# Center for Biological Diversity

Please accept the attached comments filed on behalf of the Center for Biological Diversity. The Center's comments will be provided through three submissions to relay all relevant attachments. This is submission 2 of 3.

# DRAFT

## Biological Opinion
## on
## EPA's Proposed Program of Continuing Approval or Promulgation of New Cyanide Criteria in State and Tribal Water Quality Standards

**U.S. Fish and Wildlife Service**
**Arlington, Virginia**
**January 15, 2010**

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## 1.0 LIST OF ACRONYMS

| | |
|---|---|
| ACR | Acute to Chronic Ratio |
| BE | Biological Evaluation |
| BMP | Best Management Practice |
| CCC | Criteria Continuous Concentration (chronic criterion) |
| CFR | Code of Federal Regulations |
| CMC | Criteria Maximum Concentration (acute criterion) |
| CN | Cyanide |
| CPP | Continuing Planning Process |
| CWA | Clean Water Act |
| EC | Effects Concentration |
| EPA | Environmental Protection Agency |
| ESA | Endangered Species Act |
| FAV | Final Acute Value |
| FR | Federal Register |
| HCN | Hydrogen Cyanide |
| ICE | Interspecies Correlation Estimate |
| ISO | International Organization for Standardization |
| LC | Lethal Concentration |
| LCL | Lower Confidence Limit |
| LOEC | Lowest Observable Effects Concentration |
| MATC | Maximum Acceptable Toxicant Concentration |
| MLE | Maximum Likelihood Estimate |
| NOEC | No Observable Effects Concentration |
| NPDES | National Pollution Discharge Elimination System |
| NPS | Non-Point Source |
| OECD | Organisation for Economic Co-operation and Development |
| SETAC | Society for Environmental Toxicology and Chemistry |
| TMDL | Total Maximum Daily Load |
| WQM | Water Quality Management |
| WQS | Water Quality Standard |

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## 2.0 INTRODUCTION

This document transmits the U.S. Fish and Wildlife Service's (Service) biological opinion based upon our review of EPA's proposed program of continuing approval or promulgation of acute and chronic cyanide criteria in State and Tribal water quality standards and their effects on endangered and threatened species and designated critical habitats in accordance with section 7 of the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq*.)(ESA). Your March 23, 2007, request for formal consultation was received on March 26, 2007.

Your request for formal consultation also included a request for our concurrence that the proposed acute and chronic cyanide criteria are not likely to adversely affect any endangered and threatened species or their critical habitats listed in section 9.2.1 of your March 23, 2007, *Biological Evaluation of Aquatic Life Criteria—Cyanide* (CN BE). According to information presented in the CN BE, available data suggest that cyanide is toxic to aquatic plants only at concentrations well above the proposed criterion concentrations. For that reason, we concur with your finding. We determined the proposed action may affect but is not likely to adversely affect numerous threatened and endangered animal species. The species for which we make this determination and our rationale are provided in Appendix B.

This biological opinion is based on information provided in: (1) your March 23, 2007, CN BE; (2) your July 31, 2006, *Draft Framework for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual* (BE Methodology); (3) your October 29, 2004, *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual*; (4) the February 22, 2001, *Memorandum of Agreement Between the Environmental Protection Agency, Fish and Wildlife Service and National Marine Fisheries Service Regarding Enhanced Coordination Under the Clean Water Act and Endangered Species Act; Notice* (66 FR 11202); and (5) additional information contained in Service files. A complete administrative record of this consultation is on file at the Service's Headquarters Office in Arlington, Virginia.

### Scope of this Biological Opinion

This biological opinion evaluates the effects of EPA's program of continuing approval of cyanide criteria adopted in State water quality standards under the Clean Water Act (CWA) on listed species and their critical habitats. More precisely, this biological opinion evaluates at a program level the potential effects to listed species and critical habitat from exposure to cyanide at EPA's recommended CWA section 304(a) criteria. The assumptions made in this biological opinion about the potential for exposure at the criteria values may not apply consistently to all State and Tribal water quality standards or even all waters within a State or Tribal boundary. State and Tribal water quality standards are comprised of 4 elements: (1) designated uses; (2) criteria sufficient to protect those uses; (3) an antidegradation policy; and (4) general policies. The other components of a water quality standard, besides the criteria, may invalidate the assumptions made here about exposure.

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Because the assumptions made in this biological opinion about the potential for listed species' exposure at the criteria values may not apply consistently to all State and Tribal water quality standards or even all waters within a State or Tribal boundary, this biological opinion does not include incidental take exemptions. Given the uncertainty about such exposure, we are unable to analytically determine at the program scale the likelihood (as opposed to the potential) of such take occurring. Therefore, it will be necessary for EPA to conduct subsequent, step-down ESA section 7 consultations with the Service on individual State and Tribal water quality standards to determine if incidental take exemptions associated with State and Tribal implementation of cyanide criteria in their water quality standards are warranted.

We anticipate much of the analysis presented in this biological opinion will carry over, so that the tiered consultation on State and Tribal water quality standards need only focus on potential effects of elements that were not fully considered here. Because we address criteria in this biological opinion, tiered consultation would likely focus on the various General Policies being implemented by each State and Tribe and the need to incorporate considerations for listed species and critical habitats into those policies, where necessary.

The Service expects that the EPA will notify affected States and Tribes of the need, in their next triennial review, to review their water quality standards for consistency with the findings presented in this biological opinion. EPA should work with States and Tribes to coordinate and consult directly with the Service's local Field Offices in conducting this review. In this review, EPA will, in consultation with the Service, evaluate waters of the United States within the range of applicable listed species and critical habitats considered in this biological opinion to determine whether:

1. there exists a designated use to which aquatic life criteria apply;
2. State-adopted criteria are consistent with this biological opinion; and,
3. general policies adopted by States or Tribes and approved by EPA would alter the effects analysis in this biological opinion.

This tiered consultation will involve a review of the effects of the proposed cyanide criteria on listed species and critical habitats at a more refined scale that incorporates the four elements of the State's or Tribe's water quality standards as approved. As noted above, it is at this scale that we will determine if the effects analysis includes findings that support an incidental take exemption(s).

## 3.0 CONSULTATION HISTORY

On February 22, 2001, the Service and the National Marine Fisheries Service (NMFS; collectively, the Services), and EPA noticed in the Federal Register the 2001, Memorandum Of Agreement (MOA) referenced above which, among other things, described our plan for conducting ESA section 7 consultations on EPA's recommended aquatic life criteria.

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In January 2004, the Services and EPA decided to proceed with a data call for the first batch of pollutants that would be reviewed in consultation, while continuing to work on the *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria--Methods Manual.*

On May 14, 2004, the Services and EPA issued data calls to regional staff and NMFS science center staff requesting information and data on cyanide, ammonia, chromium III and chromium VI. The data call requested the regions and science centers to send relevant studies to our Headquarters Offices by June 30, 2004.

On November 12, 2004, the Services received by email from EPA a November 5, 2004, revised *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria--Methods Manual* (dated October 29, 2004, on the document). This version represented a methodology developed collaboratively, and which had been peer reviewed by subject experts outside of the Federal government.

In December 2004, the Service and EPA exchanged comments on recommended revisions to the November draft methodology. EPA also informed the Services that they had received a draft BE for cyanide from their contractor and were reviewing the document to ensure the contractor had followed the BE methodology accurately.

On January 19, 2005, EPA transmitted by email the January 19, 2005, *Biological Evaluation of Aquatic Life Criteria—Cyanide, Part 1: Toxicity Analysis and Preliminary Effects Assessment*.

On May 3, 2005, the Services transmitted by email comments on EPA's January 19, 2005, draft biological evaluation for cyanide criteria.

On January 31, 2006, the Service received a January 26, 2006, letter from EPA requesting the Service to review a January 26, 2006, draft CN BE for its "completeness" in fulfilling the information requirements for ESA section 7 consultation.

On April 18, 2006, the Service transmitted a letter to EPA responding to EPA's January 26, 2006, consultation request.

In a June 29, 2006, letter, EPA requested the Service's concurrence that the proposed approvals of cyanide criteria were not likely to adversely affect 455 threatened and endangered species and their critical habitats.

On August 1, 2006, EPA electronically transmitted a July 31, 2006, *Draft Framework for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual*, which they used to support their effects determinations.

On November 28, 2006, the Service responded by letter to EPA's consultation request with a recommendation that EPA initiate formal consultation.

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On March 23, 2007, EPA requested formal consultation and provided the CN BE, which concluded their action was not likely to adversely affect listed species or designated critical habitats.

On May 29, 2007, the Service transmitted a letter acknowledging the initiation of formal consultation.

On August 6, 2007, the Service transmitted a letter to EPA requesting an extension of formal consultation to November 6, 2007.

On August 17, 2007, the Service received an August 15, 2007, letter from EPA offering to extend the consultation period to September 30, 2007.

On May 5-9, 2008, the Services met with EPA to conduct a “Kaizen” “lean event.” The purpose of the meetings was to analyze the cyanide consultation process from the development of a biological assessment through the anticipated completion of formal consultation in an effort to find efficiencies in the process. The Services and EPA also discussed coordination and communication with respect to the national consultation on cyanide and local consultation on EPA promulgation of Oregon water quality standards.

On June 12, 2008, the Services and EPA met to follow up on the Kaizen lean event. Subsequent follow up meetings were cancelled until the Services completed draft biological opinions.

## BIOLOGICAL OPINION

This biological opinion does not rely on the regulatory definition of “destruction or adverse modification” of critical habitat at 50 C.F.R. 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat.

## 4.0 DESCRIPTION OF THE PROPOSED ACTION

*Background*

Section 304(a)(1) of the CWA directs EPA to publish criteria for water quality that accurately reflect the latest scientific knowledge on a number of factors, including “…the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches…which may be expected from the presence of pollutants in any water body, including groundwater;” “on the concentration and dispersal of pollutants, or their byproducts,” and on “the effects of pollutants on biological community diversity, productivity, and stability.” EPA’s CWA section 304(a) cyanide aquatic life criteria recommendations are published in *Ambient Water Quality Criteria for Cyanide – 1984* (EPA 1985). EPA describes the published criteria as a criterion maximum concentration (CMC) and criterion continuous concentration (CCC) for freshwater and saltwater:

Freshwater CMC (as free cyanide) = 22 µg/L
Freshwater CCC (as free cyanide) = 5.2 µg/L
Saltwater CMC (as free cyanide) = 1.0 µg/L
Saltwater CCC (as free cyanide) = 1.0 µg/L

For cyanide, the CMC (“acute” criterion) represents an estimated concentration in fresh or salt water to which aquatic organisms and their uses should not be affected unacceptably if the one-hour average concentration does not exceed this value more than once every three years on average, except possibly where locally important species are more sensitive (EPA 1985). The CCC (“chronic” criterion) represents an estimated concentration in either fresh or salt water to which aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration does not exceed the CCC more than once every three years on average, except possibly where locally important species are more sensitive (EPA 1985).

Section 303(c)(2)(B) of the CWA requires States, Tribes, and U.S. territories (hereafter referred to collectively as States as defined by EPA in 40 CFR 131.3(j)) to adopt into their water quality standards numeric criteria for toxic pollutants listed under section 307(a) of the CWA for which section 304(a) criteria have been published if the presence of these pollutants is likely to affect a water body’s use. States can adopt criteria under section 303(c) that differ from EPA’s 304(a) criteria values whenever adequately justified, but States generally choose to adopt, verbatim, EPA’s 304(a) criteria and rely on the criteria document for their scientific justification. Once adopted into State or Tribal water quality standards, criteria help form the legal basis for implementing CWA programs to control pollution and achieve the goals and requirements of the CWA.

*Proposed Program Action*

EPA would approve State or Tribal-adopted water quality standards for cyanide criteria, or promulgate Federal water quality standards for cyanide criteria for U.S. waters that are identical to or more stringent than EPA’s recommended CWA section 304(a) aquatic life criteria for cyanide (EPA 1985). EPA’s cyanide criteria document (EPA 1985) serves as the States’, Tribes’ and EPA’s scientific justification that the recommended criteria values are sufficient to achieve designated uses that protect aquatic life under the CWA.

All States and Tribes have already either adopted into their state water quality standards cyanide criteria identical to EPA’s recommended 304(a) cyanide aquatic life criteria (EPA 1985) and EPA has approved them, or EPA promulgated cyanide criteria for the States and Tribes identical to their 304(a) criteria. For some water bodies, States and Tribes have adopted and EPA has approved site-specific criteria less stringent than the CWA 304(a) criteria, but EPA has not included continuing approval of less stringent criteria as part of the proposed action considered in this consultation.

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This biological opinion does not evaluate or provide ESA section 7 compliance for past EPA actions involving cyanide criteria for water quality standards or the past, present, and future effects of those actions on listed species and critical habitats. Past actions and their effects are relevant to describing the Status and Environmental Baseline of listed species and critical habitats. In this biological opinion the Service evaluates the proposed action in the context of EPA's continuing oversight role under the CWA and its discretion and authority to approve or promulgate State and Tribal water quality standards and affect implementation of CWA programs to achieve the goals and requirements of the CWA and the requirements of section 7(a)(2) of the ESA.

*Interrelated and Interdependent Activities*

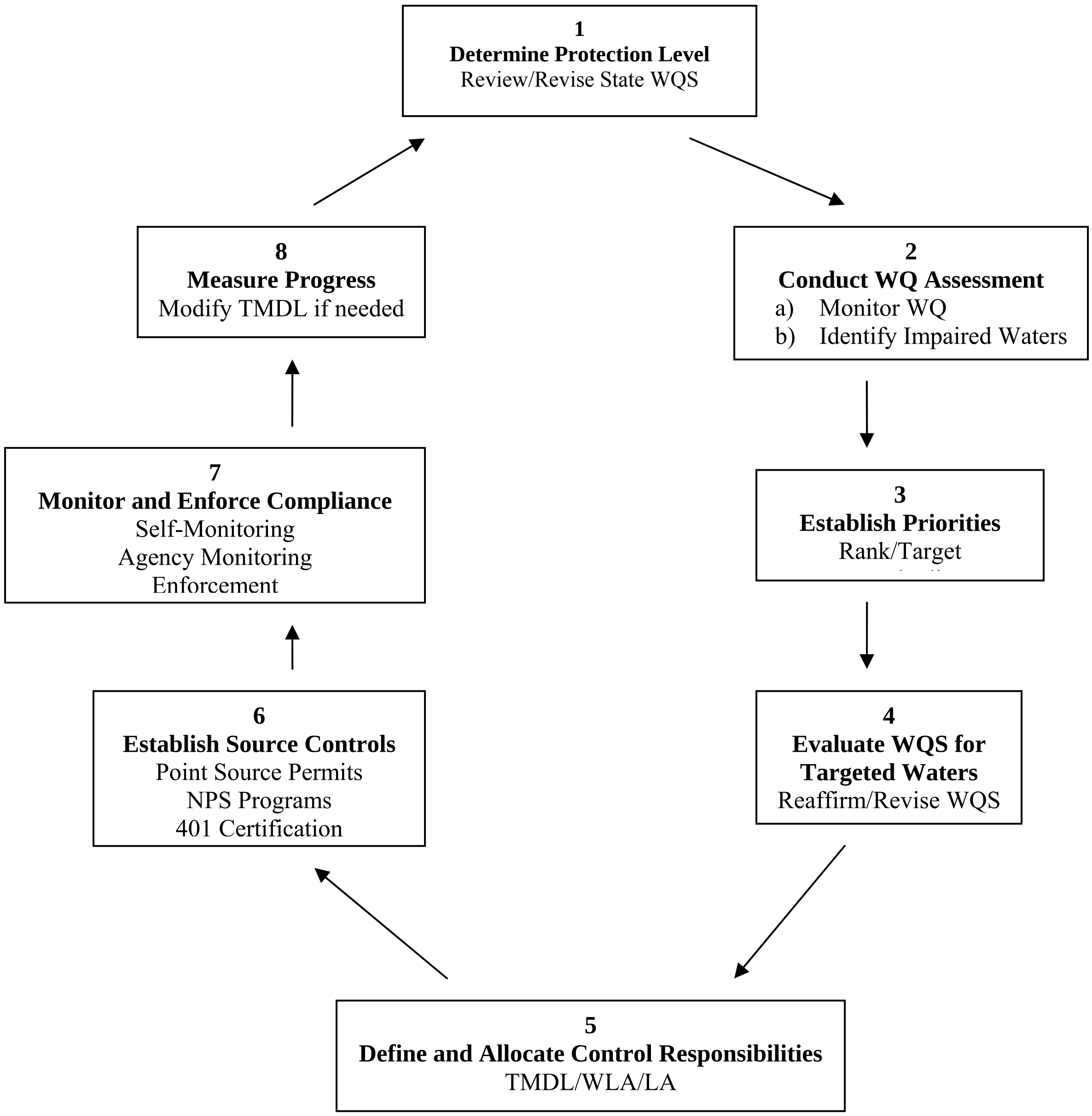
The effects of EPA's approval or promulgation of cyanide criteria in State and Tribal water quality standards must be understood in the larger context of the CWA. This larger context is framed by Congress' stated objective, goals, and policies of the CWA and the programs and activities authorized by the CWA and implemented by EPA, States, Tribes and local governments.

The objective of the CWA is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters (CWA section 101(a)). In order to achieve this objective, the CWA articulates goals to ultimately eliminate the discharge of pollutants into navigable waters, with an interim goal of water quality that provides for the protection and propagation of fish, shellfish, and wildlife. Congress also articulated in section 101 of the CWA its policy to prohibit the discharge of pollutants in toxic amounts, and its policy that programs be developed and implemented so that the goals of the CWA could be met through the control of point and nonpoint sources of pollution.

Water quality standards are the States' and Tribes' goals for individual water bodies and provide the regulatory basis for control decisions under the CWA (40 CFR 130.0(b)). States, Tribes and EPA plan and implement CWA programs to manage point and non-point sources of pollution to attain the goals established in State and Tribal water quality standards. Control measures are implemented by issuing permits, building publicly-owned treatment facilities, instituting best management practices for nonpoint sources of pollution, and other means.

In the Water Quality Standards Handbook (EPA 1994), EPA illustrates how the various elements of the CWA and its regulations work together to manage pollution. Figure 1, reproduced from the handbook, describes the eight stages of the water quality-based approach to pollution control. Each stage represents a major CWA program with specific regulatory requirements and guidance. This figure is intended to illustrate how the different programs fit into the overall water quality control scheme.

**Figure 1. Eight stages of EPA's approach to water quality-based pollution control.**




1. **Determine Protection Level**. States and Tribes adopt, and EPA approves, water quality standards to protect public health or welfare, enhance the quality of water, and serve the purposes of the CWA. A water quality standard defines the water quality goals of a water body, or portion thereof, by designating a use or uses, by setting criteria necessary to protect the uses, and by preventing degradation of water quality through antidegradation provisions. States and Tribes may, at their discretion, include general policies which allow for exceedances in discharges

under specific circumstances. Development and review/revision of standards is directed by the Water Quality Standards Regulation (40 CFR part 131), described in greater detail below.

2. **Conduct Water Quality Assessment**. Once State and Tribal water quality standards establish the level of protection to be afforded to a water body, States and Tribes are required to conduct water quality monitoring to identify waters not meeting the standards. Section 305(b) of the CWA requires States and Tribes to prepare a water quality inventory every 2 years to document the status of water bodies that have been assessed. Under section 304(l), States and Tribes identify all surface waters adversely affected by toxic, conventional, and nonconventional pollutants from point and non-point sources. Under section 314(a), States and Tribes identify publicly-owned lakes for which uses are known to be impaired by point and non-point sources. Section 319 requires States and Tribes to perform nonpoint source (NPS) assessments of navigable waters, including the identification of impaired and threatened waters and the activities causing impairment. NPS assessment reports and management programs are subject to EPA approval and oversight. The collective assessment efforts contribute to the States' and Tribes' section 303(d) listing of waters for which effluent limitations and other pollution control requirements are not stringent enough to implement a water quality standard.

3. **Establish Priorities**. Once waters needing additional controls have been identified, States and Tribes are required to submit for EPA review their priority rankings of waters in need of total maximum daily load (TMDL) development.

4. **Evaluate WQS for Targeted Waters**. At this point in the water quality management process, States and Tribes have targeted priority water quality-limited water bodies. EPA recommends that States and Tribes re-evaluate the appropriateness of the water quality standards for the targeted waters if: 1) States or Tribes have not conducted in-depth analyses of appropriate uses and criteria; 2) changes in the uses of the water body may require changes in the standard; 3) more recent water quality monitoring show the standard is being met; and, 4) site-specific criteria may be appropriate because of specific local environmental conditions or the presence of species more or less sensitive than those included in the national criteria data set.

5. **Define and Allocate Control Responsibilities**. For water quality limited waters, States and Tribes must establish a TMDL that quantifies pollutant sources, and a margin of safety, and allocates allowable loads to the contributing point and non-point source discharges so that the water quality standards are attained. EPA recommends States and Tribes develop TMDLs on a watershed basis.

6. **Establish Source Controls**. Once a TMDL has been established and the appropriate source loads developed, implementation should proceed. The first step is to update the "water quality management plan," described below. Next, point

and nonpoint source controls should be implemented to meet waste load allocations and load allocations, respectively. The NPDES permitting process is used to limit effluent from point sources. Construction decisions regarding publicly-owned treatment works must also be based on the more stringent of technology-based or water quality-based limitations. In the case of nonpoint sources, State, Tribal and local laws may authorize the implementation of nonpoint source controls, such as best management practices (BMPs) or other management measures.

7. **Monitor and Enforce Compliance**. Monitoring is essential to water quality-based decision making. Point source dischargers are required to provide reports on compliance with NPDES permit limits. A monitoring requirement can be put into the permit as a special condition as long as the information is collected for the purposes of writing a permit limit. Effective monitoring programs are also required for evaluating nonpoint source control measures and EPA provides guidance in implementing and evaluating nonpoint source control measures. EPA and States and Tribes are authorized to bring civil or criminal action against facilities that violate their NPDES permits. State nonpoint source programs are enforced under State law and to the extent provided by State law.

8. **Measure Progress**. If the water body achieves the applicable State or Tribal water quality standards it may be removed from the section 303(d) list of waters needing TMDLs. If water quality standards are not met, the TMDL and allocation of load and waste loads must be modified.

The water quality-based approach to pollution control is implemented by EPA, and the States and Tribes consistent with specific statutory and regulatory requirements. The regulatory requirements are articulated in: the *Water Quality Planning and Management* regulation (40 CFR 130) (50 FR 1779, January 11, 1985, as amended 54 FR 14359 April 11, 1989; 57 FR 33049, July 24, 1992; 59 FR 13817, March 23, 1994; 65 FR 17170, March 31, 2000; and, 66 FR 53048, October 18, 2001); and the *Water Quality Standards* regulation (40 CFR 131)(48 FR 51405, November 8, 1983, as amended 56 FR 64893, December 12, 1991; 59 FR 64344, December 14, 1994; 60 FR 15386, March 23, 1995; 65 FR 24653, April 27, 2000). Separately, the *Water Quality Guidance for the Great Lakes System* regulation (40 CFR 132)(60 FR 15387, March 23, 1995, as amended 65 FR 47874, August 4, 2000; 65 FR 59737, October 6, 2000; 65 FR 66511, November 6, 2000; 65 FR 67650, November 13, 2000) constitutes the guidance for States in the Great Lakes system; and other regulations promulgated by EPA.

State Water Quality Standards

The CWA directs States and authorized Tribes to adopt water quality standards for all their waters and submit their standards to EPA for review and approval or disapproval (of all or part)(CWA section 303(a)(1)-(2), (c)). The CWA further requires that a State shall at least once every three years hold public hearings for the purpose of reviewing applicable water quality standards and, as appropriate, modifying and adopting standards (i.e., conduct triennial reviews). Consultation with EPA is one of the first steps States and Tribes take when beginning the triennial review process (EPA 1994). The triennial review process is a forum through which EPA would review and approve or disapprove proposed revisions to State and Tribal water quality standards.

By statute, State and Tribal water quality standards consist of the designated uses of waters and the water quality criteria needed to achieve the designated uses. Section 303(d) and 118(c) of the CWA requires States and Tribes to establish an antidegradation policy. The *Water Quality Standards Regulation* (40 CFR part 131) defines the regulatory requirements of the water quality standards program and adds a provision for States and Tribes to adopt general policies that affect the application and implementation of State and Tribal water quality standards. For example, States and Tribes may adopt policies concerning mixing zones, water quality standards variances, and critical flows for water quality-based permit limits. General policies provide a mechanism for permitting a discharge where numeric water quality criteria may be exceeded, or low flows below which criteria do not apply, or where a standard is not being attained, a variance from the standard would enable permit issuance as long as progress towards attainment is being made. The designated uses, criteria, antidegradation policy, and general policies, together, define the level of protection afforded to a water body or portion thereof under the CWA.

In a January 27, 2005, memorandum (EPA 2005) EPA concluded that ESA section 7 consultation does not apply to EPA's approvals of State and Tribal antidegradation policies because EPA's approval action does not meet the "Applicability" standard defined in the regulations implementing section 7 of the ESA (50 CFR 402.03). Section 402.03 of the ESA section 7 consultation regulations (50 CFR part 402) states that section 7 and the requirements of 50 CFR part 402 apply to all actions in which there is discretionary Federal involvement or control. EPA concluded that they are compelled to approve State and Tribal antidegradation policies if State or Tribal submissions meet all applicable requirements of the *Water Quality Standards Regulation* (40 CFR part 131) and lack discretion to implement measures that would benefit listed species. Consequently, in its analyses the Service can not rely on antidegradation policies in State and Tribal water quality standards to provide protection over and beyond approved water quality criteria. EPA will treat new or revised antidegradation policies as being inapplicable to consultation and thus existing antidegradation policies can be changed by States or Tribes independent of any review of existing water quality criteria.

