National Research Council, 2003) and are often specific to certain situations, but a useful generic definition is the relative facility with which a chemical is transferred from the environment to a specified location in an organism of interest.

Of particular importance to bioavailability is that many chemicals exist in a variety of forms (chemical species). Such chemical speciation affects bioavailability because relative uptake rates can differ among chemical species and the relative concentrations of chemical species can differ among exposure conditions. At equilibrium in oxygenated waters, "free" copper exists as cupric ion - Cu(II) weakly associated with water molecules ($Cu{\cdot}nH_2O^{+2}$), but this species is usually a small percentage of the total copper. Most dissolved copper is part of stronger complexes with various ligands (complexing chemicals that interact with metals), including dissolved organic compounds, hydroxides, carbonates, and other inorganic ligands. Substantial amounts of copper can also be adsorbed to or incorporated into suspended particles. More information on copper speciation in freshwater can be found in Kramer et al. (1997), Bryan et al. (2002), and Smith et al. (2002).

Copper toxicity has been reported to vary markedly due to various physicochemical characteristics of the exposure water (e.g., either laboratory or field), including temperature, dissolved organic compounds, suspended particles, pH, and various inorganic cations and anions, including those composing hardness and alkalinity (see reviews by Sprague, 1968; Hunt, 1987; Campbell, 1995; Allen and Hansen, 1996; Paquin et al., 2002). Many of these physicochemical factors affect copper speciation, and their effects on copper toxicity therefore could be due to effects on copper bioavailability. That bioavailability is an important factor is evident from uptake of copper by aquatic organisms being reduced by various organic compounds and inorganic ligands known to complex copper (Muramoto, 1980; Buckley et al., 1984; Playle et al., 1993 a,b; MacRae et al., 1999).

A "ligand" is a complexing chemical (ion, molecule, or molecular group) that interacts with a metal like copper to form a larger complex. A "biotic ligand" is a complexing chemical that is a component of an organism (e.g. chemical site on a fish gill). For certain ligands, some studies have demonstrated that the concentration of free copper associated with a specified level of accumulation or toxicity changes little as the ligand concentration is varied, despite major changes in the

proportion of copper bound to the ligand (see review by Campbell, 1995). This suggests that, even at low concentrations, free copper is more important to bioavailability than the ligand-bound copper. This is expected if accumulation and toxicity are dependent on the binding of copper to a biochemical receptor "X" on the surface of the organism, forming a chemical species X-Cu (receptor-bound metal) that is a first limiting step in accumulation and toxicity. By standard chemical equilibrium expressions, the amount of such species and the consequent biological effects would be a function of the activity of just free copper (Morel, 1983 a), a relationship commonly referred to as the free ion activity model (FIAM). Ligand-bound copper (Cu-L) would contribute to copper bioavailability if (a) a species X-Cu-L is formed that is important to copper accumulation/toxicity, (b) the microenvironment near the organism surface is such that Cu-L dissociates and increases the free copper activity interacting with "X", or (c) copper uptake is via mechanisms that do not entail binding to such a receptor and can accommodate different copper species. Some studies have indicated dissolved complexes of copper do contribute to bioavailability (reviews by Sprague, 1968; Hunt, 1987; Campbell, 1995; Allen and Hansen, 1996; Paquin et al., 2002).

The effects of physicochemical factors on copper toxicity are diverse and the specific chemistry of the exposure water will determine whether or not there are appreciable effects on copper speciation and a resulting strong relationship of toxicity to free copper. Usually copper toxicity is reduced by increased water hardness (reviews by Sprague, 1968; Hunt, 1987; Campbell, 1995; Allen and Hansen, 1996; Paquin et al., 2002), which is composed of cations (primarily calcium and magnesium) that do not directly interact with copper in solution so as to reduce bioavailability. In some cases, the apparent effect of hardness on toxicity might be partly due to complexation of copper by higher concentrations of hydroxide and/or carbonate (increased pH and alkalinity) commonly associated with higher hardness. However, significant effects on toxicity often are still present when hardness is increased in association with anions which do not interact strongly with copper (Inglis and Davis, 1972; Chakoumakos et al., 1979; Miller and Mackay, 1980; Erickson et al., 1987). Hardness cations could have some limited effect on copper speciation by competing with copper for the same dissolved ligands, but increased hardness would then increase free copper and thus increase, not decrease, toxicity. Sodium has also been reported to affect copper toxicity (Erickson et al., 1996 b) and pH effects can be partly due to effects of hydrogen ion other than on copper speciation (Peterson et al., 1984).

The effects of hardness cations could be explained by the competing with copper for the biochemical receptor "X", thus reducing copper uptake (Zitko, 1976; Zitko et al., 1976; Pagenkopf, 1983). Reduced metal bioavailability due to increased hardness cations has been experimentally demonstrated (Playle et al., 1992; Meyer et al., 1999, 2002), although this does not specifically establish cation competition as the mechanism. Pagenkopf (1983) provided a mathematical description of a Gill Surface Interaction Model (GSIM) that addressed the effects on metal toxicity of both metal speciation and cations via the interactions of gill surface biochemical receptors with the free toxic metal, other metal species, hardness cations, and hydrogen ion.

The empirical evidence demonstrates that copper toxicity is affected by exposure conditions and that much of these effects is plausibly attributed to effects of ligands and cations on copper bioavailability. However, it should not be presumed that all of the observed effects of the physicochemical factors on copper toxicity reflect effects on bioavailability, or that bioavailability

effects are just due to ligand complexation and cation competition. For example, acute copper toxicity in aquatic organisms has been related to disruption of osmoregulation, specifically sodium/potassium exchange (Lauren and MacDonald, 1986; Wood, 1992; Wood et al., 1997; Paquin et al., 2002), which can be affected by calcium other than by competition with copper for the same biochemical receptor. Similarly, reported effects of sodium and potassium on copper toxicity (Erickson et al., 1996 b) might simply reflect favorable or unfavorable ion exchange gradients, rather than any effect on copper bioavailability. Nevertheless, the effects of ligand complexation and cation competition on copper bioavailability provide a reasonable conceptual framework for improved descriptions of how copper toxicity differs across exposure conditions.

2.2 *Existing Approaches*

EPA aquatic life criteria for metals address the reported effects of hardness on metal toxicity using empirical regressions of toxic concentrations versus hardness for available toxicity data across a wide range of hardness (Stephan et al., 1985). Such regressions provided the relative amount by which the criteria change with hardness, but have certain limitations. The regressions were not just of hardness, but of any other factor that was correlated with hardness in the toxicity data set used for the regressions, particularly pH and alkalinity. Although these regressions therefore address more bioavailability issues than hardness alone, they best apply to waters in which the correlations among hardness, pH, and alkalinity are similar to the data used in the regressions. The separate effects of these factors are not addressed for exposure conditions in which these correlations are different. In addition, some physicochemical factors affecting metal toxicity, such as organic carbon, are not addressed at all.

Existing EPA metals criteria also address bioavailability by using dissolved metal as a better approximation for metal bioavailability than total metal (U.S. EPA, 1993). Although this approach accounts for the low bioavailability of metal on suspended particles, it does not address the major effects of various dissolved species on bioavailability. This approach could conceivably be further developed to include just part of the dissolved copper, but this not only requires resolving what species to include, how to weight them, and how to assess their concentrations, but also would not address the effects of cations and other factors that affect toxicity in addition to metal speciation. Such a "bioavailable fraction" approach is not justified, because no fraction of metals species provides a constant measure of toxicity.

To address more completely the modifying effects of water quality than the hardness regressions achieve, EPA issued guidance in the early 1980s on the water-effect ratio (WER) method (Carlson et al., 1984; U.S. EPA, 1983, 1992, 1994). The WER is "a biological method to compare bioavailability and toxicity in receiving waters versus laboratory test waters" (U.S. EPA, 1992). A WER is calculated by dividing the acute LC50 of the metal, determined in water collected from the receiving water of interest, by the LC50 of the metal determined in a standard laboratory water, after adjusting both test waters to the same hardness. The standard laboratory water LC50 is used as the denominator to reflect that this LC50 is measured in test water that has water quality characteristics representative of the test waters used to develop the Water Quality Criteria (WQC) toxicity database, at least as a good approximation. The national hardness-based acute criterion concentration is then multiplied by this ratio (i.e., the WER) to establish a site-specific criterion that reflects the effect of site water characteristics on toxicity. However, a WER accounts only for

interactions of water quality parameters and their effects on metal toxicity to the species tested and in the water sample collected at a specific location and at a specific time. There is also significant cost to generate a single WER.

Because of the limitations of these past approaches for addressing bioavailability in metals criteria, there is a need for an approach that (1) explicitly and quantitatively accounts for the effect of individual water quality parameters that modify metal toxicity and (2) can be applied more cost-effectively and easily, and hence more frequently across spatial and temporal scales. An assessment framework that incorporates the bioavailability mechanisms discussed in Section 2.1 was therefore used to address more comprehensively the effects of physicochemical exposure conditions on copper toxicity with lower costs than required by the WER approach.

2.3 *The Biotic Ligand Model and Its Application to Criteria Development*

The interactions of toxic metal species and other exposure water constituents with biological surface receptors described by Zitko (1976), Morel (1983), and Pagenkopf (1983) provided the basic conceptual and mathematical structure for the bioavailability model to be used here (Figure 1). Subsequent experimental work has supported various model tenets by demonstrating the effects of complexing ligands and competing cations on accumulation of toxic metals at fish gills and the relationship of toxic effects to accumulation, and has also provided estimates of various model parameters (Playle et al., 1992, 1993a,b; Janes and Playle, 1995; MacRae et al., 1999, Meyer et al., 1999, 2002; McGeer et al., 2002). Various efforts in metal speciation modeling also have provided the ability to do better speciation calculations, especially regarding complexation of metals by organic matter (e.g., Tipping, 1994). This experimental work has supported further metal toxicity model development (Meyer, 1999; Brown and Markich, 2000; McGeer et al., 2002; Di Toro et al., 2001; Santore et al., 2001; Paquin et al., 2002). This bioavailability modeling approach is now commonly termed "Biotic Ligand Models" to broaden the scope beyond gill surfaces and to acknowledge that the biochemical receptor "X" discussed in Section 2.1 is a metal-binding ligand that is treated similarly to ligands in the exposure water, except that it is on the organism and is the keystone for metal accumulation and toxicity.

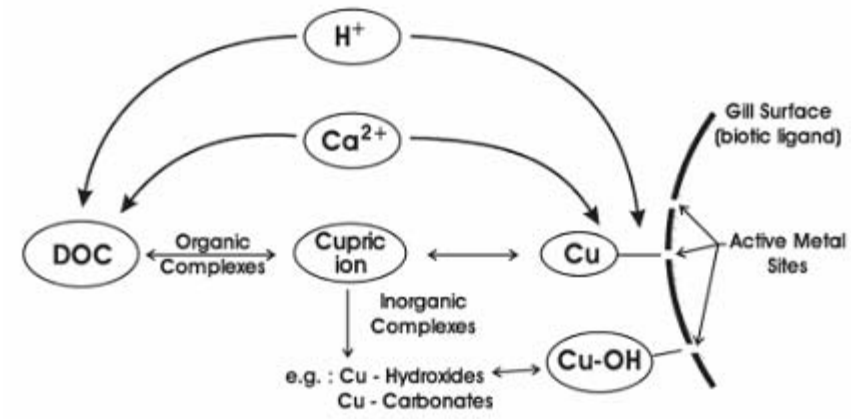

Figure1. Conceptual Diagram of Copper Speciation and Copper-Gill Model (after Pagenkopf, 1983)

Briefly, available evidence indicates that both free copper and copper monohydroxide bind to a biotic ligand "Lb" on the organism's surface (Lb-Cu and Lb-CuOH) and that death occurs when a certain amount of the total biotic ligand sites are occupied by copper. This ligand must be at the organism surface because the model describes its interactions with the external exposure water. However, this does not mean that this ligand is the site of toxic action; rather it is only necessary to assume that copper accumulation at the site(s) of toxic action is proportional to binding at the biotic ligand (i.e., the biotic ligand controls bioavailability). Other cations also will bind to the biotic ligand, affecting copper bioavailability because higher concentrations of copper are needed for copper to reach toxic levels. The binding to the biotic ligand is considered to be at equilibrium, with apparent (activity-corrected) equilibrium constants K_{LbCu}, K_{LbCuOH}, and K_{LbCj}, respectively, for free copper, copper hydroxide, and the "jth" competing cation. Chemical speciation in the exposure water is also considered to be at equilibrium, and chemical speciation calculations are conducted to compute the free copper, copper hydroxide, and competing cation activities to which the biotic ligand is exposed. Because binding to the actual biotic ligand cannot be measured, it is expected that accumulation relationships for some measurable variable (e.g., the total metal in gill tissue) provide a reasonable surrogate for the actual biotic ligand. Because criteria deal with concentrations eliciting a certain level of effects on groups of organisms (e.g., LC50s), model calculations are for an organism with characteristics appropriate for such group-wide statistics.

How the BLM is applied to criteria can be best discussed by starting with the following general expression for the BLM:

$$EC = EC_0 \cdot f_C \cdot f_L \qquad \text{Equation 1}$$

where EC is the total dissolved copper concentration eliciting an effect, EC_0 is a baseline EC in the absence of any complexing ligands and competing cations, f_C should be a factor (<1) for how much competing cations increase EC, and f_L should be a factor (<1) for how much complexing ligands increase EC. For the BLM used here:

$$EC_0 = \frac{f_{LbT}}{(1 - f_{LbT}) \cdot K_{LbCu}} \qquad \text{Equation 2}$$

$$f_C = 1 + \sum_j^m \left(K_{CjLb} \cdot [C_j]\right) \qquad \text{Equation 3}$$

$$f_L = \frac{1}{\alpha_{Cu^{+n}} + \frac{K_{LbCuOH}}{K_{LbCu}} \cdot \alpha_{CuOH}} \qquad \text{Equation 4}$$

where f_{LbT} is the fraction of the biotic ligand sites that must be occupied by copper to elicit the toxicity of interest (e.g., a lethal accumulation divided by the accumulation capacity), m is the

number of competing cations included in the model, [Cj] is the concentration of the jth competing cation, α_{Cu+2} is the ratio of free copper concentration to total dissolved copper concentration, α_{CuOH} is the ratio for the copper hydroxide complex, and the ratio K_{LbCuOH}/K_{LbCu} specifies the bioavailability of CuOH relative to free copper. Thus, in the absence of complexing ligands and competing cations, the toxic concentration is only a function of the binding strength of free copper and the copper occupied fraction of biotic ligand sites needed to elicit toxicity. The increase in the effect concentration due to competing cations is simply a sum of the products of their concentrations and binding constants. The increase in the effect concentration due to complexing ligands is the inverse of the sum of the products of the relative bioavailabilities and concentration fractions of the species that bind to the biotic ligand (free copper and copper hydroxide).

