

APPENDIX F

POREWATER BENCHMARKS

UPPER COLUMBIA RIVER

FINAL Aquatic Baseline Ecological Risk Assessment Appendix F Porewater Benchmarks

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July 2026

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ACRONYMS AND ABBREVIATIONS

ACR	acute-to-chronic ratio
ALB	aquatic life benchmark
AWQC	ambient water quality criteria
BERA	baseline ecological risk assessment
BLM	biotic ligand model
COPC	chemical of potential concern
CR	Columbia River
dph	days post hatch
DOC	dissolved organic carbon
EAE	ecological assessment endpoint
EC10	effect concentration of 10 percent
EC20	effect concentration of 20 percent
EC50	effect concentration of 50 percent
Eco-SSL	ecological soil screening level
ELS	early life stage
EPA	U.S. Environmental Protection Agency
GMCV	genus mean chronic value
GSD	genus sensitivity distribution
KR	Kootenai River
LC50	dose that is lethal to 50 percent of an exposed population
LOEC	lowest-observed-effect concentration
logX50	the log concentration associated with a 50 percent effect
MATC	maximum acceptable toxicant concentration
MLR	multiple linear regression
N/AP	not applicable
N/AV	not available
PNEC	predicted no-effect concentration
OU	operable unit

S	steepness parameter
TAI	Teck American Incorporated
TRAP	Toxicity Relationship Analysis Program
TRV	toxicity reference value
TU	toxic unit
UCR	Upper Columbia River
Windward	Windward Environmental LLC
Y ₀	the control response

UNITS OF MEASURE

d	day(s)
hr	hour(s)
µg/L	microgram(s) per liter
mg/L	milligram(s) per liter
nmol/g	nanomole(s) per gram
ww	wet weight

1 INTRODUCTION

This appendix presents toxicity benchmarks for aquatic life exposed to chemicals of potential concern (COPCs) in sediment porewater. These benchmarks are used by Teck American Incorporated (TAI) as discussed in Volume 1 of the Upper Columbia River (UCR) Aquatic Baseline Ecological Risk Assessment (BERA) for Operable Unit (OU) 1 and OU 2.

Two tiers of porewater toxicity benchmarks were developed. The first tier comprised toxicity benchmarks based on U.S. Environmental Protection Agency (EPA) ambient water quality criteria (AWQC) for the protection of aquatic life and values developed following EPA guidelines for AWQC development. As per EPA guidance, these AWQC (and “AWQC-like” values) are intended to be protective of aquatic communities; they are therefore referred to as “aquatic life benchmarks” in this report (these values are developed to be protective of 95 percent of the taxa tested). The second tier comprised taxa-specific porewater toxicity benchmarks developed using the most sensitive taxon within each of the taxonomic groups potentially exposed to porewater (i.e., benthic invertebrate, early life stage [ELS] white sturgeon and other fish, and aquatic plants).¹ If the most sensitive taxon within a taxonomic group was less sensitive than the aquatic life benchmark (see Section 2.1 for details), a benchmark specific to that taxonomic group was identified. If the most sensitive taxon within a taxonomic group had a sensitivity comparable in magnitude to the aquatic life benchmark, the benchmark for that taxon was set equal to the aquatic life benchmark.

The sensitivity of ELS white sturgeon to COPCs—particularly cadmium, copper, lead, and zinc—is reviewed in more detail in this appendix. Also, for copper, lead, and zinc, white sturgeon toxicity tests were published after EPA’s AWQCs had been developed. As such, white sturgeon-specific benchmarks were then developed, as appropriate, for application in the Aquatic BERA.

The benchmark approach presented in this appendix addresses chemicals identified as COPCs for benthic invertebrate, fish, and aquatic plants in UCR porewater in the COPC

¹ In conducting the Aquatic BERA, it may also be possible to develop receptor-specific toxicity benchmarks when toxicity data are available for a given site-specific receptor. The toxicity data sets used to develop the Tier 1 and 2 toxicity benchmarks in this report are tabulated in Attachment F1 and are thus available to risk assessors conducting the Aquatic BERA.

Refinement (TAI 2020)². The identified chemicals were ammonia³, aluminum, antimony, cadmium, copper, iron, lead, manganese, molybdenum⁴, selenium, and zinc. In the Aquatic BERA, the benchmarks are compared to COPC concentrations in porewater collected in the field. The measured porewater COPC concentrations are used to estimate porewater exposure point concentrations for benthic invertebrate, ELS fish (including sturgeon), and aquatic plants.

Note that the porewater benchmarks developed in this appendix and applied in the Aquatic BERA are for individual metals. They do not explicitly account for mixtures of metals (as directed by USEPA [2020], pers. comm.). The uncertainty associated with evaluating individual metals only is addressed in the uncertainty analyses conducted in the Aquatic BERA for the relevant receptor groups (see Appendix O). For example, in the uncertainty analysis for the assessment of ELS white sturgeon directly exposed to porewater, the findings from five metals mixtures models, as presented by (Balistreri 2023), are qualitatively compared to the individual hazard quotients calculated for each COPC with sturgeon-specific porewater benchmarks presented in this appendix. The objective of this comparison is to determine whether considering metals mixtures (as opposed to individual metals) leads to different conclusions about the potential for adverse effects on ELS sturgeon exposed to porewater. The benchmarks presented for field-collected porewater are based on the following models, which consider parameters affecting the bioavailability of metals and metalloids in porewater:

- EPA's hardness-based statistical regression models used to establish AWQC for cadmium, lead, and zinc
- EPA's biotic ligand model (BLM) used to establish AWQC for copper; the BLM predicts the toxicities of bioavailable metals by considering chemistry parameters including pH, dissolved organic carbon (DOC), alkalinity, specific cations and anions, and sulfide

² The COPC Refinement (TAI 2019) included an addendum that was finalized in January 2020 (TAI 2020). This citation (TAI 2020) is used in the remainder of this appendix to cite the COPC Refinement document.

³ Ammonia was identified as a COPC based on the inclusion of laboratory porewater data in the COPC Refinement. However, porewater benchmarks are compared to field porewater data in the Aquatic BERA, and field porewater data are not available for ammonia, so a benchmark for ammonia has been not derived. Instead, a qualitative discussion of the potential toxicity of ammonia to aquatic organisms in the UCR is included in the Aquatic BERA.

⁴ Molybdenum was identified as a COPC because no benchmark had been identified as part of the COPC Refinement process.

- Other statistical regression models, including:
 - Multiple linear regression (MLR) models for aluminum and iron with DOC, pH, and hardness as parameters affecting bioavailability
 - A hardness model for manganese developed following EPA guidance.

In the absence of a bioavailability model, benchmarks are based on available toxicity data sets that have not been adjusted for bioavailability.⁵ These models and data sets have varying degrees of uncertainty, which are discussed in Section 4 and considered in the weight-of-evidence analysis in the Aquatic BERA.

Section 2 of this appendix describes methods for deriving porewater benchmarks based on the BLM, hardness-based models, other statistically based regression models, or available toxicity data if no model accounting for bioavailability has been developed. Section 3 presents the benchmarks derived using methods presented in Section 2; for benchmarks derived using models, example values are presented using representative DOC, hardness, and pH conditions. Section 3 also includes white sturgeon-specific toxicity data evaluations for cadmium, copper, lead, and zinc, as well as the development of concentration-response relationships for copper and zinc. Section 4 discusses the uncertainties associated with deriving porewater benchmarks, and Section 5 provides a summary of the benchmarks used in the Aquatic BERA. Section 6 provides a list of the documents referenced in this appendix.

⁵ This applies to antimony in this appendix.

2 METHODS FOR DERIVING BENCHMARKS

This section summarizes the methods used to develop the porewater benchmarks. Section 2.1 describes the general approach for developing the porewater benchmarks, while Section 2.2 describes technical considerations that varied among individual COPCs or subsets of COPCs, including COPC-specific methods for adjusting benchmarks to account for site-specific bioavailability.

2.1 GENERAL APPROACH FOR CALCULATING BENCHMARKS

A 20 percent effect level was targeted for developing porewater benchmarks when using the BLM and regression model approaches. This effect level is consistent with EPA guidance and development of other toxicity benchmarks, as follows:

- To derive chronic AWQC for ammonia (USEPA 2013) and cadmium (USEPA 2016a), EPA used effect concentrations of 20 percent (i.e., EC20s) in water to derive chronic criteria. EPA also considered EC20s to be acceptable and appropriate chronic toxicity values when deriving the carbaryl AWQC (USEPA 2012). As noted by USEPA (2016a), “The endpoint for chronic exposure is the EC20, 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 EC5 or EC10, which is rarely statistically significantly different from the control treatment. EPA selected an EC20 to estimate a low level of effect that would be statistically different from control effects, but not severe enough to cause chronic effects at the population level (USEPA 1999).”
- USEPA (2003) prioritized EC20s to derive plant and soil invertebrate benchmarks in the ecological soil screening level (Eco-SSL) development process. This default approach was based on the assumptions that most experimental studies cannot detect smaller changes with acceptable power, and that changes of 20 percent or less will often not result in population-level impacts for many endpoints (USEPA 2003). EPA’s Eco-SSLs are used as soil benchmarks in the Aquatic BERA. For COPCs analyzed in the Aquatic BERA that lack Eco-SSLs, EPA has approved the use of soil screening levels developed using EPA’s Eco-SSL derivation process, which includes using the prioritized effect value of 20 percent (Attachment G1 of Appendix G).
- Suter and Tsao (1996) used EC20s to define chronic toxicity values for the development of toxicity benchmarks for aquatic life. The authors noted that the

benchmarks were intended to be indices of population production. The benchmarks represented the approximate mean levels of effect on individual response parameters observed at chronic values (defined as the geometric mean of the no-observed-effect concentration and lowest-observed-effect concentration [LOEC]) and the minimum detectable differences in population characteristics in the field.

In addition, the 20 percent effect level was the preferred threshold used to develop the EPA-approved wildlife toxicity reference values (TRVs) used in the Aquatic BERA (Attachment H1 of Appendix H).

The overall process for deriving the porewater benchmarks used in the Aquatic BERA is summarized in Figure 2-1. The first step was to identify an “aquatic life benchmark”; for most COPCs, this value was either EPA’s existing chronic AWQC for a COPC or, in the absence of an EPA AWQC, a value derived following EPA guidance for AWQC development (Stephan et al. 1985). Developing an AWQC required synthesizing EPA’s AWQC and associated genus sensitivity distributions (GSDs) used to derive the AWQC. For two of the COPCs—antimony and iron—GSDs were derived in support of benchmark development for this BERA, because they had not been derived previously by EPA. These GSDs were derived following EPA guidelines for AWQC development (Stephan et al. 1985). GSDs were not developed for molybdenum, for which an aquatic life benchmark was identified from the scientific literature (see Section 2.2.7), or selenium, for which no porewater benchmarks were identified because porewater was considered a weak line of evidence relative to other metrics (see Section 2.2.8).

For benchmarks derived using a bioavailability model (e.g., the BLM, hardness model, an MLR model), GSDs were normalized to a constant set of bioavailability conditions. If the data are not normalized, the different test conditions under which different toxicity studies are conducted can confound a comparison of organism sensitivity. For example, if a sensitive organism is tested in hard, alkaline water and an insensitive organism is tested in soft, acidic water, the less sensitive organism may appear to be more sensitive simply because of test conditions. Strictly speaking, it does not matter to what water chemistry the data are normalized, but it is helpful to select a standard chemistry that is relevant to biological testing. In this evaluation, the standard test water selected had a DOC of 1 mg/L, a hardness of 100 mg/L, and a pH of 7.

The aquatic life benchmarks based on EPA’s AWQC, or derived based on EPA’s methods for developing AWQC (Stephan et al. 1985), were generally equal to the 5th percentile of

the GSD (Figure 2-1),⁶ where the GSDs included data for both invertebrates⁷ and fish, as EPA does not include aquatic plants in GSDs used to develop AWQC. If sufficient chronic data were not available but acute data were sufficient, an acute-to-chronic ratio (ACR) was used to estimate the chronic GSD, per EPA's guidelines (Stephan et al. 1985).⁸

Taxa-specific benchmarks for benthic invertebrates or fish were developed if the lowest genus mean chronic value (GMCV) for either taxa in the GSD was more than a factor of two of the aquatic life benchmark, and if GMCVs included data for at least three species for the relevant taxa (Figure 2-1). The rationale for separate benthic invertebrate- and fish-specific benchmarks in these cases is based on the general observation that toxicity results for repeated tests in the same laboratory may reasonably vary by a factor of two (Moore et al. 2000). If the lowest GMCV for invertebrates or fish was more than a factor of two greater than the 5th percentile of the GSD, it was considered evidence that invertebrates or fish were less sensitive than the genera that were the "drivers" of the 5th percentile of the GSD.⁹ When the lowest GMCV for invertebrates or fish was at least twice the 5th percentile of the GSD, the lowest GMCV was used as the taxa-specific benchmark for invertebrates or fish, as long as data were available for at least three species and, for fish, as long as at least one of those species was a salmonid. Note that for fish, an additional step was included to consider the relative sensitivity of white sturgeon (see below, Figure 2-1).

⁶ The one exception was iron, for which the EPA's criterion was initially developed in the 1970s—the criterion is principally based on field observations rather than a GSD. An iron GSD was identified herein to support development of a porewater benchmark.

⁷ All macroinvertebrates, including planktonic macroinvertebrates such as daphnids, were considered benthic macroinvertebrate surrogates, consistent with USEPA (2005) equilibrium partitioning sediment benchmark guidance, which notes that "The distribution of sensitivities of benthic organisms to chemicals is similar to that of water column organisms."

⁸ Because AWQC are developed based on the 5th percentile of the GSD, EPA guidance dictates estimating the 5th percentile of the chronic GSD by applying an ACR to just the 5th percentile of the acute GSD. This approach is used because ACRs sometimes vary among acutely sensitive and acutely insensitive species; therefore, the ACR is selected based on sensitive species. The same concept applies to the present evaluation, in which benchmarks are being developed for sensitive invertebrate and fish species.

⁹ Because the GSDs were almost entirely composed of invertebrates and fish, the 5th percentile of the GSD was usually identified as either the benthic macroinvertebrate benchmark or the fish benchmark, and sometimes both. Antimony, for which the 5th percentile of the GSD was driven by an amphibian, was an exception.

White sturgeon-specific benchmarks were also derived for COPCs with acceptable ELS toxicity data, namely cadmium, copper, lead, and zinc (Figure 2-1).^{10,11} Acute and chronic cadmium, copper, lead, and zinc toxicity data were compiled for white sturgeon. Data for all life stages were compiled to provide a complete picture of white sturgeon sensitivity to these metals. Although porewater benchmarks are being developed to evaluate chronic exposures, acute data were compiled to confirm that EPA's chronic AWQC are protective of potentially sensitive ELS individuals over short exposure durations. Details on the development of white sturgeon-specific benchmarks are provided in Section 3.

Lastly, a plant-specific benchmark was developed if toxicity data were available for at least three plant species;¹² in that case, the plant benchmark was based on the lowest chronic genus value (Figure 2-1). Aquatic plant toxicity data were compiled from sources used to develop GSDs based on invertebrates and fish, as available, and from searches of EPA's ECOTOX database (USEPA 2018a). Following EPA's AWQC derivation guidance (Stephan et al. 1985), preference was given to chronic tests in which the test chemical was analytically measured before or during the test. If toxicity data were available for fewer than three plant species, no taxa-specific benchmark was derived for aquatic plants.

All benchmarks—aquatic life, benthic invertebrates, fish, white sturgeon, and aquatic plants—are reported to two significant figures herein (consistent with EPA's AWQC).

¹⁰ ELS sturgeon exhibit epibenthic behavior (ELS being the only life stage during which white sturgeon do so) and therefore may be exposed to porewater, although the relevance of porewater to this life stage has not been demonstrated.

¹¹ In the absence of white sturgeon-specific toxicity data, the benchmark for evaluating white sturgeon was set equal to the fish benchmark.

¹² Porewater benchmarks for algae were not developed because the porewater exposure pathway for periphyton/algae is considered incomplete in the conceptual site model for the Aquatic BERA (TAI 2012).

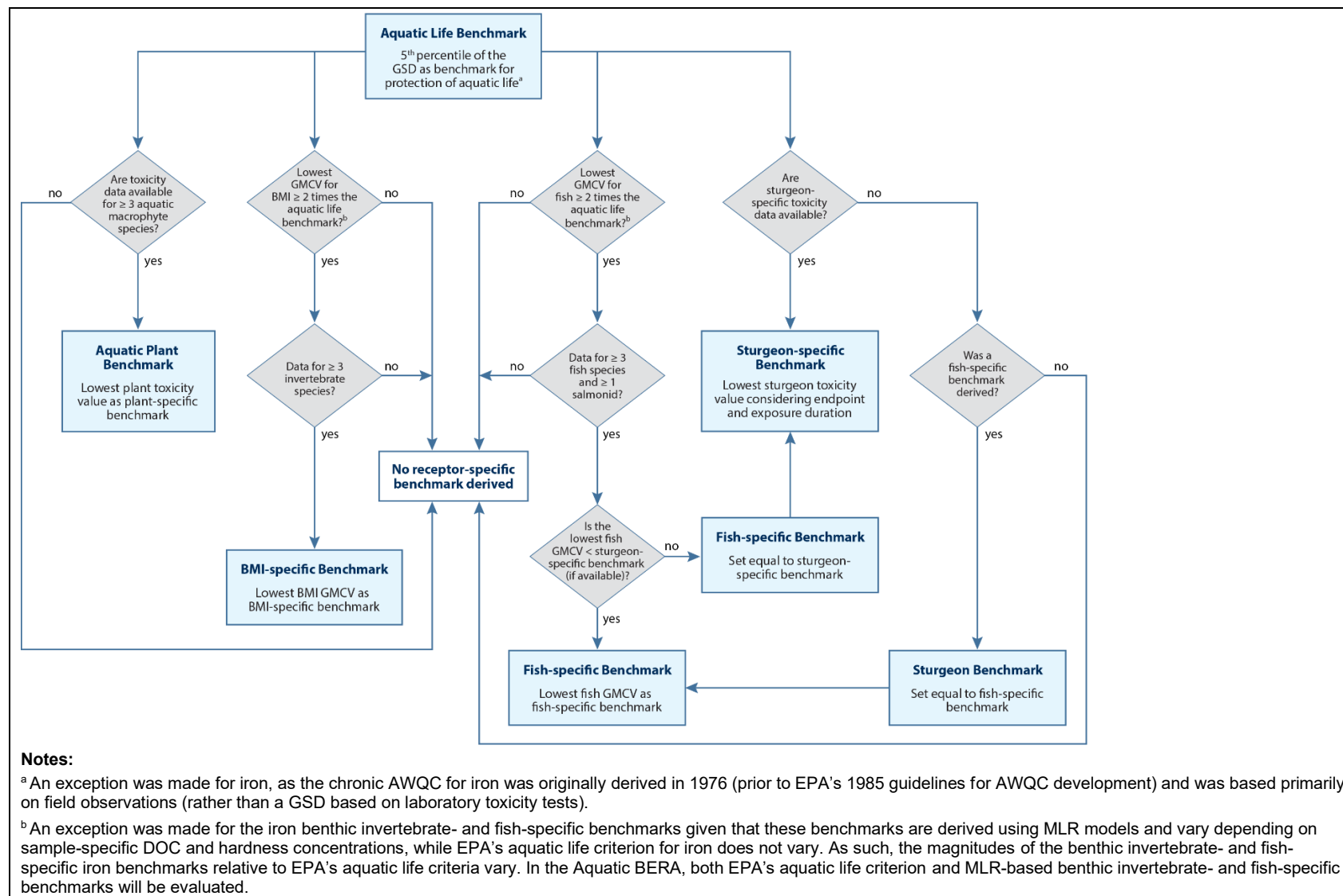


Figure 2-1. Process for Deriving Porewater Benchmarks

2.2 METHODS FOR CALCULATING AQUATIC LIFE, BENTHIC INVERTEBRATE, AND FISH BENCHMARKS

This section presents the methods used to derive benchmarks for individual COPCs or subsets of COPCs, including different methods used to adjust benchmarks to account for site-specific bioavailability. The methods are presented as follows:

- Section 2.2.1: Presents the hardness-based models used by EPA to derive its nationally recommended AWQC for cadmium, lead, and zinc
- Section 2.2.2: Presents the BLM used by EPA to derive its nationally recommended AWQC for copper
- Sections 2.2.3 and 2.2.4: Present the MLR models used to develop benchmarks for aluminum and iron, respectively
- Section 2.2.5: Presents the hardness-based model used to develop the manganese benchmark, which was developed following EPA methods for the development of hardness-based AWQC for other metals
- Section 2.2.6: Describes the use of an ACR to derive a chronic antimony benchmark. No bioavailability models are available for antimony and chronic toxicity data for antimony are limited.
- Section 2.2.7: Presents a literature-based benchmark to demonstrate that measured molybdenum concentrations in porewater are orders of magnitude less than concentrations of concern for aquatic life.
- Section 2.2.8: Summarizes selenium fate and effects in aquatic systems and the basis for why a porewater benchmark was not developed. Selenium is unique among the porewater COPCs, in that its associated risk is most appropriately addressed by evaluating selenium concentrations in fish tissue.

2.2.1 EPA AWQC Hardness-Based Models

EPA's hardness-based criteria for cadmium, lead, and zinc were derived by compiling toxicity and hardness data for multiple biological species and developing a slope of the relationship between toxicity and hardness when data for all species were combined (i.e., a pooled slope). These hardness-adjusted toxicity data were used to develop a GSD, from which the 5th percentile associated with a given effect level (e.g., 20 percent reduction in survival, growth, or reproduction) could be obtained and used to derive the intercept in the hardness-based criterion equation (Equation 1) (Stephan et al. 1985):

$$\text{Criterion} = e^{(\text{slope} \times \ln[\text{hardness}] + \text{intercept})} \times \text{CF} \quad \text{Equation 1}$$

Where:

e = exponential function
 ln = natural logarithm
 CF = total-to-dissolved conversion factor

These metal-specific GSDs were used to determine whether benchmarks for benthic invertebrate and fish should be based on the 5th percentile of the GSD or the lowest GMCV, following the process outlined in Figure 2-1. The specific hardness-based equations for each metal are presented in Section 3.

2.2.2 EPA BLM for Copper

The AWQC for copper was derived using the BLM (USEPA 2007). The BLM is a modeling framework that can be used to evaluate the toxicity of metals to aquatic organisms (Di Toro et al. 2001; Santore et al. 2001; Paquin et al. 2002). It is an adaptation of the gill surface interaction model (Pagenkopf 1983) and the free ion activity model (Morel and Hering 1993; Campbell 1995), which were developed under the premise that concentrations of various bioavailable metal species are influenced by the composition of exposure media. The BLM calculates the binding strengths of metals to ligands in porewater relative to ligands on the biological surfaces of benthic organisms (Di Toro et al. 2001). Thus, the BLM can be used to evaluate various types of biotic ligand interactions for metal ions, such as Cu^{2+} , based on the chemistry of the exposure medium.

Bioavailable metal ions interact with biological membranes, such as gills and the membranes of benthic organisms; when they accumulate to critical concentrations, the ions elicit toxicological effects (MacRae et al. 1999; MacRae 1994). Accumulated metals can cause toxicity by binding to biological ligands, thereby altering the function of membrane proteins that are important for cellular regulation (Wood et al. 1999; Grosell et al. 2007). Much of the work to evaluate the influence of exposure media on metal interactions with biological surfaces was conducted by Playle and peers (e.g., Playle et al. 1992, 1993; Hollis et al. 1996, 1997; Playle 1998). These studies were important in that they systematically investigated the effects of water chemistry on metal accumulation on toxicologically relevant tissues. The BLM treats the biotic ligand as a chemical species in a chemical equilibrium context, wherein cationic metals bind with DOC and anions—such as carbonate, bicarbonate, hydroxide, and chloride (known as “complexation”)—and compete with cations—such as calcium, magnesium, sodium, and protons—to bind at toxicologically active sites on an organism, such as the gills (i.e., the biotic ligand) (Figure 2-2). A number of studies have demonstrated that the BLM framework is

applicable to a broad range of aquatic organisms, including invertebrates (e.g., Villavicencio et al. 2005; De Schampelaere et al. 2002; De Schampelaere and Janssen 2004; DeForest and Van Genderen 2012; Heijerick et al. 2005).