Water Quality Monitoring

States and Tribes are required to establish water quality monitoring, including collection and analysis of physical, chemical, and biological data to determine abatement and control priorities; developing and reviewing water quality standards, total maximum daily loads, wasteload allocations and load allocations; assessing compliance with NPDES permits by dischargers; reporting information to the public through the section 305(b) report and reviewing site-specific monitoring efforts (40 CFR 130.4).

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Continuing Planning Process

Section 303(e) of the CWA requires States and Tribes to have in place a "continuing planning process" (CPP) approved by EPA and requires EPA to periodically review a State's or Tribe's planning process for conformity to the requirements of the CWA. Section 303(e)(3) states that EPA shall approve any CPP that will result in plans for all navigable waters that include, but are not limited to, eight elements. EPA's regulations implementing this section (40 CFR 130) add a ninth element:

1. effluent limitations and schedules of compliance at least as stringent as those required by section 301(b)(1), section 301(b)(2), section 306, and section 307, and at least as stringent as any requirements contained in any applicable water quality standard in effect under section 303;
2. the incorporation of all elements of any applicable areawide waste management plans (CWA section 208) and applicable basin plans (CWA section 209);
3. total maximum daily load for pollutants in accordance with section 303(d);
4. procedures for revision;
5. adequate authority for intergovernmental cooperation;
6. adequate implementation, including schedules of compliance, for revised or new water quality standards under CWA section 303(c);
7. controls over the disposition of all residual waste from any water treatment processing;
8. an inventory and priority ranking of need for waste treatment works to meet applicable requirements of CWA sections 301 and 302; and,
9. a process for determining the priority of permit issuance.

By statute and regulation, an approved CPP is a prerequisite for EPA's approval of a permitting program under Title IV of the CWA.

Water Quality Management Plans

The CWA and its implementing regulations (40 CFR 130) require States and Tribes to establish Water Quality Management (WQM) Plans to prioritize and direct implementation of CWA programs. The following are required elements of State or Tribal WQM Plans:

1. total maximum daily loads;
2. effluent limitations;
3. municipal and industrial waste treatment;
4. nonpoint source management and control;
5. identification of management agencies capable of carrying out responsibilities;
6. implementation measures necessary to carry out the plan;
7. programs for the control of dredge or fill material;
8. identification of relationship to basin plans developed under CWA section 209;
9. identification of programs for control of ground water pollution.

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WQM Plans are produced and regularly updated on the basis of water quality assessments. Consistency with WQM Plans is a requirement for issuance of permits under CWA section 402 and grants made under the municipal construction grants program. States and Tribes must certify by letter for EPA approval that WQM Plan updates are consistent with all other parts of the plan (40 CFR 130).

In summary, the effects of EPA's approval or promulgation of cyanide criteria in State or Tribal water quality standards must be understood in the larger context of the CWA. This larger context is framed by Congress' stated objective, goals, and policies of the CWA and the programs and activities authorized by the CWA and implemented by EPA, States, Tribes and local governments to achieve these objectives, goals, and policies.

State water quality standards establish the level of protection afforded to water bodies or water body segments. Water quality standards include the designated uses assigned to water bodies, criteria to achieve the designated uses, an antidegradation policy to protect higher quality waters, and general policies which provide for exceedances of criteria values in specified circumstances. The approval or promulgation of cyanide criteria is either interrelated to and/or interdependent with the other elements of State or Tribal water quality standards. The proposed action analyzed in this biological opinion must be compared with the water quality standards as they are approved for individual States and Tribes. Separate, tiered consultations would be necessary to address discrepancies and quantify and exempt incidental take, as appropriate.

The programs of the water quality-based approach to pollution control are targeted to assessing compliance with standards and instituting changes to achieve compliance through modifications to allowable discharges or to the standards themselves. The adaptive management of water quality, articulated in the water quality-based approach to pollution control, is driven by continuing planning processes and water quality management planning and implementation. Because the planning and implementation of CWA programs by EPA, States, Tribes and local governments is targeted towards achieving State or Tribal water quality standards, these activities are necessarily part of a larger action, and thus interrelated to the approval of criteria in State or Tribal water quality standards.

## 5.0 ACTION AREA

The action area for this consultation consists of all waters subject to the CWA, including "territorial seas," which extend seaward a distance of three miles from the coast, and habitats effected by administration of the CWA. The action area also includes all other parts of watersheds that serve as sources to waters subject to the CWA where activities contribute to point and non-point sources of pollution and are subject to compliance with the CWA. This action area includes such waters and habitats within and surrounding Indian country, the 50 States, and all U.S. territories.

## 6.0 STATUS OF THE SPECIES AND CRITICAL HABITAT

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Information on the status of affected listed species and critical habitat is presented in two parts. The species and critical habitats considered and the states encompassing their distribution are listed in Table 1, and individual species and critical habitat accounts are presented in Appendix A. Because the action area for this consultation encompasses the entire range of affected listed species and critical habitats, no Environmental Baseline section is needed or presented.

In the Status of the Species and Critical Habitat section of a biological opinion the Service presents biological or ecological information relevant to formulating the opinion, and characterizes the current condition of the species or critical habitat, the factors responsible for that condition, and the survival and recovery needs of the species and the intended conservation function of critical habitat. This information is derived from listing documents and recovery plans or subsequent biological opinions as new information becomes available. The Status of the Species and Critical Habitat evaluation establishes key biological context for evaluating the effects of a proposed Federal action and any cumulative effects for purposes of making an ESA section 7(a)(2) determination(s).

Aquatic life criteria address the CWA goals and policy of attaining water quality that provides for the protection and propagation of fish, shellfish, and wildlife, and prohibiting the discharge of toxic pollutants in toxic amounts (CWA Sec. 101(a)(2) & (3)). The CWA policy of prohibiting the discharge of toxic pollutants in toxic amounts provides a foundation for characterizing EPA's discretion in recommending aquatic life cyanide criteria under CWA section 304(a) and approval of cyanide criteria in state water quality standards. Although most approved water quality standards include a statement prohibiting the discharge of toxic pollutants in toxic amounts, the criteria by which this standard presumably is enforced, the CCC and the CMC, are defined in such a way that adverse effects are limited but not prohibited. \

To fully assess the impacts of the proposed action, it is important to consider the status of the species and the current state of their aquatic environments as this provides a context for the jeopardy analysis. Although the baseline condition includes the past and present impacts of activities in the action area, it does not include the future impacts of the action under review in this consultation. Because the action area for this consultation (i.e. all waters of the U.S., including territories) encompasses the entire ranges for the affected listed species and critical habitats, it is difficult to fully assess 1) the past and present impacts of all Federal, State, or private actions and other human activities in the action area; 2) the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation; and 3) the impact of State or private actions occurring simultaneously with this consultation for each species included in this opinion. As such, we will broadly describe the status of the aquatic environments within the action area as a means for establishing a baseline condition from which to analyze the effect of the action.

Section 304(a)(1) of the CWA requires EPA to develop criteria for water quality for the protection of aquatic life as well as for human health. EPA develops these criteria as numeric limits on the amounts of chemicals that can be present in river, lake, or stream water to protect aquatic organisms from death, slower growth, reduced reproduction, and the accumulation of harmful levels of toxic chemicals in their tissues that may also adversely affect consumers of such organisms. Aquatic life criteria have been developed for almost 50 pollutants to date and serve as a basis for the development of state water quality standards. Waters that do not meet these standards are considered to be "impaired" under section 303(d) of the CWA. The most recent National Water Quality Inventory Report to Congress (EPA 2009) presents a summary finding of the 2004 state water quality reports covering 16% of the nation's 3.5 million miles of rivers and streams, 39% of the nation's 41.7 million acres of lakes, ponds, and reservoirs, and 29% of the nation's 87,791 square miles of bays and estuaries. For waters included in the assessment, 44% of the rivers and streams, 64% of the lakes ponds and reservoirs, and 29% of the bays and estuaries were reported as impaired or not clean enough to support their designated uses.

The sources of degradation vary by water type but included pathogens, habitat alterations, mercury, and organic enrichment/oxygen depletion from both known and unknown or unspecified sources. An assessment by the U.S. Geological Survey (USGS) on the presence and concentration of pesticide in the nation's streams and ground water found that concentrations of pesticides were frequently greater than water-quality benchmarks for aquatic life and fish eating wildlife. Specifically, USGS found that 57% of 83 agricultural streams, 83% of 30 urban streams, and 42% of 65 mixed-land-use streams had concentrations of at least one pesticide that exceeded one or more aquatic-life benchmarks at least one time during the year. The most frequently detected pesticide compounds included atrazine, deethylatrazine, and metolachlor (USGS 2006).

Over 83,000 chemicals are currently listed in the Toxic Substances Control Act Inventory and more than 87,000 chemicals including pesticides, commodity chemicals, naturally occurring non-steroidal estrogens, food additives, cosmetics, nutritional supplements, and representative mixtures have been identified as potential endocrine disruptors by the Endocrine Disruptor Screening and Testing Advisory Committee (1998) convened by EPA. For most of these chemicals, scientific data is insufficient or unavailable for evaluating their potential individual and combined impacts on aquatic environments.

**Table 1. Listed species and critical habitats considered in this biological opinion along with their listing status and distribution on a state-by-state basis. E=endangered; T=threatened; CH=critical habitat; XN=(non-essential experimental population).**

| Common Name | Scientific Name | Status | Distribution |
|---|---|---|---|
| | **FISH** | | |
| | ***Acipenseridae*** | | |
| Gulf sturgeon | *Acipenser oxyrinchus desotoi* | T, CH | AL, FL, LA, MS |
| Kootenai River white sturgeon | *A. transmontanus* | E, CH | ID, MT |
| Pallid sturgeon | *Scaphirhynchus albus* | E | AR, IA, IL, KS, KY, LA, MO, MS, MT, ND, NE, SD, TN |

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| Common Name | Scientific Name | Status | Distribution |
|---|---|---|---|
| Alabama sturgeon | *S. suttkusi* | E, CH | AL, MS |
| | ***Amblyopsidae*** | | |
| Ozark cavefish | *Amblyopsis rosae* | T | AR, MO, OK |
| Alabama cavefish | *Speoplatyrhinus poulsoni* | E, CH | AL |
| | ***Atherinidae*** | | |
| Waccamaw silverside | *Menidia extensa* | T, CH | NC |
| | ***Catostomidae*** | | |
| Modoc sucker | *Catostomus microps* | E, CH | CA, OR |
| Santa Ana sucker | *Catostomus santaanae* | T, CH | CA |
| Warner sucker | *Catostomus warnerensis* | T, CH | OR |
| Shortnose sucker | *Chasmistes brevirostris* | E | CA, OR |
| Cui-ui | *Chasmistes cujus* | E | NV |
| June sucker | *Chasmistes liorus* | E, CH | UT |
| Lost River sucker | *Deltistes luxatus* | E | CA, OR |
| Razorback sucker | *Xyrauchen texanus* | E, CH | AZ, CA, CO, NM, NV, UT |
| | ***Cotidae*** | | |
| Pygmy sculpin | *Cottus paulus* | T | AL |
| | ***Cyprinidae*** | | |
| Blue shiner | *Cyprinella caerulea* | T | AL, GA, TN |
| Beautiful shiner | *Cyprinella Formosa* | T, CH | AZ, NM |
| Devils River minnow | *Dionda diaboli* | T, CH | TX |
| Spotfin chub | *Erimonax monachus* | T, CH, XN | AL, NC, TN, VA |
| Slender chub | *Erimystax cahni* | T, CH, XN | TN, VA |
| Mojave tui chub | *Gila bicolor mohavensis* | E | CA |
| Owens tui chub | *Gila bicolor snyderi* | E, CH | CA |
| Borax Lake chub | *Gila boraxobius* | E, CH | OR |
| Humpback chub | *Gila cypha* | E, CH | AZ, CO, UT |
| Sonora chub | *Gila ditaenia* | T, CH | AZ |
| Bonytail chub | *Gila elegans* | E, CH | AZ, CA, CO, NV, UT |
| Gila chub | *Gila intermedia* | E, CH | AZ, NM |
| Yaqui chub | *Gila purpurea* | E, CH | AZ |
| Pahranagat roundtail chub | *Gila robusta jordani* | E | NV |
| Virgin River Chub | *Gila seminuda (=robusta)* | E, CH | AZ, NV, UT |
| Rio Grande silvery minnow | *Hybognathus amarus* | E, CH | NM, TX |
| Big Spring spinedace | *Lepidomeda mollispinis pratensis* | T, CH | NV |
| Little Colorado spinedace | *Lepidomeda vittata* | T, CH | AZ |
| Spikedace | *Meda fulgida* | T, CH | AZ, NM |
| Moapa dace | *Moapa coriacea* | E | NV |
| Palezone shiner | *Notropis albizonatus* | E | AL, KY |
| Cahaba shiner | *Notropis cahabae* | E | AL |
| Arkansas River shiner | *Notropis girardi* | T, CH | AR, KS, NM, OK, |

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| Common Name | Scientific Name | Status | Distribution |
|---|---|---|---|
| | | | TX |
| Cape Fear shiner | *Notropis mekistocholas* | E, CH | NC |
| Pecos bluntnose shiner | *Notropis simus pecosensis* | T, CH | NM |
| Topeka shiner | *Notropis topeka (=tristis)* | E, CH | IA, KS, MN, MO, NE, SD |
| Oregon chub | *Oregonichthys crameri* | E | OR |
| Blackside dace | *Phoxinus cumberlandensis* | T | KY, TN |
| Woundfin | *Plagopterus argentissimus* | E; CH, XN | AZ, NM, NV, UT |
| Colorado pikeminnow (=squawfish) | *Ptychocheilus lucius* | E; CH, XN | AZ, CA, CO, NM, UT |
| Ash Meadows speckled dace | *Rhinichthys osculus nevadensis* | E, CH | NV |
| Kendall Warm Springs dace | *Rhinichthys osculus thermalis* | E | WY |
| Loach minnow | *Tiaroga cobitis* | T, CH | AZ, NM |
| | ***Gasterosteidae*** | | |
| Unarmored threespine stickleback | *Gasterosteus aculeatus williamsoni* | E | CA |
| | ***Gobiidae*** | | |
| Tidewater goby | *Eucyclogobius newberryi* | T, CH | CA |
| | ***Goodeidae*** | | |
| White River springfish | *Crenichthys baileyi baileyi* | E, CH | NV |
| Hiko White River springfish | *Crenichthys baileyi grandis* | E, CH | NV |
| Railroad Valley springfish | *Crenichthys nevadae* | T, CH | NV |
| | ***Osmeridae*** | | |
| Delta smelt | *Hypomesus transpacificus* | T, CH | CA |
| | ***Percidae*** | | |
| Slackwater darter | *Etheostoma boschungi* | T, CH | AL, TN |
| Vermilion darter | *Etheostoma chermocki* | E | AL |
| Relict darter | *E. chienense* | E | KY |
| Etowah darter | *E. etowahae* | E | GA |
| Fountain darter | *E. fonticola* | E, CH | TX |
| Niangua darter[1] | *E. nianguae* | T, CH | MO |
| Watercress darter | *E. nuchale* | E | AL |
| Okaloosa darter | *E. okaloosae* | E | FL |
| Duskytail darter | *E. percnurum* | E, XN | KY, TN, VA |
| Bayou darter | *E. rubrum* | T | MS |
| Cherokee darter | *E. scotti* | T | GA |
| Maryland darter | *E. sellare* | E, CH | MD |
| Bluemask darter | *E. sp.* | E | TN |
| Boulder darter | *E. wapiti* | E, XN | AL, TN |
| Amber darter | *Percina antesella* | E, CH | GA, TN |
| Goldline darter | *P. aurolineata* | T | AL, GA |

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| Common Name | Scientific Name | Status | Distribution |
|---|---|---|---|
| Conasauga logperch | *P. jenkinsi* | E, CH | GA, TN |
| Leopard darter | *P. pantherina* | T, CH | AR, OK |
| Roanoke logperch | *P. rex* | E | VA, NC |
| Snail darter | *P. tanasi* | T, CH | AL, GA, TN |
| | ***Poeciliidae*** | | |
| Big Bend gambusia | *Gambusia gaigei* | E | TX |
| San Marcos gambusia | *Gambusia georgei* | E, CH | TX |
| Clear Creek gambusia | *Gambusia heterochir* | E | TX |
| Pecos gambusia | *Gambusia nobilis* | E | NM, TX |
| Gila topminnow | *Poeciliopsis occidentalis* | E | AZ, NM |
| | ***Salmonidae*** | | |
| Bull trout | *Salvelinus confluentus* | T, CH | CA, ID, MT, NV, OR, WA |
| Little Kern Golden trout | *Oncorhynchus aquabonita whitei* | T, CH | CA |
| Apache trout | *O. apache* | T | AZ |
| Lahontan Cutthroat trout | *O. clarkii henshawi* | T | CA, NV, OR, UT |
| Paiute Cutthroat trout | *O. clarkii seleniris* | T | CA |
| Greenback Cutthroat Mountain trout | *O. clarkii stomias* | T | CO |
| Gila trout | *O. gilae* | T | AZ, NM |
| Atlantic salmon | *Salmo salar* | E | ME |
| | | | |
| | **AMPHIPODS** | | |
| Illinois cave amphipod | *Gammarus acherondytes* | E | IL |
| Noel's amphipod | *G. desperatus* | E | NM |
| | | | |
| | **MUSSELS** | | |
| Cumberland elktoe | *Alasmidonta atropurpurea* | E, CH | KY, TN |
| Dwarf wedgemussel | *A. heterodon* | E | CT, MA, MD, NC, NH, NJ, NY, PA, VA, VT |
| Appalachian elktoe | *A. raveneliana* | E, CH | NC, TN |
| Fat three-ridge | *Amblema neislerii* | E, CH | FL, GA |
| Ouachita rock pocketbook | *Arkansia wheeleri* | E | AR, OK |
| Birdwing pearlymussel | *Conradilla caelata* | E, XN | AL, TN, VA |
| Fanshell | *Cyprogenia stegaria* | E, XN | AL, IL, IN, KY, OH, TN, VA, WV |
| Dromedary pearlymussel | *Dromus dromas* | E, XN | AL, KY, TN, VA |
| Chipola slabshell | *Elliptio chipolaensis* | T, CH | AL, FL |
| Tar River spinymussel | *E. steinstansana* | E | NC |
| Purple bankclimber | *Elliptoideus sloatianus* | T, CH | FL, GA |
| Cumberlandian combshell | *Epioblasma brevidens* | E, XN, CH | AL, KY, TN, VA |
| Oyster mussel | *E. capsaeformis* | E, XN, CH | AL, GA, KY, NC, TN, VA |

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| Common Name | Scientific Name | Status | Distribution |
|---|---|---|---|
| Curtis pearlymussel | *(E. florentina curtisii)* | E | AR, MO |
| Yellow blossom | *E. florentina florentina* | E, XN | AL, TN |
| Tan riffleshell | *E. florentina walkeri* | E | KY, TN, VA |
| Upland combshell | *E. metastriata* | E, CH | AL, GA, TN |
| Catspaw | *E. obliquata obliquata* | E, XN | AL, IL, IN, KY, OH, TN |
| White catspaw | *E. o. perobliqua* | E | IN, OH |
| Southern acornshell | *E. othcaloogensis* | E, CH | AL, GA, TN |
| Southern combshell | *E. penita* | E | AL, MS |
| Green blossom | *E. torulosa gubernaculums* | E | TN, VA |
| Northern riffleshell | *E. t. rangiana* | E | IN, KY, MI, OH, PA, WV |
| Tubercled blossom | *E.a torulosa torulosa* | E, XN | AL, IL, IN, KY, TN, WV |
| Turgid blossom | *Epioblasma turgidula* | E, XN | AL, TN |
| Shiny pigtoe | *Fusconaia cor* | E, XN | AL, TN, VA |
| Finerayed pigtoe | *Fusconaia cuneolus* | E, XN | AL, TN, VA |
| Cracking pearlymussel | *Hemistena lata* | E, XN | AL, IN, KY, PA, TN, VA |
| Pink mucket | *Lampsilis abrupta* | E | AL, AR, IL, IN, KY, LA, MO, OH, PA, TN, WV |
| Fine-lined pocketbook | *Lampsilis altilis* | T, CH | AL, GA, TN |
| Higgins eye | *Lampsilis higginsii* | E | IA, IL, MN, MO, WI |
| Orangenacre mucket | *Lampsilis perovalis* | T, CH | AL, MS |
| Arkansas fatmucket | *Lampsilis powelli* | T | AR |
| Speckled pocketbook | *Lampsilis streckeri* | E | AR |
| Shinyrayed pocketbook | *Lampsilis subangulata* | E, CH | AL, FL, GA |
| Alabama lampmussel | *Lampsilis virescens* | E, XN | AL, TN |
| Carolina heelsplitter | *Lasmigona decorata* | E, CH | NC, SC |
| Scaleshell mussel | *Leptodea leptodon* | E | AR, MO, OK |
| Louisiana pearlshell | *Margaritifera hembeli* | T | LA |
| Alabama moccasinshell | *Medionidus acutissimus* | T, CH | AL, GA, MS |
| Coosa moccasinshell [1] | *Medionidus parvulus* | E, CH | GA, TN |
| Gulf moccasinshell | *Medionidus penicillatus* | E, CH | FL, GA |
| Ochlockonee moccasinshell | *Medionidus simpsonianus* | E, CH | FL, GA |
| Ring pink | *Obovaria retusa* | E, XN | AL, IN, KY, PA, TN |
| Littlewing pearlymussel | *Pegias fibula* | E | KY, NC, TN, VA |
| White wartyback pearlymussel | *Plethobasus cicatricosu* | E, XN | AL, IN, KY, TN |
| Orangefoot pimpleback | *Plethobasus cooperianus* | E, XN | AL, IL, IN, KY, PA, TN |
| Clubshell | *Pleurobema clava* | E, XN | AL, IN, KY, MI, OH, PA, WV |
| James spinymussel | *Pleurobema collina* | E | NC, VA, WV |
| Black clubshell | *Pleurobema curtum* | E | MS |

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| Common Name | Scientific Name | Status | Distribution |
|---|---|---|---|
| Southern clubshell | *Pleurobema decisum* | E, CH | AL, GA, MS |
| Dark pigtoe | *Pleurobema furvum* | E, CH | AL |
| Southern pigtoe | *Pleurobema georgianum* | E, CH | AL, GA, TN |
| Cumberland pigtoe | *Pleurobema gibberum* | E | TN |
| Flat pigtoe | *Pleurobema marshalli* | E | AL, MS |
| Ovate clubshell | *Pleurobema perovatum* | E, CH | AL, MS |
| Rough pigtoe | *Pleurobema plenum* | E, XN | AL, IN, KY, PA, TN, VA |
| Oval pigtoe | *Pleurobema pyriforme* | E, CH | FL, GA |
| Heavy pigtoe | *Pleurobema taitianum* | E | AL |
| Fat pocketbook | *Potamilus capax* | E | AR, IL, IN, KY, LA, MO, MS |
| Alabama heelsplitter | *Potamilus inflatus* | T | AL, LA |
| Triangular kidneyshell | *Ptychobranchus greeni* | E, CH | AL, GA, TN |
| Rough rabbitsfoot | *Quadrula cylindrica strigillata* | E, CH | TN, VA |
| Winged mapleleaf | *Quadrula fragosa* | E, XN | AR, MN, OK, WI |
| Cumberland monkeyface | *Quadrula intermedia* | E, XN | AL, TN, VA |
| Appalachian monkeyface | *Quadrula sparsa* | E, XN | TN, VA |
| Stirrupshell | *Quadrula stapes* | E | AL, MS |
| Pale lilliput | *Toxolasma cylindrellus* | E | AL, TN |
| Purple bean | *Villosa perpurpurea* | E, CH | TN, VA |
| Cumberland bean | *Villosa trabalis* | E, XN | AL, KY, TN, VA |
| | | | |
| | **AMPHIBIANS** | | |
| | ***Ambystomatidae*** | | |
| Reticulated flatwoods salamander | *Ambystoma bishop* | E, CH | AL, FL, GA |
| Frosted flatwoods Salamander | *A. cingulatum* | T, CH | FL, GA, SC |
| California tiger salamander | *A. californiese* Central California DPS | T, CH | CA |
| California tiger salamander | *A. californiese* Santa Barbara County DPS | E, CH | CA |
| California tiger salamander | *A. californiense* Sonoma County DPS | E | CA |
| Santa Cruz long-toed salamander | *A. macrodactylum croceum* | E | CA |
| Sonora tiger salamander | *A. tigrinum stebbinsi* | E | AZ |
| | ***Plethodontidae*** | | |
| San Marcos salamander | *Eurycea nana* | T, CH | TX |
| Barton Springs salamander | *E. sosorum* | E | TX |
| Texas blind salamander | *Typhlomolge rathbuni* | E | TX |
| | ***Bufonidae*** | | |
| Wyoming toad | *Bufo baxteri* | E | WY |

| Common Name | Scientific Name | Status | Distribution |
|---|---|---|---|
| Arroyo toad | *B. californicus* | E, CH | CA |
| Houston toad | *B. houstonensis* | E, CH | TX |
| | ***Eleutherodactylidae*** | | |
| Guajon | *Eleutherodactylus cooki* | T, CH | PR |
| | ***Ranidae*** | | |
| California red-legged frog | *Rana aurora draytonii* | T, CH | CA |
| Chiricahua leopard Frog | *R. chiricahuensis* | T | AZ, NM |
| Mountain yellow-legged frog | *R. muscosa* | E, CH | CA, NV |

## 7.0 EFFECTS OF THE ACTION

Our analysis of the effects of the proposed action on listed species and critical habitats includes a discussion of the sources of cyanide, an explanation of our assumptions about cyanide exposure, an overview of cyanide toxicity, an analysis of effects to listed fish, followed by analyses of effects to listed amphibians and invertebrates including freshwater mussels. The taxon-specific analyses (i.e. fish, amphibians, invertebrates, mussels) considered information on cyanide toxicity, factors influencing toxicity, approaches for estimating effects on individuals, and discussions of population-level responses. Our analyses were dependent on the available information, which varied among taxa. For fish, the available chronic toxicity data enabled us to develop a method for estimating the magnitude of effect on fish species. For other taxa we exercised best professional judgement to estimate effects based on existing data. These sections provide the basis for our discussion of effects to individual listed species and their designated critical habitat.