If toxicity to all the biological species in the criteria (at least the most sensitive ones) were determined based on measured accumulation properties and the relationship of toxicity to accumulation, the above model equations would be directly applied in criteria calculations. However, this is not the case. Although gill accumulation properties and lethal accumulations have been measured for certain species and conditions, and this has been useful in validating BLM assumptions and formulations, the data that must be applied to the criteria consists of water effect concentration (ECs) for biological species for which this accumulation information is generally not available. The BLM therefore is needed, not to make absolute calculations regarding toxic concentrations, but to extrapolate toxic concentrations from one exposure condition to another:

$$EC_A = EC_B \cdot \frac{f_{C,A} \cdot f_{I,A}}{f_{C,B} \cdot f_{I,B}} \qquad \text{Equation 5}$$

where the A and B subscripts refer to different exposure conditions. The general procedure that was followed for criteria development here was to use the above equation to normalize all available toxicity data to a reference exposure condition, calculate criteria values at the reference condition, and again use the above equation to compute criteria at other conditions.

This means that the BLM assumptions and parameters that just pertain to EC_0 are not important to its application to criteria, which actually simplifies model validation and parameterization needs. In particular, there is no need to estimate f_{LbT}, or the lethal accumulations and accumulation capacities that define this fraction. Furthermore, the absolute values of K_{LbCu} and K_{LbCuOH} do not need to be known, only their relative value (and if copper binding to the biotic ligand was dependent only on free copper, the value of K_{LbCu} would not be needed at all). Absolute values are only needed for the binding constants for the competing cations, as well as the various constants needed in speciation calculations to estimate α_{Cu+2} and α_{CuOH}. For BLM application to criteria, the important concern is whether f_C and f_L are suitably formulated and parameterized, and not with issues that relate to lethal accumulations and accumulation capacities.

2.4 *BLM Uncertainties and Performance*

The BLM employed here uses equilibrium reactions of copper and other cations with a single, simple type of surface ligand as the focus for all the effects of physicochemical exposure conditions on toxicity, and thus is a simple, approximate representation for the complex set of chemical

reactions and transfers involved with environmental copper concentrations eliciting toxicity. As already noted, cation effects might involve mechanisms other than competition for a surface ligand. The microenvironment at the gill might change copper speciation. Multiple mechanisms that do not react the same to external conditions might be involved in copper bioavailability and toxicity. Accumulation parameters based on bulk gill measurements will likely not be the same as those for the biotic ligand. Nonequilibrium processes might be important, especially regarding the relationship of copper-binding on a surface ligand to toxic action.

However, any model is a simplification of reality and the existence of uncertainties does not preclude a model from being useful and justified. Despite its simplicity, the BLM used here provides a reasonable mechanistic framework for the well-established effects of copper speciation, explicitly addressing the relative bioavailability of different copper species. It also includes a plausible mechanism that allows the effects of cations to be addressed and uses a comprehensive model for calculating the required concentrations of various chemical species. Even if the mechanistic descriptions are incomplete, this model allows the major empirical effects of complexing ligands and competing cations to be described in a more comprehensive and reasonable fashion than other approaches.

Because this model is used in criteria to predict relative effects of physicochemical exposure factors, its utility for criteria can be judged based on how well it predicts the relative effects of these factors in copper toxicity studies. Examples of BLM performance for various exposure factors and studies are provided in the technical support document for this criteria. Figure 2 shows one example from a study on the effects of various exposure conditions on the acute lethality of copper to fathead minnows. This set of exposures consisted of synthetic exposure solutions of various total ion concentrations with fixed ratios of the major cations and anions, at a fixed pH (8.0) and low dissolved organic matter (< 0.5 mg/L). Observed dissolved LC50s (solid circles with uncertainty bars) varied by 24-fold for only a 9-fold change in total ions. These large effects reflect the combined influences of increased alkalinity (copper carbonate complex formation), hardness, and sodium. Considering the wide range of the observed LC50s and that the model was not fitted to these data, BLM-predicted LC50s (open symbols) were rather accurate, ranging from 55 to 87% (average 75%) of the observed value. More importantly for criteria, the predicted relative change across the range of total ion concentration was 20-fold, very close to that observed.

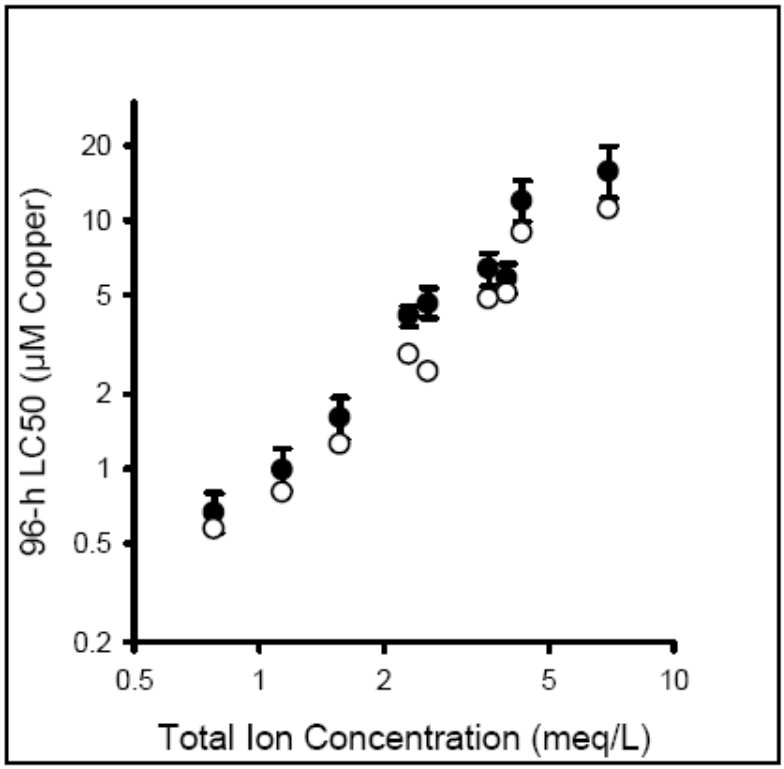

Figure 2. Effects of increasing total ion concentration on the acute lethality of copper to fathead minnows at constant pH=8 and low DOC < 0.5 mg/L. Solid symbols represent observed values, open symbols represent predicted values.

Model performance can also be judged across a variety of factors as in Figure 3, which shows predicted versus observed LC50s for a large number of exposures in the cited study, which varied hardness, alkalinity, sodium, and pH together and separately over a wide range. Observed LC50s varied by about 60-fold, but predicted values deviated from observed values by only 0.12 log units (a factor of 1.3) on average, and at worst only slightly more than a factor of 2. Again, more information on model performance is provided in the Technical Support Document and the figures here just provide some examples demonstrating the utility of this model for use in criteria.

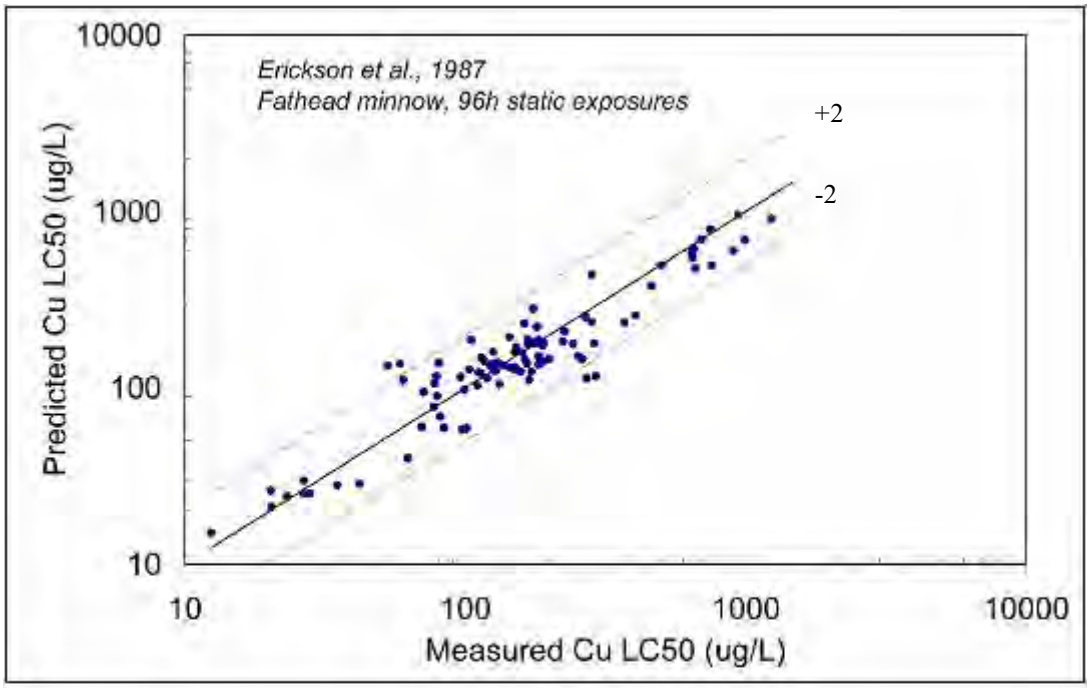

Figure 3. Comparison of Predicted and Measured Acute Copper Toxicity to *P. promelas*.

The use of the BLM to predict the bioavailability and toxicity of copper to aquatic organisms under site-specific conditions is a significant change from the previous Criterion Maximum Concentration (CMC) derivation methodology. Previous aquatic life criteria documents for copper (e.g., U.S. EPA, 1980, 1985, 1996) expressed the CMC as a function of water hardness. Now, EPA chooses to utilize the BLM to update its freshwater acute criterion because the BLM accounts for all important inorganic and organic ligand interactions of copper while also considering competitive interactions that influence binding of copper at the site of toxicity, or the "biotic ligand." The BLM's ability to incorporate metal speciation reactions and organism interactions allows prediction of metal effect levels to a variety of organisms over a wide range of water quality conditions. Accordingly, the BLM is an attractive tool for deriving water quality criteria. Application of the BLM has the potential to substantially reduce the need for site-specific modifications, such as Water Effect Ratio, to account for site-specific chemistry influences on metal toxicity.

The updated BLM-based WQC will in some cases be more stringent and in other cases less stringent than the hardness based WQC. As there is not a single WQC value to use for comparison purposes, it will only be possible to provide illustrative examples of each situation. It is the judgement of the EPA that the BLM-based WQC for Cu will provide an improved framework for evaluating a level of protection (LOP) that is consistent with the LOP that was intended by the 1985 Guidelines (i.e., a 1-in-3 year exceedance frequency that will be protective of 95% of the genera).

While the BLM is currently considered appropriate for use to derive an updated freshwater CMC for the acute WQC, further development is required before it will be suitable for use to

evaluate a saltwater CMC or a Criterion Continuous Concentration (CCC) or chronic value (freshwater or saltwater WQC).

3.0 INCORPORATION OF THE BLM INTO CRITERIA DERIVATIONS PROCEDURES

3.1 *General Final Acute Value (FAV) Procedures*

Application of the acute copper BLM to the derivation of the copper FAV is analogous to procedures already described in the Guidelines for metals criteria using empirical hardness regressions. For these hardness-dependent metals criteria, LC50s at various hardness are normalized to a reference hardness using the regression slopes. The normalized LC50s for each biological species are averaged to derive Species Mean Acute Values (SMAVs) at the reference hardness. The SMAVs within each genus are then averaged to derive Genus Mean Acute Values (GMAVs) at the reference hardness. The Guidelines' procedures for estimating the fifth percentile of the GMAVs are then used to derive the FAV at the reference hardness. FAVs for other hardness can then be derived using the hardness regression slope, and these FAVs are used to calculate the Criterion Maximum Concentration (CMC) by dividing the FAV by 2.0 and the Final Chronic Values (FCV) by dividing the FAV by the Final Acute-Chronic Ratio (FACR). Following the Guidelines, the Criterion Continuous Concentration (CCC) is set to the FCV unless other data justifies a lower value.

Extending this procedure to apply the BLM simply involves normalizing the LC50s to a reference exposure condition that includes all the physicochemical exposure factors important to the BLM, not just hardness. For this normalization, the BLM provides the factors f_C and f_L discussed in Section 2.3, these factors serving the same purpose as the hardness regression slope described above. Each LC50 to be used in criteria derivation would be normalized to the reference exposure conditions by the equation:

$$LC50_R = LC50_A \cdot \frac{f_{C,R} \cdot f_{L,R}}{f_{C,A} \cdot f_{L,A}} \qquad \text{Equation 6}$$

where the subscript A refers to the exposure conditions for the observed LC50 and the subscript R refers to the reference exposure conditions to which the LC50 is being normalized. These normalized LC50s are then used to derive the SMAVs, GMAVs, and FAV at the reference exposure condition as described above for the hardness-corrected criteria. The BLM is then used to derive FAVs at other exposures by the equation:

$$FAV_B = FAV_R \cdot \frac{f_{C,B} \cdot f_{L,B}}{f_{C,R} \cdot f_{L,R}} \qquad \text{Equation 7}$$

where the subscript B refers to the exposure conditions for which an FAV is desired. These BLM-derived FAVs are then used to derive CMCs and CCCs following standard Guidelines procedures.

For the criteria in this document, the reference exposure conditions to which LC50s are normalized and at which the reference FAV is calculated are as follows (see also footnote f in Table 1). The water chemistry used in the normalization was based on the EPA formulation for moderately-hard reconstituted water, but any other water chemistry could have been used. In this formulation the parameters included: temperature = 20°C, pH = 7.5, DOC = 0.5 mg/L, Ca = 14.0 mg/L, Mg = 12.1 mg/L, Na = 26.3 mg/L, K = 2.1 mg/L, SO_4 =81.4 mg/L, Cl = 1.90 mg/L, Alkalinity = 65.0 mg/L and S = 0.0003 mg/L.

3.2 *BLM Input Parameters*

For applying an LC50 to criteria derivations and for determining an FAV at exposure conditions of interest, the necessary water quality input parameters for BLM calculations are temperature, pH, dissolved organic carbon, major geochemical cations (calcium, magnesium, sodium, and potassium), dissolved inorganic carbon (DIC, the sum of dissolved carbon dioxide, carbonic acid, bicarbonate, and carbonate), and other major geochemical anions (chloride, sulfate). DIC measurements are typically not made in the environment, and an alternative input parameter is alkalinity, which can be used with pH and temperature to estimate DIC. There is some evidence that other metals such as iron and aluminum can have an effect on copper toxicity to aquatic organisms, which might be due to interactions of these metals with the biotic ligand, effects of these metals on organic carbon complexation of copper, or adsorption of copper to iron and aluminum colloids which are present in filtrates used to measure dissolved copper. These metals are not currently included in routine BLM inputs, but users are encouraged to measure dissolved iron and aluminum as part of monitoring efforts to support possible future criteria applications.

A number of fixed parameters are also used in the BLM but are not required user inputs in criteria derivations. These include the variety of equilibrium constants used in copper speciation calculations, and also the binding constants for copper and various cations to the biotic ligand. The values for these constants were obtained from work by Playle and coworkers (Playle et al., 1992, 1993a,b) and also by inference from the relationship of toxicity to various water quality characteristics. More information about these parameters can be obtained from the technical support document.

3.3 *Data Screening Procedures*

To use a toxicity test in the derivation of BLM-based criteria, information must be available for the various water quality parameters described in Section 3.2. This is in contrast to past metals criteria, for which the only necessary water quality parameter was hardness. Many of these parameters are not routinely measured in toxicity tests and, if measured, are not necessarily reported in the primary literature for the test, especially for older toxicity tests. However, this information might be available from supplemental sources or be estimated based on other information. Therefore, in addition to reviewing the primary sources for relevant information,

additional efforts were made to obtain or estimate the necessary water quality parameters for as many of the available LC50s as possible.