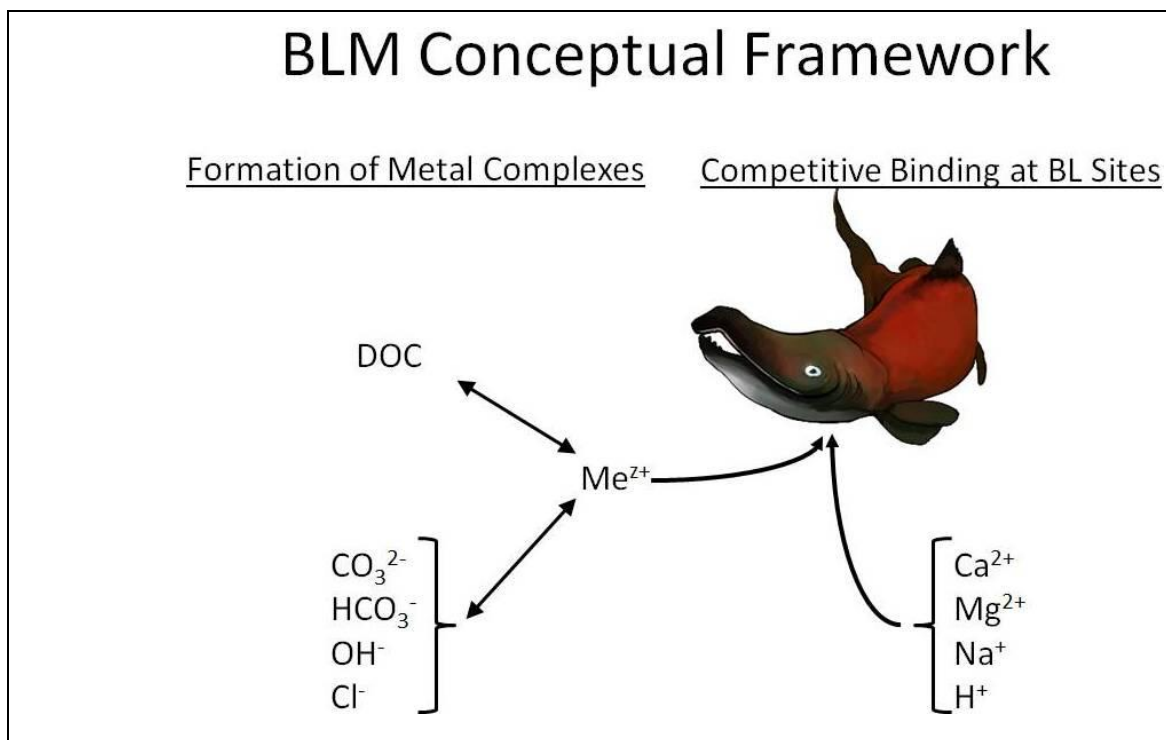


Figure 2-2. Conceptual Illustration of Key Interactions at Biotic Ligand with Multiple Metals

The complete list of input parameters for deriving EPA's BLM-based AWQC for copper is as follows: temperature, pH, DOC, percent humic acid, calcium, magnesium, sodium, potassium, sulfate, and chloride. Because humic acid is not routinely measured, EPA recommends a default value of 10 percent. The EPA copper BLM has been developed and applied using data sets encompassing a wide range of input parameter conditions, including most of those expected in UCR field-collected porewaters (see Attachment F1, Table F1-1). The software for calculating EPA's BLM-based copper AWQC can be obtained by contacting EPA (<https://www.epa.gov/wqs-tech/copper-biotic-ligand-model>, current contact: cruz.luis@epa.gov) or by downloading the software from: <http://www.windwardenv.com/biotic-ligand-model/>.

In addition to an interface with the complete set of BLM parameter inputs, the Windward Environmental LLC (Windward) version of the BLM has a "simplified" mode that requires only temperature, DOC, hardness, and pH as inputs. When using this mode, concentrations of cations, anions, and alkalinity are estimated based on geochemical ion ratios calculated from chemistry reported for the water type used in the analysis. The

simplified mode was used in the generic examples provided herein, in which a DOC of 1 mg/L, hardness of 100 mg/L, and pH of 7 were assumed.

Using the process illustrated in Figure 2-1, the EPA copper BLM was used to determine whether the benchmarks for benthic invertebrate and fish should be based on the aquatic life benchmark or whether they should be receptor specific. The BLM GSD for copper was created using the empirical water-only toxicity data normalized to the chemistry in each sample. The effect concentration in this BLM-derived GSD is the critical accumulation concentration of copper on the biotic ligand expressed on a molar basis. In simple terms, each genus value in the GSD can be considered a back-calculation from the water-based EC20, given the BLM input parameters (e.g., DOC, pH, calcium, etc.) from the toxicity test water.¹³ Once the benchmark basis is determined, the BLM can calculate the sample-specific porewater benchmark based on the BLM inputs for that sample.

2.2.3 Aluminum MLR Model

EPA's AWQC for aluminum (USEPA 2018b) uses an MLR approach to explain the relationships between multiple parameters (DOC and pH, in addition to hardness). Aluminum MLR models were developed for a cladoceran (*Ceriodaphnia dubia*) and the fathead minnow (*Pimephales promelas*), as the chronic toxicities of aluminum to these species have been tested over a range of water chemistry conditions. The resulting aluminum MLR models for predicting chronic toxicity to *C. dubia* and fathead minnow as a function of DOC, pH, and hardness are as follows:

C. dubia:

$$\ln(\text{EC20}) = -35.523 + 0.597 \times \ln(\text{DOC}) + 8.802 \times \text{pH} + 2.089 \times \ln(\text{hardness}) - 0.230 \times (\ln[\text{hardness}] \times \text{pH}) \quad \text{Equation 2}$$

Fathead minnow:

$$\ln(\text{EC20}) = -7.371 + 2.209 \times \ln(\text{DOC}) + 2.041 \times \text{pH} + 1.862 \times \ln(\text{hardness}) - 0.232 \times (\ln[\text{hardness}] \times \text{pH}) - 0.261 \times (\ln[\text{DOC}] \times \text{pH}) \quad \text{Equation 3}$$

Where:

DOC	=	dissolved organic carbon
EC20	=	concentration that causes a 20 percent effect
ln	=	natural logarithm

¹³ The BLM uses an iterative, numerical root-finding approach to determine the amount of dissolved metal required in a particular porewater sample—based on the chemistry of that sample—to reach a critical accumulation (on a molar concentration basis) at the biotic ligand resulting in specified effects (Santore et al. 2001).

Equations 2 and 3 are specific to *C. dubia* and fathead minnow, respectively. However, EPA assumed that DOC, pH, and hardness would interact in a similar manner to influence aluminum toxicity to other species. Thus, the slopes from the MLR models were used to adjust toxicity values to water chemistry characteristics of interest for species other than *C. dubia* and fathead minnow. This approach is analogous to using a pooled slope—which is developed based on a subset of the species tested and then applied to all species—to derive hardness-based metals criteria. Because DOC, pH, and hardness influence aluminum toxicity to *C. dubia* differently than to fathead minnow, the slopes from the *C. dubia* MLR model were applied to invertebrates and the slopes from the fathead minnow MLR model were applied to fish. The result is an MLR-based GSD, normalized to a common DOC, hardness, and pH condition, that can be used to assess whether the aluminum benchmarks for benthic invertebrate and fish should be based on the aquatic life benchmark or should be receptor specific.

The toxicity values used to develop EPA’s draft aluminum criteria and supporting chronic GSD are expressed as total aluminum, because both dissolved and precipitated forms of aluminum are bioavailable and can contribute to aluminum toxicity. However, it is important to emphasize that total aluminum in natural, field-collected samples generally includes non-bioavailable forms of aluminum, such as suspended particulates that may contain aluminum (e.g., clay minerals). This means that measurements of total aluminum generally overestimate bioavailable aluminum concentrations.

2.2.4 Iron MLR Model

EPA’s AWQC for iron is based on a limited amount of toxicity data compiled in 1976 that have not been adjusted for bioavailability (USEPA 1976). However, like aluminum, the chronic toxicities of iron to *C. dubia* and fathead minnow have been tested over a range of DOC, pH, and hardness conditions (Cardwell et al. 2023).¹⁴ Using these data, Brix et al. (2023) developed MLR models for predicting chronic iron toxicities to *C. dubia* and fathead minnow as a function of DOC, pH, and hardness conditions. These models were developed using methods similar to those used to develop the aluminum MLR models. The resulting MLR models for *C. dubia* and fathead minnow are as follows:

C. dubia:

$$\ln(\text{EC}_{20}) = 7.925 + 0.507 \times \ln(\text{DOC}) \quad \text{Equation 4}$$

¹⁴ As for aluminum, both dissolved and precipitated forms of iron are available to aquatic organisms and can contribute to toxicity.

Fathead minnow:

$$\ln(\text{EC20}) = 2.607 + 0.920 \times \ln(\text{DOC}) + 0.830 \times \text{pH} \quad \text{Equation 5}$$

Where:

DOC = dissolved organic carbon
 EC20 = concentration that causes a 20 percent effect
 ln = natural logarithm

As for aluminum, the slopes from the *C. dubia* model were applied to other invertebrate species and the slopes from the fathead minnow model were applied to other fish species. As discussed in Section 3.5, inadequate iron toxicity data were available to develop a GSD, so the porewater benchmarks for benthic invertebrate and fish were developed based on the lowest MLR-adjusted GMCVs for invertebrates and fish, respectively.

2.2.5 Manganese

EPA has not developed nationally recommended AWQC for manganese, but both New Mexico and Colorado have adopted hardness-based manganese criteria that have been approved by EPA (CDPHE 2012; NMED 2011). A BLM for predicting chronic manganese toxicity has also been developed by Peters et al. (2011). The authors noted that manganese toxicity to invertebrates is mitigated by calcium and magnesium, and manganese toxicity to fish is modified by calcium, while DOC and pH do not have a strong mitigating effect on manganese toxicity to either receptor. Due to the importance of hardness ions (i.e., calcium and magnesium) on manganese bioavailability, hardness-based porewater benchmarks for manganese are considered appropriate for use in the Aquatic BERA.

For the Portland Harbor remedial investigation and feasibility study, the Lower Willamette Group developed a hardness-based regression model for manganese (Windward 2014) following EPA guidelines for criteria development (Stephan et al. 1985); this model was approved by EPA Region 10. Because the chronic toxicity database for the model did not meet the required minimum number of different families, an ACR of 5.267 was used to derive the chronic criterion. The ACR was based on the geometric mean of ACRs for six species, which ranged from 2.963 to 18.93. Details of the approach are presented by Windward (2014). The resulting chronic criterion equation is:

$$\text{Chronic Mn benchmark} = e^{(0.7424 \times \ln[\text{hardness}] + 4.124)} \quad \text{Equation 6}$$

Where:

e = exponential function
 ln = natural logarithm

Mn = manganese

The GSD developed using this model was used to determine whether the benchmark would be based on the 5th percentile of the GSD or the lowest GMCV.

No hardness-based model is available for plants; instead, the lowest chronic genus value will be used as the plant benchmark, as long as the value is at least twice the benthic invertebrate benchmark.

2.2.6 Antimony

There is no AWQC for antimony for the protection of aquatic life. Therefore, the antimony benchmark was derived by generally following EPA guidelines for AWQC development (Stephan et al. 1985), as described in the COPC Refinement (TAI 2020). Because chronic toxicity data were limited for antimony, the benchmarks were derived based on the 5th percentile of the acute GSD for antimony divided by a generic ACR of 8.3 (Raimondo et al. 2007). A generic ACR was used because insufficient data were available to derive an antimony-specific ACR.

To develop porewater benchmarks for the protection of benthic invertebrate and fish, the process shown in Figure 2-1 was used. For aquatic plants, the one available GMCV (for *Lemna* [duckweed]) was used to derive the benchmark.

2.2.7 Molybdenum

Molybdenum was identified as a chemical of interest because no benchmark had been identified as part of the COPC Refinement process. However, a freshwater predicted no-effect concentration (PNEC) of 38.2 mg/L (38,200 µg/L), expressed as dissolved molybdenum, was reported by De Schamphelaere et al. (2010), based on the 5th percentile of a species sensitivity distribution. The distribution was composed of EC10s for 10 freshwater species, including plants, invertebrates, fish, and amphibians, with the lowest EC10 being 43,200 µg/L for rainbow trout (*O. mykiss*). The aquatic life benchmark for porewater is the PNEC of 38,000 µg/L (rounded to two significant figures), a value that is orders of magnitude greater than the maximum porewater molybdenum concentration of 52 µg/L measured in site samples.

2.2.8 Selenium

EPA's waterborne selenium AWQC criteria for the protection of aquatic life are 1.5 µg/L for lentic (non-flowing) water bodies and 3.1 µg/L for lotic (flowing) water bodies

(USEPA 2016b). These criteria are back-calculated from EPA's fish tissue-based selenium criterion of 15.1 µg/g dry weight for fish eggs/ovaries using conservative bioaccumulation assumptions. EPA's selenium criteria are based entirely on the sensitivity of fish, as they are the most sensitive aquatic species. The fish tissue-based selenium criteria supersede the waterborne selenium criteria when fish tissue data are available (USEPA 2016b), because waterborne selenium concentrations are poor predictors of the potential selenium exposure of fish and poor indicators of potential selenium toxicity.¹⁵ Selenium did not screen in as a COPC in the COPC Refinement (TAI 2020) using the fish tissue-based selenium criteria from USEPA (2016b); therefore, unacceptable levels of risk to fish in the UCR are not expected from selenium exposure. Because fish are the most sensitive of the aquatic species to selenium, unacceptable levels of risk are also not expected for other, less sensitive classes of organisms, including benthic invertebrates and plants. For this reason, porewater benchmarks were not developed for the assessment of selenium risks to aquatic species in the Aquatic BERA.

¹⁵ Fish are predominantly exposed to selenium via their diet, and the primary forms of selenium in their diet are organic selenium compounds (e.g., selenomethionine) (Janz et al. 2010). The amount of organic selenium in fish diets is, in turn, highly dependent on site-specific factors that influence selenium speciation. More reducing environments, such as a productive wetland, have much greater selenium bioaccumulation potential than do oxic environments, such as high-energy streams. Thus, the bioaccumulation potential of selenium at a given waterborne concentration can vary greatly among different sites.

3 RESULTS OF WHITE STURGEON BENCHMARK DERIVATION

The four metals for which white sturgeon-specific benchmarks could be derived are cadmium, copper, lead, and zinc. The benchmarks, as well as details on how they were derived, are summarized in this section.

3.1 CADMIUM

Four studies, comprising a total of two chronic tests and eight acute tests, have evaluated cadmium toxicity to white sturgeon (Table F1-2). The two chronic tests were a 53-day cadmium exposure initiated with 2-day-old (i.e., 2 days post hatch [dph]) larvae (Wang et al. 2014) and a 66-day cadmium exposure initiated with embryos (Vardy et al. 2011). The chronic EC20s based on the most sensitive endpoint from each study—dry weight for Wang et al. (2014) and survival for Vardy et al. (2011)—were 7-fold and 16-fold greater than their respective hardness-based chronic cadmium criteria (Figure 3-1). The eight acute tests were based on 96-hour exposures initiated with life stages ranging from 2 to 89 dph. The ratios of acute values to chronic cadmium criteria, including two acute LC50s reported by Vardy et al. (2014) and six acute EC50s reported by Calfee et al. (2014), were within the range of or, in some cases, much greater than those reported for the two chronic tests (Figure 3-1). Therefore, chronic cadmium criteria (i.e., the basis for the aquatic life benchmark) are protective of white sturgeon.

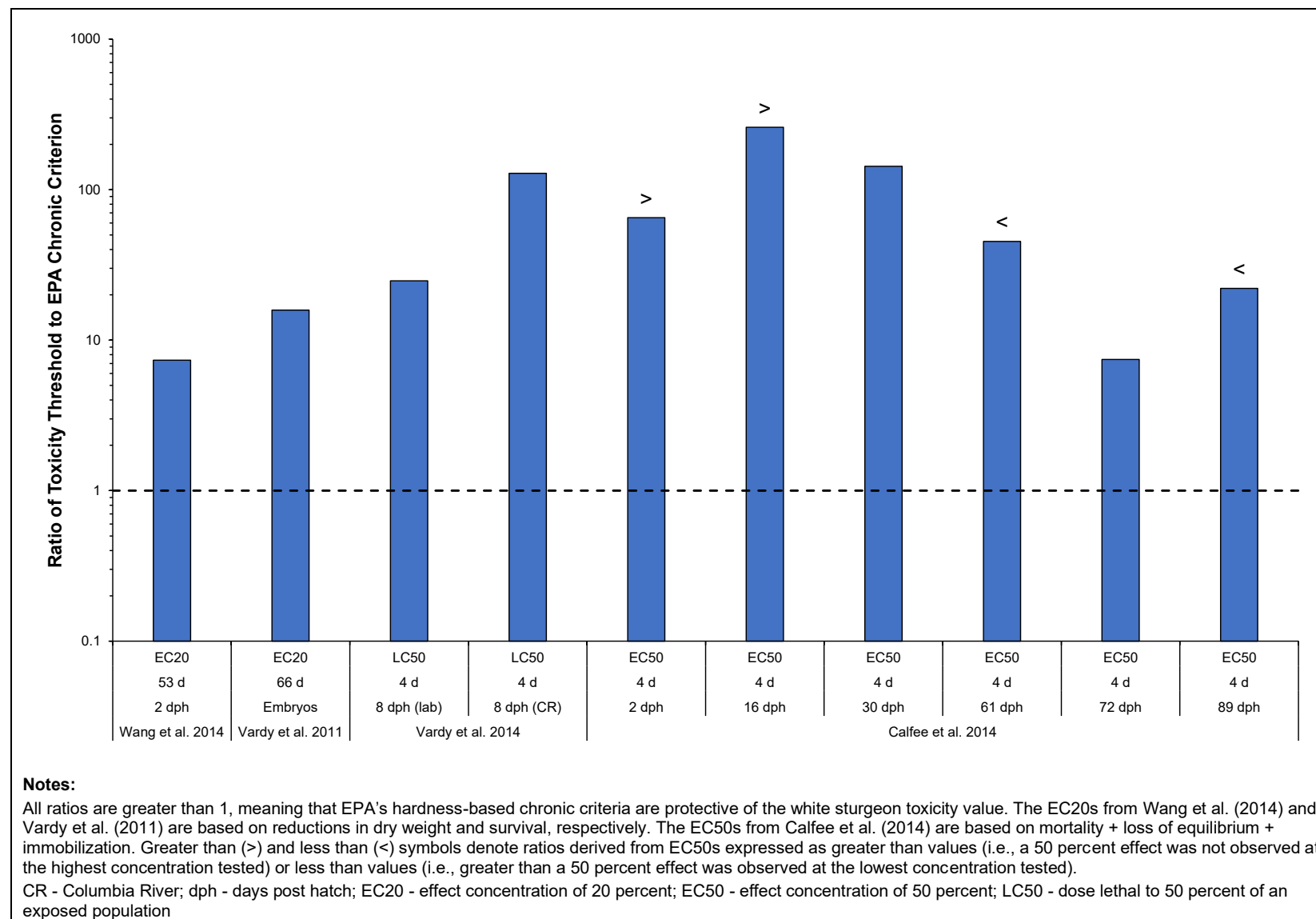


Figure 3-1. Ratios of White Sturgeon Toxicity Values for Cadmium to EPA's Hardness-Based Chronic Cadmium Criterion

To support sturgeon-specific evaluations in the Aquatic BERA, a sturgeon-specific porewater benchmark for cadmium was derived based on the EC20 of 5.4 µg/L for larval dry weight (Wang et al. 2014). This was the lower EC20 from the two chronic studies (Figure 3-1). To derive the sturgeon-specific benchmark equation, the intercept was first calculated based on the EC20 of 5.4 µg/L, the test hardness of 103 mg/L, and EPA's hardness-based model with a slope of 0.7977:

$$\text{Intercept} = \ln(5.4 \text{ } \mu\text{g/L}) - 0.7977 \times \ln(103 \text{ mg/L}) = -2.01 \quad \text{Equation 7}$$

This intercept was then used to develop a sturgeon-specific hardness-based EC20 model:

$$\text{Sturgeon-specific Cadmium EC20} = \exp(0.7977 \times \ln(\text{hardness}) - 2.01) \quad \text{Equation 8}$$

Because all porewater cadmium concentrations were less than the respective hardness-based white sturgeon benchmarks derived using Equation 8, a cadmium concentration-response model was not developed for cadmium and white sturgeon.

It should be noted that white sturgeon (genus *Acipenser*) is not represented in EPA's chronic GSD for cadmium, as in Section 4 (Figure 4-3). This is because 1) Wang et al. (2014) did not meet the minimum ELS study requirements for inclusion in the chronic GSD, and 2) there are questions as to whether Vardy et al. (2011) had a true control group, since cadmium concentrations in the control water (mean of 0.05 µg/L) were similar to those in the lowest exposure group water (mean of 0.06 µg/L). Data from these two studies were tabulated as "other data" in EPA's AWQC document for cadmium (USEPA 2016a), meaning that the studies were not technically deficient but simply did not meet the strict requirements for inclusion in the GSD. Following guidelines from Stephan et al. (1985) for AWQC, "other data" may be used to derive AWQC if toxicity thresholds are more sensitive than AWQC derived based on the 5th percentile of the GSD (which was not the case for cadmium and white sturgeon).

3.2 COPPER

Eight studies, comprising a total of 2 chronic tests, 1 sub-chronic test, and 29 acute tests, have evaluated copper toxicity to white sturgeon (Table F1-2; Figure 3-2). The two chronic tests (Vardy et al. 2011; Wang et al. 2014) and the sub-chronic test (Puglis et al. 2019) were evaluated to develop a copper benchmark for white sturgeon. Ultimately, the "time spent moving" endpoint reported by Puglis et al. (2019) was used to develop the copper benchmark (as described further below). The following summarizes the white sturgeon studies reviewed; then the copper benchmark and associated concentration-response model are described.

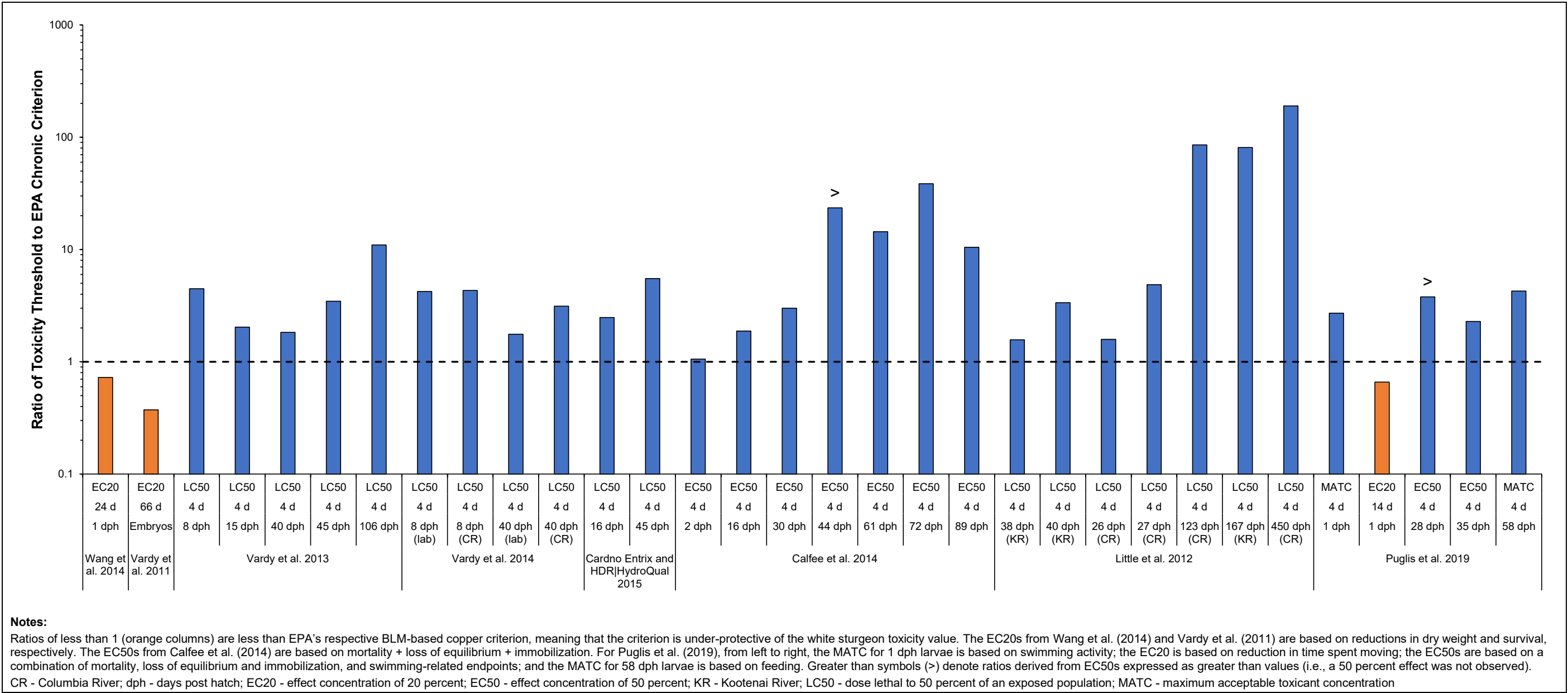


Figure 3-2. Ratios of White Sturgeon Toxicity Values for Copper to EPA's BLM-Based Chronic Copper Criterion

The two chronic tests were a 66-day copper exposure initiated with white sturgeon embryos (Vardy et al. 2011) and a 24-day copper exposure initiated with 1 dph white sturgeon larvae (Wang et al. 2014). The most sensitive endpoint in the Vardy et al. (2011) study was an EC20 of 3.4 µg/L for mortality. Wang et al. (2014) conducted three tests in which they were able to derive EC20s for growth endpoints: a 53-day test initiated with 2 dph larvae, a 28-day test initiated with 27 dph larvae, and the 24-day test initiated with 1 dph larvae. The EC20 for larval dry weight based on the last test was selected for inclusion in this evaluation because it was the most sensitive EC20 reported (1.4 µg/L). The ratios of copper EC20s from both Wang et al. (2014) and Vardy et al. (2011) to their respective chronic BLM-based copper AWQC were less than 1, meaning that the chronic AWQC are under-protective of these endpoints (Figure 3-2).