*General Sources of Cyanide*

Cyanide is ubiquitous in the environment, and enters waterways as a point source and nonpoint source of pollution. Cyanide is produced synthetically to support industrial uses, it is found naturally in such foods as oil of bitter almonds, cassava, cherry pits, and various microorganisms produce cyanide (Leduc 1984, Eisler 2000, Dzombak et al. 2006). Cyanides are used widely in ore-extraction, steel and heavy metal industries (e.g., electroplating), the manufacture of synthetic fabrics and plastics, some chlorination treatment plants, as a pesticide and as an intermediate ingredient in herbicides, in road salts, and until recently some fire retardants (see Table 2). Anthropogenic sources contribute the vast majority of cyanide in the environment. Certain activities directly release cyanide into the environment, or cyanide may be produced as the byproduct of activities. Industrial activities that produce cyanide as a byproduct include municipal waste and sludge incineration and coking and gasification of coal, to name a few. Metal industries and organic chemical industries are major contributors of cyanide into the freshwater aquatic environment, whereas, atmospheric cyanide, a by-product of forests fires, may be the primary source of oceanic cyanide (Leduc 1984, EPA 2005, Dzombak et al. 2006).

Compared to storm water runoff, urban roadway snow exposed to traffic and winter maintenance practices has a much greater capacity to accumulate and retain heavy metals and other pollutants. In a study of urban highway sites, concentrations of cyanide and metals were orders of magnitude higher than at the control sites and exceeded storm water runoff concentrations by one to two orders of magnitude. Cyanide levels, although demonstrating some variability, remained relatively constant at all sites, averaging 154 µg/L, or cyanide concentrations increased according to the increased application of deicing salts that contained cyanide compounds as anti-caking agents (Glenn and Sansalone 2002).

A study on the effect of cyanide on the anaerobic treatment of synthetic wastewater noted that cyanide is produced on an industrial scale of 2–3 million tons per year and, therefore is in many different industrial waste waters. The concentrations encountered in industrial waste generally are in the range 0.01–10,000 mg/L, most of it in complexed species of cyanide, which are less toxic than free cyanide. Cyanide contamination also occurs in the processing of agricultural crops containing high concentrations of this compound, such as cassava (Gijzen et al. 2000). Systematic surveys of large wastewater effluents in Southern California support a low estimate of free cyanide in wastewaters. In different years reported from 1992 – 2002, mean cyanide concentrations in effluents ranged from <2 to 30 µg/L (Steinberger and Stein 2003).

A more widespread risk of cyanide poisoning of aquatic life is likely from biomass burning (e.g., burning waste biomass for energy conversion, crop burning, prescribed forest fires and wildfires). Barber et al. (2003) examined releases of cyanides from biomass burning and their effect on surface runoff water. In laboratory test burns, available cyanide concentrations in leachate from residual ash were much higher than in leachate from partially burned and unburned fuel and were similar to or higher than a 96-h median lethal concentration (LC50) for rainbow trout (45 µg/L). Free cyanide concentrations in stormwater runoff collected after a wildfire in North Carolina averaged 49 µg/L, an order of magnitude higher than in samples from an adjacent unburned area (Barber et al. 2003).

According to the Toxics Release Inventory, cyanide compound releases to land and water totaled about 1.5 million lbs. between 1987 and 1993, of which about 65 percent was to water. The largest releases (combined land and water releases) occurred in California and Pennsylvania. Releases in California were land-based, while the releases in Pennsylvania occurred primarily in water (EPA 2005). A review of STORET data in 1981, reported the mean cyanide concentration in surface waters of the United States was less than 3.5 µg/L, with concentrations in some industrial areas exceeding 200 µg/L (Dzombak et al. 2006). Coincidentally, production of HCN, one of most toxic forms of cyanide in water, has more than doubled in the United States since the early 1980s (from 330,000 tons per year in 1983 to 750,000 tons per year in 2001).

**Table 2. Forms of cyanides and their uses.**

| Form | Uses | Reference |
|---|---|---|
| Cyanide salts (potassium | Steel manufacturing & heat-treating facilities Metal cleaning, electroplating | IPCS; Leduc 1984; EPA 2005 |

| | | |
|---|---|---|
| cyanide, sodium cyanide) | Ore-extraction (gold-mining, coke extraction) | IPCS; Leduc 1984; EPA 2005 |
| | Dyeing, printing of photographs | IPCS; EPA 2005 |
| | Production of resin monomers (acrylates) | IPCS |
| | Fire retardants[1] | Little and Calfee 2002 |
| | Anti-caking agent for road salts | Dzombak et al. 2006 |
| | Pharmaceuticals (antibiotics, steroids, chemotherapy) | Dzombak et al. 2006 |
| Hydrogen cyanide, organocyanides, metallo-cyanide compounds | Fumigant/pesticide | IPCS, Dzombak et al. 2006 |
| | Herbicides (dichlobenil, bromoxynil, bantrol) | EPA 2005, Dzombak et al. 2006 |
| | Road salts | EPA 2005 |
| | Production of other cyanides (e.g., sodium cyanide for gold mining) | EPA 2005, Dzombak et al. 2006 |
| | Pyrolysis of paper, wool, polyurethane | IPCS |
| | Chelating agents for water and wastewater treatment | EPA 2005, Dzombak et al. 2006 |
| | Production of clear plastics | Dzombak et al. 2006 |
| | Methionine[2] for animal food supplement | Dzombak et al. 2006 |

[1] As of 2007 sodium ferrocyanide was no longer an accepted ingredient in fire retardants ( Long Term Retardents) used in the U.S. (U.S. Forest Service. 2007. Specification 5100-304c Long Term Retardant, Wildland Firefighting)
[2] Hydrogen cyanide is used in the manufacturing of methionine

The risk to aquatic environments from cyanide releases depends on several factors including: the form and concentration of cyanide released, water pH, the presence of metallic trace elements like iron, degree of solar radiation, air and water temperatures, and the presence of natural cyanide sinks (Dzombak et al. 2006). Aqueous cyanide readily evolves from hydrogen cyanide, metallocyanide complexes, and organocyanides, and from metal-cyanide solids. Solid forms of cyanide may exist in the soil of sites for years, and once exposed to water may result in dissolved cyanide reaching ground water and eventually surface waters (see Dzombak et al.'s [2006] discussion about the industrial legacy of cyanide box wastes at thousands of former manufactured gas plants in the United States). Free cyanide readily biodegrades to carbon dioxide and the ammonia ion but its fate depends upon water temperatures, dissolved oxygen levels, mixing, and nutrients (Young et al. 2006).

While available free cyanide is the primary toxic agent in water and is the form expressed by the aquatic life criteria reported by EPA (1985; see Table 1), total cyanide is most commonly measured in discharges (EPA 1985, Dzombak et al. 2006). Measurements are frequently conducted via colorimetric, titrimetric, or electrochemical finish techniques (Dzombak et al. 2006). Measurements of total cyanide are limited to detection in a reagent water matrix of about 1 to 5 µg/L and do not measure: cyanates, thocyanates, most organic-cyanide compounds, and most cobalt and platinum cyanide complexes (Dzombak et al. 2006). Problems with sample storage, regulatory criteria, and the methods for testing and their sensitivity are a concern (Eisler 2000, Dzombak et al. 2006). Eisler (2000) notes that due to the volatilization of cyanide, periodic monitoring is not informative (for example, monitoring once per quarter [see EPA 2006]). Consequently, Eisler (2000) and others recommend that continuous monitoring systems are necessary, with particular emphasis on industrial dischargers, to understand the fate and transport, critical exposures, and relative contributions of human and natural sources of cyanide in the aquatic environment.

It should be noted that EPA's request for ESA section 7 consultation on the aquatic life criteria for cyanide is the first consultation request on its recommended aquatic life criteria. Cyanide will co-occur in waters of the United States in mixture with numerous other pollutants. These other pollutants may or may not have criteria established under section 304(a) of the CWA.

*Basis for Assuming Exposure to Cyanide at Criteria Concentrations*

We based our effects analysis in this Biological Opinion on the premise that the proposed action could be implemented fully. Listed aquatic species and their critical habitats could be exposed to cyanide concentrations in the water column at concentrations consistent with the CMC and CCC. These criteria, as approved by EPA in State and Tribal water quality standards, become the "protection level" to which the water quality-based approach to pollution control is applied. In freshwater systems, this protection level is met if the one-hour average concentration in waters does not exceed 22 ug/L cyanide more than once every three years on average; AND, the four-day average concentration does not exceed 5.2 ug/L cyanide more than once every three years on average. For saltwater systems, the one-hour average concentration should not exceed 1.0 ug/L more than once every three years on the average.

Our rationale for assuming exposure to cyanide at the CMC and CCC includes the following:

1. Section 7 consultation is future-oriented, focusing on the potential effects of the *proposed* action.
2. The CMC and CCC represent the basis for administering water quality programs under the water quality-based approach to pollution control, including monitoring to determine whether waters are attaining designated uses, 303(d) listing of impaired waters, and the development and implementation of TMDLs.
3. We can make no reasonable assumption that any additional protections afforded by antidegradation policies will persist in the future, given EPA's conclusion that section 7 consultation does not apply to EPA approval of new or revised antidegradation policies.
4. There are no other elements of the action proposed that would limit the timing, duration, frequency, or magnitude of exposure of listed species and critical habitats to cyanide pollution other than those defined by the CMC and CCC. General Policies could have the effect of relaxing limits on exposure.

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5. Approval of cyanide concentrations at the CMC and CCC can result in exposures that vary through time and space or occur uniformly throughout waterbodies. Exposure to cyanide pollution at the CMC and CCC could occur to portions of populations, whole populations, or the full range of a species.

*Physiology of Cyanide Toxicity*

The adverse effects of cyanide on fish and other aquatic species have been well documented (Leduc 1984, Eisler 2000, Gensemer et al. 2006, Lanno and Menzie 2006) and are discussed in subsequent sections of this document. Our knowledge of the basic mechanisms by which cyanide exerts its toxic effects, however, has been gained largely as a result of human health-related studies with rodents and other mammals. Recent reviews of the mammalian literature (ATSDR 2006, Borowitz et al. 2006) describe the physiological mechanisms for cyanide toxicity and are summarized below.

Cyanide is known to affect several organ systems including the central nervous system, heart, liver and kidney, although, the principal sites of action are the brain and heart. Cyanide is a potent respiratory toxin which inhibits mitochondrial respiration and can induce rapid lethal responses following acute exposures. Cyanide preferentially binds to the ferric iron atom of the metalloenzyme, cytochrome c oxidase, which is the terminal oxidase in the mitochondrial membrane respiratory chain (electron transport system). The binding of cyanide to the ferric iron atom inhibits the reaction by which electrons are transferred from reduced cytochrome to oxygen thereby blocking oxidative metabolism and ATP production. Blockage of oxidative metabolism can disrupt calcium homeostasis by stimulating calcium release from intracellular stores and enhance influx of extracellular calcium. Elevated cellular calcium activates numerous biochemical pathways including those that lead to nerve cell death via necrosis and apoptosis (programmed cell death). Cytotoxicity is common following acute lethal cyanide exposures. Impaired cellular oxygen utilization results in hypoxia, a shift from aerobic to anaerobic metabolism, and a build up of lactic acid. Because neural tissues have a high oxygen demand and cannot function under low oxygen, the brain is particularly sensitive. The effects of cellular hypoxia and lactate acidosis depress the central nervous system and, at lethal cyanide concentrations, lead to respiratory arrest and death.

Cyanide also binds to other metalloenzymes such as catalase, peroxidase, methemoglobin, hydroxocobalamin, phosphatase, tyrosinase, ascorbic acid oxidase, xanthine oxidase, and succinic dehydrogenase (ATSDR 2006). Modulation of these enzymes contributes to other manifestations of cyanide toxicity. For example, oxidative stress plays an important role in cyanide-induced neurotoxicity. Inhibition of antioxidant enzymes (e.g., catalase) promotes oxidative stress because these enzymes provide protection from the adverse effects (such as lipid peroxidation) that are caused by reactive oxygen species (ROS). In addition, elevated cellular calcium levels (described above) have been associated with increased concentrations of ROS (Borowitz et al. 2006). The effects of rising ROS and antioxident enzyme inhibition can exacerbate nerve cell damage. Cyanide can also cause the release of excitatory neurotransmitters (e.g., dopamine) in the brain and can initiate the release of catecholamines from the adrenals and adrenergic nerve terminals (Smith 1996).

Modulation of catecholamine biosynthesis and neurotransmitter release may help explain some of the acute and chronic neurological effects of cyanide (Borowitz et al. 2006).

The major route for cyanide elimination from the body is via enzyme-mediated thiocyanate formation in the kidney and excretion in urine. Rhodanase is the enzyme that catalyzes the transformation of thiosulfate to thiocyanate and its activity is limited by the availability of sulfur. Although thiocyanate ($SCN^-$) is the principle form of cyanide that is eliminated, it can also accumulate in tissues and is known to have antithyroidal properties. $SCN^-$ inhibits iodine uptake by thyroid tissues and disrupts thyroid hormone homeostasis which can result in the development of goiter.

In the aquatic environment, the primary route of exposure for fish, amphibians, and aquatic invertebrates is via water. Cyanide enters the body through semipermiable membranes such as gills, egg capsules and other sites where gas exchange and osmoregulatory processes occur (Eisler 2000). Cyanide does not tend to bioaccumulate in aquatic biota so dietary uptake (at criteria concentrations) is not considered to be an important exposure pathway (Lanno and Menzie 2006).

## 7.1 Effects to Fish

*Acute Toxicity to Fish*

Knowledge of the acute lethal effects of cyanide on fish has been gained through observations following accidental spills, intentional field applications for lake/stream management, and controlled laboratory studies. Massive kills of fish and other aquatic organisms have been observed following accidental spills of cyanide from storage reservoirs, overturned rail tank cars, and other sources (Leduc 1984). Some of the most catastrophic releases have been from gold mine heap leaching operations, where cyanide is used in the gold extraction process (Wong-Chong et al. 2006). Releases from waste heap leaching pads and tailing storage ponds have discharged large quantities of cyanide into surface waters. In 1997, 245,000 gallons of cyanide solution leaked from a heap leach pad at the Gold Quarry Mine, NV into two nearby creeks, and 7 million gallons of treated leach solution (0.2 ppm cyanide) was released from storage ponds at the USMX Mine, UT into the East Fork of Beaver Dam Wash (Wong-Chong et al. 2006). One of the largest spills occurred in Baia Mare, Romania where 26 million gallons of cyanide-bearing tailings were released due to a failure in the tailings dam, killing an estimated 1,240 tons of fish in the Hungarian portion of the Tisza River alone (Wong-Chong et al. 2006). Because of its high toxicity and relatively short half-life, cyanide has been used by fishery managers for lake restoration (Leduc 1984) and for collecting fish from ponds, lakes and steams (Lewis and Tarrant 1960). Following treatment, poisoned fish exhibit several symptoms including increased ventilation, surfacing, gulping for air, frantic swimming in circles, convulsions, tremors and finally death (Leduc 1984). At high levels of exposure the onset of acute toxicity occurs rapidly, however, live fish that are rescued and transferred to clean water may survive (Leduc 1984).

Laboratory tests under controlled conditions have revealed that not all life stages of fish are equally sensitive to acute cyanide exposure, that cyanide toxicity can be modulated by abiotic factors, and that there is a wide range in sensitivity among aquatic organisms. Smith et al. (1978) conducted 96-hr acute toxicity tests with fathead minnow, bluegill, brook trout, and yellow perch and found that juveniles and fry were more sensitive to the lethal effects of cyanide than eggs. Bluegill, yellow perch, and brook trout juveniles were more sensitive than newly-hatched fry, where as, swim-up fry were the most sensitive fathead minnow life stage.

Smith et al. (1978) also reported that dissolved oxygen (DO) concentration and water temperature affect the susceptibility of these species to cyanide toxicity. Cyanide was more toxic when fish were held in water with lower DO concentrations. Considering that cyanide is a respiratory toxin that inhibits oxidative metabolism, it is not surprising that the effects are exacerbated under conditions where oxygen availability is limited. The authors also tested the effect of temperature on cyanide toxicity. $LC_{50}$ values for juvenile brook trout, bluegill and yellow perch were lower at low temperatures than at higher temperatures, indicating a heightened sensitivity at lower temperatures. The combination of low DO and low temperature tended to produce conditions that rendered juveniles most susceptible to cyanide toxicity.

Kovacs (1979) found that the sensitivity of juvenile rainbow trout to cyanide was similarly influenced by temperature. He conducted 96-hr acute toxicity tests at three different temperatures (6, 12 and 18˚ C) and reported $LC_{50}$ values of 28, 42, and 68 ug HCN/L, respectively. Thus, trout held at 6˚ C were 2.4 times more sensitive to cyanide than trout held at 18˚ C. These studies demonstrate that life stage as well as abiotic factors (DO and temperature) can influence acute cyanide toxicity. There is also considerable variability among aquatic taxa in terms of their intrinsic sensitivity to cyanide.

Standardized acute toxicity tests with cyanide have been conducted with numerous aquatic species. EPA compiled toxicity data for 83 species of aquatic animals and plants (61 freshwater species and 22 saltwater species) as part of their cyanide BE (EPA 2007). Based on this compilation, there appears to be a large range in sensitivity between the most sensitive (rock crab $LC_{50}$ 4.89 ug CN/L) and the least sensitive species tested (river snail $LC_{50}$ 760,000 ug CN/L). Freshwater species represented 9 phyla, 15 classes, 29 orders, 36 families, and 52 genera. Fishes were among the most sensitive freshwater taxa although there was substantial variability in sensitivity. Among the 24 freshwater fish species included in the list, there was a 33-fold difference in sensitivity between the most sensitive (rainbow trout, *Oncorhynchus mykiss*, $LC_{50}$ 59 ug CN/L) and the least sensitive (bata, *Labeo bata*, $LC_{50}$ 1970 ug CN/L). The 8 most sensitive fish species belong to 3 different families: Salmonidae (3 species, 3 genera); Percidae (2 species, 1 genera); and Centrarchidae (3 species, 3 genera). Because of the relatively low number of species that have been tested within these families, it is difficult to get a sense of the amount of intra-family variability in species sensitivity on the low end of the species sensitivity distribution. By contrast, the family Cyprinidae was well represented with 10 different species representing 8 genera. Among those 10 species, there is an 18-fold difference in sensitivity between the most sensitive (roach $LC_{50}$ 108 ug CN/L) and the least sensitive (bata, *Labeo bata*, $LC_{50}$ 1970 ug CN/L) species. Because of pronounced intra-family variation it is unlikely that the 8 species within the 3 most sensitive families represent the most sensitive species within those families.

Dwyer et al. (2005) conducted acute toxicity tests with 5 different water pollutants (carbaryl, copper, 4-nonylphenol, pentachlorophenol, and permethrin), 3 common test species (rainbow trout, fathead minnow, and sheepshead minnow) and 17 federally threatened, endangered or candidate fish species. They found that for some pollutants, threatened and endangered species in the following families were as or more sensitive than rainbow trout: Salmonidae (Apache trout, Greenback Cutthroat trout, and Lahontan Cutthroat trout), Percidae (Fountain darter and Greenthroat darter), Cyprinidae (Cape Fear shiner, Spotfin chub, and Colorado pikeminnow), and Acipenseridae (Atlantic sturgeon and Shortnose sturgeon). Based on these results, the authors recommended that for listed fish species which require greater protection (i.e., species that may be more sensitive than rainbow trout), a factor of 0.63 can be applied to the geometric mean of the rainbow trout $LC_{50}$, and that if even greater protection is desired a factor of 0.46 can be used (0.63 – 1 standard deviation). For cyanide, applying the adjustment factors of 0.63 and 0.46 to the rainbow trout $LC_{50}$ (59 ug CN/L) would result in $LC_{50}$ estimates of 27 to 37 ug CN/L for sensitive listed fish species.

*Chronic Toxicity to Fish*

Chronic cyanide toxicity tests have been conducted with relatively few fish species. However, available data indicate that cyanide not only reduces survival but also affects reproduction, growth, swimming performance, condition, and development (Table 3). Reproduction appears to be one of the most sensitive endpoints. Full and partial life cycle tests with fathead minnows and brook trout have shown that fish exposed to sublethal concentrations of cyanide spawned fewer eggs than non-exposed fish (Koenst et al. 1977, Lind et al. 1977). Fecundity was reduced by 57.8% and 46.9% (compared to controls) in female fathead minnows exposed to cyanide at 19.6 ug HCN/L (the LOEC) and 12.9 ug HCN/L (the NOEC), respectively. Similarly, the mean number of eggs spawned by brook trout was reduced by 53.3% at 11.2 ug HCN/L and by 17.7% at 5.7 ug HCN/L.

Kimball et al. (1978) exposed bluegill to cyanide (5.2 – 80 ug HCN/L) for 289 days and reported that no eggs were spawned in any of the cyanide treatments except for one spawning that occurred at the highest concentration. Although the single spawning is difficult to explain, the fact that spawning was completely inhibited in 42 of 43 cyanide-exposed females suggests that bluegill may be particularly sensitive. It is surprising, considering the overwhelming effects on reproduction, that a confirmatory study with bluegills has not been conducted over the past 30 years.

Cheng and Ruby (1981) studied the effects of pulsed exposures of cyanide on flagfish reproduction. Unlike the studies describe above, where fish were exposed over an extended period of time to a constant concentration, flagfish were exposed to sublethal concentrations of cyanide for 5-day pulses. Flagfish exposed to cyanide (65 ug/L) for 5 days following fertilization (i.e., as eggs) and then reared to maturity in clean water, spawned 25.6% fewer eggs than flagfish that had not been exposed. In another experiment by the same authors, flagfish that received a second 5-day pulse of cyanide as juveniles had an even greater reduction (39.3%) in the number of eggs spawned. These studies demonstrate that cyanide can affect an apical reproductive endpoint in fish.

The mechanism by which cyanide induces these reproductive effects is not fully understood. However, key physiological, biochemical, histological (morphological), and endocrine functions known to be involved in sexual maturation are affected by cyanide (Table 3).