A detailed description of these efforts is provided in Appendix C, Estimation of Water Chemistry Parameters for Acute Copper Toxicity Tests, and are summarized as follows. Reports of acute copper toxicity tests identified in literature searches were reviewed to identify LC50s for possible inclusion in the criteria derivation. In addition to test acceptability standards specified in the Guidelines, the current effort also required that the LC50s be based on measured copper concentrations. LC50s based on nominal concentrations have been used in previous criteria, but there are enough measured LC50s for copper that this was considered to be no longer warranted, especially considering the more advanced bioavailability assessments represented by the BLM. For the identified LC50s, the primary reports were reviewed to record all reported information on dilution and test water chemistry. Any additional references specified by the authors were also obtained and reviewed. If test waters were synthetically prepared based on specified formulas, these were used to estimate parameters as appropriate. When critical water chemistry parameters were not available, authors were contacted regarding unpublished information or to measure missing water chemistry parameters in dilution source waters. If primary or corresponding authors could not be contacted, an attempt was made to contact secondary authors or personnel from the laboratories where the studies were conducted. Where actual water chemistry data were unavailable, data from other studies with the same water source were used as surrogate values if appropriate. Absent this, the U.S. Geological Survey's National Stream Quality Accounting Network (NASQAN) and the EPA STOrage and RETrieval (STORET) were used to obtain data for ambient surface waters which were the source of water for a test. In some instances other available sources were contacted to obtained water chemistry data (e.g., city drinking water treatment personnel). The acquired data were scrutinized for representativeness and usefulness for estimating surrogate values to complete the water quality information for the dilution and/or test water that was used in the original studies. When the above sources could not be used, geochemical ion inputs were based on reported hardness measurements and regressions relationships constructed for the relationship of various ions to hardness from NASQAN data.

As with any modeling effort, the reliability of model output depends on the reliability of model inputs. Although the input data have been closely scrutinized, the reliability of the BLM-normalized LC50s are subject to the uncertainties of the estimation procedures described above. Therefore, a ranking system was devised to rank the quality of the chemical characterization of the test water. Studies with a rank of 1 contain all of the necessary parameters for BLM input based on measurements from either the test chambers or the water source. In general, studies in which the BLM input parameters were reported for test chamber samples take precedence over studies in which the parameters were reported only for the source water. A characterization ranking of 2 denotes those studies where not all parameters were measured, but reliable estimates of the requisite concentrations could be made. Similarly, a rank of 3 denotes studies in which all parameters except DOC were measured, but reliable estimates of DOC could be made. For the majority of the tests, a chemical characterization of 4+ was assigned because hardness, alkalinity, and pH were measured, and the ionic composition could be reliably estimated or calculated. A 4- was assigned to those studies conducted using standard reconstituted water in which hardness, alkalinity, or pH was either measured or referenced, and the recipe for the water is known (ASTM, 2000; U.S. EPA, 1993). The chemical characterization rank of 5 was ascribed to studies in which

one of the key parameters (DOC, Ca, pH, alkalinity) was not measured, and when it could not be reliably estimated. If two or more key parameters (DOC, Ca, pH, alkalinity) were not measured and could not be reliably estimated, a study was given a chemical characterization rank of 6. Studies receiving a quality rating of greater than 4+ (i.e., higher than 4) were not used in the criteria development procedures because the estimates for some of the key input parameters were not thought to be reliable, all other studies were used.

3.4 *Conversion Factors*

The LC50s used in deriving previous EPA metals criteria were based on total metal concentration (measured or nominal) and the criteria were consequently for total metals concentration. EPA afterwards made the decision that metals criteria should be based on dissolved metal because it was thought to better represent the bioavailable fraction of the metal (U.S. EPA, 1993). It was thus necessary to convert the criteria to a dissolved concentration basis. However, at that time, most toxicity tests reported only total concentration, so that a procedure was necessary to estimate the likely fractions of metals that were dissolved in typical toxicity tests. Studies were therefore conducted to determine these fractions under a variety of test conditions that mimicked the conditions in the tests used to derive the metals criteria (University of Wisconsin-Superior, 1995). These tests demonstrated high fractions of dissolved copper and resulted in a conversion factor (CF) of 0.96 for converting both the CMC and CCC for copper from a total to dissolved basis (Stephan, 1995). The BLM-derived criteria developed here also uses dissolved copper as the basis for criteria, assuming a negligible bioavailability for particulate copper. The conversion factor of 0.96 was also used to convert total to dissolved copper for any toxicity test for which dissolved copper measurements were not available.

3.5 *Final Chronic Value (FCV) Procedures*

Because the minimum eight family data requirements for chronic toxicity data were not met in order to calculate the FCV by the fifth percentile method used for the FAV and because insufficient information was available to develop a chronic BLM, EPA derived the CCC utilizing the Acute to Chronic Ratio (ACR) approach from the Guidelines (Stephan et al., 1985). To calculate the FCV at a specific water chemistry, the FAV at that chemistry is divided by the FACR. This entails the assumption that the acute BLM reasonably approximates the bioavailability relationships for chronic toxicity. Limited data available regarding effects of water chemistry on sublethal effects and chronic lethality do show substantial effects of organic matter, alkalinity, pH, and sodium (Winner, 1985; Erickson et al., 1996 a,b) similar to those in the acute BLM used here. For hardness, apparent effects are limited and uncertain, but the use of the acute BLM does not introduce major uncertainties in this regard because the effects of hardness by itself in the acute BLM are also limited.

4.0 DATA SUMMARY AND CRITERIA CALCULATION

4.1 *Summary of Acute Toxicity to Freshwater Animals and Criteria Calculation*

The screening procedure outlined in Sec. 3.3 (high quality data = 1, low quality data > 4, e.g. 4+) identified approximately 600 acute freshwater toxicity tests with aquatic organisms and copper

potentially acceptable for deriving criteria. Of these tests, approximately 100 were eliminated from the criteria derivation process because they did not report measured copper concentrations. Nearly 150 additional tests were eliminated from the calculation of the FAV because they received a quality rating of greater than 4 in the quality rating scheme described in section 3.3 described above.

Data from approximately 350 tests were used to derive normalized LC50 values, including 15 species of invertebrates, 22 species of fish, and 1 amphibian species (Table 1), representing 27 different genera. Species Mean Acute Values (SMAVs) at the reference chemistry were calculated from the normalized LC50s and Genus Mean Acute Values (GMAVs) at the normalization chemistry were calculated from the SMAVs.

SMAVs ranged from 2.37 µg/L for the most sensitive species, *Daphnia pulicaria*, to 107,860 µg/L for the least sensitive species, *Notemigonus crysoleucas*. Cladocerans were among the most sensitive species, with *D. pulicaria, D. magna*, *Ceriodaphnia dubia*, and *Scapholeberis sp.* being four out of the six most sensitive species. Invertebrates in general were more sensitive than fish, representing the 10 lowest SMAVs.

The 27 GMAVs calculated from the above-mentioned SMAVs ranged from 4.05 µg/L for *Daphnia* to 107,860 µg/L for *Notemigonus* (Table 3a). Nine of the 10 most sensitive genera were invertebrates. The salmonid genus *Oncorhynchus* was the most sensitive fish genus, with a GMAV of 31.39 µg/L and an overall GMAV ranking of 10.

The ranked GMAVs are presented in Figure 4. Pursuant to procedures used to calculate the FAV, a FAV of 4.67 µg/L was derived from the four GMAVs with cumulative probabilities closest to the 5th percentile toxicity value for all the tested genera (Table 3b). The presumption is that this

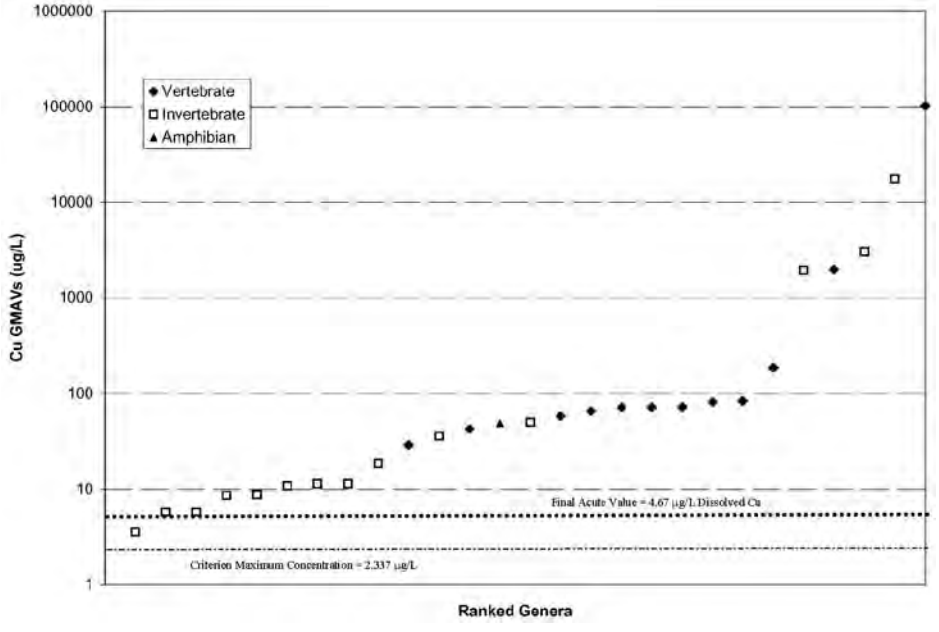

Figure 4. Ranked Freshwater Genus Mean Acute Values (GMAVs)

acute toxicity value represents the LC50 for an organism that is sensitive at the 5th percentile of the GMAV distribution. The CMC is the FAV divided by two. Therefore, the freshwater dissolved copper CMC for the reference chemistry presented is 2.337 µg/L.

Site-water chemistry parameters are needed to evaluate a criterion. This is analogous to the situation that previously existed for the hardness-based WQC, where a hardness concentration was necessary in order to derive a criterion. Examples of CMC calculations at various water chemistry conditions are presented in Figure 5 and Appendix G.

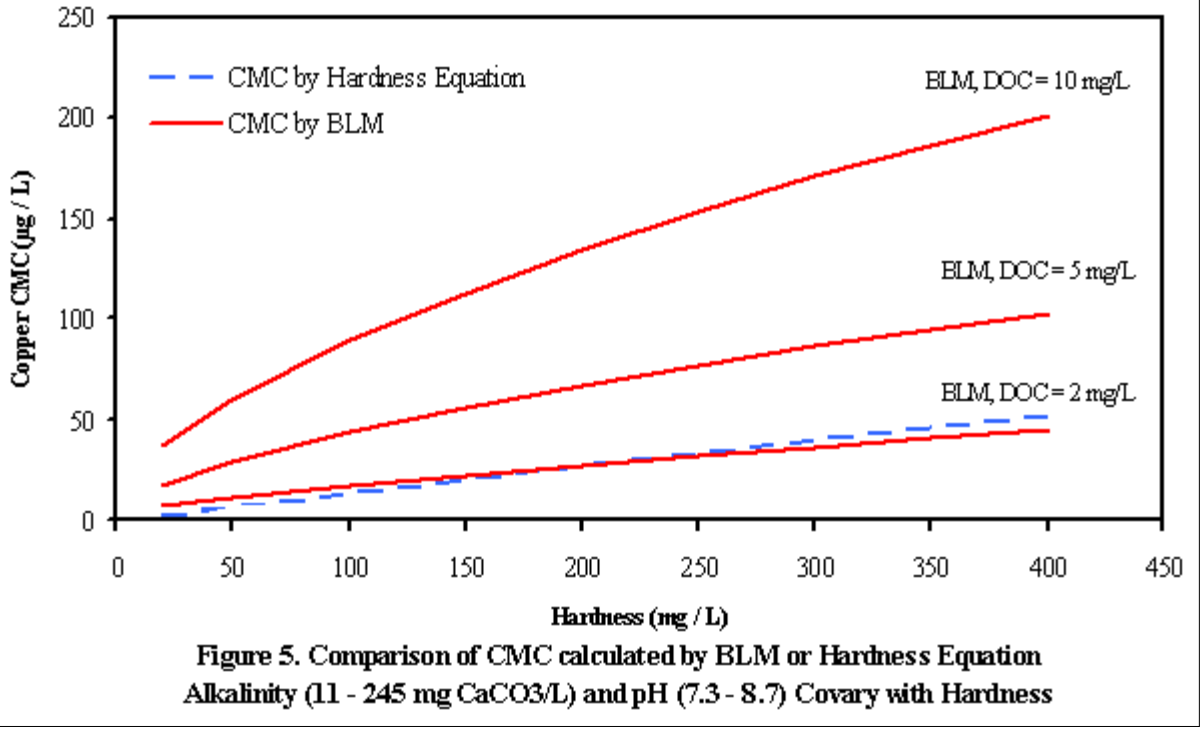

Figure 5. Comparison of CMC calculated by BLM or Hardness Equation Alkalinity (11 - 245 mg CaCO3/L) and pH (7.3 - 8.7) Covary with Hardness

4.1.1 *Comparison With Earlier Hardness-Adjusted Criteria*

EPA's earlier freshwater copper criteria recommendations were hardness-dependent values. One would expect a BLM-based criterion calculation procedure to yield the more appropriate criterion—appropriate in the sense that it accounts for the important water chemistry factors that affect toxicity, including DOC complexation, where the hardness correction does not. Application of the BLM in field situations where DOC is expected to be present at higher concentrations than those observed in laboratory studies would likely improve the performance of the BLM compared with the hardness adjustment. The reason is that the BLM would reasonably account for the typically observed increase in effect levels under such conditions, while the hardness-based approach would not (Figure 5).

As a comparison between the hardness typical of the previous copper criterion and this revised criterion using the BLM, both procedures were used to calculate criterion values for waters with a range in hardness as specified by the standard EPA recipes (U.S. EPA, 1993). The EPA formulations specify the concentration of various salts and reagents to be used in the synthesis of

laboratory test waters with specific hardness values (e.g., very soft, soft, moderately hard, hard, or very hard). As the water hardness increases in these recipes, pH and alkalinity also increase. This has implications for the BLM because the bioavailability of copper would be expected to decrease with increasing pH and alkalinity due to the increasing degree of complexation of copper with hydroxides and carbonates and decreasing proton competition with the metal at both DOM and biotic ligand binding sites. The BLM criterion for these waters agrees very well with that calculated by the hardness equation used in previous copper criterion documents (Figure 5). However, alkalinity and pH change as hardness changes in the EPA recipes. The BLM prediction is taking all of these changes in water quality into account.

It is possible to use the BLM to look only at the change in predicted WQC with changes in hardness (e.g., alkalinity and pH remaining constant). The hardness equation is based on waters where changes in hardness are accompanied by changes in pH and alkalinity. However, there are many possible natural waters where changes in hardness are not accompanied by changes in pH and alkalinity (such as water draining a region rich in gypsum). In these cases, the hardness equation based criterion will still assume a response that is characteristic of waters where hardness, alkalinity, and pH co-vary, and will likely be underprotective relative to the level of protection intended by the Guidelines, in high hardness waters. Conversely, in waters where the covariation between hardness, pH, and alkalinity is greater than is typical for data in Table 1, the hardness equation based criteria may be overprotective. Appendix G shows representative water quality criteria values using both the BLM and the hardness equation approaches for waters with a range in pH, hardness, and DOC concentrations. The hardness approach does not consider pH and DOC while the BLM approach takes those water quality parameters into consideration.