The 14-day sub-chronic toxicity test was initiated with 1 dph larvae and included endpoints related to both mortality and behavior (Puglis et al. 2019). The mortality-related endpoints included direct mortality and loss of equilibrium and immobilization, both of which are considered precursors to mortality. Behavioral endpoints were swimming related: total distance traveled, average velocity, and time spent moving. The behavioral endpoints were evaluated during 10-minute activity trials from days 3 to 14 of the test, using digital cameras to record forward movements by individual larvae (n = 4 to 12 daily, depending on the endpoint). For example, on day 14, control larvae on average spent approximately 60 percent of their time moving from one location in the tank to another during the 10-minute observation window. Effects on swimming behavior, such as a reduction in time spent moving, can impact larval survival and growth by affecting their ability to find food and hide from predators, among other effects. The most sensitive behavioral endpoint was time spent moving, which had a copper EC20 of 1.3 µg/L, whereas the EC20 for the combined mortality endpoint (i.e., mortality + loss of equilibrium + immobilization) was less sensitive at 2.1 µg/L. The time spent moving EC20 was more sensitive than the corresponding chronic BLM-based copper criterion of 2.0 µg/L (Figure 3-2), while the combined mortality endpoint EC20 was slightly greater than the corresponding BLM-based copper criterion (not shown in Figure 3-2 because only the most sensitive endpoint from each test was plotted). The time spent moving EC20 of 1.3 µg/L meant that on average, larval sturgeon were predicted to spend 20 percent less time moving than would the control larval sturgeon.

The 29 acute tests were based on 96-hour exposures and were initiated with life stages ranging from 1 to 450 dph. In general, white sturgeon sensitivity to copper decreased with increasing life stage (Figure 3-2). All acute EC50-to-chronic copper AWQC ratios were

greater than 1, although the ratio for the test initiated with 2 dph larvae was near 1 (Figure 3-2) (Calfee et al. 2014). This pattern indicates that copper-related effects in longer-term tests are driven by the sensitivity of very ELS sturgeon.

In summary, three tests had white sturgeon EC20s less than their respective chronic AQWC. Critical copper accumulations on the biotic ligand were 0.00250 nmol/g ww for the mortality EC20 from Vardy et al. (2011), 0.00461 nmol/g ww for the time spent moving EC20 from Puglis et al. (2019), and 0.00530 nmol/g ww for the larval dry weight EC20 from Wang et al. (2014). The bioavailability-normalized EC20 from Puglis et al. (2019), based on the time spent moving endpoint, was intermediate to the other two EC20s and within a factor of two of each. When there are multiple tests for an individual species, EPA calculates the geometric mean to define the sensitivity of a species for AWQC development (Stephan et al. 1985). The Vardy et al. (2011) and Wang et al. (2014) studies were similar in survival and growth, while the most sensitive endpoint in Puglis et al. (2019) related to behavior. The geometric mean critical accumulation from Vardy et al. (2011) and Wang et al. (2014) is 0.00364 nmol/g ww, which is slightly less than the critical accumulation of 0.00461 nmol/g ww from Puglis et al. (2019). However, the study by Vardy et al. (2011) had an exposure duration of 66 days while that of Puglis et al. (2019) was 14 days. The latter is a more relevant exposure duration for ELS sturgeon in porewater, and therefore more appropriate for defining a benchmark applicable to porewater exposures. As such, the critical accumulation derived from Puglis et al. (2019)—based on the time spent moving EC20—was selected as the benchmark for evaluating white sturgeon exposures to copper in porewater.

Because copper concentrations in some UCR porewater samples exceeded the BLM-adjusted porewater benchmark, a concentration-response model for the time spent moving endpoint (i.e., the basis for the benchmark) was developed. The following nonlinear regression models were evaluated in EPA's Toxicity Relationship Analysis Program (TRAP) (USEPA 2015): the logistic model, the threshold sigmoid model, and the piecewise linear model. TRAP was also used by Puglis et al. (2019) to evaluate concentration-response data. Among the three nonlinear models, the logistic model resulted in the EC20 of 1.3 µg/L reported by Puglis et al. (2019). All three models had high R² values (0.985, 0.979, and 0.974 for the logistic, threshold sigmoid, and piecewise linear models, respectively). For consistency with Puglis et al. (2019), and because it had the highest R², the logistic model was selected to describe the concentration-response relationship for the time spent moving endpoint (Figure 3-3A). To apply the concentration-response model to porewater with varying bioavailability conditions,

EPA's copper BLM was applied in speciation mode to calculate copper concentrations on the biotic ligand from those in the water tested by Puglis et al. (2019). The concentration-response model for the time spent moving endpoint based on copper concentrations on the biotic ligand is provided in Figure 3-3B.

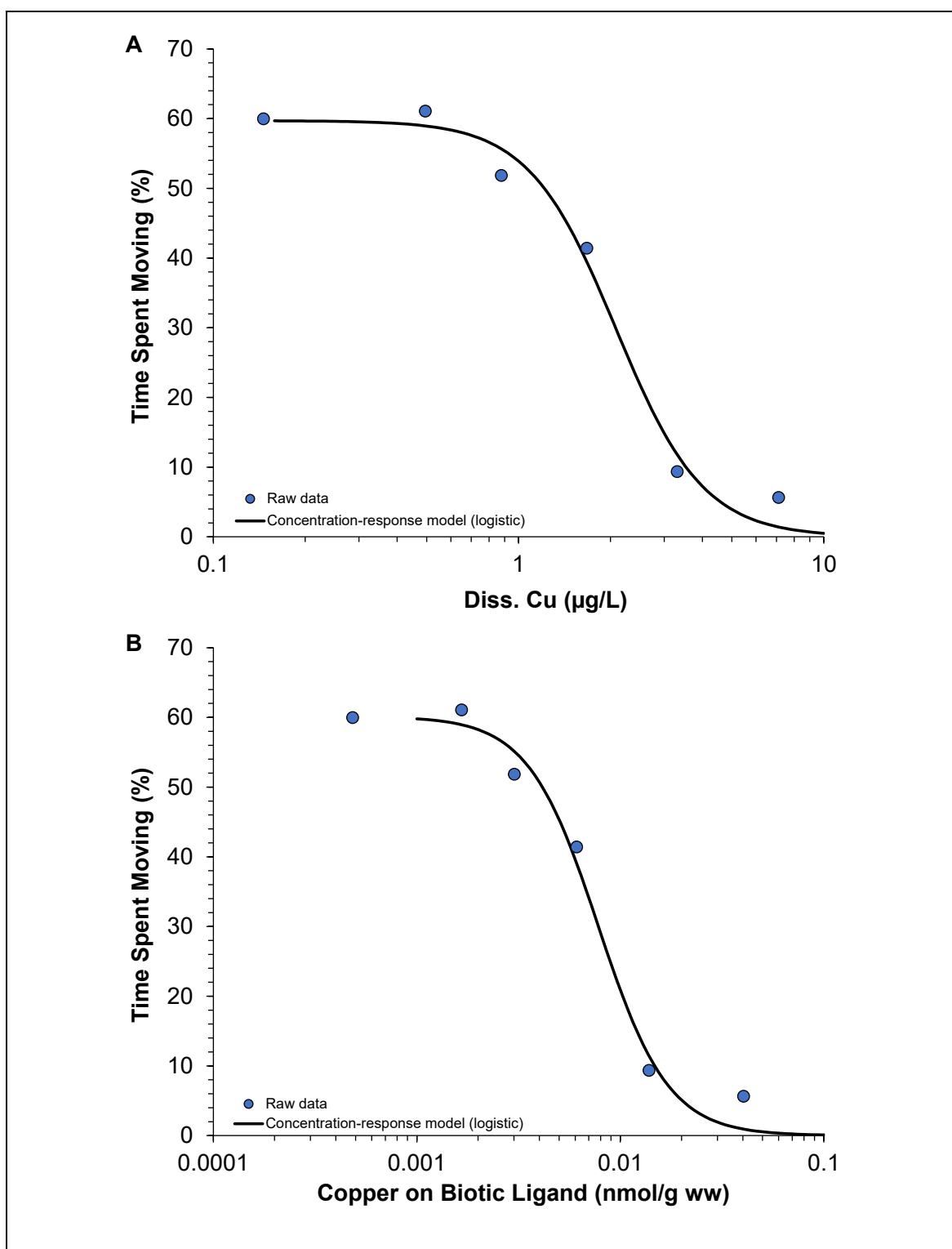


Figure 3-3. Copper Concentration-Response Model for White Sturgeon Time Spent Moving Endpoint Based on Puglis et al. (2019): A) Dissolved Copper Concentrations and B) Copper Concentrations on the Biotic Ligand

To calculate the predicted percent response for a porewater sample, the copper concentration on the biotic ligand is first calculated for the porewater sample (applying EPA's copper BLM in speciation mode). Then, the predicted percent response is calculated from the concentration-response model in Figure 3-3B using Equation 9.

$$\text{Predicted Reduction in Time Spent Moving (\%)} = 1 - \frac{1}{(1 + \exp(4 \times S \times (\log(\text{BL-Cu}) - \log(\text{X50})))}$$

Equation 9

Where:

S = relative slope at X50 (1.472; output from TRAP)
 BL-Cu = copper concentration on the biotic ligand (output from BLM)
 X50 = BL-Cu concentration with 50% effect (-2.105; output from TRAP)

Equation 9 is constructed to predict the time spent moving response relative to the control.

Using shallow porewater UCR data as an example, the relationship between the predicted control-normalized reduction in time spent moving and toxic units (TUs) based on the EC20 for the same endpoint is provided in Figure 3-4. As shown, the predicted effect is 20 percent when the TU equals 1 (because the benchmark used to calculate the TU is based on an EC20). TUs of 2, 3, and 4 are associated with predicted reductions in time spent moving of approximately 63, 88, and 94 percent, respectively.

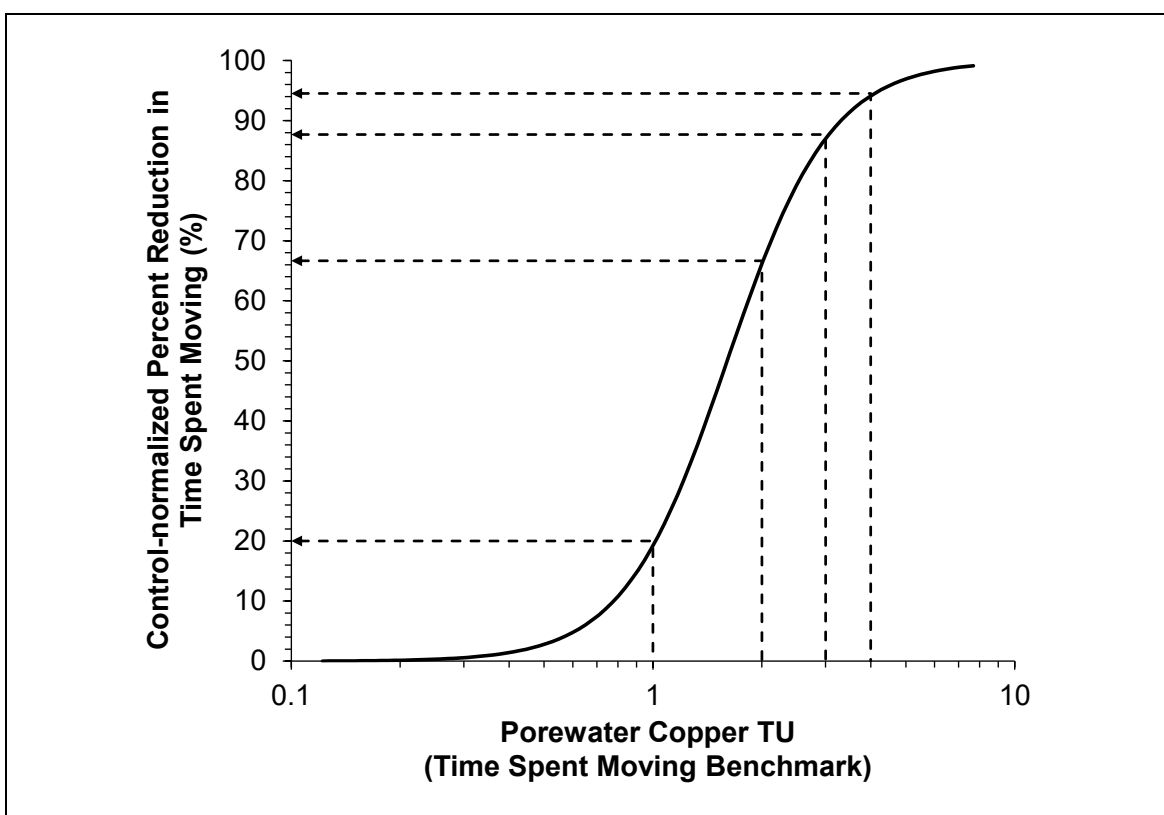


Figure 3-4. Relationship Between Control-Normalized Percent Reduction in Time Spent Moving and Copper TUs Based on Time Spent Moving Benchmark

3.3 LEAD

Five studies, comprising a total of one chronic test and four acute tests, have evaluated lead toxicity to white sturgeon (Table F1-2). The chronic test was a 53-day lead exposure initiated with 2 dph larvae; there were no effects on dry weight or length at a dissolved lead concentration of 27 $\mu\text{g/L}$ (Wang et al. 2014). Thus, the EC20 is $> 27 \mu\text{g/L}$, which is 10-fold greater than the hardness-based chronic lead criterion (Figure 3-5). The four acute tests were based on 96-hour exposures initiated with life stages of 8 or 40 dph; these tests were conducted in either laboratory water or Columbia River (CR) water. The ratios of acute values to chronic lead criteria were greater than 100 for all tests (Figure 3-5). Therefore, chronic lead criteria (i.e., the basis for the aquatic life benchmark) were determined to be protective of white sturgeon.

To support sturgeon-specific evaluations in the Aquatic BERA, a sturgeon-specific porewater benchmark for lead was derived based on the EC20 of $> 27 \mu\text{g/L}$ from Wang et al. (2014), the test hardness of 103 mg/L , and EPA's hardness-based model with a slope of 1.273:

$$\text{Intercept} = \ln(27 \mu\text{g/L}) - 1.273 \times \ln(103 \text{ mg/L}) = -2.60 \quad \text{Equation 10}$$

This intercept was then used to develop a sturgeon-specific hardness-based EC20 model:

$$\text{Sturgeon-specific Lead EC20} = \exp(1.273 \times \ln(\text{hardness}) - 2.604)^{16} \quad \text{Equation 11}$$

A concentration-response model for lead and white sturgeon could be not developed because a concentration-response relationship was not observed over the range of lead concentrations tested by Wang et al. (2014). Therefore, the benchmark is expressed as a “greater than” value.

¹⁶ The sturgeon-specific lead EC20 derived from this equation is a “greater than” value, as a 20 percent effect was not observed in the toxicity test used to derive this benchmark (see text).

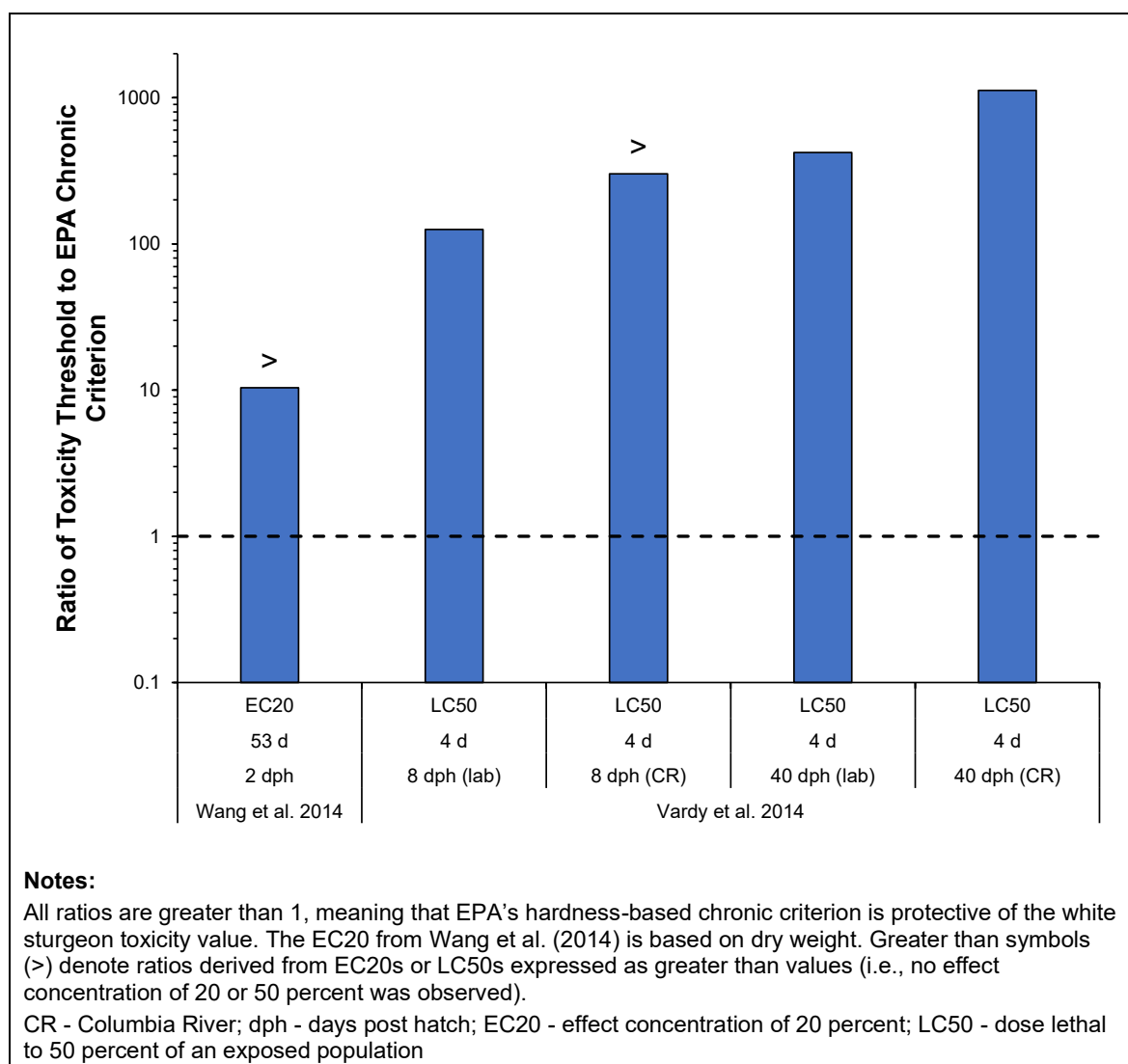


Figure 3-5. Ratios of White Sturgeon Toxicity Values for Lead to EPA's Hardness-Based Chronic Lead Criterion

3.4 ZINC

Four studies, comprising a total of two chronic tests and nine acute tests, have evaluated zinc toxicity to white sturgeon (Table F1-2; Figure 3-6). The nine acute tests were based on 96-hour exposures initiated with life stages ranging from 2 to 89 dph (Figure 3-6). All acute EC50-to-chronic zinc AWQC ratios were greater than 1, although the ratio for the test initiated with 2-dph larvae, as reported by Calfee et al. (2014), was near 1 (Figure 3-6). This situation was similar to that observed for copper (Figure 3-2), similarly indicating that zinc-related effects in longer-term tests are driven by the sensitivity of very ELS sturgeon. The two chronic tests—a 53-day zinc exposure initiated with 2-dph larvae

(Wang et al. 2014) and a 66-day zinc exposure initiated with embryos (Vardy et al. 2011)—were considered for benchmark development. The toxicity data from Wang et al. (2014) were used to develop the porewater zinc benchmarks for white sturgeon, as discussed below.

Wang et al. (2014) conducted two tests in which they were able to derive EC20s for growth endpoints, but the EC20 for larval dry weight based on the 53-day test was selected for inclusion in the present evaluation, because it was the most sensitive EC20 reported in the study. However, whereas Wang et al. (2014) reported an EC20 of 99 µg/L, an alternative EC20 of 160 µg/L was developed as part of this benchmark development evaluation.

The reported EC20 of 99 µg/L was originally identified as the white sturgeon benchmark for zinc. However, during efforts to develop the complete concentration-response model for the larval dry weight endpoint using TRAP (the program used to derive the EC20 of 99 µg/L), no models or data transformations were identified that could duplicate this EC20.¹⁷ Upon being contacted, Dr. Ning Wang helpfully provided (USGS 2023, pers. comm.) the TRAP inputs and outputs that had been the basis for the concentration-response model resulting in the EC20 of 99 µg/L. Wang et al. (2014) had selected the TRAP piecewise regression model fit to individual replicate response data. When fitting nonlinear regression models, TRAP uses iterative mathematical search methods to find parameter values that minimize the squared deviations of the data from the estimated model curve. TRAP also allows the user to provide initial guesses for model parameters: the log concentration associated with a 50 percent effect (logX50), the steepness parameter (S), and the control response (Y₀). Based on the TRAP inputs provided by Dr. Wang, the following initial guesses were applied: logX50 of 3.317, S of 0.19283, and Y₀ of 0.2543. The basis for the initial guesses for logX50 and S were unknown, but the Y₀ of 0.2543 was the mean dry weight of control larvae.

An alternative to the user providing initial guesses is to let TRAP provide the initial guesses. This approach was also used so the results could be compared to those from Wang et al. (2014). The results of the piecewise regression models developed by Wang et al. (2014) and those developed for the present analysis are compared in Figure 3-7A. The EC20 derived from the piecewise regression analysis was 211 µg/L, a value primarily driven by an increase in the inflection point in the piecewise model (Figure 3-7A). Figure 3-7A clearly shows that the effect of zinc on larval growth is not observed until the

¹⁷ The concentration-response data for this initial model evaluation were not reported in Wang et al. (2014), but mean response data were available from Ingersoll and Mebane (2014).

highest zinc treatment of 390 µg/L. The final Y_0 for larval dry weight differed between the two TRAP analyses, with the final Y_0 values being 0.2572 and 0.2323 in the Wang et al. (2014) analysis and present analysis, respectively (Figure 3-7A). The TRAP results also provided two error messages for the model fit when the initial model input guesses provided by Wang et al. (2014) were used,¹⁸ but no error messages occurred for the present analysis (when TRAP provided the initial guesses for the model inputs). Given all these results, the piecewise model from the present analysis was considered more reliable than that selected by Wang et al. (2014).

In addition, the other two nonlinear regression models in TRAP—the logistic and threshold sigmoidal—were evaluated as part of the present analysis. When TRAP provided the initial guesses on model parameters, these models resulted in similar fits to the data and similar EC20s of 160 and 145 µg/L for the logistic and threshold sigmoidal models, respectively (Figure 3-7B). The logistic model was selected to define the porewater benchmark for zinc and white sturgeon, as it is a common concentration-response model and used to describe concentration-response relationships for metal toxicity to aquatic organisms (Ritz 2010).