**Table 3. Chronic effects of cyanide on various fish species.**

| Species | Life stage | Exposure duration | Temp (C) | pH | HCN (ug/L) | CN (ug/L) | Effect | Reference |
|---|---|---|---|---|---|---|---|---|
| Atlantic Salmon *Salmo salar* | Adult females during late vitellogensis | 12 days | 7 | 6.8 | 5 | 4.8 | Inhibition of vitellogenin uptake by ovaries: observed increased plasma vitellogenin, decline in gonad vitellogenin, and no change in liver vitellogenin. | Ruby et al. 1987 |
| Atlantic Salmon *Salmo salar* | Egg/fry | 103-112 day incubation plus 58 days post hatch | 3.5 – 8.3 | 7.6 | 10 – 100 (nominal) | | Teratagenic effects observed in fry: malformation and/or absence of the eyes, defects in mouth and vertebral column and yolk-sac dropsy (note: CN concentrations not measured in exposure tanks) | Leduc 1977 |
| Bluegill *Lepomis macrochirus* | Adult and early life stages | 289 days[1] | 24.9 | 8.1 | 5.2 - 80 | 5.4 – 82.7 | No spawning occurred among fish exposed to CN except for one female in the 82.7 ug/l treatment. There were 5 and 8 spawnings in the two controls, respectively. | Kimball et al. 1978 |
| Bluegill *Lepomis macrochirus* | Egg/larvae | 57 days | 25 | 8.02 | 4.8 – 82.1 | 4.9 – 84.4 | NOEC for early life stage survival 9.4 ug CN/L. LOEC was 19.9 ug CN/L (88% reduction in survival compared to controls). | Kimball et al. 1978 |
| Brook Trout *Salvelinus fontinalis* | 19 month-old adults | 144 days | 12.5 | 7.9 | 5.7 – 75.3 | 5.6 – 74.4 | NOEC for fecundity (number of eggs spawned) 5.6 ug CN/L | Koenst et al. 1977 |
| Brook Trout *Salvelinus fontinalis* | Sac fry/juvenile | 90 days (from hatch to 90 post hatch) | 9 | 7.9 | 5.6 – 77.2 | 5.5 – 76 | NOEC for growth (weight of juveniles at 90 days post hatch) 21.4 ug CN/L. | Koenst et al. 1977 |
| Brook Trout *Salvelinus fontinalis* | Sac fry/juvenile | 90 days (from hatch to 90 post hatch) | 9 | 7.9 | 5.5 - 77 | 5.4 – 75.6 | Survival reduced by 15%, 25% and 70%, at 55, 66 and 77 ug CN/L, respectively. | Koenst et al. 1977 |
| Fathead Minnow *Pimephales promelas* | Larvae through adult | 256 days | 24.95 | 8.075 | 5.8 – 100.7 | 6.0 – 103.9 | NOEC for fecundity (number of eggs spawned) 13.3 ug CN/L. LOEC 20.2 ug CN/L (58% reduction in spawning compared to controls). | Lind et al. 1977 |
| Flagfish *Jordanella floridae* | Egg through adult | 5 days during embryo/ larvae stage | 25 | 8.05 | 65 – 87 | 66.8 – 89.5 | Onset of spawning delayed, estrus cycle[2] shortened, fecundity reduced in CN treatments 26% to 35% compared to controls | Cheng and Ruby 1981 |

| Species | Life stage | Exposure duration | Temp (C) | pH | HCN (ug/L) | CN (ug/L) | Effect | Reference |
|---|---|---|---|---|---|---|---|---|
| Flagfish *Jordanella floridae* | Egg through adult | 5 days & 5 days[3] | 25 | 8.05 | 65 – 87 | 66.8 – 89.5 | Onset of spawning delayed, estrus cycle[2] shortened, fecundity reduced in CN treatments 39 to 47% compared to controls | Cheng and Ruby 1981 |
| Flagfish *Jordanella floridae* | Egg/larvae | From fertilization through hatching (~5 – 9 days) | 25 | 8.05 | 65 – 150 | 66.8 – 154.3 | Hatching Success: 89% (control), 86%, - 3% in CN treatments. Hatching delayed and pituitary gland size reduced in all CN treatments<br>Eye Malformations (microphthalmia - reduced eye size, and monophthalmia - disintigration of the eye), 30%, (66.8 ug/L), 40% (77 and 89.5 ug/L) | Cheng and Ruby 1981 |
| Rainbow Trout *Oncorhynchus mykiss* | Juvenile males | 18 days | 12.5 | 7.9 | 10, 30 | 9.8, 29.5 | Spermatogenesis: reduced number of dividing spermatogonia, 13% reduction at 10 ug/L, 50% reduction at 30 ug/L. | Ruby et al. 1979 |
| Rainbow Trout *Oncorhynchus mykiss* | 150 – 300 g vitellogenic females | 12 days | 12.5 | 7.2 | 10 | 9.7 | Reduction in plasma vitellogenin and GSI compared to controls | Ruby et al. 1986 |
| Rainbow Trout *Oncorhynchus mykiss* | 2–3 year-old | 12 days | 12.5[4]<br>10.7[5] | 7.83[4]<br>7.78[5] | 10 | 9.8 | Increase in brain dopamine compared to control fish, decrease in mean oocyte diameter (19%) of vitellogenic females, and higher numbers of spermatogonial cysts in males. Two identical experiments, one in July the other in August. | Szabo et al. 1991 |
| Rainbow Trout *Oncorhynchus mykiss* | 200 - 350g vitellogenic female rainbow trout | 7 days | 12 | 7.2 | 10, 20, 30 | 9.7, 19.3, 29 | Decrease in serum calcium at 9.7 and 19.3 ug/L | Da Costa, H. and Ruby, S.M. 1984 |
| Rainbow Trout *Oncorhynchus mykiss* | 2.5 year old vitellogenic female rainbow trout | 12 days | 12 | 7.6 | 10 | 9.7 | 20% reduction in female GSI, 33% reduction in plasma vitellogenin, 55% reduction in plasma 17B-estradiol (E2), 48% reduction in oocyte diameter, and a 70% reduction in plasma T3 compared to controls. Plasma T4 levels were lower but difference was not statistically significant. | Ruby, S.M. et al. 1993a. |
| Rainbow Trout *Oncorhynchus mykiss* | juveniles (35 - 43g) | 20 days | 10 | 7.8 | 10, 20 | 9.8, 19.5 | At 9.8: 65% reduction in the frequency of stage 5 (most mature) oocytes, 2 fold increase in the frequency of atretic follicles. At 19.5: 62% reduction in the frequency of stage 5 oocytes, 1.9 fold increase in the frequence of atretic follicles. | Lesniak, J.A. and Ruby, S.M. 1982. |

| Species | Life stage | Exposure duration | Temp (C) | pH | HCN (ug/L) | CN (ug/L) | Effect | Reference |
|---|---|---|---|---|---|---|---|---|
| Rainbow Trout *Oncorhynchus mykiss* | 2 year old sexually maturing males | 12 days | 11.5 | 7.9 | 10 | 9.8 | Reduced # of spermatocytes, increased # of spermatogonial cysts, decreased # of basophils in pituitary, hypothesise effect of CN on hypthalmic-pituarity-gonadal axis | Ruby, S.M. et al. 1993b. |
| Rainbow Trout *Oncorhynchus mykiss* | Juveniles (3 g) | 18 days | 12.5 | 7.9 | 10 | 9.8 | LOEC for growth measured as wet weight change over 18 days. NOEC < 9.8 ug CN/L | Dixon, D.G. and G. Leduc. 1981. |
| Rainbow Trout *Oncorhynchus mykiss* | Juveniles (12 g) | 18 days | 12.5 | 7.9 | 10 | 9.8 | NOEC for growth measured as wet weight change over 18 days. | Dixon, D.G. and G. Leduc. 1981. |
| Rainbow Trout *Oncorhynchus mykiss* | Juveniles (3 g & 12 g) | 18 days | 12.5 | 7.9 | 10 – 30 | 9.8 – 29.5 | Increased resting metabolic rate<br>Liver damage:degenerative hepatic necrosis | Dixon, D.G. and G. Leduc. 1981. |
| Rainbow Trout *Oncorhynchus mykiss* | Juveniles (20g) | 20 days | 6<br>12<br>18 | 7.9 | 5<br>20<br>30 | 4.9<br>19.7<br>29.8 | NOEC for mean specific growth rate (MSGR) based on dry weight. Author estimated thresholds at 6C, 12C and 18C to be < 4.9, 9.8 and 29.8 ug CN/L, respectively. | Kovacs, T.G. 1979. |
| Rainbow Trout *Oncorhynchus mykiss* | Juveniles (20g) | 20 days | 6,12, 18 | 7.9 | 5 – 45 | 4.9 – 44.7 | Reduced swimming performance | Kovacs, T.G. 1979. |
| Sheepshead Minow *Cyprinodon variegatus* | Embryo/larvae | 28 days | 22.4 | | 29 - 462 | | Survival of 28 day-old larvae reduced from 22.5% (29 ug/L) to 54.5% (462 ug/L) compared to controls. Author states that MATC[6] lies between 29 and 45 ug /L | Schimmel et al. 1981 |

[1]Exposure of first and second year spawners to HCN followed by 90-day exposure of eggs/larvae (second generation).
[2]Estrous cycle as defined by Cheng and Ruby 1981: "In this study, estrous cycle is defined as the duration of egg-laying starting from the first appearance of more than ten eggs through the occurrence of less than ten eggs on the mats."
[3] 5-day exposure during embryo/ larvae stage followed by a second 5 day exposure during juvenile stage.
[4]Experiment conducted in July.
[5]Experiment conducted in August.
[6]MATC – Maximum Acceptable Toxicant Concentration (the MATC is typically the geometric mean of the NOEC and LOEC).

Lesniak and Ruby (1982) reported abnormal oocyte development in sexually maturing female rainbow trout exposed to cyanide (10 and 20 ug HCN/L) for 20 days. Ovaries from cyanide-exposed fish contained fewer mature oocytes, exhibited altered patterns of secondary yolk deposition (in developing oocytes), had nearly twice the frequency of atresia (oocyte resorption), and had an overall reduction in the number of viable eggs.

Ruby et al. (1986) reported that vitellogenic female rainbow trout exposed for 12 days to 10 ug HCN/L had lower levels of plasma vitellogenin and a lower gonadosomatic index (GSI) compared to controls. In two similar studies, oocyte diameter (an indicator of gonadal growth and development) was reduced in sexually maturing female rainbow trout exposed for 12 days to 10 ug HCN/L (Ruby et al. 1993a, Szabo et al. 1991). Reduced oocyte diameter was accompanied by reductions in plasma vitellogenin,17β-estradiol (E2), and GSI (Ruby et al. 1993a), as well as increased whole brain dopamine levels (Szabo et al. 1991).

Dopamine has an inhibitory effect on gonadotropin-releasing hormone (GnRH) neurons in some fish species and it is GnRH that stimulates the release of gonadotropins (GtH I and GtH II) from the pituitary (Saligaut et al. 1999; Patino, R. 1997). GtH I and GtH II are believed to function similar to follicle-stimulating hormone and luteinizing hormone, respectively, in tetrapods (Patino, R. 1997). In female fish, GtH I acts on target cells in the gonad, stimulating E2 synthesis. E2 induces vitellogenin synthesis in the liver. Vitellogenin is the egg yolk precursor in fish that is produced by the liver, transported via blood, taken up by the ovaries, and incorporated into developing oocytes. GtH II also acts on the gonad by inducing the synthesis of maturation-inducing steroid (MIS). MIS induces oocyte maturational competence and ovulation (Park et al. 2007; Patino, R. 1997).

The control exerted by dopamine over gonadal maturation has been recognized by fish culturists, who have been successful in treating captive-reared fish with anti-dopaminergic drugs (which block dopamine receptors), such as pimozide and domperidone, to induce ovulation (Szabo et al. 2002, Park et al. 2007, Jensen 1993, Patino 1997). Thus, oocyte development, maturation and ovulation are under the control of gonadotropins and E2 which in turn, are modulated in part by GnRH and dopamine. This interaction between the neuroendocrine system and reproductive organs is referred to as the hypothalamus-pituitary-gonadal (HPG) axis (IPCS 2002).

Cyanide has also been shown to affect male reproductive processes. Exposure of male rainbow trout to cyanide concentrations of 10 and 30 ug HCN/L for 18 days disrupted spermatogenesis as evidenced by a reduction in the number of dividing spermatogonia and a blockage of mitotic progress (Ruby et al. 1979). Exposure of rainbow trout for 12 days to 10 ug HCN/L resulted in higher numbers of spermatogonial cysts in testes of male trout as well as higher levels of whole brain dopamine (Szabo et al. 1991). Ruby et al. (1993b) reported similar results where the number of spermatocytes decreased and the number of spermatocyte precursors (spermatogonial cysts) increased in two-year-old sexually maturing rainbow trout after a 12-day exposure to 10 ug HCN/L. There are indications that the transformation of spermatogonial cysts to spermatocytes is hormonally regulated through GtH along the HPG axis and that, within the pituitary, GtH is released from type I granular basophils (Ruby et al. 1993b). Histological examination of pituitary glands from cyanide-exposed fish showed a reduction in the number of type I granular basophils. Ruby et al. (1993b) suggested that elevated levels of brain dopamine may be responsible for the selective loss of type I granular basophils and subsequent alteration of spermatocyte formation.

Ruby et al. (1979, 1993b) and Szabo et al. (1991) hypothesized that cyanide acts through the HPG axis to affect reproduction in fish. Their studies (described above) demonstrated (1) that cyanide caused an increase in brain dopamine levels, consistent with neuronal effects observed on mammals, (2) that levels of reproductive hormones (E2) and egg-yolk precursors (vitellogenin) were altered following exposure to cyanide, (3) the selective loss of putative GtH releasing pituitary cells (type I granular basophils) and (4) retarded gonad development in cyanide-exposed male and female rainbow trout. Taken together, these results appear to be consistent with HPG axis involvement. Ruby et al. (1979, 1993b) and Szabo et al. (1991) also reported that these effects occurred following relatively short (12 to 18 days), sublethal exposures to cyanide. Whether these effects would result in the same type of reduced fecundity and spawning, as was observed in cyanide-exposed female fathead minnows (Lind et al. 1977), bluegill (Kimball et al. 1978), and brook trout (Koenst et al. 1977), was not addressed in the rainbow trout studies because they were terminated before the fish reached full sexual maturity. However, it does seem likely that these effects would occur.

Results from Cheng and Ruby (1981) indicate that continuous exposure of fish to cyanide through the spawning period may not be necessary to affect fecundity. Short-term, pulsed exposures of cyanide to flagfish were sufficient to induce subsequent effects on the number of eggs spawned, and exposed fish did not appear to recover once the exposure had ceased. Even exposure of eggs, one of the most tolerant life stages in terms of acute toxicity (Smith et al. 1979), resulted in latent effects on fecundity once embryos hatched and survived to maturity. This finding is not unexpected given that it is during early developmental stages that the HPG endocrine axis is set up and feedback sensitivity of the hypothalamus and pituitary gonadotropes to gonadal steroids is established (IPCS 2002). Although Cheng and Ruby (1981) did not measure specific indicators of endocrine axis function, they did find that the pituitary gland of cyanide-exposed flagfish embryos was significantly smaller than the pituitaries from control fish. It appears that cyanide, like many EDCs (endocrine disrupting compounds, IPCS 2002), may affect the “set up” of the HPG axis and that these early developmental effects may have long-term consequences on reproduction.

Chronic exposure of eggs and larvae to cyanide can result in reduced embryo/larvae survival and altered development. Leduc (1978) exposed newly fertilized Atlantic salmon eggs to cyanide at concentrations of 10 to 100 ug HCN/L, and observed teratogenesis, as well as, delayed hatching and reduced hatching success at higher concentrations. There was a dose-dependent increase in the frequency of abnormal fry, ranging from 5.8% to 18.5%. Abnormalities included malformed and/or absence of eyes, defects in the mouth and vertebral column and yolk-sac dropsy (*Hydrocoele embryonalis*, also known as blue sac disease). Similar eye abnormalities were reported by Cheng and Ruby (1981) in flagfish larvae exposed, as eggs, to cyanide at concentrations of 65, 75, 87, and 150 ug

HCN/L. Egg hatching success was also reduced and time to hatch was delayed in all cyanide treatments. In a 28-day embryo/juvenile toxicity test, sheepshead minnow survival was significantly reduced in all treatments >29 ug HCN/L (Schimmel 1981). Schimmel (1981) noted there was considerable embryonic mortality and that there was no larval mortality during the last two weeks of exposure, indicating a greater sensitivity during early development. Kimball et al. (1978) exposed bluegill eggs and larvae to cyanide at concentrations of 4.8 to 82.1 ug/HCN/L, and reported that most deaths occurred within the first 30 days after hatching. Survival was reduced in all cyanide treatments and the effects were statistically significant at cyanide concentrations >9.1 ug HCN/L.

As previously mentioned, cyanide effects oxidative metabolism, energy production, and thyroid function; all are important for normal growth and performance. Therefore, it is not unexpected that sublethal exposure of fish to cyanide has been shown to impact growth, condition and swimming performance. There is also evidence that the effect of cyanide on these physiological endpoints can be modulated by other factors such as diet/ration and temperature. When cichlids (*Cichlasoma bimaculatum*) were fed unlimited rations and exposed to cyanide for 24 days, those fish exposed to lower concentrations of cyanide (< 0.06 ug HCN/L) were larger than controls, where as, at higher concentrations weight gain was depressed (Leduc 1984). The increased weight gain in the low-dose treatments was attributed to higher food consumption, which was allowed to occur because ration was not restricted. Low-dose stimulation is a common effect across a broad range of chemical and non-chemical stressors (Calabrese 2008).

Dixon and Leduc (1981) held juvenile rainbow trout on restricted rations and exposed them to cyanide at concentrations of 10, 20, and 30 ug HCN/L for 18 days and observed significantly reduced weight gain in all treatments compared to controls. The effect was characterized by an initial decrease in specific growth during the first 9 days followed by a significant increase from day 9 through day 18. The growth surge during the latter half of the exposure period was not sufficient to offset early reductions. Cyanide-affected juvenile rainbow trout were in poorer condition, as indicated by lower fat content, and had higher respiration rates for several days post-exposure. In addition, juvenile rainbow trout in all cyanide treatments exhibited degenerative necrosis of hepatocytes (i.e., liver tissue damage) that increased in severity with the level cyanide exposure.

Kovacs (1979) held juvenile rainbow trout on restricted rations and exposed them to cyanide for 20 days. The results were similar to those reported by Dixon and Leduc (1981). Cyanide reduced the mean specific growth rate (MSGR) and affected-fish gained less fat during the exposure period. Kovacs (1979) also examined the effects of temperature on rainbow trout growth and sensitivity to cyanide. He found that the growth rate of rainbow trout was inversely related to holding temperature (6, 12 and 18˚C), as would be expected, and that trout held at colder temperatures were more sensitive to cyanide. The NOECs for MSGR were 5, 20, and 30 ug HCN/L for trout held at 6, 12, and 18˚C, respectively. Based on the exposure response curves, Kovacs (1979) estimated thresholds for effects on rainbow trout growth to be <5 ug HCN/L at 6˚C, 10 ug HCN/L at 12˚C, and 30 ug HCN/L at 30˚C.

Kovacs (1979) also evaluated the effects of cyanide on fish swimming performance. His results indicated that swimming performance was affected by cyanide and that the effect was also temperature-sensitive. Fish from the growth study described above were placed in swimming chambers and tested for swimming stamina. Among non-exposed trout, swimming stamina, measured as distance travelled (in meters), decreased with decreasing temperature (i.e., fish held at 6˚C travelled a shorter distance than fish held at 18˚C). Cyanide-exposed fish had reduced swimming stamina compared to non-exposed fish and the effect was more severe at colder temperatures. Based on the exposure-response regression equations reported by Kovacs (1979), the predicted reduction in swimming stamina (compared to controls) for fish exposed to cyanide at the chronic water quality criterion (5.2 ug CN/L) would be 52% at 6˚C, 20% at 12˚C, and 3% at 18˚C.

Cyanide has been shown to affect the swimming performance of other salmonid species, as well. Leduc (1984) calculated the cyanide concentration causing a 50% reduction in swimming ability based on original data collected for brook trout (Neil 1957) and coho salmon (*Oncorhynchus kisutch*) (Broderius 1970) to be 5 ug HCN/L and 7 ug HCN/L, respectively. Thus, chronic exposure of fish to cyanide at sublethal concentrations can affect growth, body condition, and swimming performance. Factors such as temperature and diet/ration can modulate cyanide toxicity, and, for some species-endpoint combinations, these effects may occur at or below the chronic cyanide criterion.

*Effects of Pollutant Mixtures*

Relatively few studies have been performed to measure the effects of free cyanide in combination with other contaminants. Concurrent exposure to cyanide and ammonia produced greater than additive effects to acute lethality in rainbow trout, salmon, and chub (Smith et al. 1979, Alabaster et al. 1983, and Douderoff 1976), and to chronic sublethal effects to growth in rainbow trout (Smith et al 1979). In rainbow trout and salmon, effects to acute lethality were 1.2 and 1.63 times greater, respectively, than would be expected by additivity. Concurrent exposure to cyanide and zinc also resulted in synergistic effects to acute lethality in fathead minnows, where toxicity was 1.4 times that predicted by additivity (Smith et al 1979). Although we are unable to quantify the effect of these synergistic mechanisms for this analysis, they should be considered when assessing effects of cyanide to aquatic organisms in waterways with elevated concentrations of ammonia and zinc.

*Acute Effects Estimation for Listed Fish Evaluation Species*

Of the 103 listed fish species that were evaluated for acute cyanide effects, only the 4 most sensitive (i.e. fountain darter, Apache trout, Lahontan cutthroat trout, and bull trout) were considered likely to be adversely affected (Appendix B). For these 4 species, the magnitude of effect resulting from acute exposure to cyanide at the CMC (22.4 ug CN/L) was estimated using regression analysis.

<u>Estimated acute effects for Fountain Darter</u>: Of the acute exposure-response regression equations listed in Appendix G there are six equations for juvenile mortality in a species of

fish, Yellow Perch, from the family Percidae (Smith et al. 1978). Because toxicity is inversely related to water temperature (Eisler 2000), the equation for the lowest tested temperature, 15 C, would be the most appropriate equation for estimating acute effects in the Fountain Darter. That equation of:

Probit (% juv. Mortality) = -17.790 + 11.650 (log (ug HCN /L))

yields an estimated Yellow Perch acute $LC_{50}$ of 90.4 ug HCN /L, or 88.9 ug CN /L (at the test pH of 7.82). The estimated acute $LC_{50}$ for Fountain Darter from the ICE lower 95% confidence value is 21.5 ug CN/L (Appendix B). This yields an acute $SSEC_x$ estimate of 92.6 ug CN/L, or 94.2 ug HCN /L (see chronic effects section and Appendix C for explanation of $SSEC_x$s). Entering that $SSEC_x$ value on an HCN basis into our probit-log regression equation yields an estimated effect level for Fountain Darter of 58.2% juvenile mortality at the CMC criterion.

<u>Estimated acute effects for Apache Trout</u>: Of the acute exposure-response regression equations listed in Appendix G there is only one for an *Oncorhynchus* species of salmonid (Broderius and Smith 1979). That equation is for tests with juvenile Rainbow Trout. That equation of:

Probit (% juv. Mortality) = 33.63 + 23.04 (log (mg CN /L))

yields an estimated Rainbow Trout acute $LC_{50}$ of 57.2 ug CN /L. The estimated acute $LC_{50}$ for Apache Trout from the ICE lower 95% confidence value is 16.5 ug CN/L (Appendix B). This yields an acute $SSEC_x$ estimate of 77.7 ug CN/L (see chronic effects section Appendix C for explanation of $SSEC_x$s). Entering that $SSEC_x$ value into our probit-log regression equation yields an estimated effect level for Apache Trout of >99.9% juvenile mortality at the CMC criterion.

<u>Estimated acute effects for Lahontan Cutthroat Trout</u>: Of the acute exposure-response regression equations listed in Appendix G there is only one for an *Oncorhynchus* species of salmonid (Broderius and Smith 1979). That equation is for tests with juvenile Rainbow Trout. That equation of:

Probit (% juv. Mortality) = 33.63 + 23.04 (log (mg CN /L))

yields an estimated Rainbow Trout acute $LC_{50}$ of 57.2 ug CN /L. The estimated acute $LC_{50}$ for Lahontan Cutthroat Trout from the ICE lower 95% confidence value is 22.8 ug CN/L (Appendix B). This yields an acute $SSEC_x$ estimate of 56.2 ug CN/L (see chronic effects section and Appendix C for explanation of $SSEC_x$s). Entering that $SSEC_x$ value into our probit-log regression equation yields an estimated effect level for Lahontan Cutthroat Trout of 43% juvenile mortality at the CMC criterion.

<u>Estimated acute effects for Bull Trout</u>: Of the acute exposure-response regression equations listed in Appendix G there are nine equations for juvenile mortality in a species of fish, Brook Trout, from the genus *Salvelinus* (Smith et al. 1978). Because toxicity is

inversely related to water temperature (Eisler 2000), the equation for the lowest tested temperature, 4 C, would be the most appropriate equation for estimating acute effects in the Bull Trout. That equation of:

Probit (% juv. Mortality) = -28.849 + 19.626 (log (ug HCN /L))

yields an estimated Brook Trout acute $LC_{50}$ of 53.1 ug HCN /L, or 51.2 ug CN /L (at the test pH of 7.19). The estimated acute $LC_{50}$ for Bull Trout from the ICE lower 95% confidence value is 15.7 ug CN/L (Appendix B). This yields an acute $SSEC_x$ estimate of 73.0 ug CN/L, or 75.6 ug HCN /L (see chronic effects section and Appendix C for explanation of $SSEC_x$s). Entering that $SSEC_x$ value on an HCN basis into our probit-log regression equation yields an estimated effect level for Bull Trout of 99.9% juvenile mortality at the CMC criterion.

Citing DeForest et al. (*in prep.*), Gensemer et al. (2007) noted that exposure-response curves for cyanide acute toxicity in fish are "quite steep". Gensemer et al.'s (2007) observation is supported by the low $LC_{50}/LC_{10}$ ratios derived for this biological opinion from the exposure-response regression equations in Smith et al. (1978) and Broderius and Smith (1979). A combination of the steep response curves and the low estimated $LC_{50}$ values for the listed evaluation species relative to the surrogate toxicity test species results in substantive estimates of adverse effects at the acute (CMC) criterion of 22.4 ug CN /L even though the acute effects assessment $EC_a$s (13.77-20.00) were not far below the CMC (22.4).

*Chronic Effects Estimation for Listed Fish Evaluation Species*

Ideally, concentration (dose)-response data suitable for predictive modeling would be available for sensitive chronic endpoints for each of the listed species evaluated in this analysis (hereafter referred to as "listed evaluation species"). Such data do not exist for cyanide for any of our listed evaluation species. As recently reviewed by Gensemer et al. (2007), the current inventory of concentration-response data from chronic toxicity testing with cyanide consists of four datasets: one each for reproductive endpoints among the fathead minnow (*Pimephales promelas*; Lind et al. 1977) and the brook trout (Koenst et al. 1977); one for juvenile survivorship among bluegill (Kimball et al. 1978); and one for the sheepshead minnow (Schimmel et al. 1981). Upon closer inspection, Gensemer et al. (2007) found the dataset for the sheepshead minnow to be insufficient for meaningful predictive modeling and we agree with that conclusion. Thus, we are left with three datasets as the best available scientific basis for estimating toxic effects (or the lack thereof) at the chronic criterion value of 5.2 ug CN/L. In addition to our three useable concentration-response datasets, we also possess estimates of $LC_{50}$ values for our listed evaluation species as per the procedures described in Appendix B.

Based on the above information, we took the following approach to evaluating the effects of the proposed action on the listed fish evaluation species:

(1) The three concentration-response data sets were transformed into the most precise, predictive concentration-response models that the data can support; these models were used to predict the response of chronic toxicity test species to proposed chronic CN exposure levels.