4.2 Formulation of the CCC

4.2.1 *Evaluation of Chronic Toxicity Data*

In aquatic toxicity tests, chronic values are usually defined as the geometric mean of the highest concentration of a toxic substance at which no adverse effect is observed (highest no observed adverse effect concentration, or NOAEC) and the lowest concentration of the toxic substance that causes an adverse effect (lowest observed adverse effect concentration, or LOAEC). The significance of the observed effects is determined by statistical tests comparing responses of organisms exposed to low-level and control concentrations of the toxic substance against responses of organisms exposed to elevated concentrations. Analysis of variance is the most common test employed for such comparisons. This approach, however, has the disadvantage of resulting in marked differences between the magnitudes of the effects corresponding to the individual chronic values, because of variation in the power of the statistical tests used, the concentrations tested, and the size and variability of the samples used (Stephan and Rogers, 1985).

An alternative approach to calculating chronic values focuses on the use of point estimates such as from regression analysis to define the dose-response relationship. With a regression equation or probit analysis, which defines the level of adverse effects as a function of increasing concentrations of the toxic substance, it is possible to determine the concentration that causes a specific small effect, such as a 5 to 30 percent reduction in response. To make chronic values reflect a uniform level of effect, regression and probit analyses were used, where possible, both to demonstrate that a significant concentration-effect relationship was present and to estimate chronic

values with a consistent level of effect. The most precise estimates of effect concentrations can generally be made for 50 percent reduction (EC50); however, such a major reduction is not necessarily consistent with criteria providing adequate protection. In contrast, a concentration that causes a low level of reduction, such as an EC5 or EC10, might not be statistically significantly different from the control treatment. As a compromise, the EC20 is used here to represent a low level of effect that is generally significantly different from the control treatment across the useful chronic datasets that are available for copper. The EC20 was also viewed as providing a level of protection similar to the geometric mean of the NOEC and LOEC. Since the EC20 is not directly dependent on the tested dilution series, similar EC20s should be expected irrespective of the tested concentrations, provided that the range of tested concentrations is appropriate.

Regression or probit analysis was utilized to evaluate a chronic dataset only in cases where the necessary data were available and the dataset met the following conditions: (1) it contained a control treatment (or low exposure data point) to anchor the curve at the low end, (2) it contained at least three concentrations, and (3) two of the data points had effect variable values below the control and above zero (i.e., "partial effects"). Control concentrations of copper were estimated in cases where no measurements were reported. These analyses were performed using the Toxicity Relationship Analysis Program software (version 1.0; U.S. EPA, Mid-Continental Ecology Division, Duluth, MN, USA). Additional detail regarding the aforementioned statistical procedures is available in the cited program.

When the data from an acceptable chronic test met the conditions for the logistic regression or probit analysis, the EC20 was the preferred chronic value. When data did not meet the conditions the chronic value was usually set to the geometric mean of the NOAEC and the LOAEC. However, when no treatment concentration was an NOAEC, the chronic value is reported as less than the lowest tested concentration.

For life-cycle, partial life-cycle, and early life stage tests, the toxicological variable used in chronic value analyses was survival, reproduction, growth, emergence, or intrinsic growth rate. If copper apparently reduced both survival and growth (weight or length), the product of variables (biomass) was analyzed, rather than analyzing the variables separately. The most sensitive of the toxicological variables was generally selected as the chronic value for the particular study.

A species-by-species discussion of each acceptable chronic test on copper evaluated for this document is presented in Appendix F. Figures that present the data and regression/probability distribution line for each of the acceptable chronic test which contained sufficient acceptable data are also provided in Appendix F.

4.2.2 *Calculation of Freshwater CCC*

Acceptable freshwater chronic toxicity data from early life stage tests, partial life-cycle tests, and full life-cycle tests were available for 29 tests including data for 6 invertebrate species and 10 fish species (Table 2a). The 17 chronic values for invertebrate species range from 2.83 (*D. pulex)* to 34.6 µg/L (*C. dubia*); and the 12 chronic values for the fish species range from <5 (brook trout) to 60.4 µg/L (northern pike). Of the 29 chronic tests, comparable acute values are available for 18 of the tests (Table 2c). The relationship between acute toxicity values and ACRs is presented in Figure 6. The supporting acute and chronic test values for the ACRs and the species mean ACRs are

presented in Table 2c. For the 11 tests in Table 2a with chronic values both from a regression EC20 and the geometric mean of the NOAEC and LOAEC, the EC20 averaged 81% of the geometric mean, demonstrating the similar level of protection for the two approaches.

Overall, individual ACRs varied from <1 (0.55) for *C. dubia* (Oris et al., 1991) to 191.6 for the snail, *Campeloma decisum* (Arthur and Leonard, 1970). Species mean acute-chronic ratios ranged from 1.48 in saltwater for the sheepshead minnow (Hughes et al., 1989) to 171.2 in freshwater for the snail, *C. decisum*. Pursuant to the Guidelines (Stephan et al., 1985), consideration was given to calculating the FACR based on all ACRs within a factor of 10, but because there appeared to be a relationship between acute sensitivity and ACRs (Figure 6), the FACR was derived from data for species whose SMAVs were close to the FAV. The FACR of 3.22 was calculated as the geometric mean of the ACRs for sensitive freshwater species, *C. dubia, D. magna, D. pulex, O. tshawytscha*, and *O. mykiss* along with the one saltwater ACR for *C. variegatus* (Table 2b). Based on the normalization water chemistry conditions used for illustrative purposes in the document, the freshwater site specific FAV value is 4.67 µg/L, which divided by the FACR of 3.22 results in a freshwater FCV of 1.45 µg/L dissolved Cu.

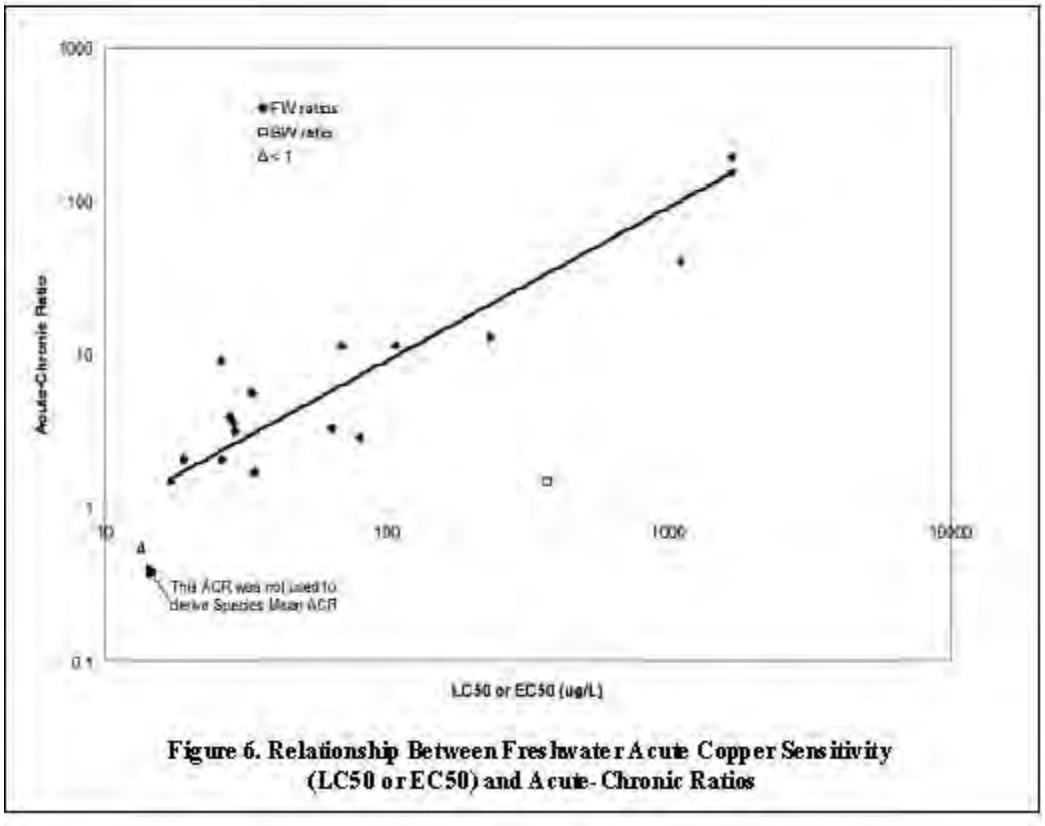

Figure 6. Relationship Between Freshwater Acute Copper Sensitivity (LC50 or EC50) and Acute-Chronic Ratios

5.0 PLANT DATA

Copper has been widely used as an algicide and herbicide for nuisance aquatic plants (McKnight et al., 1983). Although copper is known as an inhibitor of photosynthesis and plant growth, toxicity data on individual species suitable for deriving aquatic life criteria (Table 4) are not numerous.

The relationship of copper toxicity to the complexing capacity of the water or the culture medium is now widely recognized (Gächter et al., 1973; Petersen, 1982), and several studies have used algae to "assay" the copper complexing capacity of both fresh and salt waters (Allen et al., 1983; Lumsden and Florence, 1983; Rueter, 1983). It has also been shown that algae are capable of excreting complexing substances in response to copper stress (McKnight and Morel, 1979; Swallow et al., 1978; van den Berg et al., 1979). Foster (1982) and Stokes and Hutchinson (1976) have identified resistant strains and/or species of algae from copper (or other metal) impacted environments. A portion of this resistance probably results from induction of the chelate-excretion mechanism. Chelate excretion by algae may also serve as a protective mechanism for other aquatic organisms in eutrophic waters; that is, where algae are capable of maintaining free copper activities below harmful concentrations.

Copper concentrations from 1 to 8,000 µg/L have been shown to inhibit growth of various freshwater plant species. Very few of these tests, though, were accompanied by analysis of actual copper exposure concentrations. Notable exceptions are freshwater tests with green alga including *Chlamydomonas reinhardtii* (Schafer et al., 1993; Winner and Owen, 1991b), which is the only flow-through, measured test with an aquatic plant, *Chlorella vulgaris* and *Selenastrum capricornutum* (Blaylock et al., 1985). There is also a measured test with duckweed, *Lemna minor* (Taraldsen and Norberg-King, 1990).

A direct comparison between the freshwater plant data and the BLM derived criteria is difficult to make without a better understanding of the composition of the algal media used for different studies (e.g., DOC, hardness, and pH) because these factors influence the applicable criteria comparison. BLM derived criteria for certain water conditions, such as low to mid-range pH, hardness up to 100 mg/L as $CaCO_3$, and low DOC are in the range of, if not lower than, the lowest reported toxic endpoints for freshwater algal species and would therefore appear protective of plant species. In other water quality conditions BLM-derived criteria may be significantly higher (see Figure 5).

Two publications provide data for the red algae *Champia parvula* that indicate that reproduction of this species is especially sensitive to copper. The methods manual (U.S. EPA 1988) for whole effluent toxicity (WET) testing contains the results of six experiments showing nominal reproduction LOECs from 48-hr exposures to 1.0 to 2.5 µg/L copper (mean 2.0 µg/L); these tests used a mixture of 50 percent sterile seawater and 50 percent GP2 medium copper. The second study by Morrison et al. (1989) evaluated interlaboratory variation of the 48-hr WET test procedure; this six-test study gave growth EC50 values from 0.8 to 1.9 µg/L (mean 1.0 µg/L). Thus, there are actually 12 tests that provide evidence of significant reproductive impairment in *C. parvula* at nominal copper concentrations between 0.8 and 2.5 µg/L. For these studies though, the dilution water source was not identified.

One difficulty in assessing these data is the uncertainty of the copper concentration in the test solutions, primarily with respect to any background copper that might be found in the dilution water, especially with solutions compounded from sea salts or reagents. Thus, with a CCC of 1.9 µg/L dissolved copper, the significance of a 1 or 2 µg/L background copper level to a 1 to 3 µg/L nominal effect level can be considerable.

The reproduction of other macroalgae appears to be generally sensitive to copper, but not to the extent of *Champia*. Many of these other macroalgae appear to have greater ecological significance than *Champia*, several forming significant intertidal and subtidal habitats for other saltwater organisms, as well as being a major food source for grazers. Reproductive and growth effects on the other species of macroalgae sometimes appear to occur at copper concentrations between 5 and 10 µg/L (Appendix B, Other Data). Thus, most major macrophyte groups seem to be adequately protected by the CMC and CCC, but appear similar in sensitivity to some of the more sensitive groups of saltwater animals.

6.0 OTHER DATA

Many of the data identified for this effort are listed in Appendix B, Other Data, for various reasons, including exposure durations other than 96 hours with the same species reported in Table 1, and some exposures lasting up to 30 days. Acute values for test durations less than 96 hours are available for several species not shown in Table 1. Still, these species have approximately the same sensitivities to copper as species in the same families listed in Table 1. Reported LC50s at 200 hours for chinook salmon and rainbow trout (Chapman, 1978) differ only slightly from 96-hour LC50s reported for these same species in the same water.

A number of other acute tests in Appendix B were conducted in dilution waters that were not considered appropriate for criteria development. Brungs et al. (1976) and Geckler et al. (1976) conducted tests with many species in stream water that contained a large amount of effluent from a sewage treatment plant. Wallen et al. (1957) tested mosquitofish in a turbid pond water. Until chemical measurements that correlate well with the toxicity of copper in a wide variety of waters are identified and widely used, results of tests in unusual dilution waters, such as those in Appendix B, will not be very useful for deriving water quality criteria.

Appendix B also includes tests based on physiological effects, such as changes in appetite, blood parameters, stamina, etc. These were included in Appendix B because they could not be directly interpreted for derivation of criteria. For the reasons stated in this section above, data in Appendix B was not used for criteria derivation.

A direct comparison of a particular test result to a BLM-derived criterion is not always straightforward, particularly if complete chemical characterization of the test water is not available. Such is the case for a number of studies included in Appendix B. While there are some test results with effect concentrations below the example criteria concentrations presented in this document, these same effect concentrations could be above criteria derived for other normalization chemistries, raising the question as to what is the appropriate comparison to make. For example, Appendix B includes an EC50 for *D. Pulex* of 3.6 µg/L (Koivisto et al., 1992) at an approximate hardness of 25 mg/L (33 mg/L as $CaCO_3$). Yet, example criteria at a hardness of 25 mg/L (as $CaCO_3$) (including those in Figure 6) range from 0.23 µg/L (DOC = 0.1 mg/L) to 4.09 µg/L (DOC = 2.3 mg/L) based

on the DOC concentration selected for the synthetic water recipe. The chemical composition for the Koivisto et al. (1992) study would dictate what the appropriate BLM criteria comparison should be.

Based on the expectation that many of the test results presented in Appendix B were conducted in laboratory dilution water with low levels of DOC, the appropriate comparison would be to the criteria derived from low DOC waters. Comparing many of the values in Appendix B to the example criteria presented in this document, it appears that a large proportion of Appendix B values are above these concentration levels. This is a broad generalization though and as stated previously, all important water chemistry variables that affect toxicity of copper to aquatic organisms should be considered before making these types of comparisons.

Studies not considered suitable for criteria development were placed in Appendix G, Unused Data.

7.0 NATIONAL CRITERIA STATEMENT

The available toxicity data, when evaluated using the procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that freshwater aquatic life should be protected if the 24-hour average and four-day average concentrations do not respectively exceed the acute and chronic criteria concentrations calculated by the Biotic Ligand Model.