As noted, the zinc EC20 of 99 µg/L reported by Wang et al. (2014) was initially identified as the benchmark for white sturgeon, but this value was revised to 160 µg/L based on the present analysis. The revised zinc EC20 of 160 µg/L is 12 percent greater than the EC20 from Vardy et al. (2011) when adjusted for hardness.¹⁹ For example, at the test hardness of 103 mg/L in the Wang et al. (2014) study, the hardness-adjusted EC20 from Vardy et al. (2011) is 141 µg/L (based on EPA's hardness slope of 0.8473). Even though the EC20 from Vardy et al. (2011) was slightly less than that from Wang et al. (2014), the EC20 from Wang et al. (2014) was used to define the sturgeon benchmark for the following reasons:

- The 95 percent confidence intervals on the EC20s from Wang et al. (2014) and Vardy et al. (2011) overlap, even when the tests were conducted at different hardness concentrations (103 and 70 mg/L, respectively).
- There was greater uncertainty in the mortality measurements by Vardy et al. (2011) than by Wang et al. (2014) due to density-dependent mortalities (for which Vardy et al. (2011) attempted to adjust).

¹⁸ The two TRAP error messages based on Wang et al. (2014) analysis are: "Maximum iterations reached without convergence" and "X50 at maximum or minimum limit."

¹⁹ The EC20 reported by Vardy et al. (2011) was based on larval mortality, the most sensitive endpoint evaluated.

The sturgeon-specific porewater benchmark was therefore derived based on the EC20 of 160 µg/L from the logistic model, the test hardness of 103 mg/L, and EPA's hardness-based model with a slope of 0.8473, as follows:

$$\text{Intercept} = \ln(160 \text{ } \mu\text{g/L}) - 0.8473 \times \ln(103 \text{ mg/L}) = 1.15 \quad \text{Equation 12}$$

This intercept was then used to develop a sturgeon-specific hardness-based EC20 model:

$$\text{Sturgeon-specific Zinc EC20} = \exp(0.8473 \times \ln(\text{hardness}) + 1.15) \quad \text{Equation 13}$$

To apply the concentration-response model to porewater with varying bioavailability conditions, intercepts for each of the zinc treatments in the Wang et al. (2014) test were calculated for the hardness model using Equation 12.²⁰ The concentration-response model based on intercepts is provided in Figure 3-8. To calculate the predicted percent response for a porewater sample, first the intercept based on the zinc and hardness concentrations measured in the porewater sample is calculated. Then, the predicted percent response is calculated from the concentration-response model in Figure 3-8 using on the following equation:

$$\text{Reduction in Larval Dry Weight (\%)} = 1 - \frac{1}{(1 + \exp(4 \times S \times (\log(\text{intercept}) - \log(X50))))} \quad \text{Equation 14}$$

Where:

S	=	relative slope at X50 (0.4301; output from TRAP)
Intercept	=	back-calculated intercept based on hardness model slope and porewater zinc and hardness concentrations
log(X50)	=	intercept associated with 50% effect (1.956; output from TRAP)

²⁰ The zinc EC20 of 160 µg/L in Equation 12 was replaced with each of the zinc concentrations in the sturgeon test, which were 1.5, 22.7, 45.9, 92.5, 180, and 390 µg/L.

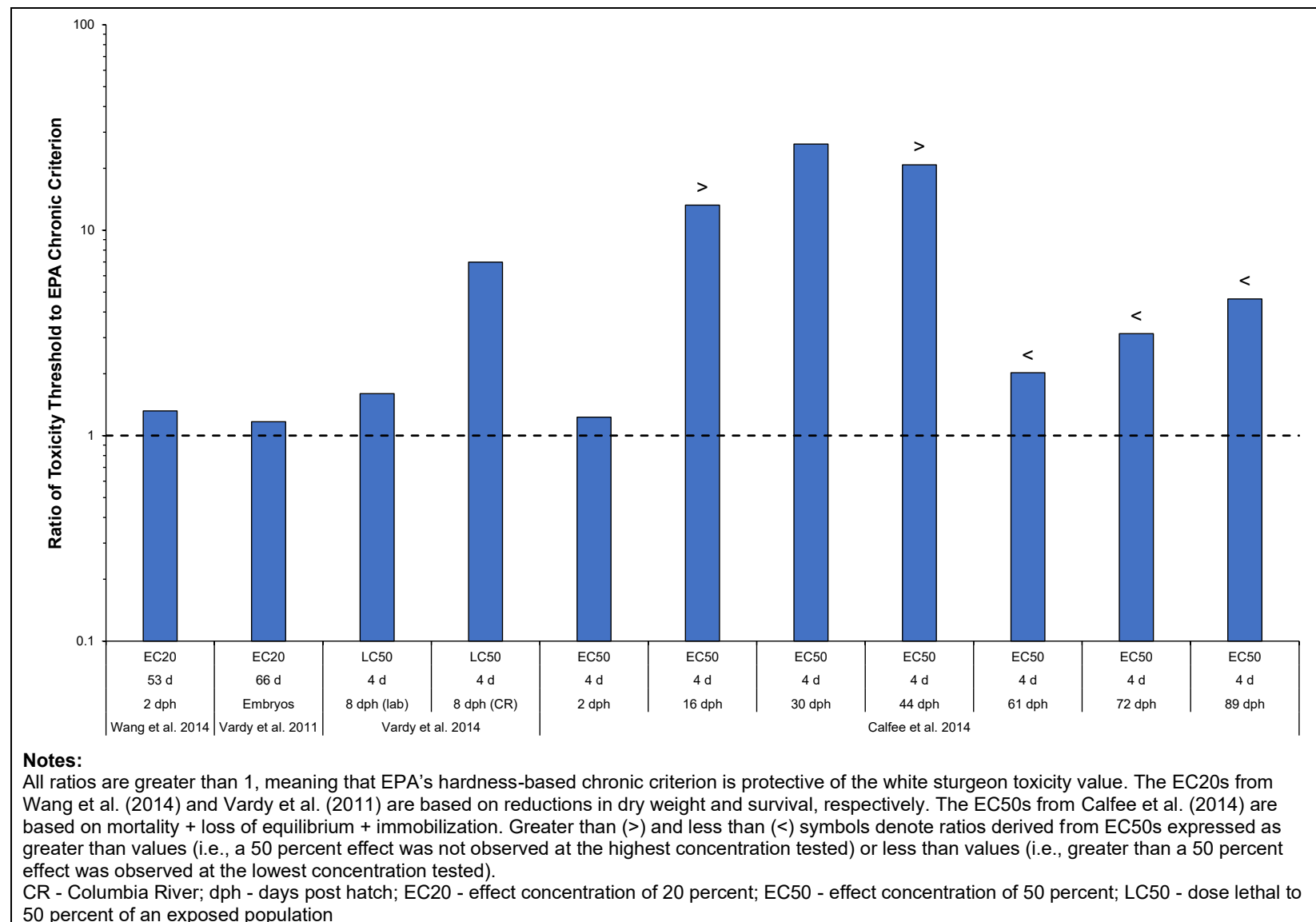


Figure 3-6. Ratios of White Sturgeon Toxicity Values for Zinc to EPA's Hardness-Based Chronic Zinc Criterion

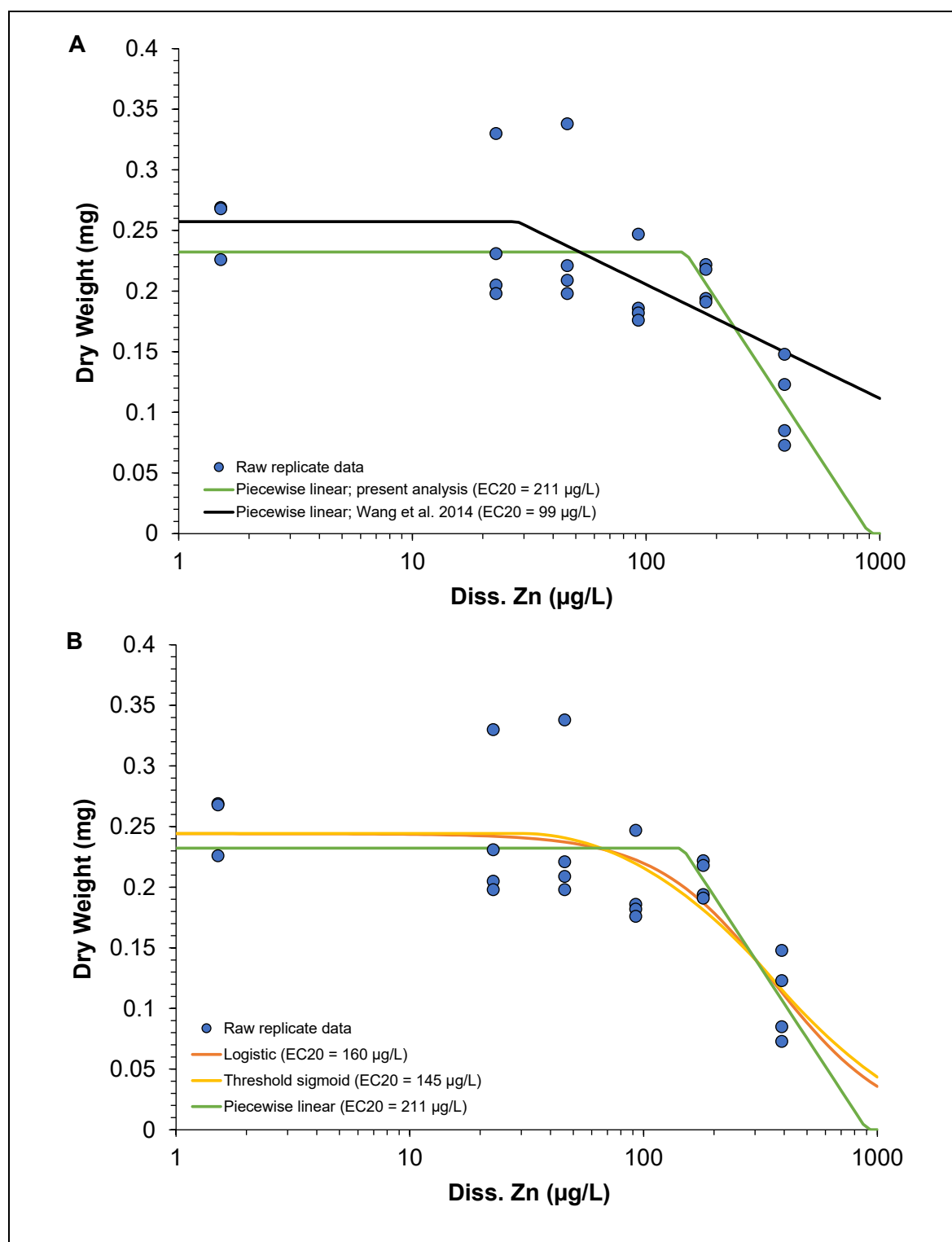


Figure 3-7. Zinc Concentration-Response Model for White Sturgeon Larval Dry Weight:
 A) Comparison of Piecewise Linear Regression Models from Wang et al. (2014) and
 the Present Analysis and B) Comparison of Logistic, Threshold Sigmoid, and
 Piecewise Linear Regression Models

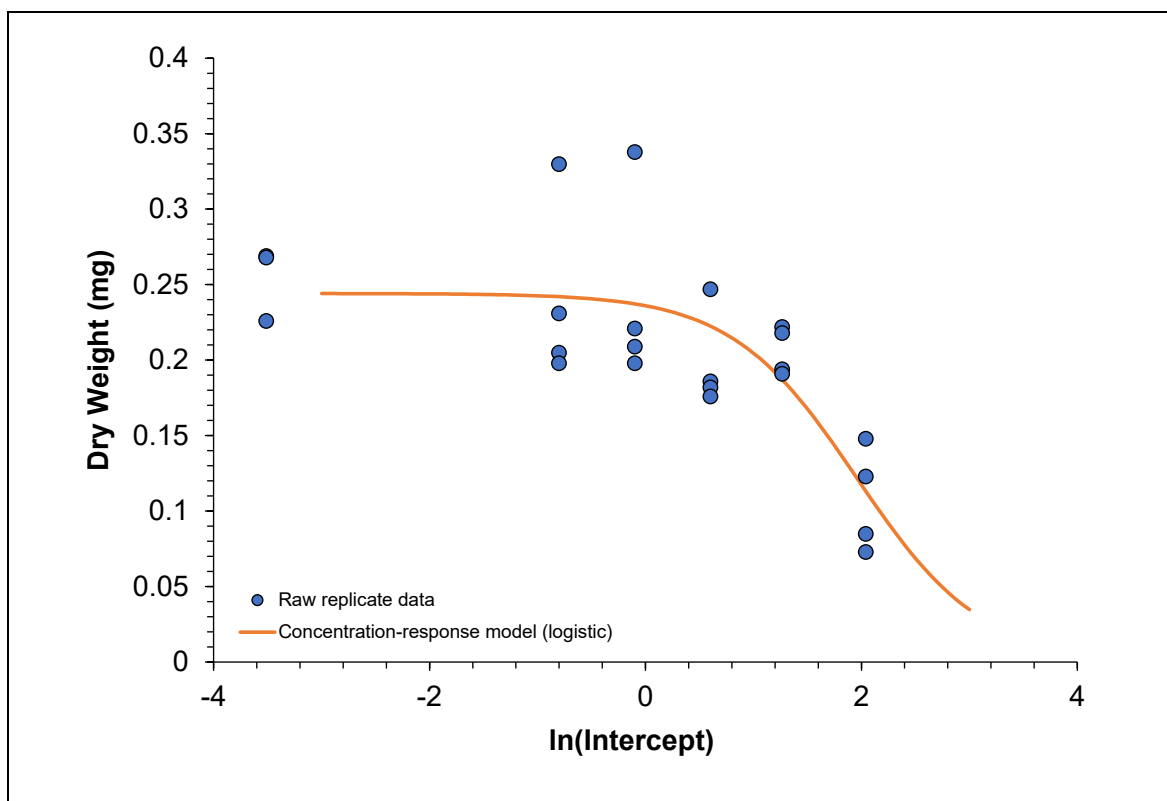


Figure 3-8. Zinc Concentration-Response Model for White Sturgeon Larval Dry Weight Versus Hardness Model Intercept

4 RESULTS OF BENCHMARK DERIVATION

This section describes the results of the benchmark derivation process for aluminum, antimony, cadmium, copper, iron, lead, manganese, and zinc. The general methods for deriving the benchmarks are provided in Section 2 and summarized schematically in Figure 2-1. In addition, details on the development of white sturgeon-specific benchmarks for cadmium, copper, lead, and zinc, are presented in Section 3.

4.1 ALUMINUM

Selected porewater benchmarks for aluminum (expressed as total aluminum) are presented in Table 4-1, followed by a more detailed description of the benchmark selection process for each receptor.

Table 4-1. MLR Model-Derived Aluminum Porewater Benchmarks for Aquatic Life, Benthic Invertebrates, Fish, and Plants

Receptor	Benchmark Basis	Benchmark Value Equation or Lowest Chronic Value	Example Benchmark
Aquatic life	EPA chronic criterion for aquatic life (i.e., 5th percentile of the GSD)	use EPA calculator (https://www.epa.gov/wqc/aquatic-life-criteria-aluminum)	380 µg/L ^a
Benthic invertebrate	ALB	ALB	380 µg/L ^a
Fish	ALB	ALB	380 µg/L ^a
White sturgeon	ALB	ALB	380 µg/L ^a
Aquatic plants	ALB	ALB	ALB

Notes:

^a Assuming DOC = 1 mg/L, hardness = 100 mg/L, and pH = 7
 ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)
 DOC - dissolved organic carbon
 GSD - genus sensitivity distribution
 MLR - multiple linear regression

EPA's final chronic aluminum criterion for the protection of aquatic life (USEPA 2018b) is based on a GSD with toxicity data for 13 genera, including 4 fish, 8 invertebrates, and

1 amphibian (Table F1-3). However, there are separate MLR models for invertebrates and fish, and the relative sensitivities of invertebrates and fish vary as a function of DOC, hardness, and pH conditions, as shown in Figure 4-1 for two sets of conditions. As a result, the calculated AWQC can be driven by the sensitivity of either invertebrates or fish. Under some DOC, hardness, and pH conditions, the most sensitive invertebrate and fish genera both have GMCVs within a factor of two of the 5th percentile of the GSD (Figure 4-1A); however, this is not true for all DOC, hardness, and pH conditions (Figure 4-1B). Because the GMCVs for the most sensitive invertebrate and fish genera are within a factor of two of the 5th percentile of the GSD under some conditions, benchmarks specific to benthic invertebrate or fish were not derived.

- **Aquatic life benchmark.** The total aluminum benchmark for aquatic life is EPA's chronic aluminum criterion, which is the 5th percentile of the GSD (380 µg/L at a DOC of 1 mg/L, hardness of 100 mg/L, and pH of 7 [Figure 4-1A] and 340 µg/L at a DOC of 10 mg/L, hardness of 25 mg/L, and pH of 6 [Figure 4-1B]).
- **Benthic invertebrate benchmark.** As noted, depending on the DOC, hardness, and pH conditions in the water, the most sensitive invertebrate genus may be the most sensitive genus in the GSD. Accordingly, the benthic invertebrate benchmark is set equal to the aquatic life benchmark.
- **Fish benchmark.** As for invertebrates, the most sensitive fish genus may be the most sensitive genus in the GSD, depending on the DOC, hardness, and pH conditions in the water. Accordingly, the fish benchmark is set equal to the aquatic life benchmark.
- **Aquatic plant benchmark.** Aluminum toxicity data for aquatic plants are limited. The only test in which DOC, hardness concentrations, and pH were all reported was for duckweed, which had a total aluminum EC20 of 4,093 µg/L in water with a DOC of 0.3 mg/L, hardness of 52 mg/L, and pH of 6 (Table F1-4) (Cardwell et al. 2018). This test was conducted under high aluminum bioavailability conditions (i.e., low DOC, hardness, and pH), and the EC20 is still approximately an order of magnitude greater than the aquatic life benchmark (Figure 4-1). However, because data were available for fewer than three aquatic plant species, no plant-specific benchmark was derived.

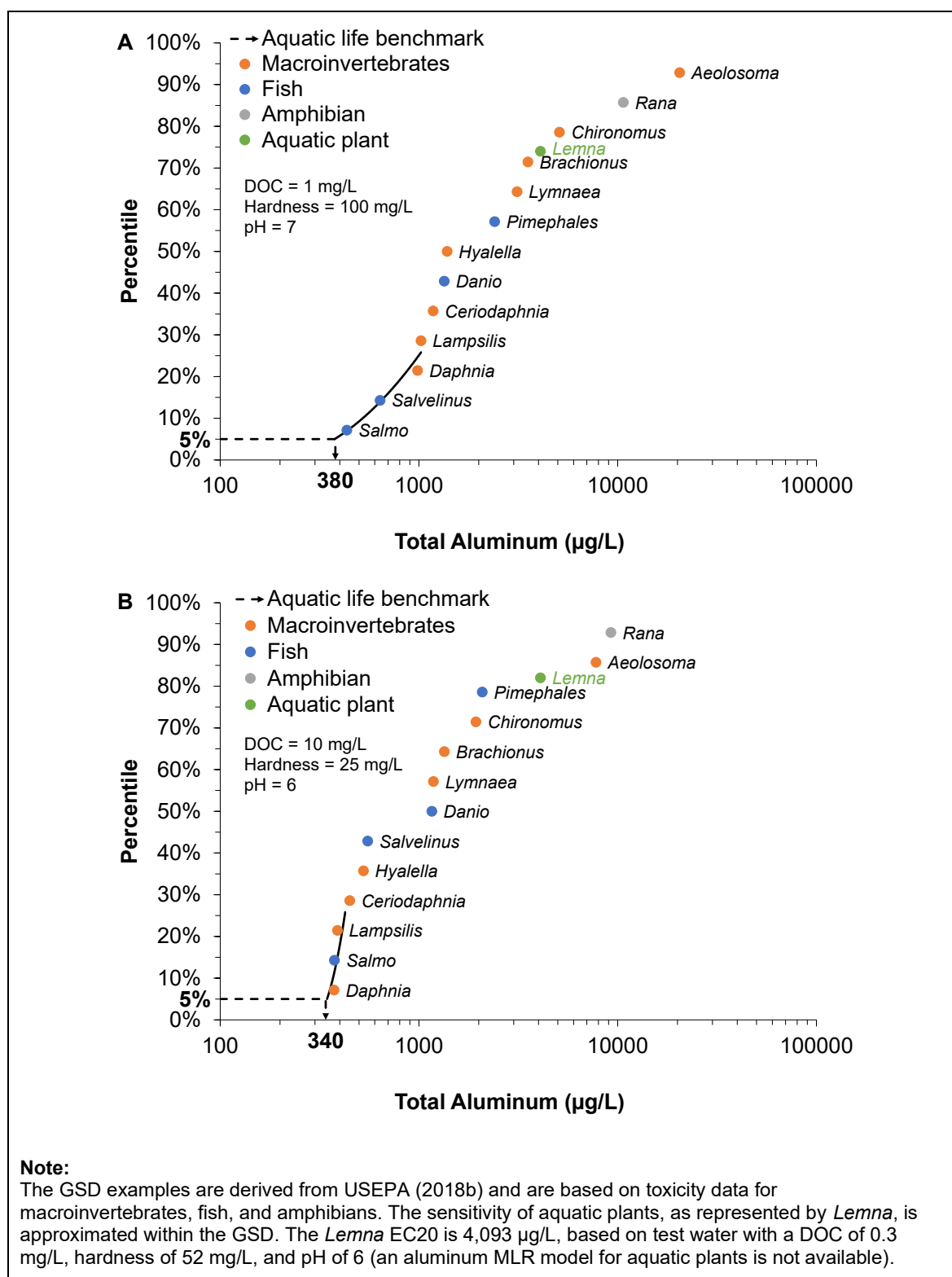


Figure 4-1. Data Used to Derive MLR-Based Total Aluminum Porewater Benchmarks at A) DOC of 1 mg/L, Hardness of 100 mg/L and pH of 7, and B) DOC of 10 mg/L, Hardness of 25 mg/L, and pH of 6, Including Chronic Value for Plants

4.2 ANTIMONY

Selected porewater benchmarks for antimony are presented in Table 4-2, followed by a more detailed description of the benchmark selection process for each receptor.

Table 4-2. Antimony Porewater Benchmarks for Aquatic Life, Benthic Invertebrates, Fish, and Plants

Receptor	Benchmark Basis	Benchmark
Aquatic life	5th percentile of the chronic GSD	39 µg/L
Benthic invertebrate	most sensitive invertebrate genus	83 µg/L
Fish	most sensitive fish genus	2,800 µg/L
White sturgeon	most sensitive fish genus	2,800 µg/L
Aquatic plants	ALB	39 µg/L

Notes:

ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)

GSD - genus sensitivity distribution

The GSD for antimony is shown in Figure 4-2; this GSD was derived as part of the COPC Refinement (TAI 2020) (data are presented in Table F1-5). The most sensitive genus in the acute antimony GSD was *Gastrophryne*, as represented by the eastern narrow-mouthed toad (*G. carolinensis*) (Figure 4-2). This species had a 7-day LC50 of 300 µg/L in a test initiated with eggs that concluded 4 days after the eggs had hatched (Table F1-5) (Birge 1978). This LC50 was the driver of the 5th percentile of the acute GSD, which was 324.9 µg/L. The second most sensitive species was the amphipod *Hyalella azteca*, which had a 7-day LC50 of 687 µg/L (Table F1-5) (Borgmann et al. 2005). The LC50s for these two species were considered acute toxicity values in the COPC Refinement, because a 7-day exposure does not technically meet EPA's definition of a chronic exposure; however, a 7-day exposure is longer than the 96-hour exposure duration that EPA considers acute for criteria derivation for vertebrates. The division of the 5th percentile of the GSD by the ACR of 8.3 results in a value of 39 µg/L.

Based on the data plotted in Figure 4-2, the antimony benchmarks for aquatic life, benthic invertebrate, fish, and aquatic plants are as follows (as summarized in Table 4-2):

- **Aquatic life benchmark.** The aquatic life benchmark is the 5th percentile of the acute GSD adjusted using an ACR (39 µg/L).

- **Benthic invertebrate benchmark.** The estimated chronic value for the most sensitive invertebrate genus (*Hyalella*) is more than twice the 5th percentile of the GSD, and data are available for at least three invertebrate species. Therefore, the benthic invertebrate benchmark is based on the lowest chronic mean genus value for *Hyalella* (83 µg/L).
- **Fish benchmark.** The estimated chronic value for the most sensitive fish genus (*Pimephales*) is more than twice the 5th percentile of the GSD, and data are available for at least three fish species, including a salmonid. Therefore, the fish benchmark is based on the lowest chronic mean genus value for *Pimephales* (2,823 µg/L).
- **Aquatic plant benchmark.** The only available estimated genus chronic value is for duckweed (2,151 µg/L). Because antimony toxicity data were available for fewer than three plant species, no plant-specific benchmark was derived.