(2) The predicted response of a listed fish evaluation species to chronic CN exposures was considered to be the same as the response observed for a chronic toxicity test species at an adjusted chronic CN exposure level based on the ratio of their respective $LC_{50}$ values (see example below).

Two assumptions were relied upon to predict the effect of proposed chronic CN exposure levels on the listed fish evaluation species:

(1) The relative differences in sensitivity to chronic CN exposures between the listed evaluation species and the chronic toxicity test species (fathead minnow, brook trout, and bluegill) are approximated by the ratio of their respective $LC_{50}$ values; and
(2) The slopes of the concentration-response curves are also approximately comparable between the listed evaluation species and the chronic toxicity test species.

These assumptions create a clearly defined basis for a default hypothesis that allows for an analysis of the effects of the proposed action on listed fish evaluation species to proceed within the constraints of minimal data until such time as more data become available. As more data become available appropriate modification (or validation) of our default approach will be facilitated.

An example of applying the above methodology: suppose that one of our chronic toxicity test species is predicted to exhibit a 20% adverse effect from being exposed to a concentration of 5.2 ug CN/L. If a listed evaluation species happens to have an estimated $LC_{50}$ value equal to that of the chronic toxicity test species, then a 20% adverse effect would also be predicted for the listed evaluation species. If the ratio of $LC_{50}$ values was 1.5 (rather than 1.0) in the direction of greater sensitivity for the listed evaluation species than the chronic toxicity test species, then the predicted response at our concentration of interest of 5.2 ug/L for our listed evaluation species would be the same as the response observed for the chronic toxicity test species at a CN concentration 1.5 times 5.2 ug/L, which equals 7.8 ug/L. We refer to such predicted response values as surrogate currency equivalents (or $SSEC_x$ or $SS_x$ values) for our listed evaluation species. In this example, the predicted adverse effect for our chronic toxicity test species at the $SSEC_x$ of 7.8 ug/L would be our surrogate currency predicted effect for the listed evaluation species at 5.2 ug CN/L (for one of three prediction models) for the purposes of this Biological Opinion. A more detailed derivation and explanation of the $SSEC_x$/ $SS_x$ concept is provided in Appendix C.

Because groups of taxonomically related listed evaluation species were assigned identical $LC_{50}$ values from the same ICE or SSD model, there are only 17 $SSEC_x$ values that need to be evaluated for any given (chronic toxicity test species) prediction model, but they are different for each prediction model (3 models x 17 values = 51 total $SSEC_x$ values of interest).

For the prediction model based on fathead minnow chronic toxicity data, the $SSEC_x$ values range from 6.7 to 45.8 ug CN/L (Table 4). As indicated by the entire range of $SSEC_x$ values being greater than 5.2 ug CN/L, all listed evaluation species have $LC_{50}$ values that are more sensitive to cyanide than the fathead minnow $LC_{50}$ value.

For the prediction model based on brook trout chronic toxicity data, the $SSEC_x$ values range from 4.2 to 28.4 ug CN/L (Table 4).

For the prediction model based on bluegill chronic toxicity data, the $SSEC_x$ values range from 6.1 to 41.7 ug CN/L (Table 4).

The $SSEC_x$ ranges indicated above define for each prediction model the range of cyanide concentrations over which model fit will be of most relevance to the effects of the proposed action on the listed evaluation species considered in this Biological Opinion. Detailed $SSEC_x$ -related results and the origins of the $LC_{50}$ values used to calculate the $SSEC_x$ values are presented in Table 4 and Appendix D.

**Table 4. Surrogate currency equivalents (SSECx) for each LC50 surrogate taxon/chronic toxicity test species combination. SSECx values were calculated using equation 5 in Appendix C. Surrogate taxa were used to estimate LC50 values for listed evaluation species except when measured values for the listed species were available (i.e. *Salmo salar*).**

| | | | Effects on Fecundity | | Effects on Early Life Stage Survival |
|---|---|---|---|---|---|
| | | | Fathead Minnow SS $LC_{50}$=138.4 (ug CN/L) | Brook Trout SS $LC_{50}$=85.7 (ug CN/L) | Bluegill SS $LC_{50}$=126.1 (ug CN/L) |
| Surrogate taxa used to estimate listed species (LS) $LC_{50}$ | $LSEC_x$ (ug CN/L) | LS $LC_{50}$ (ug CN/L) | $SSEC_X$ (ug CN/L) | $SSEC_X$ (ug CN/L) | $SSEC_X$ (ug CN/L) |
| Actinopterygii (class) | 5.2 | 66.5[1] | 10.8 | 6.7 | 9.9 |
| Cypriniformes (order) | 5.2 | 84.55[1] | 8.5 | 5.3 | 7.8 |
| Family Catostomidae | | | | | |
| *Xyrauchen texanus* (species) | 5.2 | 83.8[2] | 8.6 | 5.3 | 7.8 |
| Cyprinidae (family) | 5.2 | 101.7[2] | 7.1 | 4.4 | 6.4 |
| *Cyprinella monacha* (species) | 5.2 | 36.81[2] | 19.6 | 12.1 | 17.8 |
| *Gila elegans* (species) | 5.2 | 50.9[2] | 14.1 | 8.8 | 12.9 |
| *Notropis mekistocholas* (species) | 5.2 | 48.5[2] | 14.8 | 9.2 | 13.5 |
| *Ptychocheilus lucius* (species) | 5.2 | 43.5[2] | 16.6 | 10.3 | 15.1 |
| Perciformes (order) | 5.2 | 90.8[1] | 7.9 | 4.9 | 7.2 |
| Percidae (family) | 5.2 | 42.3[2] | 17.0 | 10.5 | 15.5 |
| *Etheostoma* (genus) | 5.2 | 40.0[2] | 18.0 | 11.1 | 16.4 |
| *Etheostoma fonticola* (species) | 5.2 | 21.5[2] | 33.4 | 20.7 | 30.5 |
| Salmoniformes, Salmonidae | | | | | |

| | | | | | |
|---|---|---|---|---|---|
| *Oncorhynchus* (genus) | 5.2 | 47.0[2] | 15.3 | 9.5 | 13.9 |
| *O. apache* (species) | 5.2 | 16.5[2] | 43.6 | 27.0 | 39.7 |
| *O. clarki henshawi* (species) | 5.2 | 22.8[2] | 31.5 | 19.5 | 28.7 |
| *Salmo salar* (species) | 5.2 | 90[3] | 8 | 5 | 7.3 |
| *Salvelinus* (genus) | 5.2 | 15.7[2] | 45.8 | 28.4 | 41.7 |

[1] $LC_{50}$ based on 5th percentile estimate from species sensitivity distribution (SSD), Table 2 – Cyanide BE.
[2] $LC_{50}$ estimate based on lower bound of the 95% CI from ICE model (Appendix D).
[3] $LC_{50}$ based on measured value from the Cyanide BE (Table 1).

Prediction Models

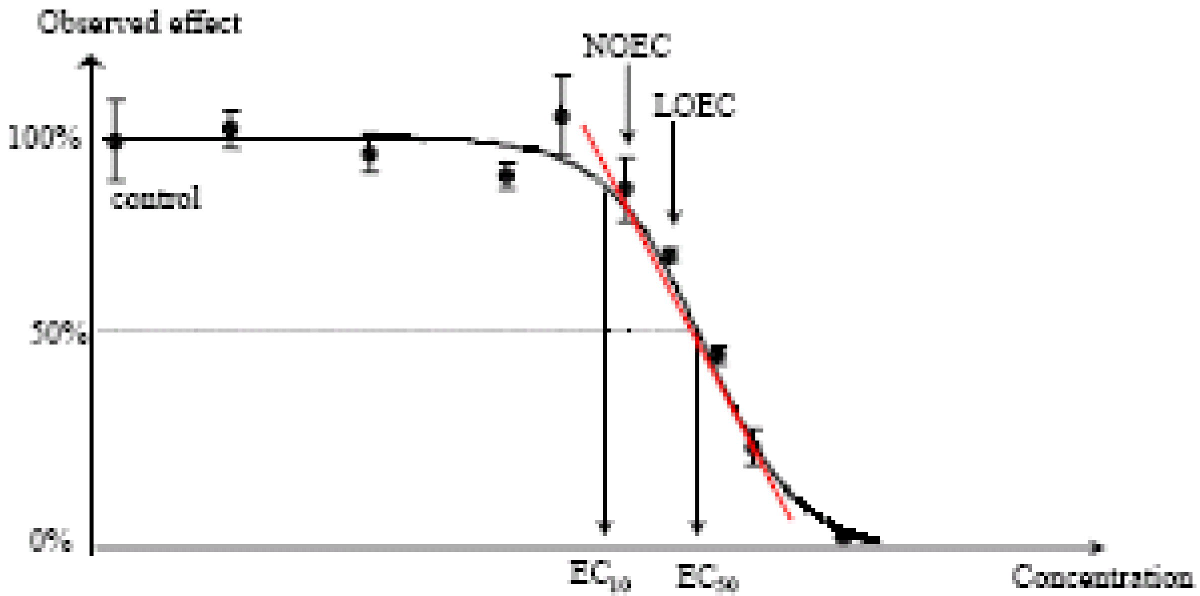
Statistical regression techniques were used to model or “fit” the relationship between cyanide concentrations and toxic effects to listed evaluation species based on data for the chronic toxicity test species. For nuances of statistical regression specific to toxicological applications, we relied substantively on two recent technical guidance documents: (1) Enivronment Canada (2005: “Guidance Document on Statistical Methods for Environmental Toxicity Tests”); and (2) OECD (2006: “Current Approaches in the Statistical Analysis of Ecotoxicity Data: A Guidance to Application”). We also reviewed other relevant guidance such as that provided in the documentation for EPA’s Toxicity Relationship Analysis Program (TRAP) (EPA 2002) and in discipline-specific statistical textbooks such as Gad and Weil (1988) and Sparks (2000).

As noted by Environment Canada (2005), an important principle of regression techniques is to keep the model simple, if that can reasonably be done. In completing this analysis, we have further incentive to follow that principle because we have a strong interest in evaluating the uncertainty (confidence) associated with point estimates and therefore an interest in avoiding what Environment Canada (2005) noted as the “...obstacle of calculating confidence intervals around nonlinear regression estimates...” Throughout this analysis we have been mindful of that because our models are not based on biological or chemical mechanisms of action, but are purely statistical constructs that have no mechanistic meaning. A statistical concentration-response model only serves to smooth the observed concentration-response to estimate effect concentrations by interpolating between treatment concentrations, and to provide a tool for assessing confidence intervals. Therefore, the choice of model is to some extent arbitrary (OECD 2006). That being noted, we constructed models that conformed to the non-arbitrary characteristics of the data we are working with and with statistical standard practices (such as data transformations). The degree of model fit achieved is an artifact of those specific decisions not the result of *post hoc* “model shopping” (EPA 2002).

Generic Concentration-reponse Relationship

Figure 1 illustrates a generic concentration-response relationship that typically takes on a sigmoidal form due to threshold effects on the low concentration end of the x-axis and to asymptotic effects at the high concentration end of the x-axis.

**Figure 2. Generalized concentration-response relationship adapted from OECD (2006:Figure 3.2). Note that the illustrated curve is a plot fitted to a real dataset, thus the identification of NOEC and LOEC concentrations. For the purposes of this figure, consider the y-axis as a positive attribute that becomes diminished by toxicity, such as percent survivorship.**




Note that the superimposed straight line in Figure 2 represents the region of concentrations that induce an intermediate toxic response that are well approximated by a linear fit. This "linear region" is strongest within one probit (also known as "normal equivalent deviate") either side of the median response concentration ($EC_{50}$), or roughly for concentrations that induce 16 to 84% response (Environment Canada 2005). The narrow ranges of $SSEC_x$ values that we need to evaluate can be expected to overwhelmingly fall within those boundaries as a result of the methods EPA used to set the chronic criterion at 5.2 ug CN/L; see the next section titled "Derivation of the Criterion Continuous Concentration (CCC)". Our approach is conceptually similar to the TRAP program's Piecewise Linear regression option (EPA 2002). Even with regard to the nonlinear regression options in TRAP, EPA (2002) provides a recommendation for segmented analysis when there is a focal region (or subset) of test concentrations of particular concern:

*Within the limitations of this program, one useful approach can be to exclude (censor) high effects data from the analysis if (a) only low levels of effect are of interest and (b) there are sufficient low-to-moderate effects data to support a good analysis.*

<u>Prediction Model based on the Fathead Minnow Dataset</u>

Lind et al. (1977) examined fathead minnow fecundity (number of eggs per spawn) and egg hatchability in relation to a series of cyanide treatments (concentrations). The experimental structure, as well as the fecundity results, is summarized in Table 5. There were five control replicates, and two replicates each for ten exposure concentrations. The response data are reasonably monotonic, especially within the intermediate response range covered by the lowest six treatments. Those treatments ranged (on a free cyanide basis) from 6 to 45.6 ug/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 4).

**Table 5. Egg production of adult fathead minnows exposed for 256 days (from larvae through adult) to various concentrations of cyanide (from Lind et al. 1977; Table II).**

| Treatment HCN (ug/L) | Mean HCN (ug/L) | Free cyanide as CN (ug/L) | Mean eggs per female | Mean eggs per female per treatment | Reduction in the number of eggs per female - percent of control |
|---|---|---|---|---|---|
| Control | | | 2530 | 3476 | |
| Control | | | 4483 | | |
| Control | | | 3990 | | |
| Control | | | 2718 | | |
| Control | | | 3660 | | |
| 5.7 | 5.8 | 6.0 | 1886 | 2512 | 27.7% |
| 5.9 | | | 3138 | | |
| 13.0 | 12.9 | 13.3[N] | 1701 | 1845 | 46.9% |
| 12.7 | | | 1989 | | |
| 19.6 | 19.6 | 20.2[L] | 1694 | 1468 | 57.8% |
| 19.6 | | | 1241 | | |
| 27.1 | 27.3 | 28.2 | 1093 | 1367 | 60.7% |
| 27.5 | | | 1640 | | |
| 36.0 | 35.8 | 36.9 | 678 | 1010 | 71.0% |
| 35.6 | | | 1341 | | |
| 43.7 | 44.2 | 45.6 | 2054 | 1124 | 67.7% |
| 44.7 | | | 194 | | |
| 62.5 | 63.5 | 65.6 | 74 | 72 | 97.9% |
| 64.5 | | | 70 | | |
| 73.1 | 72.8 | 75.1 | 573 | 319 | 90.8% |
| 72.4 | | | 64 | | |
| 81.5 | 80.7 | 83.3 | 266 | 243 | 93.0% |
| 79.8 | | | 219 | | |
| 96.1 | 100.8 | 103.9 | 0 | 0 | 100.0% |
| 105.4 | | | 0 | | |

[N] NOEC
[L] LOEC

To "build" our prediction model we transformed both the concentration data and the fecundity data for *a priori* reasons. We log-transformed the concentration data for two reasons: (1) statistically, toxicological tolerance distributions have long been confirmed as log-normal (OECD 2006); and (2) biologically, organisms experience toxicants on a log scale. Toxicological custom is to use log base-10 for the log transformations of test concentrations (Environment Canada 2005). Count data, such as “number of eggs per spawn” typically conform to a Poisson distribution rather than a normal distribution. To normalize such data for regression analysis a square-root transformation is recommended (EPA 2002). We used the square-root transformed response data for statistical analysis and then back-transformed the data for reporting results. This transformation does not change the model, but affects what the best parameter estimates and confidence limits are (EPA 2002). Thus, our model of choice is a log-square root linear regression over our focal segment (subset) of test concentrations.

In agreement with Gensemer et al.’s (2007) treatment of the same dataset, we collapsed the fecundity and egg hatchability endpoints into a single endpoint: “eggs hatched per spawn” which is the product of (eggs per spawn) x (egg hatchability) at each treatment concentration. We went a step further than Gensemer et al. (2007) and additionally applied a data-smoothing procedure to meet the assumption of monotonicity of response inherent in a linear regression. We did that by calculating three-point moving averages for both the fecundity and hatchability endpoints. This is a standard statistical technique for separating the “signal” from the “noise” in epidemiological and earth sciences (e.g., Rothman et al. 2008, Borradaile 2003).

**TABLE 6. Fathead minnow input data for effects modeling.**

| Treatment (free ug CN/L) | Pooled mean eggs/female | Pooled Proportion Hatch[a] | Unsmoothed Pooled mean hatch/female[b] | 3-pt moving average of proportion hatch | Smoothed Pooled mean hatch/female[b] | SQRT transform |
|---|---|---|---|---|---|---|
| | | | | | | |
| Control Mean | 3476 | 0.842 | 2927 | 0.763[c] | 2652 | 51.5 |
| 6.00 | 2512 | 0.606 | 1522 | 0.754 | 1894 | 43.52 |
| 13.30 | 1845 | 0.813 | 1500 | 0.682 | 1258 | 35.47 |
| 20.20 | 1468 | 0.626 | 919 | 0.612 | 898 | 29.97 |
| 28.20 | 1367 | 0.396 | 541 | 0.527 | 720 | 26.83 |
| 36.90 | 1010 | 0.559 | 565 | 0.354 | 358 | 18.92 |
| 45.60 | 1124 | 0.108 | 121 | 0.271 | 305 | 17.46 |
| 65.60 | 72 | 0.147 | 11 | 0.149 | 11 | 3.31 |
| 75.10 | 319 | 0.192 | 61 | 0.181 | 58 | 7.62 |
| 83.30 | 243 | 0.204 | 50 | 0.132 | 32 | 5.66 |
| 103.90 | 0 | 0 | 0 | 0.068[c] | 0 | 0 |

[a]Means weighted by replicate sample sizes; excludes hatchability result for Control B as per recommendation by Lind et al. (1977:264-265).

[b]Rounded to the nearest whole number.

[c]Based on double-weighted observed value; assuming any doses to the left of 0% response will be constant and any points to the right of 100% response will be constant.

[d]Final effects model based upon the shaded subset of data.

Although we didn’t use the control data in our focal segment linear regression, we estimated where the smoothed data would cross the y-axis by double-weighting the control value, which then (along with its nearest neighboring data point) provided the basis of a three-point moving average for the "endpoint" of the concentration series. This double-weighting is justified conceptually because a treatment to the left of the controls on the concentration axis would be expected to respond the same as the controls (Environment Canada 2005). This enabled us to avoid comparing point estimates of eggs hatched per spawn from models fitted to smoothed data with "unsmoothed" control reference points. Note that our "smoothed" estimate of a control reference point was obtained using the actual data nearest to the y-axis and is not extrapolated from our estimated regression equation. Also note that we did not control-adjust the results prior to model fitting, a practice that leads to serious upward bias in $EC_x$ point estimates (Environment Canada 2005, OECD 2006). A summary of response data smoothing and transformation is presented in Table 6.

The resulting log-square root focal segment linear regression model shows a very close fit to the data with an adjusted r-square of 0.964. The regression equation is:

Square-root (hatched eggs per spawn) = -30.19(LOG CN) + 68.36

The regression plot (Figure 3) and summary regression statistics (Table 7) are presented below. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples (six points in this case), we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

**Figure 3. Log- square root focal segment regression plot for fathead minnow fecundity x hatchability (= eggs hatched per spawn).**

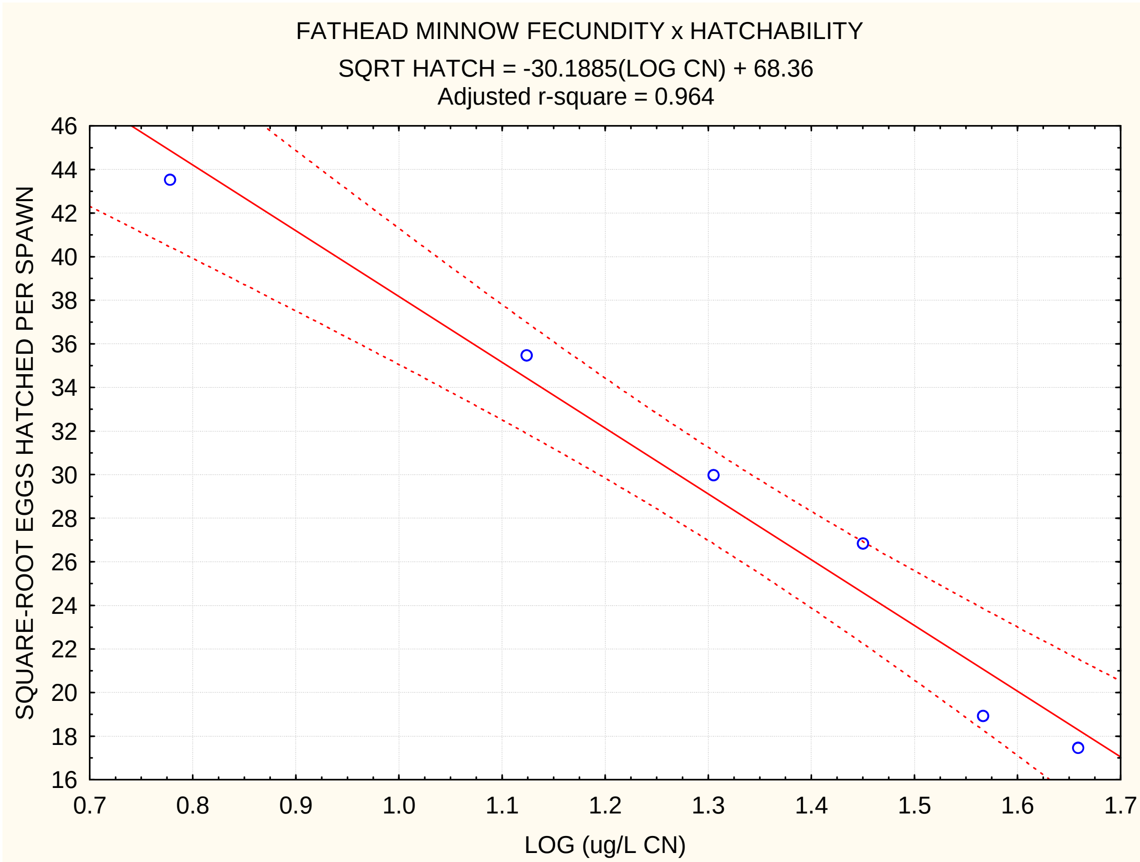



**TABLE 7. Summary regression statistics.**

| Effects Surrogate | N | F value | p-level | Intercept | Std Err | p-level | Slope | Std Err | p-level |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | |
| Fathead Minnow | 6 | 134.6 | <0.00032 | 68.36 | 3.505 | 0.000041 | -30.19 | 2.602 | 0.00032 |
| | | | | | | | | | |
| Brook Trout | 5 | 12.34 | <.039 | 24.85 | 2.595 | 0.0024 | -6.594 | 1.877 | 0.039 |
| | | | | | | | | | |
| Bluegill | 5 | 11.75 | <0.042 | 0.3514 | 0.9277 | 0.73 | -2.533 | 0.7919 | 0.042 |

<u>Prediction Model based on the Brook Trout Dataset</u>

Koenst et al. (1977) examined brook trout fecundity (number of eggs per spawn) and egg viability in relation to a series of cyanide treatments (concentrations). The experimental structure, as well as the fecundity results, is summarized below (Table 8). There were two control replicates, and seven cyanide treatments. The lowest five treatments produced intermediate effects responses and covered a range of concentrations from 5.6 to 53.2 ug/L

CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 4). There was substantive variability in the results for the two control replicates. This lead to Koenst et al. (1977) excluding control replicate B, but noting that additional testing might indicate that the control results should be averaged. As noted in the footnote to Table 8, subsequent studies with brook trout (Holcombe et al. 2000) have confirmed that control replicate B should be averaged with control replicate A. For that reason, we used the control mean as our reference point for evaluating model predictions.

**Table 8. Egg production of adult brook trout exposed to HCN for 144 days prior to the start of spawning (from Koenst et al., 1977).**

| HCN (ug/L) | Free cyanide as CN (ug/L) | Mean eggs spawned per female | Reduction in the number of eggs per female - percent of control* |
|---|---|---|---|
| Control A | | 502 | |
| Control B | | 744 | |
| Control Mean | | 623 | |
| 5.7 | 5.6 | 513 | 17.7% |
| 11.2 | 11.1 | 291 | 53.3% |
| 32.3 | 31.9 | 246 | 60.5% |
| 43.6 | 43.1 | 442 | 29.1% |
| 53.9 | 53.2 | 262 | 57.9% |
| 64.9 | 64.1 | 124 | 80.1% |
| 75.3 | 74.4 | 0 | 100.0% |

* Reductions in the number of eggs spawned relative to controls were calculated using the Control mean (623 eggs per female). Koenst et al. (1977) performed the same calculation using only Control A (502 eggs per female) and reported that the MATC (Maximum Acceptable Toxicant Concentration) lies between 5.7 and 11.2 ug HCN/L. However, the authors went on to say that "When compared to the mean of the two controls, 5.7 ug/L HCN would appear to show a substantial reduction in eggs spawned per female, but due to the high variability in spawning in the two controls, further study would be required to reach this conclusion." Since that time other studies with brook trout have been conducted (Holcombe et al. 2000). The mean number of eggs spawned per female observed by Koenst et al. (1977) is within the range reported for these other studies, which supports the use of data from both controls in estimating the effect of cyanide on brook trout fecundity.

Again, in agreement with Gensemer et al.'s (2007) treatment of the same dataset, we collapsed the fecundity and egg viability endpoints into a single endpoint: "viable eggs per spawn" which is the product of (eggs per spawn) x (egg viability) at each treatment concentration. In the five-point segment of the data that we focus on, there was a substantive deviation from monotonicity at the 43.1 ug/L CN concentration. Therefore, once again we employed data-smoothing with a 3-point moving average to restore a monotonic progression of responses. Because the endpoint here is virtually the same as the endpoint for the fathead minnow dataset, other aspects of our treatment of the data for "building" a prediction model are the same as previously presented for the fathead minnow model. A summary of response data-smoothing and transformation is presented in Table 9 below.