A return interval of 3 years between exceedances of the criterion continues to be EPA's general recommendation. However, the resilience of ecosystems and their ability to recover differ greatly. Therefore, scientific derivation of alternative frequencies for exceeding criteria may be appropriate.

8.0 IMPLEMENTATION

The use of water quality criteria in designing waste treatment facilities and appropriate effluent limits involves the use of an appropriate wasteload allocation model. Although dynamic models are preferred for application of these criteria, limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. EPA recommends the interim use of 1B3 or 1Q10 for criterion maximum concentration stream design flow and 4B3 or 7Q10 for the criterion continuous concentration design flow in steady-state models. These matters are discussed in more detail in the Technical Support Document for Water Quality-Based Toxics Control (U.S. EPA, 1991).

With regard to BLM-derived freshwater criteria, to develop a site-specific criterion for a stream reach, one is faced with determining what single criterion is appropriate even though a BLM criterion calculated for the event corresponding to the input water chemistry conditions will be time-variable. This is not a new problem unique to the BLM—hardness-dependent metals criteria are also time-variable values. Although the variability of hardness over time can be characterized, EPA has not provided guidance on how to calculate site-specific criteria considering this variability. Multiple input parameters for the BLM could complicate the calculation of site-specific criteria because of their combined effects on variability. Another problem arise from potential scarcity of data from small stream reaches with small dischargers. The EPA is currently exploring two

approaches to fill data gaps in such situations. One potential approach is the selection of values based on geography, the second approach is based on correlations between measured parameters and missing parameter measurements. A companion document in the form of Supplementary Training Materials, addressing issues related to data requirements, implementation, permitting, and monitoring will be released via EPA's website following the publication of this criteria document.□□

Table 1. Acute Toxicity of Copper to Freshwater Animals

Species[a]	Organism Age, Size, or Lifestage	Method[b]	Chemical[c]	Reported LC50 or EC50 (total µg/L)[d]	Reported LC50 or EC50 (Diss. µg/L)[e]	BLM Data Label	BLM Normalized LC50 or EC50 (µg/L)[f]	Species Mean Acute Value (µg/L)[g]	Reference
Worm,	adult (mixed age)	S,M,T	N	130	---	LUVA01S	37.81	48.41	Schubauer-Berigan et al. 1993
Lumbriculus variega	adult (mixed age)	S,M,T	N	270	---	LUVA02S	55.39		Schubauer-Berigan et al. 1993
	adult (mixed age)	S,M,T	N	500	---	LUVA03S	54.18		Schubauer-Berigan et al. 1993
Snail,	1.1-2.7 cm	F,M,T	S	2000	---	CADE01F	4319	3573	Arthur and Leonard 1970
Campeloma	1.1-2.7 cm	F,M,T	S	1400	---	CADE02F	2956		Arthur and Leonard 1970
Snail, *Juga plicifera*	adult	F,M,T	C	15	---	JUPL01F	12.31	12.31	Nebeker et al. 1986b
Snail, *Lithoglyphus virens*	adult	F,M,T	C	8	---	LIVI01F	6.67	6.67	Nebeker et al. 1986b
Snail,	0.4-0.7 cm	F,M,T	S	41	---	PHIN01F	21.81	20.41	Arthur and Leonard 1970
Physa integra	0.4-0.7 cm	F,M,T	S	37	---	PHIN02F	19.09		Arthur and Leonard 1970
Freshwater mussel,	juvenile	S,M,T	S	27	---	ACPE01S	10.36	11.33	Keller unpublished
Actinonaias	juvenile	S,M,T	S	<29	---	ACPE02S	12.39		Keller unpublished
Freshwater mussel,	1-2 d juv	S,M,T	S	86	---	UTIM01S	177.9	52.51	Keller and Zam 1991
Utterbackia imbecilli	1-2 d juv	S,M,T	S	199	---	UTIM02S	172.3		Keller and Zam 1991
	juvenile	S,M,T	N	76	---	UTIM03S	40.96		Keller unpublished
	juvenile	S,M,T	N	85	---	UTIM04S	43.22		Keller unpublished
	juvenile	S,M,T	N	41	---	UTIM05S	24.12		Keller unpublished
	juvenile	S,M,T	S	79	---	UTIM06S	39.04		Keller unpublished
	juvenile	S,M,T	S	72	---	UTIM07S	39.96		Keller unpublished
	juvenile	S,M,T	S	38	---	UTIM08S	28.31		Keller unpublished
Cladoceran,	<4 h	S,M,T	C	19	---	CEDU01S	10.28	5.93	Carlson et al. 1986
Ceriodaphnia dubia	<4 h	S,M,T	C	17	---	CEDU02S	9.19		Carlson et al. 1986
	<12 h	S,M,D	---	-	25	CEDU03S	7.98		Belanger et al. 1989
	<12 h	S,M,D	---	-	17	CEDU04S	5.25		Belanger et al. 1989
	<12 h	S,M,D	---	-	30	CEDU05S	9.80		Belanger et al. 1989
	<12 h	S,M,D	---	-	24	CEDU06S	7.63		Belanger et al. 1989
	<12 h	S,M,D	---	-	28	CEDU07S	9.06		Belanger et al. 1989
	<12 h	S,M,D	---	-	32	CEDU08S	10.56		Belanger et al. 1989
	<12 h	S,M,D	---	-	23	CEDU09S	7.28		Belanger et al. 1989
	<12 h	S,M,D	---	-	20	CEDU10S	6.25		Belanger et al. 1989
	<12 h	S,M,D	---	-	19	CEDU11S	5.91		Belanger et al. 1989
	<12 h	S,M,D	---	-	26	CEDU12S	3.10		Belanger et al. 1989
	<12 h	S,M,D	---	-	21	CEDU13S	2.46		Belanger et al. 1989
	<12 h	S,M,D	---	-	27	CEDU14S	3.24		Belanger et al. 1989
	<12 h	S,M,D	---	-	37	CEDU15S	4.66		Belanger et al. 1989
	<12 h	S,M,D	---	-	34	CEDU16S	4.22		Belanger et al. 1989
	<12 h	S,M,D	---	-	67	CEDU17S	5.50		Belanger et al. 1989
	<12 h	S,M,D	---	-	38	CEDU18S	2.72		Belanger et al. 1989
	<12 h	S,M,D	---	-	78	CEDU19S	6.74		Belanger et al. 1989
	<12 h	S,M,D	---	-	81	CEDU20S	7.10		Belanger et al. 1989
	<12 h	S,M,D	---	-	28	CEDU21S	4.10		Belanger and Cherry 1990

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	<12 h	S,M,D	---	-	84	CEDU22S	10.74		Belanger and Cherry 1990
	<12 h	S,M,T	S	13.4	---	CEDU23S	6.19		Oris et al. 1991
	<24 h	R,M,T,D	S	6.98	5.54	CEDU24R	5.03		Diamond et al. 1997b
Cladoceran,	1 d	S,M,T	C	9.1	---	DAMA01S	3.42	6.00	Nebeker et al. 1986a
Daphnia magna	1 d	S,M,T	C	11.7	---	DAMA02S	4.43		Nebeker et al. 1986a
	<2 h	S,M,T	C	6.6	---	DAMA03S	2.50		Nebeker et al. 1986a
	<2 h	S,M,T	C	9.9	---	DAMA04S	3.78		Nebeker et al. 1986a
	1 d	S,M,T	C	11.7	---	DAMA05S	13.46		Nebeker et al. 1986a
	<4 h	S,M,T	C	6.7	---	DAMA06S	8.21		Nebeker et al. 1986a
	1 d	S,M,T	C	9.1	---	DAMA07S	4.40		Nebeker et al. 1986a
	<2 h	S,M,T	C	5.2	---	DAMA08S	2.16		Nebeker et al. 1986a
	<24 h	S,M,T	S	41.2	---	DAMA09S	21.55		Baird et al. 1991
	<24 h	S,M,T	S	10.5	---	DAMA10S	5.63		Baird et al. 1991
	<24 h	S,M,T	S	20.6	---	DAMA11S	11.31		Baird et al. 1991
	<24 h	S,M,T	S	17.3	---	DAMA12S	9.48		Baird et al. 1991
	<24 h	S,M,T	S	70.7	---	DAMA13S	33.58		Baird et al. 1991
	<24 h	S,M,T	S	31.3	---	DAMA14S	16.90		Baird et al. 1991
	<24 h	S,M,I	S	7.1	---	DAMA15S	2.67		Meador 1991
	<24 h	S,M,I	S	16.4	---	DAMA16S	4.26		Meador 1991
	<24 h	S,M,I	S	39.9	---	DAMA17S	5.18		Meador 1991
	<24 h	S,M,I	S	18.7	---	DAMA18S	3.39		Meador 1991
	<24 h	S,M,I	S	18.9	---	DAMA19S	1.99		Meador 1991
	<24 h	S,M,I	S	39.7	---	DAMA20S	3.04		Meador 1991
	<24 h	S,M,I	S	46	---	DAMA21S	8.93		Meador 1991
	<24 h	S,M,I	S	71.9	---	DAMA22S	9.97		Meador 1991
	<24 h	S,M,I	S	57.2	---	DAMA23S	5.76		Meador 1991
	<24 h	S,M,I	S	67.8	---	DAMA24S	4.16		Meador 1991
	<24 h	S,M,T	C	26	---	DAMA25S	10.34		Chapman et al. Manuscript
	<24 h	S,M,T	C	30	---	DAMA26S	9.04		Chapman et al. Manuscript
	<24 h	S,M,T	C	38	---	DAMA27S	9.84		Chapman et al. Manuscript
	<24 h	S,M,T	C	69	---	DAMA28S	12.31		Chapman et al. Manuscript
	<24 h	S,M,T,D	S	4.8	---	DAMA29S	1.22		Long's MS Thesis
	<24 h	S,M,T,D	S	7.4	---	DAMA30S	16.29		Long's MS Thesis
	<24 h	S,M,T,D	S	6.5	---	DAMA31S	2.11		Long's MS Thesis
Cladoceran,	---	S,M,T	S	11.4	---	DAPC01S	1.63	2.73	Lind et al. Manuscript (1978)
Daphnia pulicaria	---	S,M,T	S	9.06	---	DAPC02S	1.04		Lind et al. Manuscript (1978)
	---	S,M,T	S	7.24	---	DAPC03S	0.88		Lind et al. Manuscript (1978)
	---	S,M,T	S	10.8	---	DAPC04S	1.13		Lind et al. Manuscript (1978)
	---	S,M,T	S	55.4	---	DAPC05S	8.81		Lind et al. Manuscript (1978)
	---	S,M,T	S	55.3	---	DAPC06S	6.03		Lind et al. Manuscript (1978)
	---	S,M,T	S	53.3	---	DAPC07S	4.12		Lind et al. Manuscript (1978)
	---	S,M,T	S	97.2	---	DAPC08S	3.94		Lind et al. Manuscript (1978)

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	---	S,M,T	S	199	---	DAPC09S	3.01		Lind et al. Manuscript (1978)
	---	S,M,T	S	213	---	DAPC10S	7.63		Lind et al. Manuscript (1978)
	---	S,M,T	S	165	---	DAPC11S	5.78		Lind et al. Manuscript (1978)
	---	S,M,T	S	35.5	---	DAPC12S	1.83		Lind et al. Manuscript (1978)
	---	S,M,T	S	78.8	---	DAPC13S	2.36		Lind et al. Manuscript (1978)
	---	S,M,T	S	113	---	DAPC14S	1.06		Lind et al. Manuscript (1978)
	---	S,M,T	S	76.4	---	DAPC15S	2.36		Lind et al. Manuscript (1978)
	---	S,M,T	S	84.7	---	DAPC16S	6.62		Lind et al. Manuscript (1978)
	---	S,M,T	S	184	---	DAPC17S	7.14		Lind et al. Manuscript (1978)
	---	S,M,T	S	9.3	---	DAPC18S	1.11		Lind et al. Manuscript (1978)
	---	S,M,T	S	17.8	---	DAPC19S	2.11		Lind et al. Manuscript (1978)
	---	S,M,T	S	23.7	---	DAPC20S	2.67		Lind et al. Manuscript (1978)
	---	S,M,T	S	27.3	---	DAPC21S	2.77		Lind et al. Manuscript (1978)
	---	S,M,T	S	25.2	---	DAPC22S	2.81		Lind et al. Manuscript (1978)
	---	S,M,T	S	25.1	---	DAPC23S	2.60		Lind et al. Manuscript (1978)
	---	S,M,T	S	25.1	---	DAPC24S	2.31		Lind et al. Manuscript (1978)
Cladoceran, *Scapholeberis sp.*	adult	S,M,T	C	18	---	SCSP01S	9.73	9.73	Carlson et al. 1986
Amphipod,	1-3 d	F,M,T	S	22	---	GAPS01F	10.39	9.60	Arthur and Leonard 1970
Gammarus	1-3 d	F,M,T	S	19	---	GAPS02F	8.86		Arthur and Leonard 1970
Amphipod,	7-14 d	S,M,T	N	17	---	HYAZ01S	12.19	12.07	Schubauer-Berigan et al. 1993
Hyalella azteca	7-14 d	S,M,T	N	24	---	HYAZ02S	9.96		Schubauer-Berigan et al. 1993
	7-14 d	S,M,T	N	87	---	HYAZ03S	15.77		Schubauer-Berigan et al. 1993
	<7 d	S,M,T	S	24.3	---	HYAZ04S	8.26		Welsh 1996
	<7 d	S,M,T	S	23.8	---	HYAZ05S	8.09		Welsh 1996
	<7 d	S,M,T	S	8.2	---	HYAZ06S	15.49		Welsh 1996
	<7 d	S,M,T	S	10	---	HYAZ07S	18.80		Welsh 1996
Stonefly, *Acroneuria lycorias*	---	S,M,T	S	8300	---	ACLY01S	20636	20636	Warnick and Bell 1969
Midge, *Chironomus*	4th instar	S,M,T	S	739	---	CHDE01S	1987	1987	Kosalwat and Knight 1987
Shovelnose sturgeon, *Scaphirhynchus*	fry, 6.01 cm, 0.719 g	S,M,T	S	160	---	SCPL01S	69.63	69.63	Dwyer et al. 1999
Apache trout, *Oncorhynchus*	larval, 0.38 g	S,M,T	S	70	---	ONAP01S	32.54	32.54	Dwyer et al. 1995
Lahontan cutthroat	larval, 0.34 g	S,M,T	S	80	---	ONCL01S	34.26	32.97	Dwyer et al. 1995
Oncorhynchus clarki henshawi	larval, 0.57 g	S,M,T	S	60	---	ONCL02S	24.73		Dwyer et al. 1995