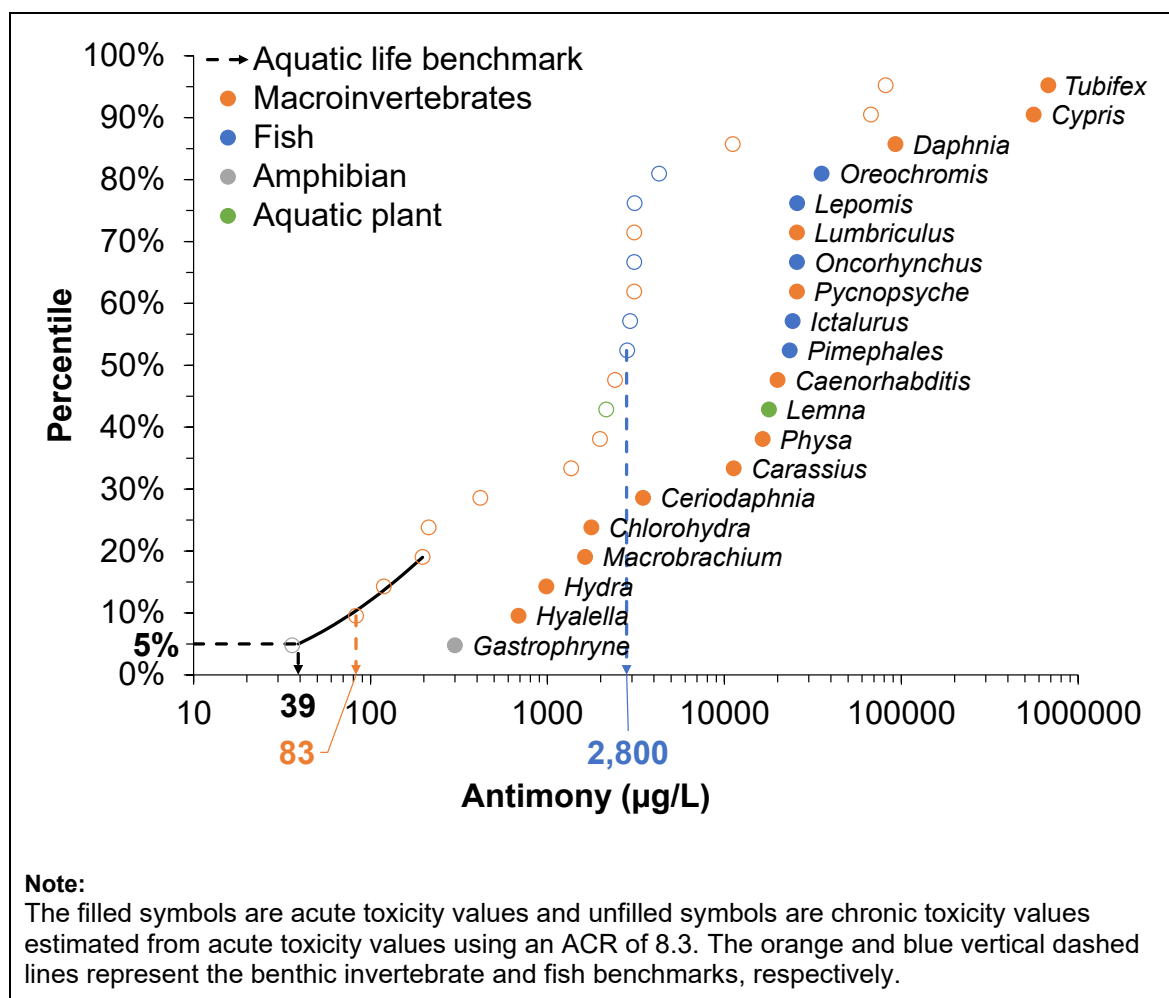


Figure 4-2. Data Used to Derive Antimony Porewater Benchmarks

4.3 CADMIUM

Selected porewater benchmarks for cadmium are presented in Table 4-3, followed by a more detailed description of the benchmark selection process for each receptor.

Table 4-3. Cadmium Porewater Benchmarks for Aquatic Life, Benthic Invertebrates, Fish, and Plants

Method	Receptor	Benchmark Basis	Hardness Equation	Benchmark
EPA AWQC hardness-based model	aquatic life	5th percentile of GSD	$\text{benchmark } (\mu\text{g/L}) = \exp(0.7977 \times \ln[\text{hardness}] - 3.909) \times (1.101672 - \ln[\text{hardness}] \times 0.041838)$	0.72 µg/L ^a

Method	Receptor	Benchmark Basis	Hardness Equation	Benchmark
	benthic invertebrate	ALB	ALB	0.72 µg/L ^a
	fish	ALB	ALB	0.72 µg/L ^a
	white sturgeon	EC20	benchmark (µg/L) = $\exp(0.7977 \times \ln(\text{hardness}) - 2.01)$	5.3 ^a
No model; lowest chronic value	aquatic plants	most sensitive plant genus	NA	56 µg/L ^b

Notes:

^a Assumed hardness = 100 mg/L

^b Aquatic plant benchmark is not adjusted for hardness

ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)

AWQC - ambient water quality criteria

DOC - dissolved organic carbon

EC20 - concentration that causes a 20 percent effect

GSD - genus sensitivity distribution

NA - not applicable

EPA's AWQC for cadmium are derived from a hardness-based GSD composed of invertebrates and fish (USEPA 2016a) (Table F1-6). The 5th percentile of the GSD adjusted for a hardness of 100 mg/L is 0.79 µg/L (Figure 4-3). Based on the data plotted in Figure 4-3, cadmium benchmarks for aquatic life, benthic invertebrates, fish (generally), white sturgeon (specifically), and aquatic plants are as follows (Table 4-3):

- **Aquatic life benchmark.** EPA's hardness-based cadmium criterion for aquatic life is the 5th percentile of the GSD (0.79 µg/L at a hardness of 100 mg/L) (Figure 4-3).
- **Benthic invertebrate benchmark.** The estimated chronic value for the most sensitive invertebrate genus (*Hyalella*) is within a factor of two of the 5th percentile of the GSD. Therefore, the benthic invertebrate benchmark was set equal to the aquatic life benchmark.
- **Fish benchmark.** The estimated chronic value for the most sensitive fish genus (*Cottus*) is within a factor of two of the 5th percentile of the GSD. Therefore, the fish benchmark was set equal to the aquatic life benchmark.
- **White sturgeon benchmark.** The sturgeon-specific benchmark is based on a hardness-dependent equation, as described in Section 3.1 and provided in

Table 4-3. At a hardness of 100 mg/L, the sturgeon-specific benchmark is 5.3 µg/L, which is near the 45th percentile of the chronic cadmium GSD (Figure 4-3).

- Aquatic plant benchmark.** Acceptable cadmium toxicity data were available for four aquatic plant species (Table F1-4). The lowest genus mean survival- or growth-related toxicity threshold for aquatic plants, from a test in which cadmium concentrations were measured, was a 7-day EC50 of 112.4 µg/L for pondweed (*Elodea canadensis*) (USEPA 2016a). Division of this EC50 by 2 (Stephan et al. 1985) results in a “low effect” cadmium benchmark of 56 µg/L (limited to 2 significant figures) (Figure 4-3). This value is not adjusted for bioavailability, as there are no published bioavailability models for cadmium and aquatic plants.

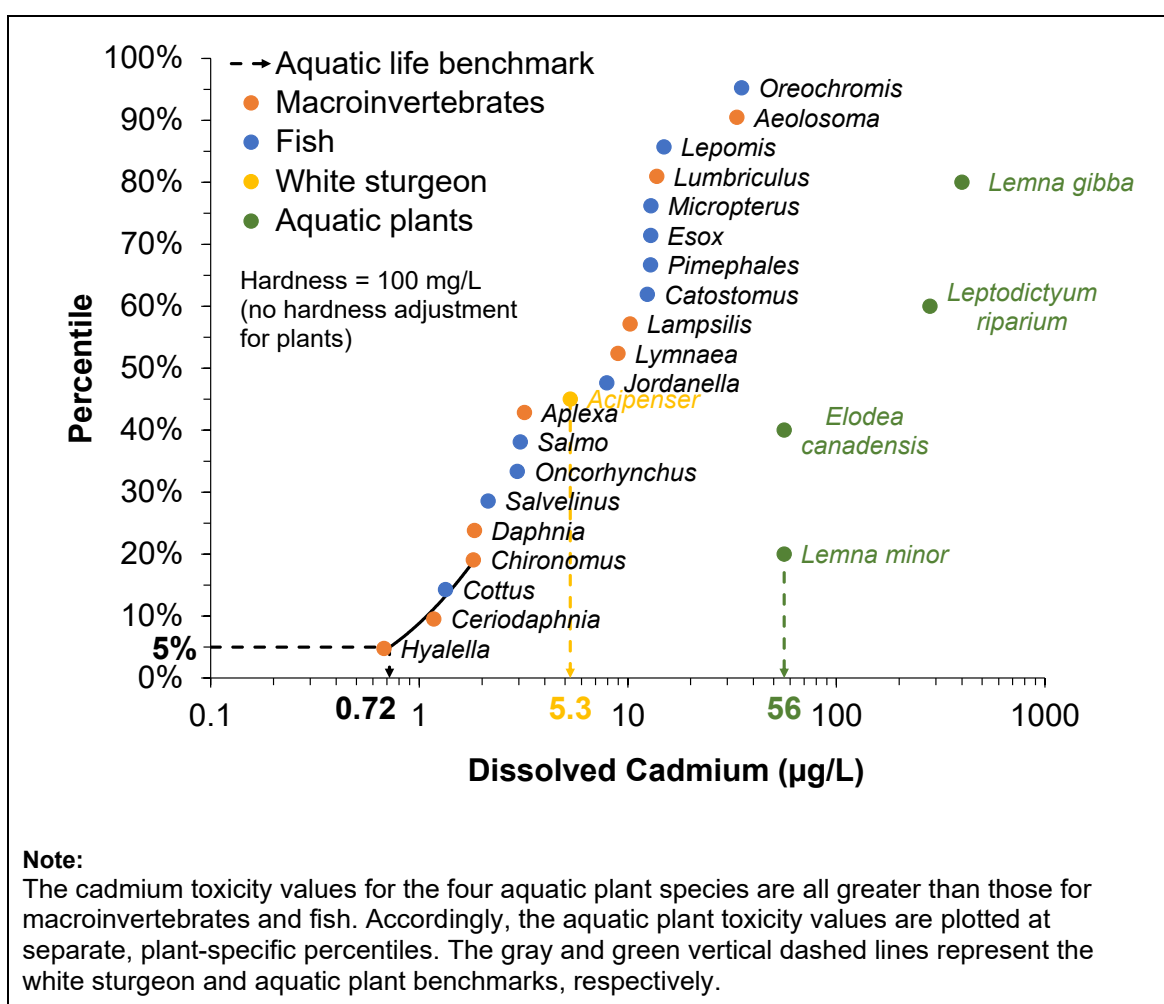


Figure 4-3. Data Used to Derive Hardness-Based Cadmium Porewater Benchmarks at Hardness Value of 100 mg/L, Including Chronic Values for Plants

4.4 COPPER

Selected porewater benchmarks for copper are presented in Table 4-4, followed by a more detailed description of the benchmark selection process for each receptor.

Table 4-4. EPA AWQC BLM-Derived Copper Porewater Benchmarks for Aquatic Life, Benthic Invertebrates, Fish, and Plants

Receptor	Benchmark Basis	Critical Accumulation Concentration	Benchmark
Aquatic life	5th percentile of GSD	0.03395 nmol/g ww	1.5 µg/L ^{a,b}
Benthic invertebrate	ALB	ALB	1.5 µg/L ^a
Fish	white sturgeon	0.00461 nmol/g ww	0.68 µg/L ^a
White sturgeon	EC20 ^c	0.00461 nmol/g ww	0.68 µg/L ^a
Aquatic plants	ALB	ALB	1.5 µg/L ^a

Notes:

^a Assuming DOC = 1 mg/L, hardness = 100 mg/L, and pH = 7 (simplified chemistry mode; see Section 2.2.2)

^b The critical accumulation for aquatic life is based on acute toxicity, so the resulting water-based benchmark is divided by an ACR of 3.22 (USEPA 2007). This ACR is based on the geometric mean of ACRs for six species, which range from 1.48 to 5.59 (USEPA 2007).

^c EC20 is for the time spent moving the endpoint from Puglis et al. (2019) (see text). The aquatic life benchmark is used as the basis for the aquatic plant benchmark (see text).

ACR - acute-to-chronic ratio

ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)

AWQC - ambient water quality criteria

BLM - biotic ligand model

DOC - dissolved organic carbon

EC20 - effect concentration of 20 percent

GSD - genus sensitivity distribution

ww - wet weight

EPA's AWQC for copper is based on the BLM (USEPA 2007). However, insufficient data were available at the time the criteria were developed to support a GSD based on chronic toxicity data; instead, chronic criteria were derived by applying an ACR to the 5th percentile of the acute GSD (Table F1-7). The GSD for invertebrates and fish based on critical accumulation concentrations on the biotic ligand is provided in Figure 4-4A. Figure 4-4B provides the estimated chronic GSD as dissolved copper concentrations after adjustment to a DOC of 1 mg/L, hardness of 100 mg/L, and pH of 7 (based on simplified chemistry; see Section 2.2.2). The 5th percentile of the GSD based on critical accumulation concentrations for copper on the biotic ligand is 0.03395 nmol/g ww (Figure 4-4A). Based on the data plotted in Figure 4-4, the copper benchmarks for aquatic life, benthic

invertebrates, fish (generally), white sturgeon (specifically), and aquatic plants are as follows (Table 4-4):

- **Aquatic life benchmark.** EPA's BLM-based AWQC copper criterion for aquatic life is the 5th percentile of the GSD (1.5 µg/L at a DOC of 1 mg/L, hardness of 100 mg/L, and pH of 7) (Figure 4-4B).
- **Benthic invertebrate benchmark.** The GSD is driven by the sensitivity of *Daphnia* with a lowest chronic value within a factor of two of the 5th percentile of the GSD. Therefore, a benthic invertebrate-specific benchmark was not derived.
- **Fish benchmark.** The most sensitive fish genus in the chronic GSD is *Oncorhynchus*, which has a chronic value more than twice the 5th percentile of the GSD. There are at least three fish species in the GSD, and *Oncorhynchus* is a salmonid. Therefore, *Oncorhynchus* is the basis for the most sensitive non-sturgeon fish genus (9.7 µg/L at a DOC of 1 mg/L, hardness of 100 mg/L, and pH of 7) (Figure 4-4B). However, as discussed in Section 3.4.2, the sensitivity of white sturgeon to copper is greater than that of *Oncorhynchus*. Accordingly, the porewater benchmark for fish is set equal to the benchmark for white sturgeon (Table 4-4).
- **White sturgeon benchmark.** The sturgeon-specific benchmark is based on the BLM, calibrated to the sensitive of white sturgeon, as described in Section 3.2. At a DOC of 1 mg/L, hardness of 100 mg/L, and pH of 7, the sturgeon specific benchmark is 0.68 µg/L (Table 4-4). This benchmark is less than the 5th percentile of the chronic copper GSD (Figure 4-4B).
- **Aquatic plant benchmark.** Acceptable copper toxicity data were available for one aquatic plant species (common duckweed) (Table F1-4). Toxicity data were available from three studies, with highly variable toxicity values. The lowest chronic toxicity values identified were from a study by Antunes et al. (2012).²¹ The magnitudes of the toxicity values reported in Antunes et al. (2012) varied with test water chemistry. In a test using water with a DOC of 1 mg/L, hardness of 187 mg/L, and final test pH of 8.7, the copper EC10 for common duckweed was 0.72 µg/L. In three additional common duckweed tests with DOCs of 1 or 2 mg/L,

²¹ Acceptable copper toxicity values for duckweed were also identified from Bishop and Perry (1981) and Girling et al. (2000), but these values were at higher concentrations ranging from an maximum acceptable toxicant concentration (MATC) of 179 µg/L to an EC50 of 1,200 µg/L. Preference was given to the toxicity data from Antunes et al. (2012), which included greater characterization of parameters that influence copper bioavailability.

EC10s ranged from 2.1 to 5.9 µg/L; in two other tests with a DOC of 10 mg/L, EC10s ranged from 21 to 95 µg/L. Although bioavailability models for copper and aquatic plants have not been published, EC10s in these tests increased with increasing DOC concentrations (consistent with the copper BLM), demonstrating that copper toxicity to aquatic plants varies with water chemistry conditions. Because data were available for fewer than three aquatic plant species, no plant-specific benchmark was derived.

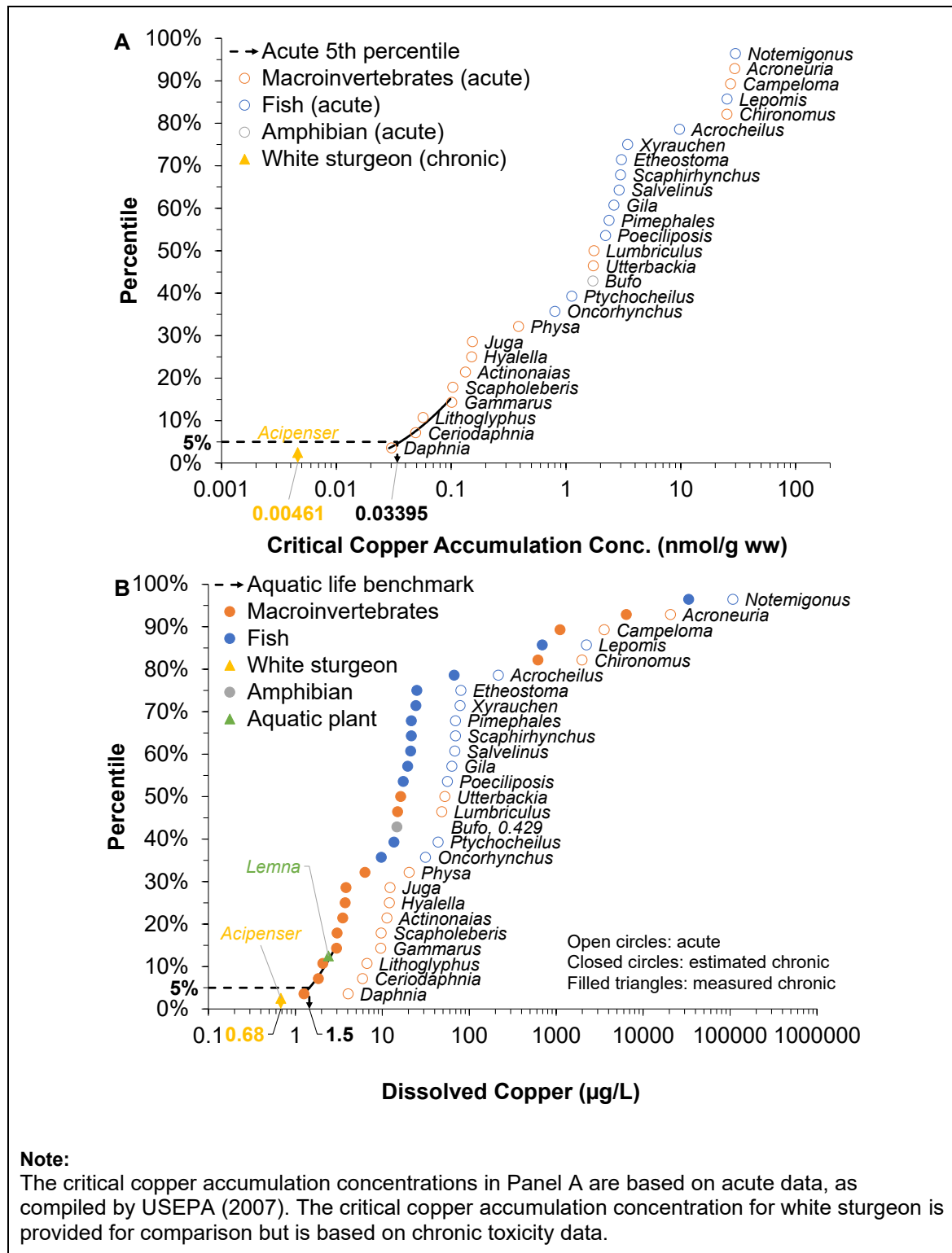


Figure 4-4. A) EPA BLM-Based Critical Copper Accumulation Concentrations and B) Data Used to Derive EPA BLM-Based Copper Porewater Benchmarks at DOC of 1 mg/L, Hardness of 100 mg/L, and pH of 7, Including Chronic Values for White Sturgeon and Plants

4.5 IRON

Selected porewater benchmarks for iron (expressed as total iron) are presented in Table 4-5, followed by a more detailed description of the benchmark selection process for each receptor.

Table 4-5. Iron Porewater Benchmarks for Aquatic Life, Benthic Invertebrates, Fish, and Plants

Method	Receptor	Benchmark Basis	Benchmark Value Equation	Benchmark
Field observations	aquatic life	EPA AWQC	NA	1,000 µg/L
MLR	benthic invertebrate	most sensitive invertebrate genus	benchmark (µg/L) = $\exp(6.412 + 0.507[\ln(\text{DOC})])$	609 µg/L ^a
	fish	most sensitive fish genus	benchmark (µg/L) = $\exp(0.3383 + 0.920[\ln(\text{DOC})] + 0.830[\text{pH}])$	474 µg/L ^a
	white sturgeon	most sensitive fish genus	benchmark (µg/L) = $\exp(0.3383 + 0.920[\ln(\text{DOC})] + 0.830[\text{pH}])$	474 µg/L ^a
Lowest chronic genus value	aquatic plants	ALB	ALB	1,000 µg/L

Notes:

^a Assuming DOC = 1 mg/L (invertebrate model) and DOC = 1 mg/L and pH = 7 (fish model). Note that the benthic invertebrate- and fish-specific benchmarks based on these DOC and pH conditions are less than EPA's chronic criterion of 1,000 µg/L. However, the DOC and pH conditions in UCR porewater samples result in benthic invertebrate- and fish-specific benchmarks that are sometimes less than and sometimes greater than the EPA criterion.

ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)

AWQC - ambient water quality criteria

DOC - dissolved organic carbon

MLR - multiple linear regression

NA - not applicable

EPA's current chronic criterion for iron is 1,000 µg/L (USEPA 1986). This criterion was initially developed in the 1970s and is not based on a GSD. Rather, it is based principally on field observations. This chronic criterion was identified as the aquatic life benchmark. However, as noted in Section 2.2.4 and discussed further in the following paragraph, MLR models for iron are now available and have been applied to more recent chronic iron toxicity data to develop benchmarks for benthic invertebrate and fish. Because the MLR models result in iron benchmarks that vary depending on DOC and pH conditions in porewater, the magnitudes of these benchmarks relative to the EPA criterion of 1,000 µg/L vary. Accordingly, the MLR-based benchmarks may be less than or greater than

1,000 µg/L depending on porewater DOC and pH conditions. In the Aquatic BERA, both the EPA criterion and MLR-based benchmarks are applied.

The MLR-based benthic invertebrate and fish benchmarks were derived based on the MLR models for predicting chronic iron toxicity (Section 2.2.4) and the chronic iron toxicity data from both single-species laboratory toxicity tests and mesocosm experiments with naturally colonized benthic communities; the data from all of these tests and experiments were synthesized by Brix et al. (2023). For the mesocosm data, 14 aquatic insect taxa were sufficiently abundant in control samples to calculate iron EC20s. However, for consistency with EPA's current guidelines for developing AWQC (Stephan et al. 1985), which give preference to single-species laboratory toxicity tests, the mesocosm data were not considered for porewater benchmark development. The resulting chronic GSD based on single-species laboratory tests included five invertebrate genera, five fish genera, and one amphibian genus (Table F1-8; Figure 4-5). Although chronic data were available for a total of 11 genera, those data did not meet EPA's minimum genus data requirements for AWQC development, as chronic iron toxicity data were unavailable for a benthic crustacean (Stephan et al. 1985). Therefore, a GSD could not be developed to derive an aquatic life benchmark as the 5th percentile value of the distribution, and benthic invertebrate and fish benchmarks were derived based on the most sensitive invertebrate and fish genera, respectively. Selected benchmarks are summarized in Table 4-5, followed by a discussion of the benchmark selection process.

As summarized in Section 2.2.4, MLR modeling retained DOC for invertebrates and DOC and pH for fish. When all genera were adjusted to common DOC and pH conditions, *Lumbriculus* (oligochaete worms) and *Prosopium* (whitefish) were identified as the most sensitive invertebrate and fish genera, respectively (Figure 4-5).