**TABLE 9. Brook trout input data for effects modeling.**

| Treatment (free CN ug/L) | Mean eggs/female | 3-pt moving average of mean eggs/spawn | Proportio n Viable | 3-pt moving average of proportion viable | Smoothed mean viable/female[a] | SQRT transform |
|---|---|---|---|---|---|---|
| | | | | | | |
| Control Mean | 623 | 586[b] | 0.935 | 0.923[b] | 541 | 23.26 |
| 5.60 | 513 | 476 | 0.899 | 0.872 | 415 | 20.37 |
| 11.10 | 291 | 350 | 0.781 | 0.803 | 281 | 16.76 |
| 31.90 | 246 | 326 | 0.729 | 0.792 | 258 | 16.06 |
| 43.10 | 442 | 317 | 0.866 | 0.745 | 236 | 15.36 |
| 53.20 | 262 | 276 | 0.641 | 0.502 | 139 | 11.79 |
| 64.10 | 124 | 129 | 0 | 0.214 | 28 | 5.29 |
| 74.40 | 0 | 41[b] | 0 | 0[b] | 0 | 0 |

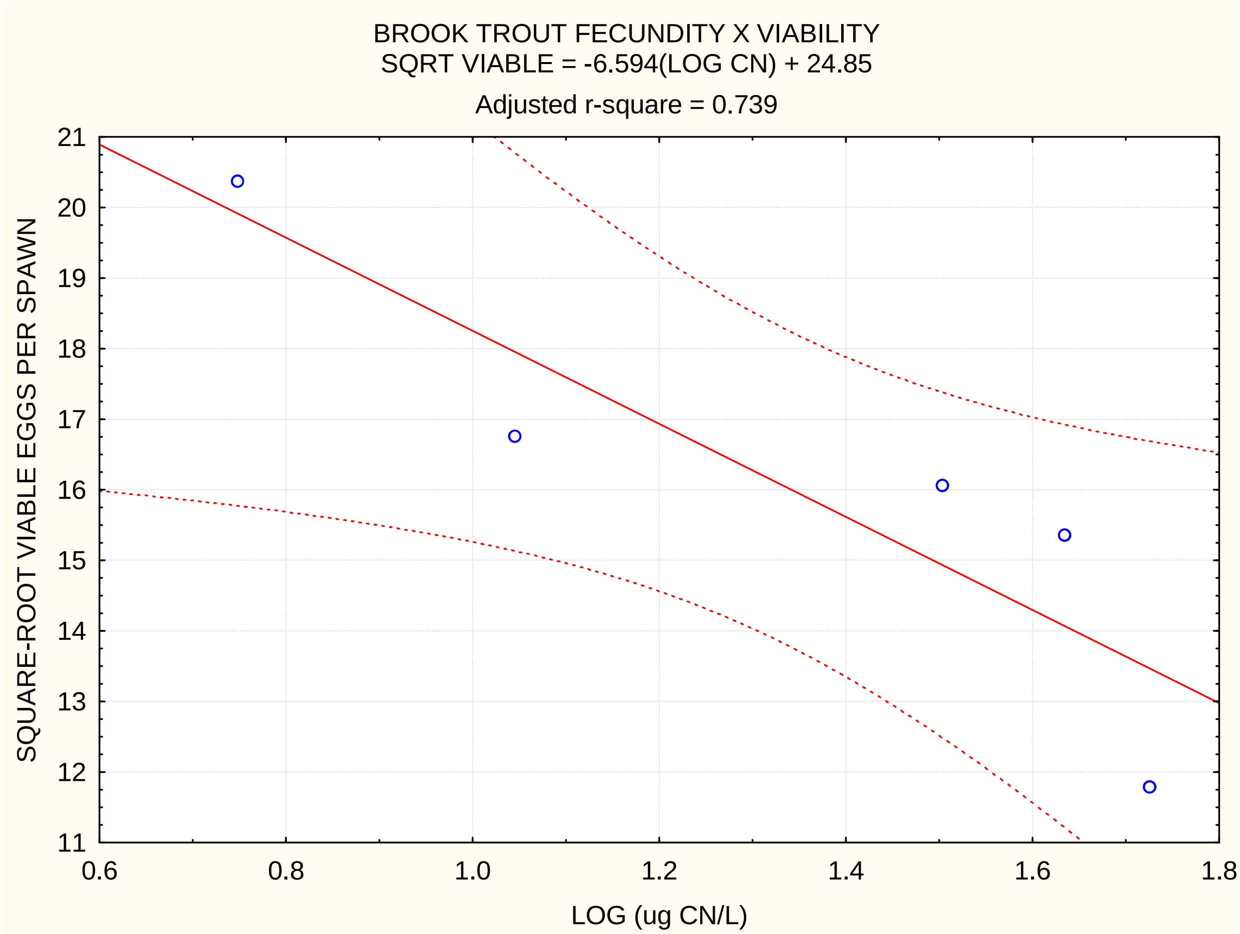
[a]Rounded to the nearest whole number.
[b]Based on double-weighted observed value; assuming any doses to the left of 0% response will be constant and any points to the right of 100% response will be constant.
[c]Final effects model based upon the shaded subset of data.

The resulting log-square root focal segment linear regression model does not show as strong a fit to the data as the fathead minnow model does, but still shows a reasonably good fit with an adjusted r-square of 0.739. The regression equation is:

Square-root (viable eggs per spawn) = -6.594(LOG CN) + 24.85

The regression plot is presented in Figure 4 and summary regression statistics are presented in Table 7. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples, i.e., five points in this case, we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

**<u>Figure 4.</u> Log-square root focal segment regression plot for brook trout fecundity x viability (= viable eggs per spawn).**




<u>Prediction Model based on the Bluegill Dataset</u>

Kimball et al. (1978) examined bluegill juvenile survivorship in relation to a series of cyanide treatments (concentrations). The experimental structure, as well as the survivorship results, is summarized in Table 10. There were four control replicates, and two replicates each for eight cyanide treatments. The lowest five treatments produced intermediate effects responses and covered a range of concentrations from 4.9 to 40.6 ug/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 4).

**Table 10.** **Survival of bluegills from fertilized egg to the 57-day juvenile state in various HCN concentrations (from Kimball et al. 1978; Table 3).**

| HCN (ug/L) | Mean HCN (ug/L) | Free cyanide as CN (ug/L) | Percent survival | Number of surviving juveniles * | Mean percent survival | Reduction in survival compared to controls |
|---|---|---|---|---|---|---|
| Control | | | 37.5 | 75 | 23.3 | |
| Control | | | 20.0 | 40 | | |
| Control | | | 10.0 | 20 | | |
| Control | | | 25.5 | 51 | | |
| 4.8 | 4.8 | 4.9 | 18.5 | 37 | 18.5 | 20.6% |
| 5.2 | | | lost | | | |
| 8.9 | 9.1 | 9.4[N] | 25.0 | 50 | 16.3 | 30.0% |
| 9.2 | | | 7.5 | 15 | | |
| 19.2 | 19.4 | 19.9[L] | 3.0 | 6 | 2.8 | 88.0% |
| 19.6 | | | 2.5 | 5 | | |
| 28.5 | 29.1 | 29.9 | 2.5 | 5 | 2.5 | 89.3% |
| 29.7 | | | 2.5 | 5 | | |
| 38.7 | 39.5 | 40.6 | 3.0 | 6 | 3.8 | 83.7% |
| 40.2 | | | 4.5 | 9 | | |
| 49.3 | 49.3 | 50.7 | 13.5 | 27 | 13.5 | 42.1% |
| 51.9 | | | lost | | | |
| 61.8 | 62.9 | 64.6 | 0.0 | 0 | 0.0 | 100.0% |
| 64 | | | 0.0 | 0 | | |
| 80.4 | 82.1 | 84.4 | 0.0 | 0 | 0.0 | 100.0% |
| 83.8 | | | 0.0 | 0 | | |

* Number of surviving juveniles was calculated by multiplying the reported percent survival times the starting number of fertilized eggs per treatment (200).

[N] NOEC

[L] LOEC

The bluegill dataset differs qualitatively from the fathead minnow and brook trout datasets because the response variable, juvenile survivorship, is a quantal (binary) rather than continuous variable. Quantal variables conform to a binomial distribution. Such data are typically analyzed via either probit transformation, as employed by Gensemer et al. (2007), or logit transformation of the proportions of responding and non-responding test subjects. Probits are normal equivalent deviates and logits are logistic equivalent deviates. These two transforms usually yield similar estimates of $EC_{50}$ values, but differ appreciably in their EC estimates in the tails of the distributions.

Environment Canada (2005) recommends logistic methods over probits for "... mathematical simplicity and other good reasons." Logit = ln (p/1-p), where p is the proportion of effected test subjects (e.g., if juvenile survival were 30% for a particular treatment concentration, p would equal 0.3 and the logit transform would equal -0.8473).

The logit transform linearizes the sigmoidal logistic response curve (Environment Canada 2005, StatSoft 2006). Furthermore, in fitting the logit model, the control observations can be excluded, as they do not provide any information, unless a background parameter is included (OECD 2006).

Both Environment Canada (2005) and OECD (2006) note that it is common practice to correct the data for background response prior to analysis (for example via Abbott's correction), but that such pre-treatment of the data is unsound statistical practice that can result in substantive overestimation of $EC_x$ values. The bias increases as the control effect being adjusted for increases. We fit a focal segment of the bluegill dataset to a log-logit regression using results that were not control-adjusted prior to analysis. Thus, our prediction model yields unbiased estimates of proportion effect that can be control-adjusted for reporting purposes after-the-fact. The dataset is reasonably monotonic until the highly anomalous result for the treatment at a concentration of 50.7 ug/L CN. Gensemer et al. (2007) censored that point as an outlier. Because our $SSEC_x$ range extended up to only 41.7 ug/L CN (Table 4) the 50.7 ug/L CN treatment did not fall within our focal segment of concern. The last three treatments in our focal segment produced results of greater than 84% effect which would place them in the nonlinear upper tail of the sigmoidal curve (Figure 2), but unlike a log-square root regression the logit transform will linearize points in the tails relative to intermediate effect points. Thus, for log-logit regression points that fall in tails do not have to be avoided in order to apply linear regression. The minor deviation from monotonicity in the last two points of our focal segment did not warrant data-smoothing. A summary of the logit transformed response data is presented in Table 11.

**TABLE 11. Bluegill input data for effects modeling.**

| Treatment (free CN ug/L) | Mean surviving juveniles | Proportion Survival | Logit Proportion Survival |
|---|---|---|---|
| | | | |
| Control Mean | 46.5 | 0.2325 | -1.1942 |
| 4.9 | 37 | 0.1850 | -1.4828 |
| 9.4 | 32.5 | 0.1630 | -1.6361 |
| 19.9 | 5.5 | 0.0280 | -3.5472 |
| 29.9 | 5 | 0.0250 | -3.6636 |
| 40.6 | 7.5 | 0.0380 | -3.2314 |
| 50.7 | 27 | 0.1350 | -1.8575 |
| 64.6 | 0 | 0.0000 | |
| 84.4 | 0 | 0.0000 | |

[a]Final effects model based upon the shaded subset of data.

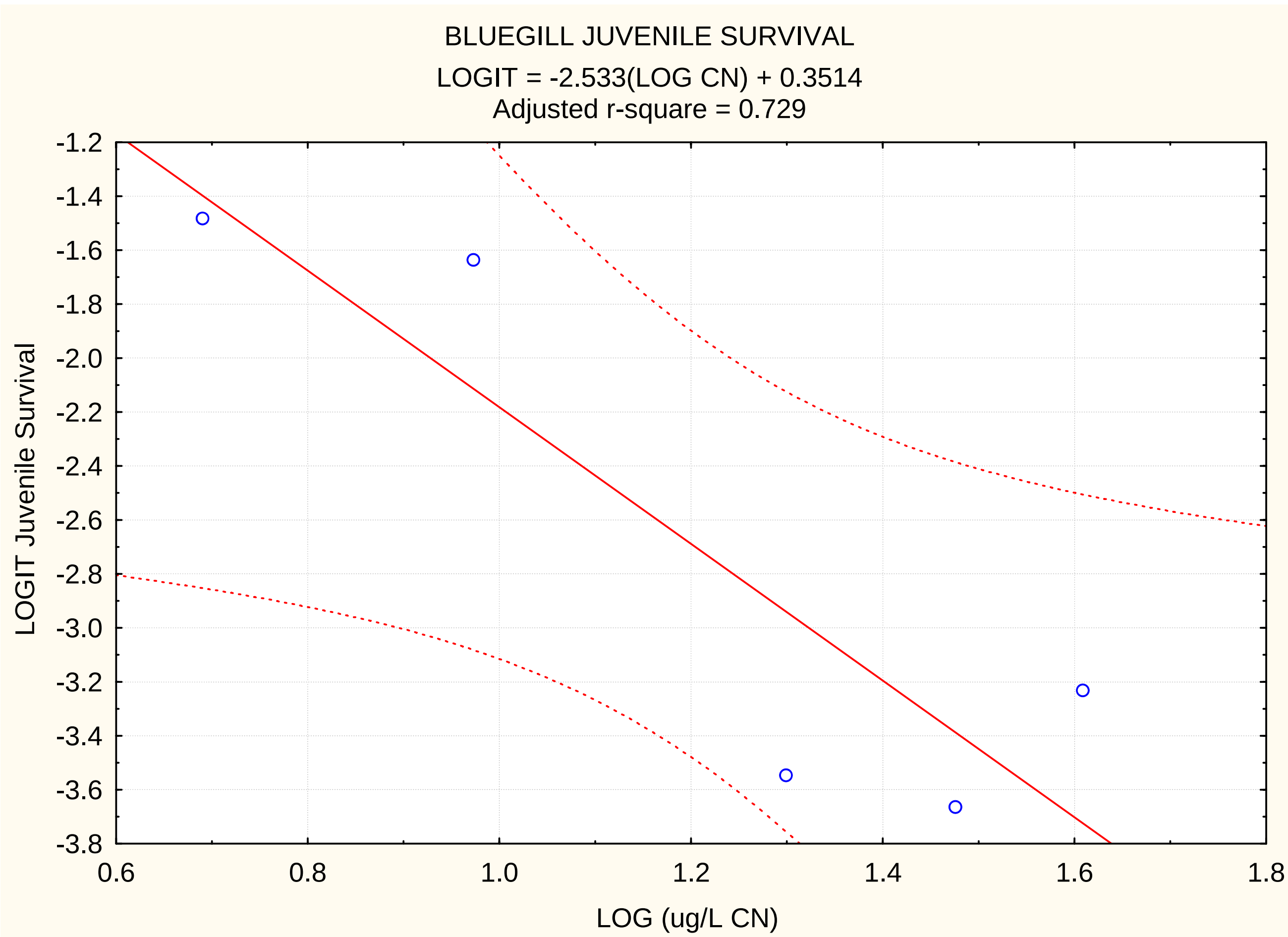
The resulting log-logit focal segment linear regression model does not show as strong a fit to the data as the fathead minnow model does, but with an adjusted r-square of 0.729

shows a reasonably good fit comparable to that achieved for the brook trout dataset. The regression equation is:

Logit (proportion juvenile survival) = -2.533 (LOG CN) + 0.3514

The regression plot is presented in Figure 5 and summary regression statistics are presented in Table 7. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples (five points in this case), we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

**FIGURE 5. Log-logit focal segment regression plot for bluegill juvenile survival.**




<u>Prediction Results</u>

Effects predictions are generated by substituting LOG ($SSEC_x$) for LOG (CN) into the prediction regression equations. This was accomplished via the "predict dependent variable" algorithm in the multiple linear regression module of Statistica (StatSoft 2006). That algorithm also uses the estimated standard error of the regression coefficient to generate 95% confidence limits for the predicted point estimates (maximum likelihood estimates). For the fathead minnow and brook trout prediction regressions, the prediction and confidence limit output are in the form of square-roots of numbers of eggs. To convert those predictions to a percent effect, the predicted results were first squared and then scaled for percent change compared to the applicable smoothed control value according to the formula:

% Effect = [1- (predicted egg count / smoothed control value)] x 100

Any predicted egg counts exceeding the smoothed control value were automatically converted to 0% effect. For the bluegill prediction regression, the prediction and confidence limit output are in the form of logit transforms for proportions of juvenile survivorship. The logit transforms are back-transformed to proportions by the formula:

$$\text{Proportion survival} = e^{(\text{logit})} / 1 + e^{(\text{logit})}$$

The predicted survival proportions are scaled for percent change compared to the reported control value according to the formula:

% Effect = [1- (predicted proportion survival / mean control proportion survival)] x 100

Again, any predicted survivorship exceeding the observed mean control survivorship results in a percent effect prediction that is automatically converted to 0% effect. The raw input and output data for effects predictions are presented in Appendix E.

A summary of predicted effects and their estimated 95% confidence limits from each of the three prediction models for each of the 17 surrogate taxa from which listed evaluation species' $LC_{50}$ values were derived are presented in Table 12. The effects estimates are presented in Table 13 for the listed evaluation species based on matching up the effects estimates for surrogate taxa in Table 12 with the listed species linked to each surrogate taxon.

The $EC_{10}$ and $EC_{20}$ concentrations for each of our three regression models were also estimated.

The fathead minnow regression yielded an estimated $EC_{10}$ of 4.4 ug/L CN (95% CI = 2.6-6.2 ug/L CN) and an estimated $EC_{20}$ of 5.5 ug/L CN (95% CI = 3.5-7.4). By comparison, Gensemer et al. (2007) estimated an $EC_{20}$ of 6.0 ug/L CN from a log-probit analysis of the fathead minnow data, but did not report confidence limits for that estimate.

The brook trout regression yielded an estimated $EC_{10}$ of 2.6 ug/L CN (95% CI = 0.0-8.4 ug/L CN) and an estimated $EC_{20}$ of 4.1 ug/L CN (95% CI = 0.0-11.1). Gensemer et al. (2007) estimated an $EC_{20}$ of 7.7 ug/L by linear interpolation of the brook trout data, and again did not report confidence limits for that estimate. It is important to note that, for their analysis, Gensemer et al. (2007) did not average the controls, as we did, but used the control which produced the lowest number of eggs (refer to the Table 8 footnote for more details).

The bluegill regression yielded an estimated $EC_{10}$ of 4.6ug/L CN (95% CI = 0.0-10.5 ug/L CN) and an estimated $EC_{20}$ of 5.3 ug/L CN (95% CI = 0.0-11.5). Gensemer et al. (2007) estimated an $EC_{20}$ of 5.6 ug/L CN from a log-probit analysis of the bluegill data, and also estimated an $EC_{20}$ of 8.9 ug/L CN for the bluegill data from EPA's TRAP program.

All of Gensemer et al.'s (2007) estimates fall within our 95% confidence limits, and in general show excellent agreement with our results even though Gensemer et al's methods differed from ours. This suggests that our results are not highly dependent on the particular statistical approach that we chose for our analysis.

**Table 12. Estimated magnitude of effect of cyanide (at the CCC, 5.2 ug CN/L) on surrogate taxa for listed fish species (± 95% CL). The magnitude of effect was estimated using the regression model for each surrogate response species and SS $EC_X$ value for each surrogate taxa (Table 1). For each surrogate taxa there were two estimates of effects on reproductive performance and one estimate of effects on early life stage survival.**

| | Surrogate species | | |
|---|---|---|---|
| | Fathead Minnow | Brook Trout | Bluegill |
| Surrogate taxa used to estimate magnitude of effect on listed species | Reduction in the mean number of hatched eggs per spawn compared to controls | Reduction in the mean number of viable eggs per spawn compared to controls | Reduction in the number of surviving larvae/juveniles compared to controls |
| Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| Cypriniformes (order) | 39% (28%, 49%) | 26% (0%, 54%) | 44% (0%, 80%) |
| Family Catostomidae | | | |
| *Xyrauchen texanus* (species) | 39% (28%, 49%) | 26% (0%, 54%) | 44% (0%, 80%) |
| Cyprinidae (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| *Cyprinella monacha* (species) | 68% (63%, 72%) | 42% (23%, 58%) | 76% (50%, 89%) |
| *Gila elegans* (species) | 57% (51%, 63%) | 36% (12%, 56%) | 66% (30%, 84%) |
| *Notropis mekistocholas* (species) | 59% (53%, 65%) | 37% (14%, 56%) | 68% (34%, 85%) |
| *Ptychocheilus lucius* (species) | 63% (57%, 68%) | 39% (18%, 57%) | 71% (41%, 86%) |
| Perciformes (order) | 36% (24%, 47%) | 24% (0%, 53%) | 40% (0%, 79%) |
| Percidae (family) | 63% (58%, 68%) | 39% (18%, 57%) | 72% (43%, 87%) |
| *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |

| Surrogate taxa used to estimate magnitude of effect on listed species | Surrogate species | | |
|---|---|---|---|
| | Fathead Minnow | Brook Trout | Bluegill |
| | Reduction in the mean number of hatched eggs per spawn compared to controls | Reduction in the mean number of viable eggs per spawn compared to controls | Reduction in the number of surviving larvae/juveniles compared to controls |
| *Etheostoma fonticola* (species) | 81%<br>(76%, 85%) | 52%<br>(37%, 64%) | 86%<br>(64%, 95%) |
| Order Salmoniformes, Family Salmonidae | | | |
| *Oncorhynchus* (genus) | 60%<br>(54%, 65%) | 37%<br>(15%, 57%) | 69%<br>(36%, 85) |
| *Oncorhynchus apache* (species) | 87%<br>(82%, 91%) | 56%<br>(42%, 68%) | 90%<br>(67%, 97%) |
| *Oncorhynchus clarki henshawi* (species) | 80%<br>(75%, 84%) | 51%<br>(36%, 63%) | 85%<br>(63%, 94%) |
| *Salmo salar* (species) | 36%<br>(24%, 47%) | 24%<br>(0%, 54%) | 41%<br>(0%, 79%) |
| *Salvelinus* (genus) | 87%<br>(83%, 92%) | 57%<br>(43%, 69%) | 90%<br>(68%, 97%) |