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Cutthroat trout,	7.4 cm, 4.2 g	F,M,T,D	C	398.91	367	ONCL03F	67.30		Chakoumakos et al. 1979
Oncorhynchus clarki	6.9 cm, 3.2 g	F,M,T,D	C	197.87	186	ONCL04F	44.91		Chakoumakos et al. 1979
	8.8 cm, 9.7 g	F,M,T,D	C	41.35	36.8	ONCL05F	21.87		Chakoumakos et al. 1979
	8.1 cm, 4.4 g	F,M,T,D	C	282.93	232	ONCL06F	51.94		Chakoumakos et al. 1979
	6.8 cm, 2.7 g	F,M,T,D	C	186.21	162	ONCL07F	111.3		Chakoumakos et al. 1979
	7.0 cm, 3.2 g	F,M,T,D	C	85.58	73.6	ONCL08F	39.53		Chakoumakos et al. 1979
	8.5 cm, 5.2 g	F,M,T,D	C	116.67	91	ONCL09F	19.63		Chakoumakos et al. 1979
	7.7 cm, 4.4 g	F,M,T,D	C	56.20	44.4	ONCL10F	18.81		Chakoumakos et al. 1979
	8.9 cm, 5.7 g	F,M,T,D	C	21.22	15.7	ONCL11F	10.60		Chakoumakos et al. 1979
Pink salmon,	alevin (newly hatched	F,M,T	S	143	---	ONGO01F	41.65	40.13	Servizi and Martens 1978
Oncorhynchus gorbu	alevin	F,M,T	S	87	---	ONGO02F	19.70		Servizi and Martens 1978
	fry	F,M,T	S	199	---	ONGO03F	78.76		Servizi and Martens 1978
Coho salmon,	6 g	R,M,T,I	---	164	---	ONKI01R	106.09	22.93	Buckley 1983
Oncorhynchus kisutc	parr	F,M,T	C	33	---	ONKI02F	20.94		Chapman 1975
	adult, 2.7 kg	F,M,T	C	46	---	ONKI03F	32.66		Chapman and Stevens 1978
	fry	F,M,T,D,I	---	61	49	ONKI04F	12.67		Mudge et al. 1993
	smolt	F,M,T,D,I	---	63	51	ONKI05F	13.19		Mudge et al. 1993
	fry	F,M,T,D,I	---	86	58	ONKI06F	11.95		Mudge et al. 1993
	parr	F,M,T,D,I	---	103	78	ONKI07F	22.98		Mudge et al. 1993
Rainbow trout,	larval, 0.67 g	S,M,T	S	110	---	ONMY01S	41.64	22.19	Dwyer et al. 1995
Oncorhynchus mykis	larval, 0.48 g	S,M,T	S	50	---	ONMY02S	25.26		Dwyer et al. 1995
	larval, 0.50 g	S,M,T	S	60	---	ONMY03S	29.46		Dwyer et al. 1995
	swim-up, 0.25 g	R,M,T,D	C	46.7	40	ONMY04R	10.90		Cacela et al. 1996
	swim-up, 0.25 g	R,M,T,D	C	24.2	19	ONMY05R	9.04		Cacela et al. 1996
	swim-up, 0.20-0.24 g	R,M,T,D	C	0	3.4	ONMY06R	5.02		Welsh et al. 2000
	swim-up, 0.20-0.24 g	R,M,T,D	C	0	8.1	ONMY07R	11.97		Welsh et al. 2000
	swim-up, 0.20-0.24 g	R,M,T,D	C	0	17.2	ONMY08R	13.80		Welsh et al. 2000
	swim-up, 0.20-0.24 g	R,M,T,D	C	0	32	ONMY09R	23.84		Welsh et al. 2000
	alevin	F,M,T	C	28	---	ONMY10F	20.30		Chapman 1975, 1978
	swim-up, 0.17 g	F,M,T	C	17	---	ONMY11F	12.54		Chapman 1975, 1978
	parr, 8.6 cm, 6.96 g	F,M,T	C	18	---	ONMY12F	9.87		Chapman 1975, 1978
	smolt, 18.8 cm, 68.19	F,M,T	C	29	---	ONMY13F	22.48		Chapman 1975, 1978
	1 g	F,M,T,D	C	-	169	ONMY14F	23.41		Chakoumakos et al. 1979
	4.9 cm	F,M,T,D	C	-	85.3	ONMY15F	10.20		Chakoumakos et al. 1979
	6.0 cm, 2.1 g	F,M,T,D	C	-	83.3	ONMY16F	9.93		Chakoumakos et al. 1979
	6.1 cm, 2.5 g	F,M,T,D	C	-	103	ONMY17F	12.71		Chakoumakos et al. 1979
	2.6 g	F,M,T,D	C	-	274	ONMY18F	44.54		Chakoumakos et al. 1979
	4.3 g	F,M,T,D	C	-	128	ONMY19F	16.51		Chakoumakos et al. 1979
	9.2 cm, 9.4 g	F,M,T,D	C	-	221	ONMY20F	33.33		Chakoumakos et al. 1979
	9.9 cm, 11.5 g	F,M,T,D	C	-	165	ONMY21F	22.70		Chakoumakos et al. 1979
	11.8 cm, 18.7 g	F,M,T,D	C	-	197	ONMY22F	28.60		Chakoumakos et al. 1979
	13.5 cm, 24.9 g	F,M,T,D	C	-	514	ONMY23F	99.97		Chakoumakos et al. 1979

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	13.4 cm, 25.6 g	F,M,T,D	C	-	243	ONMY24F	37.88		Chakoumakos et al. 1979
	6.7 cm, 2.65 g	F,M,T	C	2.8	---	ONMY25F	7.00		Cusimano et al. 1986
	parr	F,M,T,D,I	---	90	68	ONMY26F	19.73		Mudge et al. 1993
	swim-up, 0.29 g	F,M,T,D	C	19.6	18	ONMY27F	8.10		Cacela et al. 1996
	swim-up, 0.25 g	F,M,T,D	C	12.9	12	ONMY28F	32.15		Cacela et al. 1996
	swim-up, 0.23 g	F,M,T,D	C	5.9	5.7	ONMY29F	24.80		Cacela et al. 1996
	swim-up, 0.23 g	F,M,T,D	C	37.8	35	ONMY30F	16.16		Cacela et al. 1996
	swim-up, 0.26 g	F,M,T,D	C	25.1	18	ONMY31F	37.66		Cacela et al. 1996
	swim-up, 0.23 g	F,M,T,D	C	17.2	17	ONMY32F	24.19		Cacela et al. 1996
	0.64 g, 4.1 cm	F,M,T,D	C	101	---	ONMY33F	39.73		Hansen et al. 2000
	0.35 g, 3.4 cm	F,M,T,D	C	308	---	ONMY34F	85.83		Hansen et al. 2000
	0.68 g, 4.2 cm	F,M,T,D	C	93	---	ONMY35F	95.9		Hansen et al. 2000
	0.43 g, 3.7 cm	F,M,T,D	C	35.9	---	ONMY36F	50.83		Hansen et al. 2000
	0.29 g, 3.4 cm	F,M,T,D	C	54.4	---	ONMY37F	47.69		Hansen et al. 2000
Sockeye salmon,	alevin (newly hatched	F,M,T	S	190	---	ONNE01F	71.73	54.82	Servizi and Martens 1978
Oncorhynchus nerka	alevin	F,M,T	S	200	---	ONNE02F	79.52		Servizi and Martens 1978
	alevin	F,M,T	S	100	---	ONNE03F	23.74		Servizi and Martens 1978
	alevin	F,M,T	S	110	---	ONNE04F	27.22		Servizi and Martens 1978
	alevin	F,M,T	S	130	---	ONNE05F	35.36		Servizi and Martens 1978
	fry	F,M,T	S	150	---	ONNE06F	45.37		Servizi and Martens 1978
	smolt, 5.5 g	F,M,T	S	210	---	ONNE07F	87.77		Servizi and Martens 1978
	smolt, 5.5 g	F,M,T	S	170	---	ONNE08F	57.53		Servizi and Martens 1978
	smolt, 5.5 g	F,M,T	S	190	---	ONNE09F	71.73		Servizi and Martens 1978
	smolt, 4,8 g	F,M,T	S	240	---	ONNE10F	114.4		Servizi and Martens 1978
Chinook salmon,	alevin, 0.05 g	F,M,T	C	26	---	ONTS01F	14.48	25.02	Chapman 1975, 1978
Oncorhynchus tshaw	swim-up, 0.23 g	F,M,T	C	19	---	ONTS02F	10.44		Chapman 1975, 1978
	parr, 9.6 cm, 11.58 g	F,M,T	C	38	---	ONTS03F	28.30		Chapman 1975, 1978
	smolt, 14.4 cm, 32.46	F,M,T	C	26	---	ONTS04F	20.09		Chapman 1975, 1978
	3 mo, 1.35 g	F,M,T,I	C	10.2	---	ONTS05F	19.41		Chapman and McCrady 1977
	3 mo, 1.35 g	F,M,T,I	C	24.1	---	ONTS06F	30.91		Chapman and McCrady 1977
	3 mo, 1.35 g	F,M,T,I	C	82.5	---	ONTS07F	32.74		Chapman and McCrady 1977
	3 mo, 1.35 g	F,M,T,I	C	128.4	---	ONTS08F	20.66		Chapman and McCrady 1977
	swim-up, 0.36-0.45 g	F,M,T,D	C	0	7.4	ONTS09F	36.49		Welsh et al. 2000
	swim-up, 0.36-0.45 g	F,M,T,D	C	0	12.5	ONTS10F	30.85		Welsh et al. 2000
	swim-up, 0.36-0.45 g	F,M,T,D	C	0	14.3	ONTS11F	31.49		Welsh et al. 2000
	swim-up, 0.36-0.45 g	F,M,T,D	C	0	18.3	ONTS12F	48.56		Welsh et al. 2000

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Bull trout,	0.130 g, 2.6 cm	F,M,T,D	C	228	---	SACO01F	69.70	68.31	Hansen et al. 2000
Salvelinus confluent	0.555 g, 4.0 cm	F,M,T,D	C	207	---	SACO02F	63.62		Hansen et al. 2000
	0.774 g, 4.5 cm	F,M,T,D	C	66.6	---	SACO03F	74.18		Hansen et al. 2000
	1.520 g, 5.6 cm	F,M,T,D	C	50	---	SACO04F	63.60		Hansen et al. 2000
	1.160 g, 5.2 cm	F,M,T,D	C	89	---	SACO05F	71.11		Hansen et al. 2000
Chiselmouth, *Acrocheilus*	4.6 cm, 1.25 g	F,M,T	C	143	---	ACAL01F	216.3	216.3	Andros and Garton 1980
Bonytail chub, *Gila elegans*	larval, 0.29 g	S,M,T	S	200	---	GIEL01S	63.22	63.22	Dwyer et al. 1995
Golden shiner, *Notemigonus crysoleucas*	---	F,M,T	C	84600	---	NOCR01F	107860	107860	Hartwell et al. 1989
Fathead minnow,	adult, 40 mm	S,M,T	S	310	---	PIPR01S	266.3	69.63	Birge et al. 1983
Pimephales promela	adult, 40 mm	S,M,T	S	120	---	PIPR02S	105.61		Birge et al. 1983
	adult, 40 mm	S,M,T	S	390	---	PIPR03S	207.3		Birge et al. 1983; Benson & Birge
	---	S,M,T	C	55	---	PIPR04S	38.08		Carlson et al. 1986
	---	S,M,T	C	85	---	PIPR05S	70.71		Carlson et al. 1986
	<24 h	S,M,T	N	15	---	PIPR06S	11.23		Schubauer-Berigan et al. 1993
	<24 h	S,M,T	N	44	---	PIPR07S	18.03		Schubauer-Berigan et al. 1993
	<24 h	S,M,T	N	>200	---	PIPR08S	24.38		Schubauer-Berigan et al. 1993
	<24 h, 0.68 mg	S,M,T	S	4.82	---	PIPR09S	8.87		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	8.2	---	PIPR10S	16.72		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	31.57	---	PIPR11S	25.15		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	21.06	---	PIPR12S	17.67		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	35.97	---	PIPR13S	21.24		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	59.83	---	PIPR14S	16.64		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	4.83	---	PIPR15S	5.92		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	70.28	---	PIPR16S	13.34		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	83.59	---	PIPR17S	8.22		Welsh et al. 1993
	<24 h, 0.68 mg	S,M,T	S	182	---	PIPR18S	13.91		Welsh et al. 1993
	larval, 0.32 g	S,M,T	S	290	---	PIPR19S	73.92		Dwyer et al. 1995
	larval, 0.56 g	S,M,T	S	630	---	PIPR20S	157.9		Dwyer et al. 1995
	larval, 0.45 g	S,M,T	S	400	---	PIPR21S	103.2		Dwyer et al. 1995
	larval, 0.39 g	S,M,T	S	390	---	PIPR22S	161.7		Dwyer et al. 1995
	3.2-5.5 cm, 0.42-3.23	S,M,T	S	450	---	PIPR23S	152.9		Richards and Beitinger 1995
	2.8-5.1 cm, 0.30-2.38	S,M,T	S	297	---	PIPR24S	77.75		Richards and Beitinger 1995
	1.9-4.6 cm, 0.13-1.55	S,M,T	S	311	---	PIPR25S	67.56		Richards and Beitinger 1995
	3.0-4.8 cm, 0.23-1.36	S,M,T	S	513	---	PIPR26S	76.36		Richards and Beitinger 1995
	<24 h	S,M,T,D	S	62.23	53.96	PIPR27S	25.70		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	190.5	165.18	PIPR28S	87.89		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	68.58	59.46	PIPR29S	28.59		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	168.91	146.46	PIPR30S	89.18		Erickson et al. 1996a,b

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Species[a]	Organism Age, Size, or Lifestage	Method[b]	Chemical[c]	Reported LC50 or EC50 (total µg/L)[d]	Reported LC50 or EC50 (Diss. µg/L)[e]	BLM Data Label	BLM Normalized LC50 or EC50 (µg/L)[f]	Species Mean Acute Value (µg/L)[g]	Reference
	<24 h	S,M,T,D	S	94.62	82.04	PIPR31S	49.27		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	143.51	124.43	PIPR32S	104.90		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	120.65	103.76	PIPR33S	86.54		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	196.85	167.32	PIPR34S	122.0		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	133.35	120.02	PIPR35S	75.0		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	184.15	169.42	PIPR36S	122.2		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	304.8	268.22	PIPR37S	78.5		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	292.1	242.44	PIPR38S	201.5		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	133.35	113.35	PIPR39S	100.75		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	92.71	77.88	PIPR40S	72.95		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	152.4	128.02	PIPR41S	112.9		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	177.8	151.13	PIPR42S	136.3		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	203.2	166.62	PIPR43S	136.0		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	190.5	163.83	PIPR44S	147.7		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	196.85	157.48	PIPR45S	125.9		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	234.95	199.71	PIPR46S	157.4		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	146.05	128.52	PIPR47S	127.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	171.45	150.88	PIPR48S	153.9		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	152.4	131.06	PIPR49S	114.57		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	184.15	160.21	PIPR50S	131.3		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	203.2	182.88	PIPR51S	130.9		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	203.2	180.85	PIPR52S	105.76		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	203.2	176.78	PIPR53S	128.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	222.25	188.91	PIPR54S	122.1		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	146.05	125.60	PIPR55S	111.87		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	139.7	117.35	PIPR56S	85.45		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	139.7	114.55	PIPR57S	83.10		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	152.4	126.49	PIPR58S	85.82		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	203.2	172.72	PIPR59S	110.0		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	196.85	167.32	PIPR60S	106.46		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	266.7	226.70	PIPR61S	133.4		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	99.06	84.20	PIPR62S	138.0		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	111.13	97.79	PIPR63S	165.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	78.74	70.08	PIPR64S	114.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	92.71	81.58	PIPR65S	121.5		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	85.09	77.43	PIPR66S	106.69		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	123.19	110.87	PIPR67S	124.7		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	165.1	151.89	PIPR68S	114.24		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	190.5	175.26	PIPR69S	89.93		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	165.1	145.29	PIPR70S	140.2		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	127	111.76	PIPR71S	100.16		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	92.08	79.18	PIPR72S	58.74		Erickson et al. 1996a,b