To derive the chronic iron benchmark equation for benthic invertebrates, the *C. dubia* model with a DOC slope of 0.507 and the *Lumbriculus* EC20 of 843 µg/L at a DOC of 1.9 mg/L were used to derive a *Lumbriculus*-specific intercept:

$$\text{Intercept} = \ln(843 \text{ } \mu\text{g/L}) - 0.507 \times \ln(1.9 \text{ mg/L}) = 6.412 \quad \text{Equation 15}$$

This intercept was then used to develop a *Lumbriculus*-specific, DOC-based EC20 model, which results in the benthic invertebrate benchmark:

$$\text{Benthic invertebrate benchmark} = \exp(0.507 \times \ln(\text{DOC}) + 6.412) \quad \text{Equation 16}$$

For the fish benchmark, the fathead minnow model, with a DOC slope of 0.920 and pH slope of 0.830, and the whitefish EC20 of 1,279 µg/L, with a DOC of 1.9 mg/L and pH of 7.5, were used to derive a whitefish-specific intercept:

$$\text{Intercept} = \ln(1,279 \mu\text{g/L}) - 0.920 \times \ln(1.9 \text{ mg/L}) - 0.830 \times 7.5 = 0.3512 \quad \text{Equation 17}$$

This intercept was then used to develop a fish-specific DOC-based EC20 model, which results in the fish benchmark:

$$\text{Fish benchmark} = \exp(0.920 \times \ln(\text{DOC}) + 0.830 \times \text{pH} + 0.3383) \quad \text{Equation 18}$$

To calculate sample-specific benchmarks, Equation 16 will be used to adjust the benthic invertebrate benchmark to the DOC concentration in the relevant site sample for evaluation in the Aquatic BERA, and Equation 18 will be used to adjust the fish benchmark to the DOC and pH conditions in the relevant site sample. An iron MLR model is not available for plants. The selected iron benchmarks are as follows (as shown in Figure 4-5 and summarized in Table 4-5):

- **Benthic invertebrate benchmark.** When the invertebrate MLR model is applied to the most sensitive invertebrate genus (*Lumbriculus*), the result is a total iron benchmark of 609 $\mu\text{g/L}$ at a DOC of 1 mg/L (Figure 4-5).
- **Fish benchmark.** When the fish MLR model is applied to the most sensitive fish genus (*Prosopium*), the result is a total iron benchmark of 474 $\mu\text{g/L}$ at a DOC of 1 mg/L and pH of 7 (Figure 4-5).
- **Aquatic plant benchmark.** Iron toxicity data for aquatic plants are limited to two plant species: greater duckweed (*Spirodela polyrrhiza*), which has a 14-day LOEC of 11,700 $\mu\text{g/L}$ based on total iron (Sinha et al. 1994); and water hyacinth (*Eichhornia crassipes*), which had total iron concentrations of 2,105, 2,926, and 3,634 $\mu\text{g/L}$ that had no significant effect on wet mass relative to the control ($p > 0.05$) following 21-day exposures (Newete et al. 2016) (Table F1-4). However, because iron toxicity data are available for fewer than three plant species, no plant-specific benchmark was derived.

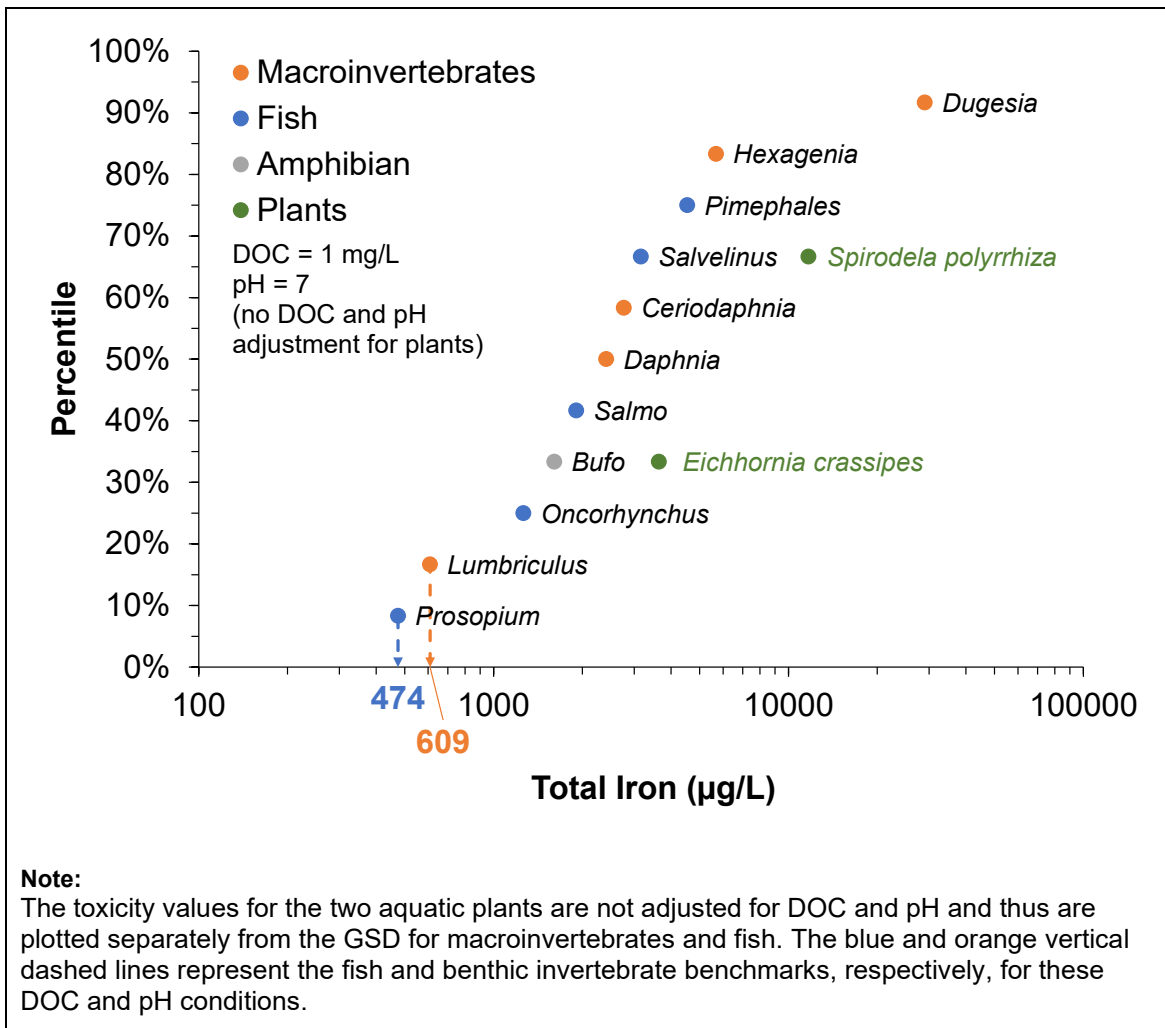


Figure 4-5. Data Used to Derive MLR-Based Total Iron Porewater Benchmarks at DOC of 1 mg/L and pH of 7, Including Chronic Values for Plants

4.6 LEAD

Selected porewater benchmarks for lead are presented in Table 4-6, followed by a more detailed description of the benchmark selection process for each receptor.

Table 4-6. Lead Porewater Benchmarks for Aquatic Life, Benthic Invertebrates, Fish, and Plants

Method	Receptor	Benchmark Basis	Hardness Equation	Benchmark
EPA AWQC hardness model	aquatic life	5th percentile of GSD	$\text{benchmark } (\mu\text{g/L}) = \exp(1.273 \times \ln(\text{hardness}) - 4.705) \times (1.46203 - \ln(\text{hardness}) \times 0.145712)$	2.5 $\mu\text{g/L}$ ^{a,b}
	benthic invertebrate	ALB	ALB	2.5 $\mu\text{g/L}$ ^a
	fish	white sturgeon	$\text{benchmark } (\mu\text{g/L}) = \exp(1.273 \times \ln(\text{hardness}) - 2.604)$	>26 $\mu\text{g/L}$ ^a
	white sturgeon	EC20 ("greater than")	$\text{benchmark } (\mu\text{g/L}) = \exp(1.273 \times \ln(\text{hardness}) - 2.604)$	>26 $\mu\text{g/L}$ ^a
No model; lowest chronic value	aquatic plants	most sensitive plant genus	NA	63 $\mu\text{g/L}$ ^c

Notes:

^a Assumed hardness = 100 mg/L

^b The aquatic life benchmark is a chronic criterion based on an ACR of 51.29, which is calculated as the geometric mean of ACRs for four species; the species mean ACRs range from 18.13 to 124.8 (USEPA 1985).

^c Aquatic plant benchmark is not adjusted for hardness

ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)

AWQC - ambient water quality criteria

EC20 - effect concentration of 20 percent

GSD - genus sensitivity distribution

NA - not applicable

EPA's AWQC for lead include a hardness-based GSD composed of invertebrates and fish (USEPA 1985) (Table F1-9). Insufficient data were available to develop a GSD based on chronic toxicity data; instead, chronic criteria were derived based on the application of an ACR to the 5th percentile of the acute GSD. The 5th percentile of the GSD for chronic toxicity adjusted for hardness is 2.5 $\mu\text{g/L}$ (Figure 4-6). Based on the benchmark selection process shown in Figure 2-1, the lead benchmarks based on EPA's AWQC model for aquatic life, benthic invertebrates, fish (generally), white sturgeon (specifically), and aquatic plants are as follows (Table 4-6):

- **Aquatic life benchmark.** EPA's hardness-based lead criterion for aquatic life is the 5th percentile of the GSD (2.5 $\mu\text{g/L}$ at a hardness of 100 mg/L) (Figure 4-6).
- **Benthic invertebrate benchmark.** The estimated chronic value for the most sensitive invertebrate genus (*Gammarus*) is within a factor of two of the 5th

percentile of the chronic toxicity GSD (Figure 4-6). Therefore, the benthic invertebrate benchmark was set equal to the aquatic life benchmark.

- **Fish benchmark.** The estimated chronic value for the most sensitive fish genus (*Oncorhynchus*) is more than twice the 5th percentile of the GSD (Figure 4-6). There are at least three fish species in the GSD and at least one is a salmonid; the chronic value for the most sensitive fish genus (*Oncorhynchus*) is 91 µg/L at a hardness of 100 mg/L (Figure 4-6). However, the chronic value for white sturgeon is less than that for *Oncorhynchus*. Accordingly, the porewater benchmark for fish is set equal to the lead benchmark for white sturgeon (Table 4-6).
- **White sturgeon benchmark.** The sturgeon-specific benchmark is based on a hardness-dependent equation, as described in Section 3.3 and provided in Table 4-6. At a hardness of 100 mg/L, the sturgeon-specific benchmark is > 26 µg/L, which is near the 22nd percentile of the chronic lead GSD (Figure 4-6). The benchmark is a “greater than” value because less than 20% effects were observed at the highest lead concentration tested (i.e., a higher lead concentration would be required to observe an effect > 20%).
- **Aquatic plant benchmark.** Models based on hardness have not been developed for aquatic plants. Therefore, the lowest chronic genus value identified for aquatic plants is used as the benchmark. Chronic lead toxicity data were identified for three aquatic plant species, with the most sensitive aquatic plant species being common duckweed (Table F1-4). The range of lead EC20s for common duckweed is 63 to more than 1,434 µg/L from test waters with varying chemistry conditions (Antunes and Kreager 2014).²² Therefore, 63 µg/L is conservatively identified as the aquatic plant benchmark (Figure 4-6).

²² Antunes and Kreager (2014) developed a speciation model to predict lead toxicity to common duckweed over a range of water chemistry conditions, but their model does not have a mechanistic basis (as noted by the authors) and does not have an available user interface.

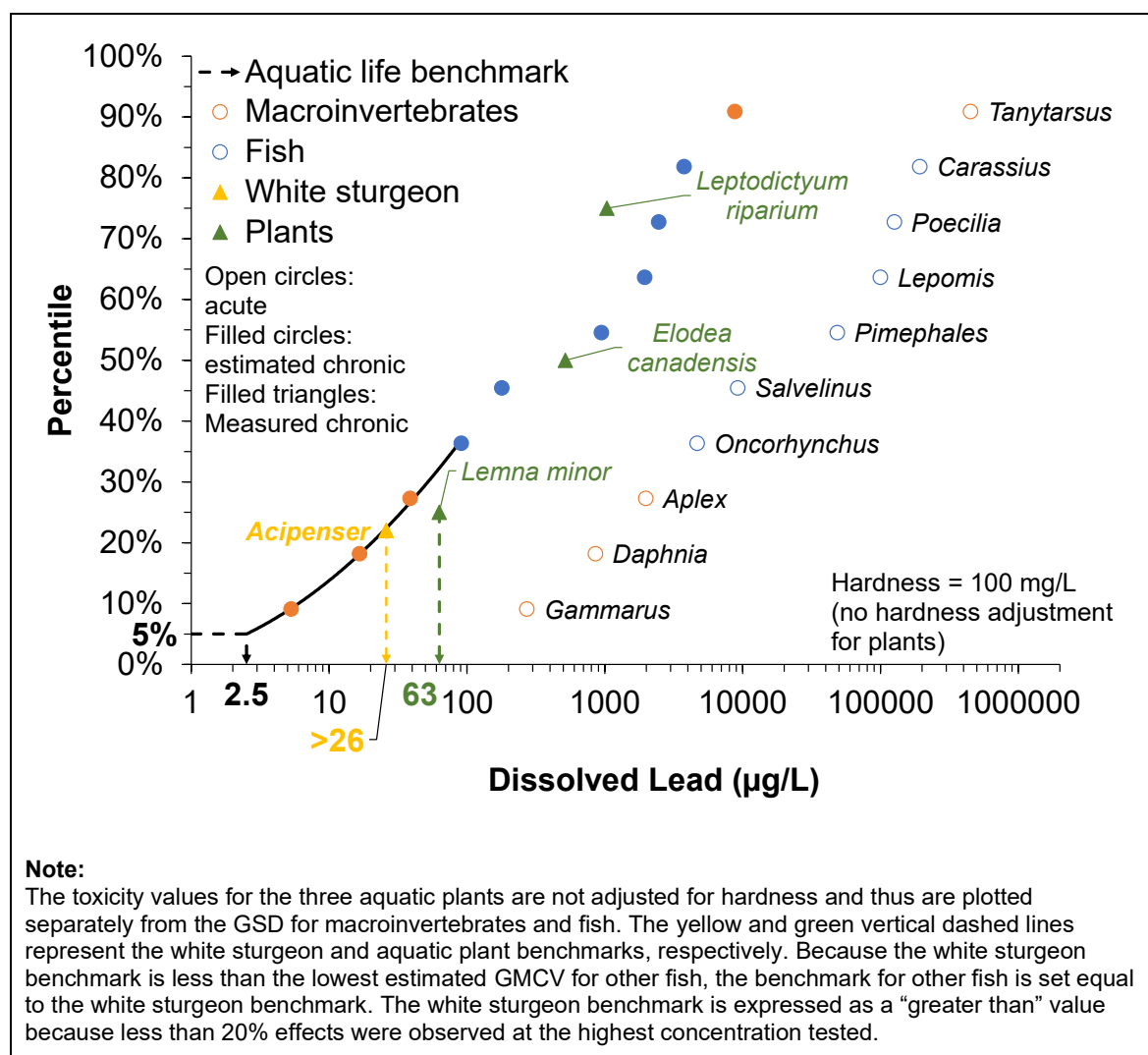


Figure 4-6. Data Used to Derive Hardness-Based Lead Porewater Benchmarks at Hardness of 100 mg/L Using EPA GSD, Including Chronic Values for Plants

4.7 MANGANESE

Selected manganese benchmarks are summarized in Table 4-7, followed by a description of the benchmark selection process for each receptor.

Table 4-7. Manganese Porewater Benchmarks for Aquatic Plants, Benthic Invertebrates, Fish, and Plants

Method	Receptor	Benchmark Basis	Benchmark Value Equation	Benchmark
Hardness model	aquatic life	5th percentile of the GSD	$\text{benchmark } (\mu\text{g/L}) = \exp(0.7424 \times \ln[\text{hardness}] + 4.124)$	1,700 $\mu\text{g/L}$ ^a
	benthic invertebrate	ALB	ALB	1,700 $\mu\text{g/L}$ ^a
	fish	ALB	ALB	1,700 $\mu\text{g/L}$ ^a
	white sturgeon	ALB	ALB	1,700 $\mu\text{g/L}$ ^a
Lowest chronic genus value	aquatic plants	ND	ND	1,700 $\mu\text{g/L}$ ^a

Notes:

^a Assumed hardness = 100 mg/L

ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)

GSD - genus sensitivity distribution

NA - not applicable

Based on methods described in Section 2.2.5, the acute GSD was derived from the hardness-based model for manganese (Figure 4-7), as was an estimated chronic GSD calculated using the ACR of 5.267 (Table F1-10). The selected manganese benchmarks are as follows (Table 4-7):

- **Aquatic life benchmark.** The aquatic life benchmark is the 5th percentile of the GSD (1,700 $\mu\text{g/L}$ at a hardness of 100 mg/L) (Figure 4-7).
- **Benthic invertebrate benchmarks.** The most sensitive invertebrate genus (*Hyaletella*) has a GMCV within a factor of two of the 5th percentile of the GSD. Therefore, the benthic invertebrate benchmark was set equal to the aquatic life benchmark.
- **Fish benchmarks.** The most sensitive fish genus (*Oncorhynchus*) has a GMCV within a factor of two of the 5th percentile of the GSD. Therefore, the fish benchmark was set equal to the aquatic life benchmark.
- **Aquatic plant benchmark.** No hardness model is available for aquatic plants. The lowest effect concentration identified was an EC50 of 31,000 $\mu\text{g/L}$ for duckweed growth (Table F1-4) (Wang 1986). Division of the EC50 by two (USEPA 1985) resulted in an estimated “low effect” concentration of 15,500 $\mu\text{g/L}$. However,

because manganese toxicity data are available for fewer than three plant species, no plant-specific benchmark was derived.

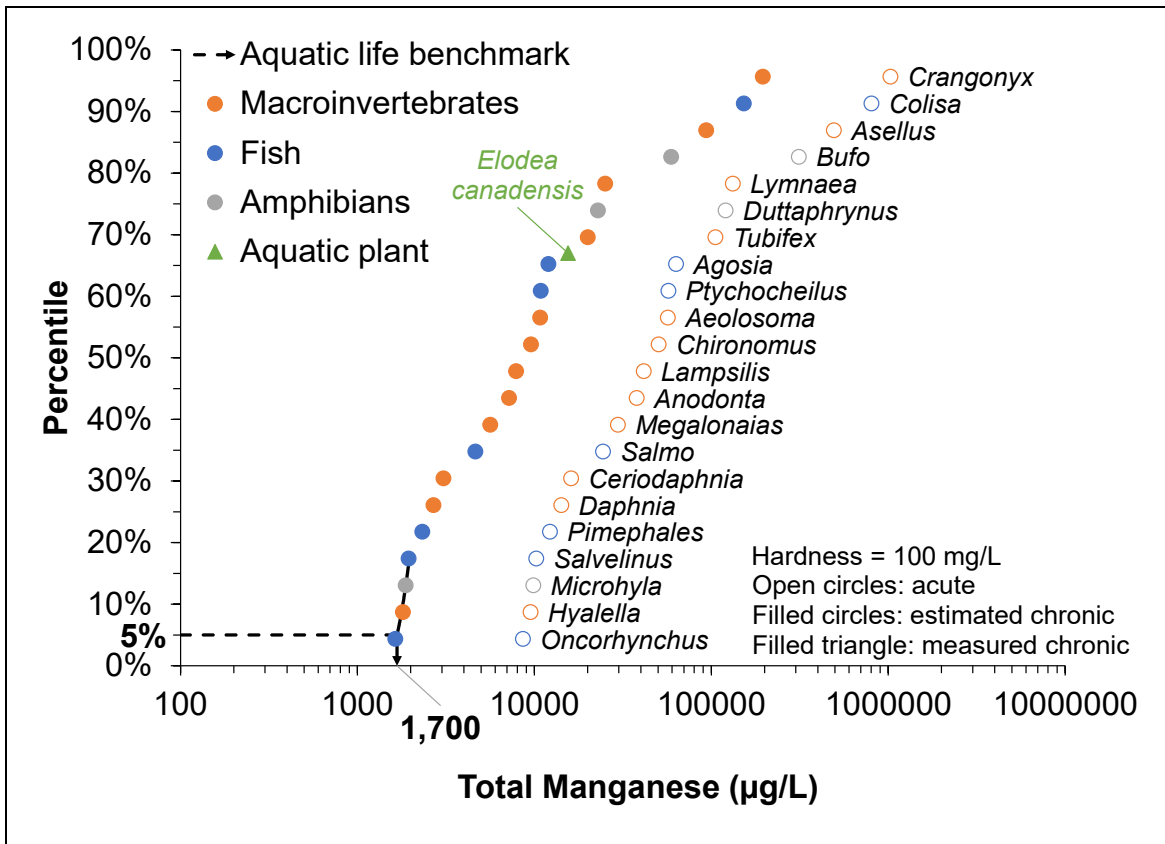


Figure 4-7. Data Used to Derive Hardness-Based Manganese Porewater Benchmarks at Hardness of 100 mg/L, Including Lowest Chronic Value for Plants

4.8 ZINC

Selected porewater benchmarks for zinc are presented in Table 4-8, followed by a more detailed description of the benchmark selection process for each receptor.

Table 4-8. Zinc Porewater Benchmarks for Aquatic Life, Benthic Invertebrates, Fish, and Plants

Method	Receptor	Benchmark Basis	Hardness Equation	Benchmark
EPA AWQC hardness model	aquatic life	5th percentile of GSD	$\text{benchmark } (\mu\text{g/L}) = \exp(0.8473 \times \ln[\text{hardness}] + 0.8840) \times 0.986$	120 $\mu\text{g/L}$ ^{a,b}
	benthic invertebrate	ALB	ALB	120 $\mu\text{g/L}$ ^a
	fish	ALB	ALB	120 $\mu\text{g/L}$ ^a
	white sturgeon	EC20	$\text{benchmark } (\mu\text{g/L}) = \exp(0.8473 \times \ln(\text{hardness}) + 1.15)$	160 $\mu\text{g/L}$ ^a
No model; lowest chronic value	aquatic plants	most sensitive plant genus	NA	160 $\mu\text{g/L}$ ^b

Notes:

^a Assumed hardness = 100 mg/L

^b The aquatic life benchmark is a chronic criterion based on an ACR of 2, which is calculated as the geometric mean of ACRs for three species; the species mean ACRs range from 0.7027 to 7.26 (USEPA 1996).

^c Aquatic plant benchmark is not adjusted for hardness

ALB - aquatic life benchmark (taxa-specific benchmark not derived, see text)

AWQC - ambient water quality criteria

GSD - genus sensitivity distribution

NA - not applicable

EPA's AWQC for zinc include a hardness-based GSD composed of invertebrates and fish (USEPA 1996) (Table F1-11). Insufficient data were available to develop a GSD based on chronic toxicity data; instead, chronic criteria were derived based on the application of an ACR to the 5th percentile of the acute GSD. The 5th percentile of the GSD for chronic toxicity is 120 $\mu\text{g/L}$ at a hardness of 100 mg/L (Figure 4-8). Based on the benchmark selection process shown in Figure 2-1, the zinc benchmarks based on EPA's hardness model are as follows (Table 4-8):

- **Aquatic life benchmark.** EPA's hardness-based AWQC zinc criterion for aquatic life is the 5th percentile of the GSD (120 $\mu\text{g/L}$ at a hardness of 100 mg/L) (Figure 4-8).
- **Benthic invertebrate benchmark.** The estimated chronic value for the most sensitive invertebrate genus (*Ceriodaphnia*) is within a factor of two of the 5th percentile of the chronic toxicity GSD (Figure 4-8). Therefore, the benthic invertebrate benchmark was set equal to the aquatic life benchmark.

- **Fish benchmarks.** The estimated chronic value for the most sensitive fish genus (*Morone*) is within a factor of two of the 5th percentile of the chronic toxicity GSD (Figure 4-8). Therefore, the fish benchmark was set equal to the aquatic life benchmark.
- **White sturgeon.** The sturgeon-specific benchmark is based on a hardness-dependent equation, as described in Section 3.4 and provided in Table 4-8. At a hardness of 100 mg/L, the sturgeon-specific benchmark is 160 µg/L, which occurs at approximately the 6th percentile of the chronic zinc GSD (Figure 4-8).
- **Aquatic plant.** Models based on hardness have not been developed for zinc and aquatic plants. Zinc toxicity data were identified for five plant species (Table F1-4). The lowest chronic genus value identified for aquatic plants—an EC50 of 327 µg/L for pondweed (Basile et al. 2012)—is used as the basis for benchmark. This EC50 was divided by two and rounded to two significant figures to estimate “a low effect” threshold of 160 µg/L (Figure 4-8).