**Table 13. Estimated magnitude of effect of cyanide (at the CCC, 5.2 ug CN/L) on listed fish species (95% CL). There are two estimates for effects on fecundity and one estimate for effects on early life stage survival. Estimates are based on analyses using surrogate taxa (Table 12). Surrogate taxa were used to estimate $LC_{50}$'s for listed species Surrogate response species (fathead minnow, brook trout, bluegill) datasets were used to estimate magnitude of chronic effects.**

| Listed Species | | Order/Family | Surrogate Taxa | Estimated reduction in fecundity and larvae/juvenile survival due to cyanide exposure (5.2 ug/L) based on surrogate species data sets. | | |
|---|---|---|---|---|---|---|
| | | | | Fathead Minnow (Reduction in the mean number of hatched eggs per spawn compared to controls) | Brook Trout (Reduction in the mean number of viable eggs per spawn compared to controls) | Bluegill (reduction in the number of surviving larvae/juveniles compared to controls) |
| Gulf sturgeon | *Acipenser oxyrinchus desotoi* | Acipenseriformes Acipenseridae | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| Kootenai River white sturgeon | *Acipenser transmontanus* | | | | | |
| Pallid sturgeon | *Scaphirhynchus albus* | | | | | |
| Alabama sturgeon | *Scaphirhynchus suttkusi* | | | | | |
| Waccamaw silverside | *Menidia extensa* | Atheriniformes Atherinopsidae | | | | |
| Modoc sucker | *Catostomus microps* | Cypriniformes Catostomidae | Cypriniformes (order) | 39% (28%, 49%) | 26% (0%, 54%) | 44% (0%, 80%) |
| Santa Ana sucker | *Catostomus santaanae* | | | | | |
| Warner sucker | *Catostomus warnerensis* | | | | | |
| Shortnose sucker | *Chasmistes brevirostris* | | | | | |
| Cui ui | *Chasmistes cujus* | | | | | |
| June sucker | *Chasmistes liorus* | | | | | |
| Lost River sucker | *Deltistes luxatus* | | | | | |
| Razorback sucker | *Xyrauchen texanus* | | *Xyrauchen texanus* | 39% (28%, 49%) | 26% (0%, 54%) | 44% (0%, 80%) |
| Spotfin chub | *Cyprinella monacha* | Cypriniformes Cyprinidae | *Cyprinella monacha* | 68% (63%, 72%) | 42% (23%, 58%) | 76% (50%, 89%) |
| Blue shiner | *Cyprinella caerulea* | | Cyprinidae (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| Beautiful shiner | *Cyprinella formosa* | | | | | |
| Devils River minnow | *Dionda diaboli* | | | | | |
| Slender chub | *Erimystax cahni* | | | | | |
| Mohave tui chub | *Gila bicolor mohavensis* | | | | | |
| Owens tui chub | *Gila bicolor snyderi* | | | | | |
| Borax Lake chub | *Gila boraxobius* | | | | | |
| Humpback chub | *Gila cypha* | | | | | |
| Sonora chub | *Gila ditaenia* | | | | | |
| Gila chub | *Gila intermedia* | | | | | |
| Pahranagat roundtail chub | *Gila robusta jordani* | | | | | |
| Virgin River chub | *Gila robusta seminuda* | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| Rio Grand silvery minnow | *Hybognathus amarus* | | | | | |
| Big Spring spinedace | *Lepidomeda mollispinis pratensis* | | | | | |
| Little Colorado spinedace | *Lepidomeda vittata* | | | | | |
| Spikedace | *Meda fulgida* | | | | | |
| Moapa dace | *Moapa coriacea* | | | | | |
| Palezone shiner | *Notropis albizonatus* | | | | | |
| Cahaba shiner | *Notropis cahabae* | | | | | |
| Arkansas River shiner | *Notropis girardi* | | | | | |
| Pecos bluntnose shiner | *Notropis simus pecosensis* | | | | | |
| Topeka shiner | *Notropis Topeka* | | | | | |
| Oregon chub | *Oregonichthys crameri* | | | | | |
| Blackside dace | *Phoxinus cumberlandensis* | | | | | |
| Woundfin | *Plagopterus agrentissimus* | | | | | |
| Ash Meadows speckled dace | *Rhinichthys osculus nevadensis* | | | | | |
| Kendall Warm Springs dace | *Rhinichthys osculus thermalis* | | | | | |
| Loach minnow | *Tiaroga cobitis* | | | | | |
| Bonytail chub | *Gila elegans* | | *Gila elegans* | 57% (51%, 63%) | 36% (12%, 56%) | 66% (30%, 84%) |
| Cape Fear shiner | *Notropis mekistocholas* | | *Notropis mekistocholas* | 59% (53%, 65%) | 37% (14%, 56%) | 68% (34%, 85%) |
| Colorado pikeminnow | *Ptychocheilus lucis* | | *Ptychocheilus lucis* | 63% (57%, 68%) | 39% (18%, 57%) | 71% (41%, 86%) |
| White River springfish | *Crenichthys baileyi baileyi* | Cyprinodontiformes Goodeidae | Actinopterygii (class**)** | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| Hiko White River springfish | *Crenichthys baileyi grandis* | | | | | |
| Railroad Valley springfish | *Crenichthys nevadae* | | | | | |
| Big Bend gambusia | *Gambusia gaigei* | Cyprinodontiformes Poeciliidae | | | | |
| San Marcos gambusia | *Gambusia georgei* | | | | | |
| Clear Creek gambusia | *Gambusia heterochir* | | | | | |
| Pecos gambusia | *Gambusia nobilis* | | | | | |
| Gila topminnow (including Yaqui) | *Poeciliopsis occidentalis* | | | | | |
| Unarmoned threespine stickleback | *Gasterosteus aculeatus williamsoni* | Gasterosteiformes Gasterosteidae | | | | |
| Delta smelt | *Hypomesus transpacificus* | Osmeriformes Osmeridae | | | | |
| Tidewater goby | *Eucyclogobius newberryi* | Perciformes Gobiidae | Perciformes | 36% (24%, 47%) | 24% (0%, 53%) | 40% (0%, 79%) |
| Slackwater darter | *Etheostoma boschungi* | Perciformes Percidae | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| Vermilion darter | *Etheostoma chermocki* | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| Relict darter | *Etheostoma chienense* | | | | | |
| Etowah darter | *Etheostoma etowahae* | | | | | |
| Niangua darter | *Etheostoma nianguae* | | | | | |
| Watercress darter | *Etheostoma nuchale* | | | | | |
| Okaloosa darter | *Etheostoma okaloosae* | | | | | |
| Duskytail darter | *Etheostoma percnurum* | | | | | |
| Bayou darter | *Etheostoma rubrum* | | | | | |
| Cherokee darter | *Etheostoma scotti* | | | | | |
| Maryland darter | *Etheostoma sellare* | | | | | |
| Bluemask darter | *Etheostoma sp.* | | | | | |
| Boulder darter | *Etheostoma wapiti* | | | | | |
| Fountain darter | *Etheostoma fonticola* | | *Etheostoma fonticola* (species) | 81% (76%, 85%) | 52% (37%, 64%) | 86% (64%, 95%) |
| Amber darter | *Percina antesella* | | Percidae (family) | 63% (58%, 68%) | 39% (18%, 57%) | 72% (43%, 87%) |
| Goldline darter | *Percina aurolineata* | | | | | |
| Conasauga logperch | *Percina jenkinsi* | | | | | |
| Leopard darter | *Percina pantherina* | | | | | |
| Roanoke logperch | *Percina rex* | | | | | |
| Snail darter | *Percina tanasi* | | | | | |
| Ozark cavefish | *Amblyopsis rosae* | Percopsiformes Amblyopsidae | Actinopterygii (class**)** | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| Alabama cavefish | *Spleoplatyrhinus poulsoni* | | | | | |
| Little Kern golden trout | *Oncorhynchus aguabonita whitei* | Salmoniformes Salmonidae | *Oncorhynchus* (genus) | 60% (54%, 65%) | 37% (15%, 57%) | 69% (36%, 85) |
| Paiute cutthroat trout | *Oncorhynchus clarki seleniris* | | | | | |
| Greenback cutthroat trout | *Oncorhynchus clarki stomias* | | | | | |
| Gila trout | *Oncorhynchus gilae* | | | | | |
| Apache trout | *Oncorhynchus apache* | | *Oncorhynchus apache* (species) | 87% (82%, 91%) | 56% (42%, 68%) | 90% (67%, 97%) |
| Lahontan cutthroat trout | *Oncorhynchus clarki henshawi* | | *Oncorhynchus clarki henshawi* (species) | 80% (75%, 84%) | 51% (36%, 63%) | 85% (63%, 94%) |
| Atlantic salmon | *Salmo salar* | | *Salmo salar* | 36% (24%, 47%) | 24% (0%, 54%) | 41% (0%, 79%) |
| Atlantic salmon | | | | | | |
| Bull trout | *Salvelinus confluentus* | | *Salvelinus* (genus) | 87% (83%, 92%) | 57% (43%, 69%) | 90% (68%, 97%) |
| Pygmy sculpin | *Cottus paulus* | Scorpaeniformes Cottidae | Actinopterygii (class**)** | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |

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Other Effects Estimates

The estimates of effects presented in Table 13 are based largely on ICE LCL (lower confidence limit) $LC_{50}$ values for listed fish evaluation species. Those are the $LC_{50}$ values that we accept as sufficiently accounting for the uncertainties inherent in relying on surrogate data and numerous other untested assumptions to estimate the sensitivity of listed species to cyanide. The Service, NMFS, and EPA agreed that using ICE LCL values was preferable to the practice of applying arbitrary uncertainty factors.

However, EPA has, at various times, questioned whether the use of ICE LCL values might not be overly conservative. Therefore, we also estimated effect levels using ICE MLE (maximum likelihood estimates) $LC_{50}$ values for listed fish evaluation species (via revised $SSEC_x$ estimates). Those results are presented in Appendix F. Based on the fathead minnow prediction model, which was the strongest model, the median levels of effect predicted for the 15 ICE surrogate taxa were 51% and 65%, respectively, for ICE MLE and ICE LCL. The number of surrogate taxa with a predicted effect of 35% or greater was 11 and 14, respectively, for ICE MLE and ICE LCL. Those differences indicate only modest conservatism conferred by ICE LCL-based effects estimates as compared to ICE MLE-based estimates. Such modest differences would not have a decision-making impact. For both sets of results, unacceptably high levels of effect would overwhelmingly be the predominant prediction.

Empirical Test of Method Performance

Because only three concentration-response datasets are available, there is almost no basis for testing our method performance (i.e., there are no known directly measured “true” values for effects to our listed fish evaluation species at a concentration of 5.2 ug/L CN). However, because the fathead minnow and brook trout datasets focused on essentially the same response variable (number of hatchable/viable eggs produced per spawn) we can perform two tests of method performance. For each species, we can directly estimate a predicted effect level at 5.2 ug/L CN using the species-specific regressions. Those would be our estimates of the “true” effect level. Next, we can use our surrogate method and estimate an $SSEC_x$ for each species on the other species’ response curve and evaluate the predicted effect level for that $SSEC_x$ value and compare the surrogate estimate to the estimated “true” value. The results are as follows:

The directly estimated fathead minnow effect level at 5.2 ug/L CN is 18% with a 95% CI of 0%-34%. The fathead minnow $SSEC_x$ value on the brook trout response curve would be 3.2 ug/L CN, which yields an effects estimate of 15%. That is nearly identical to estimated “true” value and easily within the 95% CI for the “true value”.

The directly estimated brook trout effect level at 5.2 ug/L CN is 25% with a 95% CI of 0%-54%. The brook trout $SSEC_x$ value on the fathead minnow response curve would be 8.4 ug/L CN, which yields an effects estimate of 38%. Again, that is within the 95% CI for the “true” value, although our estimate of the “true” value is not very precise and therefore the 95% CI is fairly wide.

In summary, in both test cases, the estimated effect level derived from our surrogate methodology is not significantly different from the estimated "true" value in a statistical sense, but the second comparison has low statistical power. Further validation testing of this sort should be done as concentration-response datasets become available for more species using a comparable response variable, but it is reassuring that in these test cases our method yielded results that were nearly identical to the "true" value in one case and reasonably close to the "true" value in the other case.

*Derivation of the Criterion Continuous Concentration (CCC)*

Our analysis predicts that the listed fish evaluation species considered in this Biological Opinion would be highly affected by exposure to cyanide at the CCC. These results prompted us to better understand the level of protection that aquatic life criteria in general and the cyanide criterion in particular were intended to provide.

The objective of the Clean Water Act (CWA) is to "*restore and maintain the chemical, physical and biological integrity of the Nation's waters*" with an interim goal of "*water quality which provides for the protection and propagation of fish, shellfish and wildlife and provides for recreation in and on the water*", where attainable.

Section 304(a) of the CWA requires the EPA Administrator to publish "*criteria for water quality accurately reflecting the latest scientific knowledge on the kind and extent of all identifiable effects on the health and welfare including, but not limited to, plankton, fish shellfish, wildlife, plant life….*" including information "*on the factors necessary for the protection and propagation of shellfish, fish, and wildlife for classes and categories of receiving waters…*"

Water quality criteria are developed to protect different attributes or uses of water bodies, referred to as designated uses. Aquatic life water quality criteria are intended to protect waters where the designated use includes the protection and propagation of fish, shellfish and wildlife.

EPA's *Guidelines for Deriving Numeric National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (Stephan et al. 1985) has been used to derive aquatic life water quality criteria, including the cyanide criterion, since the mid 1980's. This excerpt from the guidelines document describes, from an operational perspective, the intended purpose of national criteria and their limitations:

"*Because aquatic ecosystems can tolerate some stress and occasional adverse effects, protection of all species at all times and places is not deemed necessary. If acceptable data are available for a large number of appropriate taxa from an appropriate variety of taxonomic and functional groups, a reasonable level of protection will probably be provided if all except a small fraction of the taxa are protected, unless a commercially or recreationally important species is very sensitive. The small fraction is set at 0.05 because other fractions resulted in criteria that seemed too high or too low in* *comparison with the sets of data from which they were calculated. Use of 0.05 to calculate the Final Acute Value does not imply that this percentage of adversely affected taxa should be used to decide in a field situation whether a criterion is too high or too low or just right.* "

Based on this description it appears that National criteria are intended to protect ecosystems and ecosystem functions and not necessarily to protect all taxa or species within the ecosystem all of the time. It is expected that a small fraction (5%) of taxa or species may be adversely affected. The guidelines go on to say:

"*To be acceptable to the public and useful in field situations, protection of aquatic organisms and their uses should be defined as prevention of unacceptable long-term and short-term effects on (1) commercially, recreationally, and other important species and (2) (a) fish and benthic invertebrates assemblages in rivers and streams, and (b) fish, benthic invertebrate, and zooplankton assemblages in lakes reservoirs, estuaries, and oceans."*

Thus, the level of protection afforded to aquatic organisms should prevent *unacceptable* long-term and short term effects. The threshold for unacceptable long-term or chronic effects is estimated by the CCC. The guidelines indicate that some adverse effects may occur at the CCC but they should not rise to a level that is unacceptable:

"*However, it is important to note that this is a threshold of unacceptable effect, not a threshold of adverse effect. Some adverse effect, possibly even a small reduction in the survival, growth, or reproduction of a commercially or recreationally important species, will probably occur at, and possibly even below, the threshold. The Criterion Continuous Concentration (CCC) is intended to be a good estimate of this threshold of unacceptable effect.*"

EPAs Water Quality Standards Handbook (EPA 1994) defines the CCC as "*the EPA national water quality criteria recommendation for the highest instream concentration of a toxicant or an effluent to which organisms can be exposed indefinitely without causing unacceptable effect*".

The guidelines provide some guidance for determining what constitutes unacceptable levels of adverse effects. The following guidance is for monitoring programs designed to detect unacceptable levels of adverse effects in the field:

"*The amount of decrease in the number of taxa or number of individuals in an assemblage that should be considered unacceptable should take into account appropriate features of the body of water and its aquatic community. Because most monitoring programs can only detect decreases of more than 20 percent, any statistically significant decrease should usually be considered unacceptable. The insensitivity of most monitoring programs greatly limits their usefulness for studying the validity of criteria because unacceptable changes can occur and not be detected. Therefore, although* *limited field studies can sometimes demonstrate that criteria are underprotective, only high quality field studies can reliably demonstrate that criteria are not underprotective.*"

Here, the guidelines appear to suggest that because of the insensitivity of monitoring programs only relatively high levels of effect (>20%) may be detected as statistically significant, and suggest that lower, yet unacceptable, levels of effects may go undetected.

However, the National criteria are typically derived using chronic toxicity data from laboratory tests rather than field studies. Chronic data from individual tests are analyzed and a chronic value is computed according to the following guidance:

"*A chronic value may be obtained by calculating the geometric mean of the lower and upper chronic limits from chronic tests or by analyzing chronic data using regression analysis. A lower chronic limit is the highest tested concentration (a) in an acceptable chronic test, (b) which did not cause an unacceptable amount of adverse effect on any of the specified biological measurements, and (c) below which no tested concentration caused an unacceptable effect. An upper chronic limit is the lowest tested concentration (a) in an acceptable chronic test, (b) which did cause an unacceptable amount of adverse effect on one or more of the specified biological measurements, and (c) above which all tested concentrations also caused such an effect.*"

For most aquatic life criteria that have been derived thus far, including the cyanide criterion, chronic values have been obtained by calculating the geometric mean of the lower and upper chronic limits. In practice, the upper and lower chronic limits are often statistically determined by hypothesis testing. The lower limit is typically the No Observable Effect Concentration (NOEC), which is defined as the highest test concentration where the effects are not statistically significantly different from controls. The upper limit is typically the Lowest Observable Effect Concentration (LOEC), which is defined as the lowest test concentration where the effects are statistically significantly different from controls. The guidelines recommend that the magnitude of effect associated with the upper and lower chronic limits should be considered when determining values that appropriately estimate acceptable and unacceptable levels of adverse effect:

"*Because various authors have used a variety of terms and definitions to interpret and report results from chronic tests, reported results should be reviewed carefully. The amount of effect that is considered unacceptable is often based on a statistical hypothesis test, but might also be defined in terms of a specified percent reduction from the controls. A small percent reduction (e.g., 3%) might be considered acceptable even if it is statistically significantly different from the control, whereas, a large percent reduction (e.g., 30%) might be considered unacceptable even if it is not statistically significant.*"

Based on this guidance, the threshold for unacceptable adverse effects would be estimated by the chronic value. The magnitude of effect at the threshold would then be equivalent to the magnitude of effect at the chronic value. For chronic criteria derived using hypothesis tests, this would be the magnitude of effect occurring at a concentration equal to the geometric mean of the NOEC and LOEC, that is, somewhere between an acceptable and unacceptable level of adverse effect. The guidelines do not specify a level of adverse effect on which the threshold for unacceptability should be based. The only mention of a numeric value or range is provided in the guidance for selecting chronic limits (mentioned above) and suggests that this threshold may lie between 3% and 30%.

Thus, for a given species or test the magnitude of effect at the chronic value will depend on the magnitude of effect at the lower and upper chronic limits. We followed this approach for estimating the magnitude of effect occurring at the cyanide CCC. The freshwater cyanide CCC was derived based on chronic toxicity data for 4 species (Table 14): 3 fish (fathead minnow, brook trout, and bluegill) and 1 invertebrate (*Gammarus pseudolimnaeus*). Chronic values for each species were obtained by calculating the geometric mean of the lower and upper chronic limits. The magnitude of effect at the lower and upper chronic limits was calculated by comparing responses at the lower and upper limits to controls. For fathead minnow and brook trout these effects were expressed as reduction in the mean number of eggs spawned per female compared to controls; for the bluegill the effect was reduction in larvae/juvenile survival compared to controls; and for *G. pseudolimnaeus* the effect was a reduction in the mean number of eggs or young per gravid female relative to controls.

We then estimated the magnitude of effect at the chronic value by linear interpolation between lower and upper chronic limits (Table 14). Based on these calculations the magnitude of effect at the chronic values for the fathead minnow, brook trout, bluegill and *G. pseudolimnaeus* would be 52%, 32%, 54%, and 47%, respectively. According to the guidelines, if there were a sufficient number of chronic values (i.e., chronic values for species in 8 phylogenetic families) the chronic criterion could be computed directly from the distribution of chronic values. If there were fewer chronic values, as was the case for cyanide, the chronic criterion would be computed using Acute-Chronic Ratios (ACR). ACRs for the 4 freshwater species were reported in the cyanide criterion document and are shown in Table 14. The ACRs were calculated by dividing the species mean acute value (i.e., mean $LC_{50}$ for the species) by the chronic value. For example, the ACR for fathead minnow (7.633) was computed by dividing 125.1 ug CN/L (the mean $LC_{50}$ for the species) by 16.39 ug/ CN/L (the chronic value). Thus, the ACR is the ratio between the concentration of cyanide causing 50% lethality (following acute exposure) and the concentration following chronic exposure that causes a level of adverse effect that is at the threshold of unacceptability, i.e., 52% for fathead minnow. The guidelines require that, for criteria derivation, the geometric mean of individual species ACRs is used to obtain the Final ACR. For cyanide, the freshwater Final ACR was 8.562 (Table 14). We estimated the magnitude of chronic effects associated with the Final ACR to be about 45% (Table 14).

The Final ACR and the Final Acute Value (FAV) were then be used to derive the CCC. The guidelines describe how the FAV is computed. In short, the FAV is set equal to the $5^{th}$ percentile estimate from the distribution of genus mean acute values. In other words, the FAV represents the genus with acute sensitivity ($LC_{50}$) in the *sensitive tail* of the distribution where, theoretically, approximately 5% of the genera would be more sensitive and about 95% of the genera would be less sensitive. Based on this analysis, the FAV for cyanide was determined to be 62.68 ug CN /L. The guidelines also include provisions for adjusting the FAV to protect commercially and recreationally important species:

"*However, in some cases, if the Species Mean Acute Value of a commercially or recreationally important species is lower than the calculated Final Acute Value, then that Species Mean Acute Value replaces the calculated Final Acute Value in order to provide protection for that important species.*"

For cyanide, the FAV was lowered from 62.68 ug/L to 44.73 ug/L because the Species Mean Acute Value for rainbow trout (44.73 ug/L) was below the calculated FAV. The cyanide criterion (5.2 ug/L) was then derived by division of the FAV (44.73 ug/L) by the

**Table 14. Chronic toxicity data used by EPA to derive the freshwater chronic criterion for cyanide. Effects levels were calculated using data from the original papers.**

| Species | Chronic Limits[1] | | | | Chronic Value[2] | | $LC_{50}$[3] (ug CN/L) | ACR[3] |
|---|---|---|---|---|---|---|---|---|
| | Lower | | Upper | | | | | |
| | (ug CN/L) | Effect | (ug CN/L) | Effect | (ug CN/L) | Effect | | |
| Fathead Minnow | 13.3 | 47% | 20.2 | 58% | 16.39 | 52% | 125.1 | 7.633 |
| Brook Trout | 5.6 | 18% | 11.0 | 53% | 7.849 | 32% | 83.14 | 10.59 |
| Bluegill | 9.3 | 30% | 19.8 | 88% | 13.57 | 54% | 99.28 | 7.3 |
| Gammarus | 16 | 0% | 21 | 100% | 18.33 | 47% | 167 | 9.111 |
| Geometric mean | | | | | | 45% | | 8.562 |

[1] Lower and upper chronic limits were taken from the cyanide criteria document. For fathead minnow and bluegill these values were determined statistically (i.e., NOEC and LOEC identified via hypothesis tests). Effect levels were take from Tables 5, 8 and 10 in the Effects section of the BO and from Oseid and Smith 1979.
[2] Chronic values were taken from the cyanide criteria document. Effect levels associated with the chronic values were estimated by linear interpolation between the effects at the lower and upper chronic limits.
[3] Acute-Chronic Ratios were taken from the cyanide criteria document.

Final ACR (8.562). Thus the chronic criterion, 5.2 ug CN/L, was based on the concentration intended to protect rainbow trout from unacceptable adverse effects. Based on our estimate of the magnitude of effect associated with the Final ACR, we estimate the magnitude of adverse effects occurring to rainbow trout at the chronic criterion to be approximately 45%. This value is higher than we would have expected considering it is intended to represent the threshold for unacceptable adverse effects. However, the magnitude is in line with effects we predicted for the listed fish evaluation species, most of which were estimated to be as (or more) sensitive to cyanide as rainbow trout.

The same conclusion reached above, that NOEC/LOEC-based estimates of "chronic values" can correspond to $\geq$ 40% adverse effect, has also been reached by others. Decades ago Suter et al. (1987) reported that MATC's for fish fecundity, on average, corresponded to a 42% level of adverse effect (MATC = Maximum Acceptable Toxicant Concentration; a term for the geometric mean of the NOEC and LOEC from a given toxicity test and often assigned by EPA as the estimated "chronic value" from a test). Other response endpoints were found to correspond to average adverse effect levels of 12-35%. More recently, SETAC (Society for Environmental Toxicology and Chemistry) convened a panel of experts (Reiley et al. 2003) who concluded that "*...[toxicity] tests with high variability may result in an(sic) NOEC corresponding to a response greater than 40% different from the control."* Moore and Caux (1997) statistically examined nearly 200 toxicity data sets and found that most NOEC's (76.9%) exceeded a 10% adverse effect level and most LOEC's (62.4%) exceeded a 30% effect level. Various other researchers have noted a variety of adverse effect levels for NOEC's, such that Crane and Newman (2000) were led in summary to conclude that "*…[NOEC] effect levels from individual tests ranged from nearly 0% to nearly 100%."* For seven cyanide toxicity tests with sufficient data for comparison, Gensemer et al. (2007: Figure 3-7) found in all cases that the geometric mean of the NOEC and LOEC corresponded to an adverse effect level of $\geq$ 20% (how much greater was not reported).

Because of the highly variable and often substantive levels of effect associated with NOEC's, LOEC's, MATC's, and with the "chronic values" based on them, and for numerous other reasons, a strong professional consensus recommendation to avoid using NOEC/LOEC-based estimates for regulatory thresholds (when possible) has been expressed repeatedly. For example, there was an ISO (International Organization for Standardization) resolution (ISO TC147/SC5/WG10 Antalya 3) as well as an OECD (Organisation for Economic Co-operation and Development) workshop recommendation (OECD 1998) that the NOEC should be phased out from international standards (OECD 2006:14). Environment Canada (2005) notes, that there is a growing literature which points out many deficiencies of the NOEC approach (Suter et al. 1987, Miller et al. 1993, Pack 1993, Noppert et al. 1994, Chapman 1996, Chapman et al. 1996, Pack 1998, Suter 1996, Moore and Caux 1997, Bailer and Oris 1999, Andersen et al. 2000, Crane and Newman 2000, Crane and Godolphin 2000). Moving away from the NOEC/LOEC approach was also among the recommendations of the SETAC panel for improving the scientific basis of water-quality criteria (Reiley et al. 2003).

Accordingly, EPA has begun employing a regression approach for estimating "chronic values" whenever sufficient data are available to do so. For example, in the 1999 update for ammonia water quality criteria EPA used regression analyses to estimate 20% effect concentrations ($EC_{20}$'s) from individual toxicity tests and used those $EC_{20}$'s as estimates of "chronic values" (EPA 1999). Likewise, estimated $EC_{20}$'s have been the basis for estimating "chronic values" in recently proposed updates for copper and selenium water quality criteria (EPA 2003a, 2004). EPA's choice of the $EC_{20}$ as a basis for estimating "chronic values" was justified from statistical considerations rather than from biological or demographic considerations:

*"To make [chronic values] reflect a uniform level of effect, regression analysis was used here both to demonstrate that a significant concentration-effect relationship was present and to estimate [chronic values] with a consistent level of effect. Use of regression*

*analysis is provided for on page 39 of the 1985 Guidelines (U.S. EPA 1985b). The most precise estimates of effect concentrations can generally be made for 50 percent reduction (EC50); however, such a major reduction is not necessarily consistent with criteria providing adequate protection. In contrast, a concentration that caused a low level of reduction, such as an EC5 or EC10, is rarely statistically significantly different from the control treatment. As a compromise, the EC20 is used here as representing a low level of effect that is generally significantly different from the control treatment across the useful chronic datasets that are available for ammonia."*

Pack (1993) asserted that most ecotoxicologists consider effects in the range of 5-20% to be biologically acceptable depending on the species involved and the type of effect. However, EPA appears to have chosen the top end of that range based more on the expected statistical power of toxicity tests than on a serious examination of the typical demographic sensitivity of biotic populations to a 20% adverse effect on survival, growth, or reproduction. Furthermore, 95% statistical confidence limits for most $EC_{20}$ estimates are likely to extend well into adverse effect levels that would be of unquestionably serious demographic concern for most organisms. As evident from the above discussion, most chronic criteria derived by EPA, including for cyanide, are highly likely to be associated with $\geq$ 20% adverse effect level for species at the vulnerable end of species sensitivity distributions (such as the subset of ESA-listed species we are evaluating). Therefore, it should be no surprise that our estimated effect levels for such species at the current cyanide CCC of 5.2 ug/L are almost always higher than 20% and in some cases substantively higher.