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	<24 h	S,M,T,D	S	66.68	60.01	PIPR73S	37.67		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	393.70	370.08	PIPR74S	163.3		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	317.50	292.10	PIPR75S	252.2		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	107.95	101.47	PIPR76S	169.6		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	67.95	62.51	PIPR77S	146.5		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	45.72	42.06	PIPR78S	126.3		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	177.80	172.47	PIPR79S	197.6		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	13.97	12.43	PIPR80S	28.13		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	304.80	271.27	PIPR81S	149.2		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	71.12	71.12	PIPR82S	105.76		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	83.82	79.63	PIPR83S	108.41		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	104.78	99.54	PIPR84S	114.7		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	139.70	132.72	PIPR85S	137.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	152.40	137.16	PIPR86S	114.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	260.35	182.25	PIPR87S	114.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	488.95	268.92	PIPR88S	122.1		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	203.20	188.98	PIPR89S	147.5		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	704.85	662.56	PIPR90S	185.0		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	952.50	904.88	PIPR91S	197.1		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	1244.60	995.68	PIPR92S	188.3		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	1485.90	891.54	PIPR93S	135.5		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	781.05	757.62	PIPR94S	181.4		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	476.25	404.81	PIPR95S	172.5		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	273.05	262.13	PIPR96S	191.4		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	22.23	20.45	PIPR97S	59.14		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	24.13	23.16	PIPR98S	64.08		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	36.83	34.99	PIPR99S	97.49		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	27.94	27.94	PIPR100S	78.99		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	26.67	26.67	PIPR101S	72.86		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	20.32	20.32	PIPR102S	50.73		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	26.67	26.67	PIPR103S	68.24		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	190.50	182.88	PIPR104S	146.6		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	109.86	96.67	PIPR105S	93.76		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	203.20	182.88	PIPR106S	128.86		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	209.55	190.69	PIPR107S	113.0		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	146.05	127.06	PIPR108S	101.01		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	165.10	148.59	PIPR109S	120.9		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	254.00	223.52	PIPR110S	137.6		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	311.15	283.15	PIPR111S	142.9		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	165.10	150.24	PIPR112S	106.74		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	920.75	644.53	PIPR113S	131.9		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	1073.15	697.55	PIPR114S	116.5		Erickson et al. 1996a,b

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	<24 h	S,M,T,D	S	1003.30	752.48	PIPR115S	109.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	933.45	653.42	PIPR116S	123.2		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	742.95	646.37	PIPR117S	129.6		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	1879.60	939.80	PIPR118S	124.8		Erickson et al. 1996a,b
	<24 h	S,M,T,D	S	266.70	253.37	PIPR119S	176.1		Erickson et al. 1996a,b
	---	F,M,T	S	114.00	---	PIPR120F	17.99		Lind et al. Manuscript (1978)
	---	F,M,T	S	121.00	---	PIPR121F	19.70		Lind et al. Manuscript (1978)
	---	F,M,T	S	88.50	---	PIPR122F	13.27		Lind et al. Manuscript (1978)
	---	F,M,T	S	436.00	---	PIPR123F	78.50		Lind et al. Manuscript (1978)
	---	F,M,T	S	516.00	---	PIPR124F	50.09		Lind et al. Manuscript (1978)
	---	F,M,T	S	1586.00	---	PIPR125F	66.49		Lind et al. Manuscript (1978)
	---	F,M,T	S	1129.00	---	PIPR126F	73.03		Lind et al. Manuscript (1978)
	---	F,M,T	S	550.00	---	PIPR127F	42.76		Lind et al. Manuscript (1978)
	---	F,M,T	S	1001.00	---	PIPR128F	34.39		Lind et al. Manuscript (1978)
	30 d, 0.15 g	F,M,T,D	N	96.00	88.32	PIPR129F	39.58		Spehar and Fiandt 1986
	<24 h	F,M,T,D	S	31.75	27.94	PIPR130F	8.69		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	117.48	105.73	PIPR131F	37.88		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	48.26	40.06	PIPR132F	10.80		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	73.03	64.26	PIPR133F	22.19		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	59.06	49.02	PIPR134F	20.32		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	78.74	67.72	PIPR135F	18.51		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	22.23	18.67	PIPR136F	13.61		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	6.99	6.15	PIPR137F	10.94		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	22.23	20.45	PIPR138F	17.70		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	107.32	93.36	PIPR139F	67.09		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	292.10	245.36	PIPR140F	17.75		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	81.28	72.34	PIPR141F	41.16		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	298.45	229.81	PIPR142F	16.18		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	241.30	195.45	PIPR143F	24.40		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	133.35	109.35	PIPR144F	21.07		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	93.98	78.00	PIPR145F	50.83		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	67.95	45.52	PIPR146F	23.18		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	4.76	4.38	PIPR147F	40.09		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	13.97	12.43	PIPR148F	45.37		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	29.85	26.86	PIPR149F	59.43		Erickson et al. 1996a,b
	<24 h	F,M,T,D	S	59.69	51.33	PIPR150F	58.84		Erickson et al. 1996a,b
Northern squawfish,	larval, 0.32 g	S,M,T	S	380	---	PTLU01S	88.44	132.2	Dwyer et al. 1995
Ptychocheilus orego	larval, 0.34 g	S,M,T	S	480	---	PTLU02S	197.6		Dwyer et al. 1995

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Northern squawfish,	5.0 cm, 1.33 g	F,M,T	C	23	---	PTOR01F	17.02	14.61	Andros and Garton 1980
Ptychocheilus orego	7.2 cm, 3.69 g	F,M,T	C	18	---	PTOR02F	12.54		Andros and Garton 1980
Razorback sucker,	larval, 0.31 g	S,M,T	S	220	---	XYTE01S	63.78	78.66	Dwyer et al. 1995
Xyrauchen texanus	larval, 0.32 g	S,M,T	S	340	---	XYTE02S	97.0		Dwyer et al. 1995
Gila topminnow, *Poeciliposis*	2.72 cm, 0.219 g	S,M,T	S	160	---	POAC01S	56.15	56.15	Dwyer et al. 1999
Bluegill,	3.58 cm, 0.63 g	R,M,D	C	-	2200	LEMA01R	2202	2231	Blaylock et al. 1985
Lepomis macrochiru	12 cm, 35 g	F,M,T	S	1100	---	LEMA02F	2305		Benoit 1975
	2.8-6.8 cm	F,M,T	C	1000	---	LEMA03F	4200		Cairns et al. 1981
	3.58 cm, 0.63 g	F,M,D	C	-	1300	LEMA04F	1163		Blaylock et al. 1985
Fantail darter,	3.7 cm	S,M,T	S	330	---	ETFL01S	117.7	124.3	Lydy and Wissing 1988
Etheostoma flabellar	3.7 cm	S,M,T	S	341	---	ETFL02S	121.1		Lydy and Wissing 1988
	3.7 cm	S,M,T	S	373	---	ETFL03S	122.8		Lydy and Wissing 1988
	3.7 cm	S,M,T	S	392	---	ETFL04S	136.6		Lydy and Wissing 1988
Greenthroat darter, *Etheostoma*	2.26 cm, 0.133 g	S,M,T	S	260	---	ETLE01S	82.80	82.80	Dwyer et al. 1999
Johnny darter,	3.9 cm	S,M,T	S	493	---	ETNI01S	167.3	178.3	Lydy and Wissing 1988
Etheostoma nigrum	3.9 cm	S,M,T	S	483	---	ETNI02S	164.2		Lydy and Wissing 1988
	3.9 cm	S,M,T	S	602	---	ETNI03S	200.1		Lydy and Wissing 1988
	3.9 cm	S,M,T	S	548	---	ETNI04S	183.9		Lydy and Wissing 1988
Fountain darter, *Etheostoma rubrum*	2.02 cm, 0.062 g	S,M,T	S	60	---	ETRU01S	22.74	22.74	Dwyer et al. 1999
Boreal toad, *Bufo boreas*	tadpole, 0.012 g	S,M,T	S	120	---	BUBO01S	47.49	47.49	Dwyer et al. 1999

[a] Species appear in order taxonomically, with invertebrates listed first, fish, and an amphibian listed last. Species within each genus are ordered alphabetically. Within each species, tests are ordered by test method (static, renewal, flow-through) and date.

[b] S = static, R = renewal, F = flow-through, U = unmeasured, M = measured, T = exposure concentrations were measured as total copper, D = exposure concentrations were measured as dissolved copper.

[c] S = copper sulfate, N = copper nitrate, C = copper chloride.

[d] Values in this column are total copper LC50 or EC50 values as reported by the author.

[e] Values in this column are dissolved copper LC50 or EC50 values either reported by the author or if the author did not report a dissolved value then a conversion factor (CF) was applied to the total copper LC50 to estimate dissolved copper values.

[f] Normalization Chemistry

Temp	pH	Diss Cu	DOC	%HA	Ca	Mg	Na	K	SO_4	Cl	Alkalinity	S
Deg C		ug/L	mg/L		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
20.00	7.5	1.00	0.5	10.0	14.0	12.1	26.3	2.1	81.4	1.9	65.0	0.0003

g Underlined LC50s or EC50s not used to derive SMAV because considered extreme value.

* Table updated as of March 2, 2007

Table 2a. Chronic Toxicity of Copper to Freshwater Animals

Species	Test[a]	Chemical	Endpoint	Hardness (mg/L as CaCO3)	Chronic Limits (µg/L)	Chronic Values		Species Mean Chronic Value (Total µg/L)	Genus Mean Chronic Value (Total µg/L)	ACR	Reference
						Chronic Value[b] (µg/L)	EC20[b] (µg/L)				
Rotifer, *Brachionus calyciflorus*	LC,T	Copper sulfate	Intrinsic growth rate	85	2.5-5.0	3.54	-	3.54	3.54		Janssen et al. 1994
Snail, *Campeloma decisum (Test 1)*	LC,T	Copper sulfate	Survival	35-55	8-14.8	10.88	8.73	9.77	9.77	191.6	Arthur and Leonard 1970
Snail, *Campeloma decisum (Test 2)*	LC,T	Copper sulfate	Survival	35-55	8-14.8	10.88	10.94			153.0	Arthur and Leonard 1970
Cladoceran, *Ceriodaphnia dubia (New River)*	LC,D	-	Reproduction	179	6.3-9.9	7.90[c] (8.23)	-	19.3	19.3	3.599	Belanger et al. 1989
Cladoceran, *Ceriodaphnia dubia (Cinch River)*	LC,D	-	Reproduction	94.1	<19.3-19.3	<19.3	19.36[c] (20.17)			3.271	Belanger et al. 1989
Cladoceran, *Ceriodaphnia dubia*	LC,T	Copper sulfate	Survival and reproduction	57	-	24.50	-			0.547	Oris et al. 1991
Cladoceran, *Ceriodaphnia dubia*	LC,T	Copper sulfate	Survival and reproduction	57	-	34.60	-				Oris et al. 1991
Cladoceran, *Ceriodaphnia dubia*	LC,T,D	Copper chloride	Reproduction		12-32	19.59	9.17			2.069	Carlson et al. 1986
Cladoceran, *Daphnia magna*	LC,T	Copper chloride	Reproduction	85	10-30	17.32	-	14.1	8.96		Blaylock et al. 1985
Cladoceran, *Daphnia magna*	LC,T	Copper chloride	Carapace length	225	12.6-36.8	21.50	-				van Leeuwen et al. 1988
Cladoceran, *Daphnia magna*	LC,T	Copper chloride	Reproduction	51	11.4-16.3	13.63	12.58			2.067	Chapman et al. Manuscript
Cladoceran, *Daphnia magna*	LC,T	Copper chloride	Reproduction	104	20-43	29.33	19.89			1.697	Chapman et al. Manuscript
Cladoceran, *Daphnia magna*	LC,T	Copper chloride	Reproduction	211	7.2-12.6	9.53	6.06			11.39	Chapman et al. Manuscript
Cladoceran, *Daphnia pulex*	LC,T	Copper sulfate	Survival	57.5 (No HA)	4.0-6.0	4.90	2.83	5.68		9.104	Winner 1985
Cladoceran, *Daphnia pulex*	LC,T	Copper sulfate	Survival	115 (No HA)	5.0-10.0	7.07				3.904	Winner 1985
Cladoceran, *Daphnia pulex*	LC,T	Copper sulfate	Survival	230 (0.15 HA)	10-15	12.25	9.16			3.143	Winner 1985

Table 2a. Chronic Toxicity of Copper to Freshwater Animals

Species	Test[a]	Chemical	Endpoint	Hardness (mg/L as CaCO3)	Chronic Limits (µg/L)	Chronic Values		Species Mean Chronic Value (Total µg/L)	Genus Mean Chronic Value (Total µg/L)	ACR	Reference
						Chronic Value[b] (µg/L)	EC20[b] (µg/L)				
Caddisfly, *Clistoronia magnifica*	LC,T	Copper chloride	Emergence (adult 1st gen)	26	8.3-13	10.39	7.67	7.67	7.67		Nebeker et al. 1984b
Rainbow trout, *Oncorhynchus mykiss*	ELS,T continuous	Copper chloride	Biomass	120			27.77	23.8	11.9	2.881	Seim et al. 1984
Rainbow trout, *Oncorhynchus mykiss*	ELS,T	Copper sulfate	Biomass	160-180	12-22	16.25	20.32				Besser et al. 2001
Chinook salmon, *Oncorhynchus tshawytscha*	ELS,T	Copper chloride	Biomass	20-45	<7.4	<7.4	5.92	5.92		5.594	Chapman 1975, 1982
Brown trout, *Salmo trutta*	ELS,T	Copper sulfate	Biomass	45.4	20.8-43.8	29.91	-	29.9	29.9		McKim et al. 1978
Brook trout, *Salvelinus fontinalis*	PLC,T	Copper sulfate	Biomass	35.0	<5 -5	<5	-	12.5	19.7		Sauter et al. 1976
Brook trout, *Salvelinus fontinalis*	ELS,T	Copper sulfate	Biomass	45.4	22.3-43.5	31.15	-				McKim et al. 1978
Lake trout, *Salvelinus namaycush*	ELS, T	Copper sulfate	Biomass	45.4	22.0-43.5	30.94	-	30.9			McKim et al. 1978
Northern pike, *Esox lucius*	ELS, T	Copper sulfate	Biomass	45.4	34.9-104.4	60.36	-	60.4	60.4		McKim et al. 1978
Bluntnose minnow *Pimephales notatus*	LC,T	Copper sulfate	Egg production	172-230	<18-18	18.00	-	18.0	13.0	12.88	Horning and Neiheisel 1979
Fathead minnow, *Pimephales promelas*	ELS,T,D	-	Biomass	45			9.38	9.38		11.40	Lind et al. manuscript
White sucker, *Catostomus commersoni*	ELS, T	Copper sulfate	Biomass	45.4	12.9-33.8	20.88	-	20.9	20.9		McKim et al. 1978
Bluegill (larval), *Lepomis macrochirus*	ELS,T,D	Copper sulfate	Survival	44-50	21-40	28.98	27.15	27.2	27.2	40.52	Benoit 1975

[a] LC = life-cycle; PLC = partial life-cyle; ELS = early life state; T = total copper; D = dissolved copper.