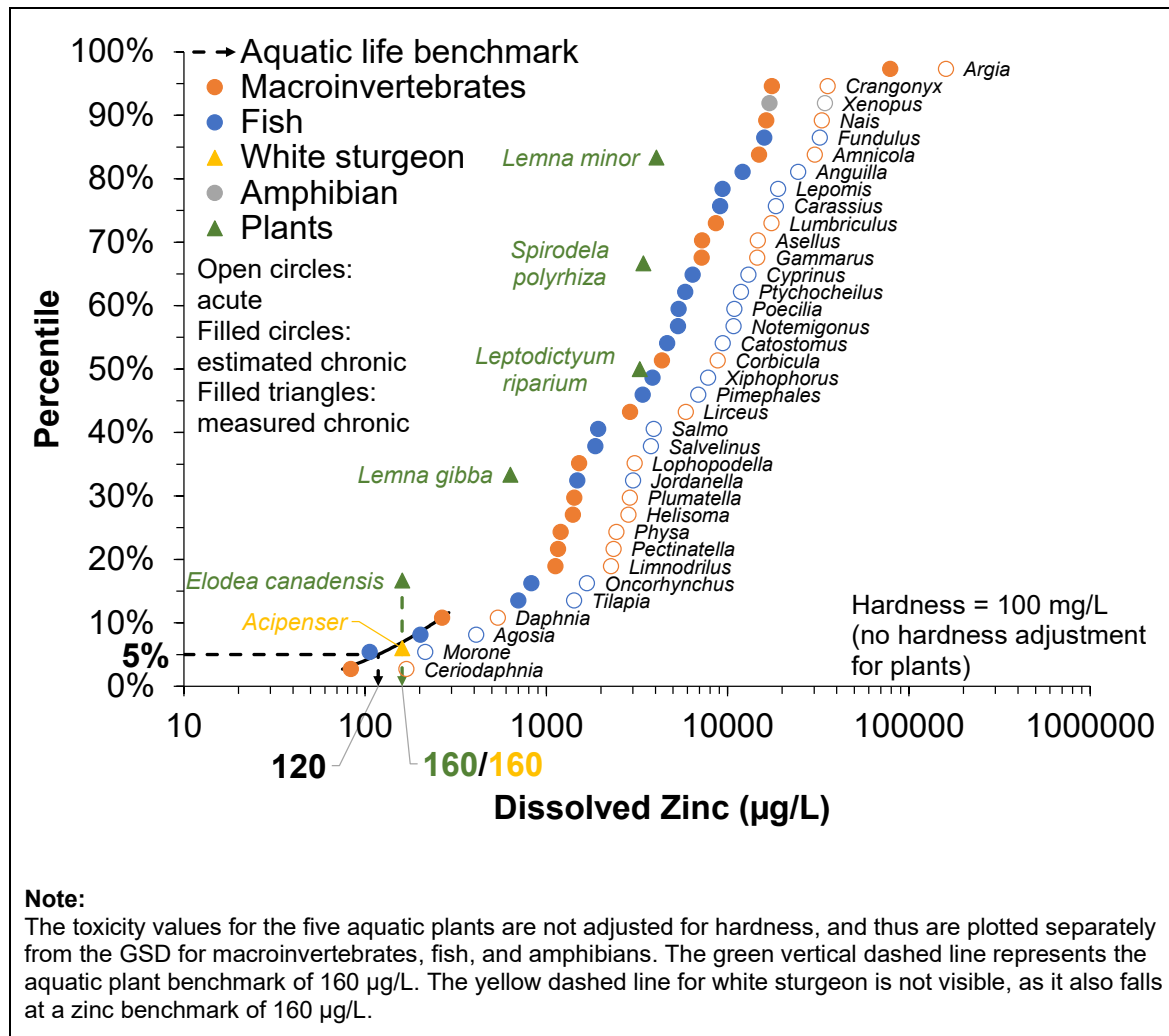


Figure 4-8. Data Used to Derive Hardness-Based Zinc Porewater Benchmarks at Hardness of 100 mg/L Using EPA GSD, Including Chronic Values for Plants

5 UNCERTAINTY

This section describes the uncertainty associated with the estimated concentrations of COPCs that cause specified toxic effects in a prescribed percentage of the organisms in an exposed test population under controlled conditions in a bioassay laboratory. In other words, it describes the limitations of the models for deriving porewater benchmarks. This section does not address uncertainties about what the porewater concentration data and porewater toxicity benchmarks indicate about risk to ecological assessment endpoints in the UCR. That topic will be addressed in the ecological assessment endpoint (EAE)-specific uncertainty analyses in Volumes 2 and 3 of the Aquatic BERA.

5.1 BENTHIC INVERTEBRATE AND FISH BENCHMARKS

Uncertainties associated with the models used to set porewater benchmarks for benthic invertebrate and fish are discussed in the sections below.

5.1.1 EPA Hardness-based Models

EPA's current freshwater AWQC for cadmium, lead, and zinc are adjusted for water hardness (USEPA 2016a, 1985, 1996), based on the observation that increasing concentrations of hardness cations (e.g., calcium and magnesium) reduce the uptake of these metals by receptors. The hardness models are based on statistical regressions between toxicity values (e.g., acute EC50s or chronic EC20s) and test water hardness. For a given metal, if it can be demonstrated that the hardness slopes are not significantly different ($p > 0.05$) among species, then EPA develops hardness models by pooling data for several species. Accordingly, hardness models are generally translatable among species, although not always. As example of an exception is that hardness does not appear to affect chronic lead toxicity to *C. dubia*, but it does affect lead toxicity to the fathead minnow (Mager et al. 2011). Hardness-based models do not account for the influence of other toxicity-modifying factors, such as DOC. This uncertainty is most important for lead, for which DOC is an important toxicity-modifying factor for sensitive species (DeForest et al. 2017); it is less important for cadmium, for which hardness is the most important toxicity-modifying factor (USEPA 2016a). For zinc, both hardness and DOC can have an important influence on toxicity (Van Sprang et al. 2009). Thus, the uncertainty in hardness models for lead and zinc is greater than the uncertainty in the hardness model for cadmium.

5.1.2 EPA Copper BLM

EPA recommended the BLM-based copper AWQC in 2007 and summarized uncertainties associated with the supporting documentation of those criteria (USEPA 2007). For example, EPA noted that the BLM used to develop its recommended AWQC was based on the equilibrium reactions of copper and other cations (e.g., calcium, magnesium, and sodium) with a single type of surface ligand, and therefore the BLM provided an approximate representation of the complex set of reactions that can result in copper toxicity. The influence of cations on copper bioavailability can involve mechanisms other than competition for the surface ligand. EPA also noted that the potential for the microenvironment of the gill (a “biotic ligand” in fish and other organisms) to influence copper speciation, as well as the potential importance of non-equilibrium processes. However, EPA stated that despite these uncertainties, the BLM is a useful mechanistic model to account for the effects of complexing ligands and competing cations that influence copper bioavailability and hence toxicity. USEPA (2007) also stated that “[e]ven if the mechanistic descriptions are incomplete, this model [the BLM] allows the major empirical effects of complexing ligands and competing cations to be described in a more comprehensive and reasonable fashion than other approaches.”

EPA also noted that the utility of the BLM can be shown most clearly by evaluating how well copper toxicity is predicted. EPA evaluated the ability of the BLM to predict copper toxicity to the fathead minnow over a wide range of bioavailability conditions. While copper toxicity values ranged by more than a factor of 60, the BLM, on average, accurately predicted copper toxicity values to within a factor of 1.3 (USEPA 2007). EPA also noted that while the BLM sometimes results in criteria that are less or greater than those produced by the earlier hardness-based model, these new criteria more consistently achieve EPA’s targeted level of protection for AWQC. In other words, using the BLM results in less uncertainty about porewater benchmarks than using the next best alternative (i.e., the hardness-based model that the BLM replaced in 2007).

5.1.3 Aluminum

The toxicity values used to develop EPA’s aluminum criteria are expressed as total aluminum, because both dissolved and precipitated forms of aluminum can be bioavailable and contribute to aluminum toxicity. However, measuring total aluminum to capture dissolved and precipitated forms includes non-bioavailable particulate aluminum forms that often occur in substantial quantities (e.g., clay minerals). This means that measuring total aluminum generally overestimates bioavailable aluminum concentrations and toxicity.

An additional source of uncertainty in the aluminum benchmarks pertains to the applicability of the MLR models for *C. dubia* and fathead minnow to other invertebrate and fish species, respectively. It is assumed that the processes influencing aluminum bioavailability to *C. dubia* and fathead minnow are similar to those influencing bioavailability to other species. However, the toxicity of aluminum to other species has not been rigorously tested over ranges of DOC, pH, and hardness, so the assumption is uncertain. EPA recommends that the MLR-based criteria be applied over the following ranges: DOC concentrations of 0.08 to 12.0 mg/L, pH levels of 5.0 to 10.5, and hardness concentrations of 0.01 to 430 mg/L. If porewater conditions fall outside of these ranges, the uncertainty in applying the MLR models to other species is great enough that EPA recommends against doing so.

5.1.4 Antimony

The acute toxicity data set for antimony is robust, with toxicity data available for 21 species: 1 plant, 13 invertebrates, 6 fish species, and 1 amphibian. These data meet EPA's minimum data requirements for the calculation of acute criteria (Stephan et al. 1985). However, the limited chronic toxicity data for antimony makes it necessary to apply a generic ACR of 8.3 to the 7-day LC50 for *H. azteca*. There is reason to suspect that this ACR is conservative for such an application. The acute data used to calculate ACRs are based on 48- to 96-hour exposure durations, depending on the species. The generic ACR of 8.3 is a median value based on a wide variety of chemicals, chronic toxicity test types, and taxa, but it is similar to the median ACR of 9.1 for metals (Raimondo et al. 2007). Use of a median ACR is a source of uncertainty, because there can be an inverse relationship between the acute sensitivity of an organism to metals and the magnitude of the ACR (USEPA 2007). This uncertainty tends to cause an ACR overestimation error, resulting in conservative chronic antimony benchmarks.

5.1.5 Iron

The iron toxicity data and associated MLR models are based on total (i.e., dissolved and precipitated) Fe(III),²³ whereas the site data represent only dissolved forms of Fe(II)²⁴ and Fe(III). In anoxic (i.e., oxygen depleted) sediments, iron is likely to be present mostly as Fe(II) (Viollier et al. 2000).

²³ Fe(III) is iron in the +3 oxidation state; it is the form of iron present in the highly oxygenated toxicity test waters.

²⁴ Fe(II) is iron in the +2 oxidation state.

The toxicity of Fe(II) is not as well studied as the toxicity of Fe(III). Given the rapid oxidation rate of Fe(II) compared to that of Fe(III) in the highly oxygenated conditions of most toxicity tests (Morgan and Lahav 2007), it would be difficult to test the toxicity of Fe(II) alone without carefully controlling for iron speciation. Fe(II) toxicity data are limited and no data on how DOC, pH, and hardness influence Fe(II) toxicity data are available. However, the available toxicity data from tests of Fe(II)—which likely included some fraction of Fe(III), which is oxidized during exposure—indicate that Fe(II) is not more toxic than Fe(III) (Rousch et al. 1997; Gerhardt 1994). That said, the fact that data are not available for the same species and from similar exposure designs is a source of uncertainty in the conclusion that Fe(II) is not more toxic than Fe(III). Current best practice is to assume that measuring total iron in porewater, including both dissolved and precipitated forms of Fe(II) and Fe(III), is conservative when assessing potential risks relative to a benchmark based on the toxicity of Fe(III). In addition, total iron in porewater may include particulate phases of iron, which are not bioavailable to aquatic organisms, adding to conservatism in this approach.

An additional source of uncertainty in the iron benchmarks is the applicability of the MLR models for *C. dubia* and fathead minnow to other invertebrate and fish species, respectively. The toxicity of iron to other species has not been tested over ranges of DOC, pH, and hardness conditions, but it is assumed that the processes influencing iron bioavailability to test species are similar to those influencing bioavailability to other species. DOC, pH, and hardness conditions in the iron toxicity tests used to develop the MLR models are within the following ranges: DOC concentrations of 0.3 to 10.5 mg/L for *C. dubia* and 0.3 to 10.9 mg/L for fathead minnow, pH levels of 6.3 to 8.4 for *C. dubia* and 6.0 to 8.5 for fathead minnow, and hardness concentrations of 10.6 to 252 mg/L for *C. dubia* and 10.3 to 247 mg/L for fathead minnow. If porewater conditions fall outside of these ranges (see Table F1-1 for UCR porewater ranges), extrapolating the MLR models is a source of uncertainty that should be carefully considered in the risk characterization for specific EAEs.

5.1.6 Manganese

Some of the uncertainties in the manganese benchmarks are similar to those identified for the aluminum and iron benchmarks. For example, it is assumed that the hardness slope based on a subset of the species tested—*C. dubia*, *Chironomus tentans* (chironomid), *D. magna*, *H. azteca*, *O. kisutch* (coho salmon), *O. mykiss* (rainbow trout), fathead minnow, *Salmo trutta* (brown trout), and *Salvelinus fontinalis* (brook trout)—is applicable to other species in the sensitivity distribution.

There is also uncertainty related to the ACR used to develop the chronic manganese benchmark. This uncertainty was evaluated by comparing the hardness-based benchmark to the available empirical chronic toxicity data for manganese. Among the 96 chronic toxicity tests conducted using 12 different species, including sensitive invertebrate species (e.g., cladocerans and snails) and fish species (e.g., salmonids), the chronic benchmark based on the ACR was under-protective in only 6 tests (i.e., 6 percent): 1 of 4 tests for rainbow trout, 1 of 43 tests for *C. dubia*, and 4 of 36 tests for fathead minnow. Overall, the high protectiveness of the chronic manganese benchmark, particularly for invertebrates, provides confidence that it is protective when evaluating porewater exposures. Therefore, uncertainty about the protectiveness of the chronic manganese benchmark, particularly for invertebrates, is low.

5.2 AQUATIC PLANT BENCHMARKS

Aquatic plant benchmarks were identified if data were available for at least three species (Figure 2-1). For several COPCs—aluminum, antimony, copper, iron, and manganese—toxicity data were available for fewer than three aquatic plant species and no aquatic plant-specific benchmark could be derived. Using the aquatic life benchmark to assess aquatic plants is likely conservative, as aquatic plant benchmarks for the three COPCs with adequate plant toxicity data—cadmium, lead, and zinc—were all greater than their respective aquatic life benchmarks. This likelihood of conservatism is also consistent with EPA’s conclusion in deriving AWQC for the COPCs included in this document, as available: that chronic criteria based on invertebrates and vertebrates are protective of aquatic plants.

The primary uncertainties in the aquatic plant benchmarks for cadmium, lead, and zinc pertain to the lack of bioavailability models for these metals and aquatic plants. Furthermore, for cadmium and lead, the most sensitive aquatic plant species is duckweed, which is a floating plant, and for zinc the most sensitive species is Canadian waterweed, which is a submergent plant with roots that may hang freely in the water or anchor the plant to bottom substrate. Both species were assumed to be reasonable surrogates for rooted aquatic plants that may take up metals from porewater via their roots.

6 SUMMARY

Porewater toxicity benchmarks were derived for benthic invertebrates, fish (including sturgeon-specific benchmarks for cadmium, copper, lead, and zinc), and aquatic plants, as summarized in Table 6-1. Many of the benchmarks were calculated using models that require sample-specific concentrations of DOC, hardness, and/or pH; therefore, representative values for these parameters (as footnoted in the table) were used to develop example benchmarks. Benchmarks were recalculated for each sample using sample-specific data, and the sample-specific benchmarks are used in the Aquatic BERA.

Table 6-1. Summary of All Porewater Benchmarks Derived for Aquatic Life, Benthic Invertebrates, Fish, and Aquatic Plants

Receptor	Benchmark (µg/L) ^a									
	Aluminum	Antimony	Cadmium	Copper	Iron	Lead	Manganese	Molybdenum	Selenium	Zinc
Aquatic life	380 ^b	39 ^c	0.72 ^d	1.5 ^e	1,000	2.5 ^d	1,700 ^f	38,000 ^c	N/AP	120 ^d
Benthic invertebrate	380 ^h	83 ^c	0.72 ^h	1.5 ^h	609 ^g	2.5 ^h	1,700 ^h	N/AP	N/AP	120 ^h
Fish	380 ^h	2,800 ^c	0.72 ^h	0.68 ⁱ	474 ^g	> 26 ⁱ	1,700 ^h	N/AP	N/AP	120 ^h
White sturgeon	380 ^h	2,800 ^j	5.3 ^d	0.68 ^e	474 ^j	> 26 ^d	1,700 ^j	N/AP	N/AP	160 ^d
Aquatic plants	380 ^h	39 ^h	56 ^c	1.5 ^h	1,000 ^h	63 ^c	1,700 ^h	N/AP	N/AP	160 ^c

Notes:

^a Inputs for benchmarks that are adjusted for DOC, hardness, and/or pH: DOC = 1 mg/L, hardness = 100 mg/L, and pH = 7

^b EPA AWQC MLR model

^c No model; lowest GMCV or PNEC (molybdenum)

^d EPA AWQC hardness model

^e EPA AWQC BLM

^f Windward (2014) hardness model

^g Brix et al. (2023) MLR model

^h Set equal to aquatic life benchmark

ⁱ Set equal to white sturgeon benchmark

^j Set equal to fish benchmark

AWQC - ambient water quality criteria

BLM - biotic ligand model

DOC - dissolved organic carbon

GMCV - genus mean chronic value

MLR - multiple linear regression

N/AP - not applicable

N/AV - not available

PNEC - predicted no-effect concentration

Windward - Windward Environmental LLC

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ATTACHMENT F1

TOXICITY AND PARAMETER DATA FOR POREWATER BENCHMARKS

Table F1-1. Range of EPA Copper BLM Parameter Values

BLM Parameter	Units	Data Sets used in Model Development and/or Validation ^a		UCR Field-Collected Porewater Data Set from Phase 2 and Phase 3 Sediment Studies			
		Lower Bound	Upper Bound	Lower Bound	Upper Bound	% of Samples Below Model Upper Bound	% of Samples Above Model Upper Bound
pH		4.9	9.2	6.38	10.2	0	4.6
DOC	(mg/L)	0.05	29.65	0.07	57.4	0	0.9
Humics	(%)	10	60	10 ^b	10 ^b	NA	NA
Calcium	(mg/L)	0.204	120.24	9.87	157	0	0.4
Magnesium	(mg/L)	0.024	51.9	0.501	37.4	0	0
Sodium	(mg/L)	0.16	236.9	0.94	12.5	0	0
Potassium	(mg/L)	0.039	156	0.46	9.12	0	0
Sulfate	(mg/L)	0.096	278.4	0.39	104	0	0
Chloride	(mg/L)	0.32	279.72	0.16	30.1	2.3	0
Alkalinity	(mg/L)	1.99	360	29	384	0	1.4

Notes:

^a Lower and upper bounds as listed in USEPA (2007).

^b Data are not available; 10% is a default value for humic acid.

BLM - biotic ligand model

DOC - dissolved organic carbon

NA - not applicable

Table F1-2. Cadmium, Copper, Lead, and Zinc Toxicity Values for White Sturgeon

Metal	Reference	Life Stage at Initiation	Duration	Effect Endpoints Measured	Most Sensitive Endpoint	Statistical Endpoint	Conc. (µg/L)
Cadmium	Wang et al. (2014)	2 dph	53 d	% survival, length, dry weight, biomass	Dry weight	EC20	5.4
Cadmium	Vardy et al. (2011)	Embryos	66 d	% hatched, % survival, body condition, behavior (mobility/swimming performance)	Survival	EC20	8.7
Cadmium	Vardy et al. (2014)	8 dph (lab water)	4 d	% mortality	Mortality	LC50	14.5
Cadmium	Vardy et al. (2014)	8 dph (Columbia River water)	4 d	% mortality	Mortality	LC50	72
Cadmium	Calfee et al. (2014)	2 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	>47.2
Cadmium	Calfee et al. (2014)	16 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	>187
Cadmium	Calfee et al. (2014)	30 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	102.7
Cadmium	Calfee et al. (2014)	61 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	<34.4
Cadmium	Calfee et al. (2014)	72 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	5.61
Cadmium	Calfee et al. (2014)	89 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	<17
Copper	Wang et al. (2014)	1 dph	24 d	% survival, length, dry weight, biomass	Dry weight	EC20	1.4
Copper	Vardy et al. (2011)	Embryos	66 d	% hatched, % survival, body condition, behavior (mobility/swimming performance)	Survival	EC20	3.4
Copper	Vardy et al. (2013)	Yolk sac (8 dph)	4 d	% mortality	Mortality	LC50	22
Copper	Vardy et al. (2013)	Swim-up (15 dph)	4 d	% mortality	Mortality	LC50	10
Copper	Vardy et al. (2013)	Juvenile (40 dph)	4 d	% mortality	Mortality	LC50	9
Copper	Vardy et al. (2013)	Juvenile (45 dph)	4 d	% mortality	Mortality	LC50	17
Copper	Vardy et al. (2013)	Later juvenile (105 dph)	4 d	% mortality	Mortality	LC50	54
Copper	Vardy et al. (2014)	8 dph (lab water)	4 d	% mortality	Mortality	LC50	22
Copper	Vardy et al. (2014)	8 dph (Columbia River water)	4 d	% mortality	Mortality	LC50	25
Copper	Vardy et al. (2014)	40 dph (lab water)	4 d	% mortality	Mortality	LC50	9
Copper	Vardy et al. (2014)	40 dph (Columbia River water)	4 d	% mortality	Mortality	LC50	18
Copper	Cardno Entrix and HDR HydroQual (2018)	16 dph	4 d	% mortality	Mortality	LC50	13.2
Copper	Cardno Entrix and HDR HydroQual (2018)	45 dph	4 d	% mortality	Mortality	LC50	20.4
Copper	Calfee et al. (2014)	2 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	2.67
Copper	Calfee et al. (2014)	16 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	4.32
Copper	Calfee et al. (2014)	30 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	6.31
Copper	Calfee et al. (2014)	44 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	>49.8
Copper	Calfee et al. (2014)	61 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	30.8
Copper	Calfee et al. (2014)	72 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	74
Copper	Calfee et al. (2014)	89 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	21.9
Copper	Little et al. (2012)	38 dph (Kootenai River)	4 d	% mortality	Mortality	LC50	4.1
Copper	Little et al. (2012)	40 dph (Kootenai River)	4 d	% mortality	Mortality	LC50	4.7
Copper	Little et al. (2012)	26 (Columbia River)	4 d	% mortality	Mortality	LC50	4.5
Copper	Little et al. (2012)	27 (Columbia River)	4 d	% mortality	Mortality	LC50	6.8
Copper	Little et al. (2012)	123 (Columbia River)	4 d	% mortality	Mortality	LC50	268.9
Copper	Little et al. (2012)	167 dph (Kootenai River)	4 d	% mortality	Mortality	LC50	103.7
Copper	Little et al. (2012)	450 dph (Columbia River)	4 d	% mortality	Mortality	LC50	269.0
Copper	Puglis et al. (2019)	1 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization, spontaneous swimming activity	Swimming activity	MATC	5.4
Copper	Puglis et al. (2019)	1 dph	14 d	% mortality, loss of equilibrium (LOE), immobilization, spontaneous swimming activity	Time spent moving	EC20	1.3
Copper	Puglis et al. (2019)	28 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization, spontaneous swimming activity, endurance	All endpoints	EC50	>7.38
Copper	Puglis et al. (2019)	35 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization, spontaneous swimming activity, endurance	Distance	EC50	4.26
Copper	Puglis et al. (2019)	58 dph	4 d	% mortality, feeding	Feeding	MATC	7.4
Lead	Wang et al. (2014)	2 dph	53 d	% survival, length, dry weight, biomass	Dry weight	EC20	>27
Lead	Wang et al. (2014)	8 dph (lab water)	4 d	% mortality	Mortality	LC50	177
Lead	Wang et al. (2014)	8 dph (Columbia River water)	4 d	% mortality	Mortality	LC50	>410
Lead	Wang et al. (2014)	40 dph (lab water)	4 d	% mortality	Mortality	LC50	528
Lead	Wang et al. (2014)	40 dph (Columbia River water)	4 d	% mortality	Mortality	LC50	1556