[b] Results are based on copper, not the chemical.

[c] Chronic values based on dissolved copper concentration.

Table 2b. Chronic Toxicity of Copper to Saltwater Animals

Species	Test	Chemical	Salinity (g/kg)	Limits (µg/L)	Chronic Value (µg/L)	Chronic Value Dissolved (µg/L)	ACR	Reference
Sheepshead minnow, *Cyprinodon variegatus*	ELS	Copper chloride	30	172-362	249	206.7	1.48	Hughes et al. 1989

Table 2c. Acute-Chronic Ratios

Species	Hardness (mg/L as $CaCO_3$)	Acute Value (µg/L)	Chronic Value (µg/L)	Ratio	Reference	Overall Ratio for Species	
Snail,	35-55	1673[a]	8.73	191.61	Arthur and Leonard 1970		
Campeloma decisum	35-55	1673[a]	10.94	152.95	Arthur and Leonard 1970	171.19	
Cladoceran,	[redacted]	28.42[b]	7.90	3.60	[redacted]		
Ceriodaphnia dubia	[redacted]	63.33[b]	19.36	3.27	[redacted]		
	57	13.4	24.5	0.55	Oris et al. 1991		
	--	[redacted]	9.17	1.96	[redacted]	2.85[g]	✓
Cladoceran,	51	26	12.58	2.07	Chapman et al. Manuscript		
Daphnia magna	104	33.76[d]	19.89	1.70	Chapman et al. Manuscript		
	211	69	6.06	11.39	Chapman et al. Manuscript	3.42	✓
Cladoceran,	57.5	25.737	2.83	9.10	[redacted]		
Daphnia pulex	115	27.6	7.07	3.90	[redacted]		
	230	28.79	9.16	3.14	[redacted]	4.82	✓
Rainbow trout, *Oncorhynchus mykiss*	120	80	27.77	2.88	Seim et al. 1984	2.88	✓
Chinook salmon, *Oncorhynchus tshawytscha*	20-45	33.1	5.92	5.59	Chapman 1975, 1982	5.59	✓
Bluntnose minnow, *Pimephales notatus*	172-230	231.9[e]	18	12.88	Horning and Neiheisel 1979	12.88	
Fathead minnow, *Pimephales promelas*	45	106.875[f]	9.38	11.40	Lind et al. 1978	11.40	
Bluegill, *Lepomis macrochirus*	21-40	1100	27.15	40.52	Benoit 1975	40.49	
Sheepshead minnow, *Cyprinodon variegatus*	-	368	249	1.48	Hughes et al. 1989	1.48	✓

[a]Geometric mean of two values from Arthur and Leonard (1970) in Table 1.

[b]Geometric mean of five values from Belanger et al. (1989) in Table 1. ACR is based on dissolved metal measurements.

[c]Geometric mean of two values from Carlson et al. (1986) in Table 1.

[d]Geometric mean of two values from Chapman manuscript in Table 1.

[e]Geometric mean of two values of three values from Horning and Neiheisel (1979) in Appendix C.

[f]Geometric mean of three values from Lind et al. (1978) in Table 1.

[g]ACR from Oris et al. (1991) not used in calculating overall ratio for species because it is <1.

FACR
Freshwater final acute-chronic ratio = 3.22
Saltwater final acute-chronic ratio = 3.22

* Table updated as of March 2, 2007

Table 3a. Ranked Freshwater Genus Mean Acute Values with Species Mean Acute-Chronic Ratios

Rank	GMAV	Species	SMAV (µg/L)	ACR
27	107,860	Golden shiner, *Notemigonus crysoleucas*	107,860	
26	20,636	Stonefly, *Acroneuria lycorias*	20,636	
25	3,573	Snail, *Campeloma decisum*	3,573	171.19
24	2,231	Bluegill sunfish, *Lepomis macrochirus*	2,231	40.49
23	1,987	Midge, *Chironomus decorus*	1,987	
22	216.3	Chiselmouth, *Acrocheilus alutaceus*	216.3	
21	80.38	Fantail darter, *Etheostoma flabellare*	124.3	
		Greenthroat darter, *Etheostoma lepidum*	82.80	
		Johnny darter, *Etheostoma nigrum*	178.3	
		Fountain darter, *Etheostoma rubrum*	22.74	
20	78.66	Razorback sucker, *Xyrauchen texanus*	78.66	
19	69.63	Fathead minnow, *Pimephales promelas*	69.63	11.40
18	69.63	Shovelnose sturgeon, *Scaphirhynchus platorynchus*	69.63	
17	68.31	Bull trout, *Salvelinus confluentus*	68.31	
16	63.22	Bonytail chub, *Gila elegans*	63.22	
15	56.15	Gila topminnow, *Poeciliposis occidentalis*	56.15	
14	52.51	Freshwater mussel, *Utterbackia imbecillis*	52.51	
13	48.41	Worm, *Lumbriculus variegatus*	48.41	
12	47.49	Boreal toad, *Bufo boreas*	47.49	
11	43.94	Colorado squawfish, *Ptychocheilus lucius*	132.2	
		Northern squawfish, *Ptychocheilus oregonensis*	14.61	
10	31.39	Apache trout, *Oncorhynchus apache*	32.54	
		Cutthroat trout, *Oncorhynchus clarki*	32.97	
		Pink salmon, *Oncorhynchus gorbuscha*	40.13	
		Coho salmon, *Oncorhynchus kisutch*	22.93	
		Rainbow trout, *Oncorhynchus mykiss*	22.19	2.88
		Sockeye salmon, *Oncorhynchus nerka*	54.82	
		Chinook salmon, *Oncorhynchus tshawytscha*	25.02	5.59
9	20.41	Snail, *Physa integra*	20.41	
8	12.31	Snail, *Juga plicifera*	12.31	
7	12.07	Amphipod, *Hyalella azteca*	12.07	
6	11.33	Freshwater mussel, *Actinonaias pectorosa*	11.33	
5	9.73	Cladoceran, *Scapholeberis sp.*	9.73	
4	9.60	Amphipod, *Gammarus pseudolimnaeus*	9.60	
3	6.67	Snail, *Lithoglyphus virens*	6.67	
2	5.93	Cladoceran, *Ceriodaphnia dubia*	5.93	2.85
1	4.05	Cladoceran, *Daphnia magna*	6.00	3.42
		Cladoceran, *Daphnia pulicaria*	2.73	

* Table updated as of March 2, 2007

Table 3b. Freshwater Final Acute Value (FAV) and Criteria Calculations

Calculated Freshwater FAV based on 4 lowest values: Total Number of GMAVs in Data Set = 27

Rank	GMAV	lnGMAV	$(lnGMAV)^2$	P = R/(n+1)	SQRT(P)
4	9.600	2.261	5.114	0.143	0.378
3	6.670	1.897	3.599	0.107	0.327
2	5.930	1.780	3.170	0.071	0.267
1	4.050	1.398	1.954	0.036	0.189
Sum:		**7.33671**	**13.83657**	**0.35714**	**1.16153**

S =	4.374
L =	0.5641
A =	1.542
Calculated FAV =	**4.674452**
Calculated CMC =	**2.337**

Dissolved Copper Criterion Maximum Concentration (CMC) = 2.337 µg/L (for example normalization chemistry see Table 1, footnote f)
Criteria Lethal Accumulation (LA50) based on example normalization chemistry = 0.03395 nmol/g wet wt
Criterion Continuous Concentration (CCC) = 4.67445/3.22 = 1.4516932 µg/L (for example normalization chemistry see Table 1, footnote f)

S = Scale parameter or slope
L = Location parameter or intercept
P = Cumulative probability
A = lnFAV

* Table updated as of March 2, 2007

Table 4. Toxicity of Copper to Freshwater Plants

Species	Method[a]	Chemical	Hardness (mg/L as $CaCO_3$)	Duration	Effect	Result[b] (Total µg/L)	Reference
Blue-green alga, *Anabaena flos-aqua*	S,U	Copper sulfate	65.2	96 hr	EC75 (cell density)	200	Young and Lisk 1972
Bllue-green alga, *Anabaena variabilis*	S,U	Copper sulfate	65.2	-	EC85 (wet weight)	100	Young and Lisk 1972
Blue-green alga, *Anabaena* strain 7120	-	-	-	-	Lag in growth	64	Laube et al. 1980
Blue-green alga, *Chroococcus paris*	S,U	Copper nitrate	54.7	10 days	Growth reduction	100	Les and Walker 1984
Blue-green alga, *Microcystis aeruginosa*	S,U	Copper sulfate	54.9	8 days	Incipient inhibition	30	Bringmann 1975; Bringmann and Kuhn 1976, 1978a,b
Alga, *Ankistrodesmus braunii*	-	-	-	-	Growth reduction	640	Laube et al. 1980
Green alga, *Chlamydomonas* sp.	S,U	Copper sulfate	68	10 days	Growth inhibition	8,000	Cairns et al. 1978
Green alga, *Chlamydomonas reinhardtii*	S,M,T	-	90 - 133	72 hr	NOEC (deflagellation)	12.2-49.1	Winner and Owen 1991a
Green alga, *Chlamydomonas reinhardtii*	S,M,T	-	90 - 133	72 hr	NOEC (cell density)	12.2-43.0	Winner and Owen 1991a
Green alga, *Chlamydomonas reinhardtii*	F,M,T	-	24	10 days	EC50 (cell density)	31.5	Schafer et al. 1993
Green alga, *Chlorella pyrenoidosa*	S,U	-	-	96 hr	ca. 12 hr lag in growth	1	Steeman-Nielsen and Wium-Andersen 1970
Green alga, *Chlorella pyrenoidosa*	S,U	-	54.7	-	Growth inhibition	100	Steeman-Nielsen and Kamp-Nielsen 1970
Green alga, *Chlorella pyrenoidosa*	S,U	Copper sulfate	365	14 days	EC50 (dry weight)	78-100	Bednarz and Warkowska-Dratnal 1985
Green alga, *Chlorella pyrenoidosa*	S,U	Copper sulfate	36.5	14 days	EC50 (dry weight)	78-100	Bednarz and Warkowska-Dratnal 1985
Green alga, *Chlorella pyrenoidosa*	S,U	Copper sulfate	3.65	14 days	EC50 (dry weight)	78-100	Bednarz and Warkowska-Dratnal 1983/1984
Green alga, *Chlorella saccharophila*	S,U	Copper chloride	-	96 hr	96-h EC50	550	Rachlin et al. 1982
Green alga, *Chlorella vulgaris*	S,U	Copper sulfate	2,000	96 hr	Growth inhibition	200	Young and Lisk 1972
Green alga, *Chlorella vulgaris*	S,U	Copper chloride		33 days	EC20 (growth)	42	Rosko and Rachlin 1977
Green alga, *Chlorella vulgaris*	F,U	Copper sulfate	-	96 hr	EC50 or EC50 (cell numbers)	62	Ferard et al. 1983
Green alga, *Chlorella vulgaris*	S,M,D	Copper sulfate	-	96 hr	IC50	270	Ferard et al. 1983
Green alga, *Chlorella vulgaris*	S,M,T	Copper chloride	-	96 hr	EC50 (cell density)	200	Blaylock et al. 1985
Green alga, *Chlorella vulgaris*	S,U	Copper sulfate	17.1	7 days	15% reduction in cell density	100	Bilgrami and Kumar 1997

Table 4. Toxicity of Copper to Freshwater Plants

Species	Method[a]	Chemical	Hardness (mg/L as $CaCO_3$)	Duration	Effect	Result[b] (Total µg/L)	Reference
Green alga, *Scenedesmus quadricauda*	S,U	Copper sulfate	68	10 days	Growth reduction	8,000	Cairns et al. 1978
Green alga, *Scenedesmus quadricauda*	S,U	Copper sulfate	181	7 days	LOEC (growth)	1,100	Bringmann and Kuhn 1977a, 1978a,b, 1979, 1980a
Green alga, *Selenastrum capricornutum*	S,U	Copper chloride	14.9	14 days	EC50 (cell volume)	85	Christensen et al. 1979
Green alga, *Selenastrum capricornutum*	S,U	Copper chloride	14.9	7 days	LOEC (growth)	50	Bartlett et al. 1974
Green alga, *Selenastrum capricornutum*	S,M,T	Copper chloride	24.2	96 hr	EC50 (cell count)	400	Blaylock et al. 1985
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	9.3	96 hr	EC50 (cell count)	48.4	Blaise et al. 1986
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	9.3	96 hr	EC50 (cell count)	44.3	Blaise et al. 1986
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	9.3	96 hr	EC50 (cell count)	46.4	Blaise et al. 1986
Green alga, *Selenastrum capricornutum*	S,U	Copper chloride	15	2-3 wk	EC50 (biomass)	53.7	Turbak et al. 1986
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	14.9	5 days	Growth reduction	58	Nyholm 1990
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	9.3	96 hr	EC50 (cell count)	69.9	St. Laurent et al. 1992
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	9.3	96 hr	EC50 (cell count)	65.7	St. Laurent et al. 1992
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	24.2	96 hr	EC50 (cell count)	54.4	Radetski et al. 1995
Green alga, *Selenastrum capricornutum*	R,U	Copper sulfate	24.2	96 hr	EC50 (cell count)	48.2	Radetski et al. 1995
Green alga, *Selenastrum capricornutum*	S,U	Copper sulfate	16	96 hr	EC50 (cell density)	38	Chen et al. 1997
Algae, mixed culture	S,U	Copper sulfate	-	-	Significant reduction in blue-green algae and nitrogen fixation	5	Elder and Horne 1978
Diatom, *Cyclotella meneghiniana*	S,U	Copper sulfate	68	10 days	Growth inhibition	8,000	Cairns et al. 1978
Diatom, *Navicula incerta*	S,U	Copper chloride	-	96 hr	EC50	10,429	Rachlin et al. 1983
Diatom, *Nitzschia linearis*	-	-	-	5 day	EC50	795-815	Academy of Natural Sciences 1960; Patrick et al. 1968
Diatom, *Nitzschia palea*	-	-	-	-	Complete growth inhibition	5	Steeman-Nielsen and Wium-Andersen 1970
Duckweed, *Lemna minor*	F	-	-	7 day	EC50	119	Walbridge 1977
Duckweed, *Lemna minor*	S,U	Copper sulfate	-	28 days	Significant plant damage	130	Brown and Rattigan 1979

Table 4. Toxicity of Copper to Freshwater Plants

Species	Method[a]	Chemical	Hardness (mg/L as $CaCO_3$)	Duration	Effect	Result[b] (Total µg/L)	Reference
Duckweed, *Lemna minor*	S,U	-	0	96 hr	EC50 (frond number)	1,100	Wang 1986
Duckweed, *Lemna minor*	S,U	Copper sulfate	78	96 hr	EC50 (chlorophyll a reduction)	250	Eloranta et al. 1988
Duckweed, *Lemna minor*	R,M,T	Copper nitrate	39	96 hr	Reduced chlorophyll production	24	Taraldsen and Norberg-King 1990
Eurasian watermilfoil, *Myriophyllum spicatum*	S,U	-	89	32 days	EC50 (root weight)	250	Stanley 1974

[a] S=Static; R=Renewal; F=Flow-through; M=Measured; U=Unmeasured; T=Total metal conc. measured; D=dissolved metal conc. measured.

[b] Results are expressed as copper, not as the chemical.

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