Table F1-2. Cadmium, Copper, Lead, and Zinc Toxicity Values for White Sturgeon

Metal	Reference	Life Stage at Initiation	Duration	Effect Endpoints Measured	Most Sensitive Endpoint	Statistical Endpoint	Conc. (µg/L)
Zinc	Wang et al. (2014)	2 dph	53 d	% survival, length, dry weight, biomass	Dry weight	EC20	160
Zinc	Vardy et al. (2011)	Embryos	66 d	% hatched, % survival, body condition, behavior (mobility/swimming performance)	Survival	EC20	102
Zinc	Vardy et al. (2014)	8 dph (lab water)	4 d	% mortality	Mortality	LC50	150
Zinc	Vardy et al. (2014)	8 dph (Columbia River water)	4 d	% mortality	Mortality	LC50	625
Zinc	Calfee et al. (2014)	2 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	146.7
Zinc	Calfee et al. (2014)	16 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	>1575
Zinc	Calfee et al. (2014)	30 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	3109
Zinc	Calfee et al. (2014)	44 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	>2610
Zinc	Calfee et al. (2014)	61 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	<253
Zinc	Calfee et al. (2014)	72 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	<391
Zinc	Calfee et al. (2014)	89 dph	4 d	% mortality, loss of equilibrium (LOE), immobilization	Mortality+LOE+Immobilization	EC50	<586

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Table F1-3. Multiple Linear Regression-Adjusted Genus Mean Chronic Values for Aluminum

DOC = 1 mg/L, Hardness = 100 mg/L, and pH = 7			DOC = 10 mg/L, Hardness = 25 mg/L, and pH = 6		
Genus	Type	GMCV (µg/L)	Genus	Type	GMCV (µg/L)
<i>Aeolosoma</i>	Invertebrate	20,514	<i>Rana</i>	Amphibian	> 9,255
<i>Rana</i>	Amphibian	10,684	<i>Aeolosoma</i>	Invertebrate	7,792
<i>Chironomus</i>	Invertebrate	> 5,099	<i>Pimephales</i>	Fish	2,085
<i>Brachionus</i>	Invertebrate	3,539	<i>Chironomus</i>	Invertebrate	1,937
<i>Lymnaea</i>	Invertebrate	3,119	<i>Brachionus</i>	Invertebrate	1,344
<i>Pimephales</i>	Fish	2,407	<i>Lymnaea</i>	Invertebrate	1,185
<i>Hyalella</i>	Invertebrate	1,387	<i>Danio</i>	Fish	1,163
<i>Danio</i>	Fish	1,342	<i>Salvelinus</i>	Fish	552.8
<i>Ceriodaphnia</i>	Invertebrate	1,181	<i>Hyalella</i>	Invertebrate	526.9
<i>Lampsilis</i>	Invertebrate	1,026	<i>Ceriodaphnia</i>	Invertebrate	448.8
<i>Daphnia</i>	Invertebrate	985.3	<i>Lampsilis</i>	Invertebrate	389.7
<i>Salvelinus</i>	Fish	638.2	<i>Salmo</i>	Fish	376.3
<i>Salmo</i>	Fish	434.4	<i>Daphnia</i>	Invertebrate	374.2

Source: USEPA (2018)

Notes:

DOC - dissolved organic carbon

GMCV - genus mean chronic value

Table F1-4. Aquatic Plant Chronic Toxicity Values

COPC	Scientific Name	Common Name	Duration	Endpoint	Effect	Chronic Value (µg/L)	Note	Reference
Aluminum	<i>Lemna minor</i>	Duckweed	7 d	EC20	Growth	4,093		Cardwell et al. (2018)
	<i>Elodea canadensis</i>	Pondweed	7 d	EC50	Growth	281		
Cadmium	<i>Lemna minor</i>	Duckweed	7 d	EC50	Growth	56.2	EC50 divided by 2	Basile et al. (2012)
	<i>Leptodictyum riparium</i>	Moss	7 d	EC50	Growth	400		
	<i>Lemna gibba</i>	Duckweed	7 d	EC50	Growth	56.2		
Copper	<i>Lemna minor</i>	Duckweed	7 d	IC10	Root Elongation	2.1	Bioavailability conditions varied among tests	Antunes et al. (2012)
						0.72		
						3.8		
						5.9		
						21.0		
						95.3		
						2105		
Iron	<i>Eichhornia crassipes</i>	Water Hyacinth	21 d	No effects ^a	Wet Mass	2926		Newete et al. (2016)
						3634		
	<i>Spirodela polyrrhiza</i>	Greater Duckweed	14 d	LOEC	Biomass	11700		
Lead	<i>Elodea canadensis</i>	Pondweed	7 d	EC50	Growth	518	EC50 divided by 2	Basile et al. (2012)
	<i>Lemna minor</i>	Duckweed	7 d	EC20	Net Root Elongation	924	Bioavailability conditions varied among tests	Antunes and Kreager (2014)
	<i>Lemna minor</i>	Duckweed	7 d	EC20	Net Root Elongation	63		Antunes and Kreager (2014)
	<i>Lemna minor</i>	Duckweed	7 d	EC20	Net Root Elongation	448		Antunes and Kreager (2014)
	<i>Lemna minor</i>	Duckweed	7 d	EC20	Net Root Elongation	236		Antunes and Kreager (2014)
	<i>Lemna minor</i>	Duckweed	7 d	EC20	Net Root Elongation	1434		Antunes and Kreager (2014)
	<i>Lemna minor</i>	Duckweed	7 d	EC20	Net Root Elongation	178		Antunes and Kreager (2014)
	<i>Lemna minor</i>	Duckweed	7 d	EC20	Net Root Elongation	118		Antunes and Kreager (2014)
	<i>Lemna minor</i>	Duckweed	7 d	EC50	Growth	518	EC50 divided by 2	Basile et al. (2012)
	<i>Leptodictyum riparium</i>	Moss	7 d	EC50	Growth	1036	EC50 divided by 2	Basile et al. (2012)
Manganese	<i>Eichhornia crassipes</i>	Water Hyacinth	21 d	MATC	Progeny Counts/Numbers	975		Newete et al. (2016)
Zinc	<i>Lemna trisulca</i>	Star Duckweed	14 d	EC50	Yield	164	EC50 divided by 2	Huebert and Shay (1992)
	<i>Elodea canadensis</i>	Pondweed	7 d	EC50	Growth	163	EC50 divided by 2	Basile et al. (2012)
	<i>Lemna gibba</i>	Duckweed	10 d	MATC	Growth	632		Megateli et al. (2009)
	<i>Lemna minor</i>	Duckweed	7 d	EC50	Growth	3269	EC50 divided by 2	Basile et al. (2012)
	<i>Leptodictyum riparium</i>	Moss	7 d	EC50	Growth	3269	EC50 divided by 2	Basile et al. (2012)

Notes:
^a Testing was conducted with single iron concentrations; no significant effects (p > 0.05) were observed relative to the control in each of three tests.
COPC - chemical of potential concern
EC20 - effect concentration of 20 percent
EC50 - effect concentration of 50 percent
IC10 - inhibition concentration of 10%
LOEC - lowest-observed-effect-concentration
MATC - maximum acceptable toxicant concentration

Table F1-5. Genus and Species Mean Chronic Values for Antimony

Scientific Name	Common Name	Form	Acute/ Chronic	Age/Size	Duration	Measured/ Unmeasured	Exposure Type	Endpoint	Effect	Hardness Mean/ Min-Max (mg/L)	pH Mean/ Min-Max	DOC Mean (mg/L)	Sb Conc. (µg/L)	Species Mean Value (µg/L)	Genus Mean Value (µg/L)	Reference
<i>Caenorhabditis elegans</i>	Nematode	trichloride	Acute	Adult (3–4 d)	96 hr	U	S	LC50	Mortality	NR	NR	NR	20,000	20,000	20,000	Williams and Dusenbery (1990)
<i>Carassius auratus</i>	Goldfish	trichloride	Acute	Egg	7 d	M	R	LC50	Mortality	195	7.4	NR	11,300	11,300	11,300	Birge (1978)
<i>Ceriodaphnia dubia</i>	Water Flea	trichloride	Acute	NR	48 hr	M	S	LC50	Mortality	NR	7.9–8.1	NR	3,470	3,470	3,470	Spehar (1987)
<i>Chlorohydra viridissima</i>	Green Hydra	trichloride	Acute	NR	96 hr	M	S	LC50	Mortality	85–103	7.6–7.8	NR	1,770	1,770	1,770	TAI (1990)
<i>Cypris subglobosa</i>	Ostracod	Antimony trioxide	Acute	NR	48 hr	U	R	EC50	Immobilization	245–230	7.6	NR	560,000	560,000	560,000	Khangarot and Das (2009)
<i>Daphnia magna</i>	Water Flea	Antimony trioxide	Acute	NR	48 hr	U	S	EC50	Immobilization	NR	7.2–7.8	NR	423,450	92,698	92,698	Khangarot and Ray (1989)
<i>Daphnia magna</i>	Water Flea	Antimony	Acute	24 hr	48 hr	U	S	LC50	Mortality	72	6.7–8.1	NR	530,000			LeBlanc (1980)
<i>Daphnia magna</i>	Water Flea	trichloride	Acute	12 hr	48 hr	U	S	LC50	Mortality	NR	8.16	NR	23,500			Kimball (1978)
<i>Daphnia magna</i>	Water Flea	trichloride	Acute	12 hr	48 hr	U	S	LC50	Mortality	NR	8.16	NR	14,000			Kimball (1978)
<i>Gastrophryne carolinensis</i>	Eastern Narrow-Mouthed Toad	trichloride	Acute	Egg	7 d	M	R	LC50	Mortality	195	7.4	NR	300	300	300	Birge (1978)
<i>Hyalella azteca</i>	Scud	Antimony	Acute	Young (1–11 d)	7 d	M	S	LC50	Mortality	18	8.27	1.4	687	687	687	Borgmann et al. (2005)
<i>Hydra oligactis</i>	Hydra	trichloride	Acute	NR	96 hr	M	S	LC50	Mortality	85–103	7.6–7.8	NR	1,950	1,950	987	TAI (1990)
<i>Hydra sp.</i>	Hydra	trichloride	Acute	Adult	96 hr	M	S	EC50	Abnormality	46.9	7.7	NR	500	500		Brooke et al. (1986)
<i>Ictalurus punctatus</i>	Channel Catfish	trichloride	Acute	NR	96 hr	M	S	LC50	Mortality	136–171	6.9–8.1	NR	24,100	24,349	24,349	TAI (1990)
<i>Ictalurus punctatus</i>	Channel Catfish	trichloride	Acute	NR	96 hr	M	S	LC50	Mortality	136–171	6.9–8.1	NR	24,600			TAI (1990)
<i>Lemna minor</i>	Duckweed	trichloride	Acute	NR	96 hr	M	S	LOEC	Abundance	60.5	7.2	NR	25,500	17,854	17,854	Brooke et al. (1986)
<i>Lemna minor</i>	Duckweed	trichloride	Acute	NR	96 hr	M	S	NOEC	Abundance	60.5	7.2	NR	12,500			Brooke et al. (1986)
<i>Lepomis macrochirus</i>	Bluegill	trichloride	Acute	NR	96 hr	M	S	LC50	Mortality	44–46	6.8–7.4	NR	25,800	25,800	25,800	Spehar (1987)
<i>Lumbriculus variegatus</i>	Oligochaete, Worm	trichloride	Acute	23 mm	96 hr	M	S	EC50	Mortality	51.7–52.9	7.12	NR	25,700	25,700	25,700	Brooke et al. (1986)
<i>Macrobrachium nipponense</i>	Oriental River Shrimp	trichloride	Acute	Juvenile (4 wk)	96 hr	U	R	LC50	Mortality	38–45	7.4–8.1	NR	1,635	1,635	1,635	Yang et al. (2010)
<i>Oncorhynchus mykiss</i>	Rainbow Trout	trichloride	Acute	Juvenile	96 hr	M	S	LC50	Mortality	52.2	7.12	NR	25,700	25,700	25,700	Brooke et al. (1986)
<i>Oreochromis mossambicus</i>	Mozambique Tilapia	trichloride	Acute	Larvae (3 d)	96 hr	U	S	LC50	Mortality	NR	NR	NR	35,500	35,500	35,500	Lin and Hwang (1998)
<i>Physa heterostropha</i>	Pond Snail, Pneumonate Snail	trichloride	Acute	Juvenile	96 hr	M	S	LC50	Mortality	85–103	7.3–7.8	NR	19,100	16,469	16,469	TAI (1990)
<i>Physa heterostropha</i>	Pond Snail, Pneumonate Snail	trichloride	Acute	Juvenile	96 hr	M	S	LC50	Mortality	85–103	7.3–7.8	NR	14,200			TAI (1990)
<i>Pimephales promelas</i>	Fathead Minnow	Antimony trioxide	Acute	NR	96 hr	U	S	LC50	Mortality	20	7.4	NR	80,000	23,432	23,432	Tarzwel and Henderson (1960)
<i>Pimephales promelas</i>	Fathead Minnow	Antimony trioxide	Acute	NR	96 hr	U	S	LC50	Mortality	400	8.2	NR	80,000			Tarzwel and Henderson (1960)
<i>Pimephales promelas</i>	Fathead Minnow	Antimony	Acute	NR	96 hr	M	F	LC50	Mortality	NR	NR	NR	22,000			Stephan (1978)
<i>Pimephales promelas</i>	Fathead Minnow	trichloride	Acute	30 d	96 hr	M	S	EC50	Mortality	48.5	7.1	NR	14,400			Brooke et al. (1986)
<i>Pimephales promelas</i>	Fathead Minnow	trichloride	Acute	NR	96 hr	U	S	LC50	Mortality	400	8.2	NR	17,000			Tarzwel and Henderson (1960)
<i>Pimephales promelas</i>	Fathead Minnow	trichloride	Acute	30 d	96 hr	M	S	LC50	Mortality	48.5	7.1	NR	14,400			Brooke et al. (1986)
<i>Pimephales promelas</i>	Fathead Minnow	trichloride	Acute	NR	96 hr	U	S	LC50	Mortality	20	7.4	NR	9,000			Tarzwel and Henderson (1960)
<i>Pimephales promelas</i>	Fathead Minnow	trichloride	Acute	Juvenile (8 wk)	96 hr	U	F	LC50	Mortality	NR	8.02	NR	21,000			Kimball (1978)
<i>Pimephales promelas</i>	Fathead Minnow	trichloride	Acute	Juvenile (8 wk)	96 hr	U	F	LC50	Mortality	NR	8.02	NR	22,700			Kimball (1978)
<i>Pimephales promelas</i>	Fathead Minnow	Antimony trioxide	Chronic	Egg (48 hpf)	30 d	M	F	NOEC	Growth, Survival	28–40	6.2–7.3	NR	7.5	7.5	7.5	LeBlanc and Dean (1984)
<i>Pycnopsyche sp.</i>	Caddisfly	trichloride	Acute	18 mm	96 hr	M	S	LC50	Mortality	52.2	7.12	NR	25,700	25,700	25,700	Brooke et al. (1986)
<i>Tubifex tubifex</i>	Tubificid Worm	Antimony trioxide	Acute	NR	96 hr	U	R	EC50	Immobilization	NR	NR	NR	678,000	678,000	678,000	Khangarot (1991)

Notes:
Dash (-) indicates no data.
Gray shading indicates chronic data.
EC50 - concentration that causes a non-lethal effect in 50% of an exposed population
F - flow-through
LC50 - concentration that causes a lethal effect in 50% of an exposed population
LOEC - lowest-observed-effect-concentration
M - measured
NOEC - no-observed-effect-concentration
NR - not reported
R - renewal
S - static
U - unmeasured

Table F1-6. Hardness-Adjusted Genus Mean Chronic Values for Cadmium

Genus	Type	GMCV (µg/L) ^a
<u><i>Invertebrate and Fish Genera in the GSD</i></u>		
<i>Oreochromis</i>	Fish	38.66
<i>Aelosoma</i>	Invertebrate	36.70
<i>Lepomis</i>	Fish	16.43
<i>Lumbriculus</i>	Invertebrate	15.16
<i>Micropterus</i>	Fish	14.22
<i>Esox</i>	Fish	14.17
<i>Pimephales</i>	Fish	14.16
<i>Catostomus</i>	Fish	13.66
<i>Lampsilis</i>	Invertebrate	11.29
<i>Lymnaea</i>	Invertebrate	9.887
<i>Jordanella</i>	Fish	8.723
<i>Aplexa</i>	Invertebrate	3.516
<i>Salmo</i>	Fish	3.36
<i>Oncorhynchus</i>	Fish	3.251
<i>Salvelinus</i>	Fish	2.356
<i>Daphnia</i>	Invertebrate	2.024
<i>Chironomus</i>	Invertebrate	2.000
<i>Cottus</i>	Fish	1.470
<i>Ceriodaphnia</i>	Invertebrate	1.293
<i>Hyaella</i>	Invertebrate	0.7453

Source: USEPA (2016)

Notes:

^a Adjusted to hardness of 100 mg/L.

GMCV - genus mean chronic value

Table F1-7. Acute Genus Mean Critical Accumulation Concentrations for Copper

Genus	Type	Genus Mean Critical Accumulation (nmol/g ww)
<i>Notemigonus</i>	Fish	30.0
<i>Acroneuria</i>	Invertebrate	29.6
<i>Campeloma</i>	Invertebrate	27.2
<i>Lepomis</i>	Fish	25.4
<i>Chironomus</i>	Invertebrate	25.3
<i>Acrocheilus</i>	Fish	9.75
<i>Xyrauchen</i>	Fish	3.45
<i>Etheostoma</i>	Fish	3.05
<i>Scaphirhynchus</i>	Fish	2.99
<i>Salvelinus</i>	Fish	2.91
<i>Gila</i>	Fish	2.62
<i>Pimephales</i>	Fish	2.37
<i>Poeciliopsis</i>	Fish	2.21
<i>Lumbriculus</i>	Invertebrate	1.76
<i>Utterbackia</i>	Invertebrate	1.73
<i>Bufo</i>	Amphibian	1.72
<i>Ptychocheilus</i>	Fish	1.12
<i>Oncorhynchus</i>	Fish	0.801
<i>Physa</i>	Invertebrate	0.386
<i>Juga</i>	Invertebrate	0.154
<i>Hyalella</i>	Invertebrate	0.151
<i>Actinonaias</i>	Invertebrate	0.134
<i>Scapholeberis</i>	Invertebrate	0.103
<i>Gammarus</i>	Invertebrate	0.101
<i>Lithoglyphus</i>	Invertebrate	0.0570
<i>Ceriodaphnia</i>	Invertebrate	0.0491
<i>Daphnia</i>	Invertebrate	0.0303

Source: USEPA (2007)

Note:

ww - wet weight

Table F1-8. Multiple Linear Regression-Adjusted Genus Mean Chronic Values for Iron

Genus	Type	GMCV (µg/L) ^a
<i>Dugesia</i>	Invert	28,986
<i>Hexagenia</i>	Invert	5,679
<i>Pimephales</i>	Fish	4,539
<i>Salvelinus</i>	Fish	3,161
<i>Ceriodaphnia</i>	Invert	2,766
<i>Daphnia</i>	Invert	2,410
<i>Salmo</i>	Fish	1,908
<i>Bufo</i>	Amphibian	1,606
<i>Oncorhynchus</i>	Fish	1,262
<i>Lumbriculus</i>	Invert	609
<i>Prosopium</i>	Fish	474

Source: Brix et al. (2023)

Notes:

^a Adjusted to DOC of 1 mg/L and pH of 7.

GMCV - genus mean chronic value

Table F1-9. Hardness-Adjusted Genus Mean Acute Values for Lead

Genus	Type	GMAV (µg/L) ^a
<i>Tanytarsus</i>	Invertebrate	450,938
<i>Carassius</i>	Fish	193,259
<i>Poecilia</i>	Fish	126,431
<i>Lepomis</i>	Fish	99,994
<i>Pimephales</i>	Fish	48,630
<i>Salvelinus</i>	Fish	9,214
<i>Oncorhynchus</i>	Fish	4,680
<i>Aplexa</i>	Invertebrate	1,988
<i>Daphnia</i>	Invertebrate	856
<i>Gammarus</i>	Invertebrate	273

Source: USEPA (1985)

Notes:

^a Adjusted to hardness of 100 mg/L.

GMAV - genus mean acute value

Table F1-10. Hardness-Adjusted Genus Mean Acute Values for Manganese

Genus	Type	GMAV (µg/L) ^a	GMCV (µg/L) ^b
<i>Crangonyx</i>	Invertebrate	1,029,070	195,381
<i>Colisa</i>	Fish	805,100	152,857
<i>Asellus</i>	Invertebrate	493,775	93,749
<i>Bufo</i>	Amphibian	312,907	59,409
<i>Lymnaea</i>	Invertebrate	132,353	25,129
<i>Duttaphrynus</i>	Amphibian	120,498	22,878
<i>Tubifex</i>	Invertebrate	105,456	20,022
<i>Agosia</i>	Fish	63,317	12,021
<i>Ptychocheilus</i>	Fish	57,290	10,877
<i>Aelosoma</i>	Invertebrate	56,833	10,790
<i>Chironomus</i>	Invertebrate	50,419	9,573
<i>Lampsilis</i>	Invertebrate	41,501	7,879
<i>Anodonta</i>	Invertebrate	37,867	7,189
<i>Megaloniaias</i>	Invertebrate	29,703	5,639
<i>Salmo</i>	Fish	24,451	4,642
<i>Ceriodaphnia</i>	Invertebrate	16,146	3,066
<i>Daphnia</i>	Invertebrate	14,193	2,695
<i>Pimephales</i>	Fish	12,268	2,329
<i>Salvelinus</i>	Fish	10,256	1,947
<i>Microhyla</i>	Amphibian	9,873	1,874
<i>Hyaella</i>	Invertebrate	9,514	1,806
<i>Oncorhynchus</i>	Fish	8,619	1,636

Source: Windward (2014)

Notes:

^a Adjusted to hardness of 100 mg/L.

^b Based on genus mean acute value (GMAV) divided by acute-to-chronic ratio (ACR) of 5.267, which is based on species mean ACRs of 2.963 for *Salmo trutta* (brown trout), 3.620 for *Oncorhynchus mykiss* (rainbow trout), 3.688 for *Ceriodaphnia dubia* (cladoceran), 4.735 for *Daphnia magna* (cladoceran), 6.019 for *Salvelinus fontinalis* (brook trout), and 18.93 for *Pimephales promelas* (fathead minnow) (Windward 2014).

GMCV - genus mean chronic value

Table F1-11. Hardness-Adjusted Genus Mean Acute Values for Zinc

Genus	Type	GMAV (µg/L) ^a
<i>Argia</i>	Invertebrate	160,051
<i>Crangonyx</i>	Invertebrate	35,623
<i>Xenopus</i>	Amphibian	34,500
<i>Nais</i>	Invertebrate	33,104
<i>Fundulus</i>	Fish	32,276
<i>Amnicola</i>	Invertebrate	30,261
<i>Anguilla</i>	Fish	24,522
<i>Lepomis</i>	Fish	18,999
<i>Carassius</i>	Fish	18,441
<i>Lumbriculus</i>	Invertebrate	17,473
<i>Asellus</i>	Invertebrate	14,676
<i>Gammarus</i>	Invertebrate	14,573
<i>Cyprinus</i>	Fish	13,013
<i>Ptychocheilus</i>	Fish	11,838
<i>Poecilia</i>	Fish	10,890
<i>Notemigonus</i>	Fish	10,795
<i>Catostomus</i>	Fish	9,406
<i>Corbicula</i>	Invertebrate	8,816
<i>Xiphophorus</i>	Fish	7,810
<i>Pimephales</i>	Fish	6,891
<i>Lirceus</i>	Invertebrate	5,874
<i>Salmo</i>	Fish	3,915
<i>Salvelinus</i>	Fish	3,778
<i>Lophopodella</i>	Invertebrate	3,071
<i>Jordanella</i>	Fish	3,008
<i>Plumatella</i>	Invertebrate	2,891
<i>Helisoma</i>	Invertebrate	2,839
<i>Physa</i>	Invertebrate	2,434
<i>Pectinatella</i>	Invertebrate	2,351
<i>Limnodrilus</i>	Invertebrate	2,274
<i>Oncorhynchus</i>	Fish	1,676
<i>Tilapia</i>	Fish	1,421
<i>Daphnia</i>	Invertebrate	539
<i>Agosia</i>	Fish	410
<i>Morone</i>	Fish	215
<i>Ceriodaphnia</i>	Invertebrate	169

Source: USEPA (1987)

Notes:

^a Adjusted to hardness of 100 mg/L.

GMAV - genus mean acute value

Appendix F Table References

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