*Population Responses to Reductions in Fecundity and Juvenile Survival*

Laboratory experiments have demonstrated that even closely related fish species can demonstrate great differences in sensitivity when exposed to the same chemical, as measured by differences in acute or chronic toxicity values. This variability in sensitivity has been related to differences in species' physiology and life history strategies. Similarly, population modeling and experimental studies have shown that variation in population-level responses to environmental toxicity can also be expected among species as a consequence of factors such as life history strategies, life stage affected, and density dependence. Studies have also demonstrated that chronic toxicity can lead to population decline and extirpation.

Under the ESA, in determining whether a proposed Federal action is likely to jeopardize the continued existence of a listed species under the ESA, we assess whether the proposed activity reasonably would be expected to appreciably reduce the likelihood of survival and recovery of a listed species by reducing its reproduction, numbers, or distribution. Two common metrics used in population modeling to assess effects of perturbations on populations are population growth rate and time to or probability of extinction.

Population growth rate is the change in a population size over a unit time period. Long-term reductions in population growth rate as low as 5% has been shown to significantly increase a population's likelihood of extinction (Snell and Serra 2000). Population growth rate can be positive when the population is increasing, negative when decreasing, or zero when the net difference between births, deaths, and migration is zero and the population is stable. For listed species, populations may exist in any of these states depending on its recovery status. Our analysis determines the relative predicted effects of the action to the population growth rate, regardless of its starting value.

Using known parameters of a species' life history, sensitivity analyses can be conducted to determine which parameters, when modified, will have the greatest impact on the species' population growth rate. Elasticity analysis is one type of sensitivity analysis that is commonly used in conservation biology to demonstrate the relative contributions to population growth rate made by life cycle transitions, based on vital rate statistics for survival, growth and fertility. While these types of analyses cannot predict absolute effects to population size, because they quantify the relative importance of an element to changes in population growth rate, they can help focus management decisions on those demographic parameters that exhibit the largest elasticity, and thus, the largest impact on population growth (de Kroon et al. 2000). However, elasticity analysis requires the development of a population model, for which adequate data are often scarce. Because this type of demographic data is often lacking for threatened and endangered species in particular, the need to develop generalized approaches for classifying population responses to perturbation for rare species has been recognized (Heppell et al. 2000, Dennis et al. 1991).

Several authors have examined the effect of life history strategies on the elasticities of various demographic measures. In evaluating demographic parameters of 50 mammal populations with different life history strategies, Heppell et al. (2000) found that phylogeny alone is often not a reliable indicator of which vital rates (survival, growth and fertility) will have the greatest impact on elasticity. Instead, the authors found that species that mature early and have high reproductive output had high fertility elasticities and low adult survival elasticities. Conversely, for those which mature late and have long lifespans, fecundity and early offspring survival are less important than survival of juveniles to maturity to changes in population growth rate. Calow et al. (1997) also found that the relative importance of juvenile fish survival can vary according to reproductive strategy. These authors concluded that reductions in juvenile survival would have the greatest impact on semelparous fish species, in which adults die after reproduction, a lesser impact on a moderately iteroparous population, in which adult postreproductive survival is intermediate, and the least impact on strongly iteroparous species, in which adult survival after reproduction is high. These assumptions held true for elasticity analysis of the green sturgeon, a fish species with life history patterns such as late-maturity and long-life that are common to other sturgeon (Heppell 2007). Juvenile survival had relatively lower elasticity values than adult and subadult survival, with compensation for the loss of adults requiring much larger increases in young-of-the-year survival than would be commensurate with the loss. However, other authors have found increased importance of juvenile survival for sturgeon, despite their lifespan (Gross et al. 2002, Paragamian and Hansen 2008). Gross et al. (2002) hypothesized that this difference was due to the vastly larger fecundity of sturgeon as compared to other long-lived species.

Vélez-Espino et al. (2006) argue the need for a broadscale summary of species' population dynamics to help guide the conservation biology of freshwater fishes, for which information on life history is often limited. Using information, on adult survival, juvenile survival, and fecundity, the authors performed elasticity analyses on 88 species of freshwater fish and found that they could be classified into 4 functional groups with regard to the sensitivity of their population growth rates:

1. species most sensitive to perturbations in adult survival
2. species most sensitive to perturbations to adult and juvenile survival
3. species most sensitive to perturbations to juvenile survival
4. species most sensitive to perturbations to juvenile survival and fecundity

These groups are characterized by decreased age at maturity, longevity, and reproductive lifespan as one moves from group 1 to group 4. Age at maturity, reproductive lifespan, fecundity, juvenile survivorship, and longevity were all correlated with adult survival and fecundity. However, the best predictors of elasticity patterns were longevity, which explained 93% of the variability in the elasticity of adult survival, and age at maturity, which explained 92% of the variability in the elasticity of fecundity. The authors also found that elasticities are highly conserved among genera within the same taxonomic family

Spromberg and Birge (2005) also found that life history strategies influence effects to populations. The five life history strategies they modeled encompassed differences in stage-specific survival, fecundity and hatch success, number of spawning events, and life-span. The authors found that regardless of strategy, changes in the number of young-of-the-year stage individuals had the greatest impact on population growth rate. However, the relative contribution of this parameter was greatest for life history strategies with multiple spawnings, high fecundity, and short lifespans as opposed to those with longer lifespan, which had increased elasticity of adult survival.

Spromberg and Meador (2005) linked toxicant effects on immune suppression, reproductive development, and growth reduction to demographic traits in Chinook salmon (*Oncorhynchus tshawytscha*) and modeled their influence on population growth rate. Overall, effects to first- and second-year survival had the greatest elasticities, with constant reductions to first year survival as low as 10% achieving population declines ranging from 35-78% compared to controls. Other studies have demonstrated the importance of first year survival in this species (Kareiva et al. 2000). Spromberg and Meador (2005) also found that models which incorporated effects to both survival and reproduction were additive, indicating the importance of evaluating the overall impact of all potential impacts to population growth.

Many listed species populations are limited by the amount of adequate habitat or resources and experience some degree of density dependence. Density-dependence at any life stage must be considered in elasticity analysis in order to yield appropriate results (Grant and Benton 2000, Hayashi et al. 2008). In a review of toxicant impacts on density-limited populations, Forbes et al. (2001) noted that the full range of interactions have been found between toxicant stress and density dependence, including less than additive, additive, and more than additive effects. Also, the type of effect may vary with increasing toxicant concentration from one that ameliorates density dependent effects at low toxicant concentrations to one that exacerbates density dependent effects at higher toxicant concentrations. Case studies which incorporate density-dependence into population modeling demonstrate this variability, with overall impacts to populations shown to be both lesser (Van Kirk and Hill 2007) and greater (Hayashi et al. 2008) than the level of effect that would be predicted from individual response depending on the situation. In time, density-dependant populations may rebound, stabilize at a lower absolute population number, or continue to decline until the population is extirpated (Forbes et al. 2001). Modeling exercises have demonstrated cases in which populations stabilize at new, lower equilibrium abundances in response to a constant impact (van Kirk and Hill 2007, Spromberg and Meador 2005).

A species' likelihood of persistence can also be estimated by modeling the species' time to extinction or probability of extinction. Population viability analysis (PVA) uses simulation modeling to identify threats to species and to assess the vulnerability of populations to these extinction risks. These models incorporate demographic parameters such as fecundity, survivorship, age structure, and population size, but can also incorporate effects to the environment such as habitat degradation and catastrophic events. As for the evaluation of population growth rate, sensitivity analysis is used to determine which factors have the greatest impact on population persistence, and many experts feel that parsing out these influential factors for management purposes is the best utilization of these models, as opposed to absolute predictions of population decline. Though PVA also requires a depth of demographic data that is often lacking for listed species, even PVAs with little data incorporated can be useful in comparative analyses of management considerations (Akçakaya and Sjögren-Gulve 2000).

Only a limited number of PVAs have been performed for listed aquatic species or other closely related species. A PVA for two darter species, the slackwater darter (*Etheostoma boschungi)* and the holiday darter (*E. brevirostrum)*, found that fertility made the largest relative contribution to population growth, with juvenile survivability a more influential contributor to fertility than egg production (Hartup 2005). This conclusion held true regardless if the species was a single or multiple-batch spawner. In modeling the contributions of population size, age structure, and migration rate on the leopard darter (*Percina pantherina*), migration had the greatest influence on persistence (Williams et al. 1999). Catastrophe also played a significant role in persistence of the species, especially considering that the species is short-lived, has only one reproductive opportunity, and is restricted to few isolated populations. A similar analysis of the freshwater rotifer *Brachionus calyciflorus* also found that reductions in growth rate resulted in substantially increased probability of extinction when coupled with catastrophic population reductions (Snell and Serra 2000). A PVA analysis for the gila trout (*Oncorynchus gilae*) revealed that the number of populations was the most influential of several life history parameters, and that the model was relatively insensitive to changes in population size and proportional abundance of age classes (Brown et al. 2001). The model was also sensitive to large change in fecundity, producing significant changes in the probability of extinction when halved.

There are few field studies of pollutant effects on populations. Kidd et al. (2007) studied the effects of 17α-ethynlestradiol (EE2) on fathead minnows in Canadian experimental lakes over a 7-year period. EE2, a synthetic estrogen found in birth control pills, was introduced at concentrations found in the receiving waters downstream from municipal wastewater dischargers. The fathead minnow, a short-lived species, was the first to show population collapse, but recovered once the pollutant stress was removed.

*Summary of Population Responses to Reductions in Fecundity and Juvenile Survival*

Modeling and experimental studies have shown that chronic toxicity to pollutants can lead to population decline and extirpation. Variation in population-level responses to environmental toxicity can be expected among species as a consequence of factors like species life history strategies, life stage affected, density dependence, and magnitude of toxicant stress. Although the degree varied among different life history strategies, fecundity and juvenile survival remained a highly influential demographic parameter throughout modeled scenarios, with adult survival taking on greater importance in long-lived species. These results must be coupled with other influences on the population status, such as the degree of density dependence and additional environmental perturbations such as catastrophes. Although population modeling often requires more demographic information than is available for threatened and endangered species, careful selection of surrogates and use of their data may allow for extrapolation from models for species with similar life histories.

*Individual Species and Critical Habitat Accounts*

### ***Acipenseridae***

### GULF STURGEON
*Acipenser oxyrinchus desotoi*

Gulf sturgeon exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Because no data for cyanide toxicity to sturgeon exist, LC50 values for sturgeon were derived from the 5% SSD concentration for the class Actinoptergyii, which encompasses all known cyanide toxicity data for fish. From this data, we developed quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13).

Compared to control populations, we estimate Gulf sturgeon exposed to cyanide at the CCC could experience an approximate 48% reduction in the number of hatched eggs. We estimate that Gulf sturgeon exposed to cyanide at the CCC could experience an approximate 56% reduction in survival of young fish through the first year. Though no cyanide-specific data exist for this species, there are data from other chemicals that support the relative sensitivity of sturgeon to contaminants. Dwyer et al. (2005) tested the relative sensitivity of 18 fish and 1 amphibian species to five chemicals. Of these, the two sturgeon tested, the Atlantic sturgeon (*A. oxyrhynchus*) and the shortnose sturgeon (*A. brevirostrum*) ranked first and second, respectively, in overall sensitivity. For all five chemicals, sturgeon were as or more sensitive than rainbow trout, for which reductions in viable eggs spawned and juvenile survival were estimated at 52% and 61%, respectively (Table 12).

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

Sturgeon have naturally high adult survival. Several authors have suggested that the rate of survival may be so high that management at the levels of these age classes is unlikely to improvement their survival or increase population growth rate (Heppell 2007, Gross et al 2002) As such, recovery efforts are often based upon increasing survival in juvenile age classes. A population viability analysis of the Suwannee River population of the gulf sturgeon found that slight changes in egg-to-age-1 mortality would strongly influence the recruitment and subsequent population (Pine et al. 2001). Decreasing the estimated 99.96% mortality for this life stage just 0.05 percentage points resulted in a 10-fold increase in population size, with a 5-fold increase in the number of recruits.

Gross et al (2002) modeled population growth rates for three species of sturgeon that varied in life history traits such as size, lifespan, age to maturity, and migration. All three sturgeons showed similar elasticity profiles, and thus the authors concluded that general interpretation could be applied to sturgeon across species. In contrast to other elasticity profiles for long-lived species, elasticity in sturgeon was highest in individual young-of-the-year and juvenile age classes, dropped at the onset of maturity, and continued to decline for each successive adult age class. Fecundity had relatively low elasticity, as the effects of changes in fecundity are shared among all adult age classes of these long-lived species, and the value of changes to egg numbers is lessened by the high mortality of the young-of-the-year age class. The authors concluded that population growth rate will show little response to improvements in fecundity, but greater responses in survival at either the young-of-the-year or juvenile age classes. However, since survival of the juvenile and adult age classes is naturally high, improvements at these stages will have smaller effects to improving population growth rate than increases to survival of young-of-the-year, when natural mortality is greater. The authors note that among biologists and managers involved in sturgeon conservation, habitat improvement was regarded as the most important conservation undertaking for sturgeon. Results from this study indicate that restoration efforts should target the survival of age classes with high elasticity, specifically young-of-year and juvenile.

Paragamian and Hansen (2008) drew similar conclusions in modeling effects on population growth of the Kootenai River white sturgeon. The authors found that subadult and adult survival (>90%) was much higher than that of juveniles (40% in the first year), and recovery was most dependant on increasing first-year survival. The authors suggested that to have the largest effect on recovery, the managers should increase the current targeted recruitment rate.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Gulf sturgeon reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survival of young fish through the first year. Gulf sturgeon may also experience effects on growth, swimming performance, condition, and development. While sturgeon have developed a life history that allows them to cope with low survivorship to maturity and occasional hits to recruitment, these adaptation are unlikely to compensate for a constant reduction in both fecundity and early life stage survival. The reductions we estimate in survival of young fish through the first year in particular would substantially decrease recovery of this species. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed.
We would anticipate a consequent reduction in numbers of Gulf sturgeon. An effected Gulf sturgeon population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Gulf sturgeon are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the gulf sturgeon.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the gulf sturgeon include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of gulf sturgeon, and cause sturgeon to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of gulf sturgeon. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the gulf sturgeon.

## KOOTENAI RIVER WHITE STURGEON
*Acipenser transmontanus*

Kootenai sturgeon exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience adverse effects on growth, swimming performance, condition, and development, as described above in the *Overview* section. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies are available for estimating the magnitude of adverse effects that could occur following Kootenai River white sturgeon exposure to cyanide at criterion concentrations. Because no data for cyanide toxicity to sturgeon exist, $LC_{50}$ values for sturgeon were derived from the 5% SSD concentration for the class Actinoptergyii, which encompasses all known cyanide toxicity data for fish. From these data, we developed quantitative estimates of the effects on sturgeon fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate that Kootenai River white sturgeon that are exposed to cyanide at the CCC are likely to be subject to an approximately 48% reduction in the number of hatched eggs. We estimate that Kootenai River white sturgeon exposed to cyanide at the CCC are likely to experience an approximately 56% reduction in the survival of young fish through the first year.

Although no cyanide-specific data exist for this species, there are data from other chemicals that support the relative sensitivity of sturgeon to contaminants. Dwyer et al. (2005) tested the relative sensitivity of 18 fish and 1 amphibian species to five chemicals. Of these, the two sturgeon tested, the Atlantic sturgeon (*A. oxyrhynchus*) and the shortnose sturgeon (*A. brevirostrum*) ranked first and second, respectively, in overall sensitivity. For all five chemicals, sturgeon were as or more sensitive than rainbow trout, for which reductions in viable eggs spawned and juvenile survival were estimated at 52% and 61%, respectively (Table 12).

As noted previously, the young first-year fish that do survive are likely to experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

Sturgeon have naturally high adult survival. Several authors have suggested that the rate of survival may be so high that management at the levels of these age classes is unlikely to improve their survival or increase the population growth rate (Heppell 2007, Gross et al. 2002). For that reason, recovery efforts are often based upon increasing survival in juvenile age classes. *The Recovery Plan for the Kootenai River White Sturgeon* lists as a Priority 1 task (tasks that must be taken to prevent extinction or to prevent the species from declining irreversibly in the foreseeable future) restoration of ecosystem functions "to ensure habitat conditions necessary for successful white sturgeon reproduction and recruitment, i.e. survival of juveniles during their first year of life and beyond" (Service 1999). The reestablishment of natural recruitment to the Kootenai River population of the white sturgeon is listed in the recovery plan as a recovery objective.

Gross et al. (2002) modeled population growth rates for three species of sturgeon that varied in life history traits such as size, lifespan, age to maturity, and migration. All three sturgeon species showed similar elasticity profiles, and thus the authors concluded that general interpretation could be applied to sturgeon across species. In contrast to other elasticity profiles for long-lived species, elasticity in sturgeon was highest in individual young-of-the-year and juvenile age classes, dropped at the onset of maturity, and continued to decline for each successive adult age class. Fecundity had relatively low elasticity, as the effects of changes in fecundity are shared among all adult age classes of these long-lived species, and the value of changes to egg numbers is lessened by the high mortality of the young-of-the-year age class. The authors concluded that population growth rate will show little response to improvements in fecundity, but greater responses in survival at either the young-of-the-year or juvenile age classes. However, since survival of the juvenile and adult age classes is naturally high, improvements at these stages will have smaller effects to improving population growth rate than increases to survival of young-of-the-year, when natural mortality is greater. The authors note that among biologists and managers involved in sturgeon conservation, habitat improvement was regarded as the most important conservation undertaking for sturgeon. Results from this study indicate that restoration efforts should target the survival of age classes with high elasticity: the young-of-the-year and juveniles.

Paragamian and Hansen (2008) drew similar conclusions in modeling effects on population growth of the Kootenai River white sturgeon. They reported that subadult and adult survival (>90%) was much higher than that of juveniles (40% in the first year), and recovery was most dependant on increasing first-year survival. Paragamian and Hansen (2008) suggested that to have the largest effect on recovery, the recruitment rate should increase.

In summary, exposure to cyanide concentrations at the chronic criterion are likely to substantially reduce Kootenai River white sturgeon reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survival of young fish through the first year. Such exposure may also subject Kootenai

River white sturgeon to adverse effects on growth, swimming performance, condition, and development. Although sturgeon have developed a life history that allows them to cope with low survivorship to maturity and occasional hits to recruitment, these adaptations are not sufficient to cope with a constant reduction in both fecundity and early life stage survival likely to be caused by exposure to cyanide concentrations at the chronic criterion. The estimated reductions in the survival of young fish through the first year in particular would significantly decrease recovery of this species in the wild. Because of the high magnitude of adverse effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed by sturgeon exposure to cyanide concentrations at the chronic criterion, which is likely to cause a consequent reduction in the numbers of Kootenai River white sturgeon. Based upon the magnitude of adverse effects that are likely to occur and given the extremely endangered status of this species, we conclude that ultimately Kootenai River white sturgeon are likely to be extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion throughout the range of this species is likely to reduce the reproduction, numbers, and distribution of the Kootenai River white sturgeon at the rangewide scale.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Kootenai River white sturgeon include water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval and cyanide in waters to these levels is likely to adversely affect the quality of water in sturgeon critical habitat to the degree that it would impair individual reproduction and survival of Kootenai River white sturgeon, and cause sturgeon to experience adverse effects to growth, swimming performance, condition, and development. Cyanide concentrations at the chronic criterion within critical habitat are likely to create habitat conditions that reduce the number of hatched sturgeon eggs by as much as 48% and reduce the survival of young sturgeon through the first year by as much as 56%. Approval of the CCC and cyanide in waters to these levels is likely to adversely affect the quality of water in sturgeon critical habitat to the degree that it is likely to preclude the intended conservation function of that habitat.

## PALLID STURGEON
*Scaphirhynchus albus*

Pallid sturgeon exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Because no data for cyanide toxicity to sturgeon exist, LC50 values for sturgeon were derived from the 5% SSD concentration for the class Actinoptergyii, which encompasses all known cyanide toxicity data for fish. From this data, we developed quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate pallid sturgeon exposed to cyanide at the CCC could experience an approximate 48% reduction in the number of hatched eggs. We estimate that pallid sturgeon exposed to cyanide at the CCC could experience an approximate 56% reduction in survival of young fish through the first year. Though no cyanide-specific data exist for this species, there are data from other chemicals that support the relative sensitivity of sturgeon to contaminants. Dwyer et al. (2005) tested the relative sensitivity of 18 fish and 1 amphibian species to five chemicals. Of these, the two sturgeon tested, the Atlantic sturgeon (*A. oxyrhynchus*) and the shortnose sturgeon (*A. brevirostrum*) ranked first and second, respectively, in overall sensitivity. For all five chemicals, sturgeon were as or more sensitive than rainbow trout, for which reductions in viable eggs spawned and juvenile survival were estimated at 52% and 61%, respectively (Table 12).

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

Sturgeon have naturally high adult survival. Several authors have suggested that the rate of survival may be so high that management at the levels of these age classes is unlikely to improve their survival or increase population growth rate (Heppell 2007, Gross et al 2002) As such, recovery efforts are often based upon increasing survival in juvenile age classes. The 2007 Five-Year Review for the pallid sturgeon found that natural recruitment is limited throughout the species' range (USFWS 2007). Wild populations in two of six Recovery Priority Management Areas (RPMAs) are comprised of old aged individuals, and three of six RPMAs are dependent on hatchery augmentation programs for recruitment. Addressing recruitment bottlenecks in the three upper Missouri River RPMAs was deemed critically important for the species to become self sustaining and be recovered in those reaches.

Gross et al (2002) modeled population growth rates for three species of sturgeon that varied in life history traits such as size, lifespan, age to maturity, and migration. All three sturgeons showed similar elasticity profiles, and thus the authors concluded that general interpretation could be applied to sturgeon across species. In contrast to other elasticity profiles for long-lived species, elasticity in sturgeon was highest in individual young-of-the-year and juvenile age classes, dropped at the onset of maturity, and continued to decline for each successive adult age class. Fecundity had relatively low elasticity, as the effects of changes in fecundity are shared among all adult age classes of these long-lived species, and the value of changes to egg numbers is lessened by the high mortality of the young-of-the-year age class. The authors concluded that population growth rate will show little response to improvements in fecundity, but greater responses in survival at either the young-of-the-year or juvenile age classes. However, since survival of the juvenile and adult age classes is naturally high, improvements at these stages will have smaller effects to improving population growth rate than increases to survival of young-of-the-year, when natural mortality is greater. The authors note that among biologists and managers involved in sturgeon conservation, habitat improvement was regarded as the most important conservation undertaking for sturgeon. Results from this study indicate that restoration efforts should target the survival of age classes with high elasticity, young-of-year and juvenile.

Paragamian and Hansen (2008) drew similar conclusions in modeling effects on population growth of the Kootenai River white sturgeon. The authors found that subadult and adult survival (>90%) was much higher than that of juveniles (40% in the first year), and recovery was most dependant on increasing first-year survival. The authors suggested that to have the largest effect on recovery, the managers should increase the current targeted recruitment rate.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the pallid sturgeon reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survival of young fish through the first year. Pallid sturgeon may also experience effects on growth, swimming performance, condition, and development. While sturgeon have developed a life history that allows them to cope with low survivorship to maturity and occasional hits to recruitment, these adaptation are not designed to withstand a constant reduction in both fecundity and early life stage survival. The reductions we estimate in survival of young fish through the first year in particular would significantly decrease recovery of this species. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed.
We would anticipate a consequent reduction in numbers of pallid sturgeon. An effected Pallid sturgeon population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Pallid sturgeon are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the pallid sturgeon.

## ALABAMA STURGEON

*Scaphirhynchus suttkusi*

Alabama sturgeon exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects

Formal Draft Biological Opinion.

Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Because no data for cyanide toxicity to sturgeon exist, LC50 values for sturgeon were derived from the 5% SSD concentration for the class Actinoptergyii, which encompasses all known cyanide toxicity data for fish. From this data, we developed quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Alabama sturgeon exposed to cyanide at the CCC could experience an approximate 48% reduction in the number of hatched eggs. We estimate that Alabama sturgeon exposed to cyanide at the CCC could experience an approximate 56% reduction in survival of young fish through the first year. Though no cyanide-specific data exist for this species, there are data from other chemicals that support the relative sensitivity of sturgeon to contaminants. Dwyer et al. (2005) tested the relative sensitivity of 18 fish and 1 amphibian species to five chemicals. Of these, the two sturgeon tested, the Atlantic sturgeon (*A. oxyrhynchus*) and the shortnose sturgeon (*A. brevirostrum*) ranked first and second, respectively, in overall sensitivity. For all five chemicals, sturgeon were as or more sensitive than rainbow trout, for which reductions in viable eggs spawned and juvenile survival were estimated at 52% and 61%, respectively (Table 12).

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

Sturgeon have naturally high adult survival. Several authors have suggested that the rate of survival may be so high that management at the levels of these age classes is unlikely to improvement their survival or increase population growth rate (Heppell 2007, Gross et al 2002) As such, recovery efforts are often based upon increasing survival in juvenile age classes. The May 5, 2000 listing rule for the Alabama sturgeon (65 FR 26438) designated the primary threat to the immediate survival of the species to be its small population size and apparent inability to offset mortality rates with current reproduction and/or recruitment rates. Its small population also makes it vulnerable to natural or human-induced events (*e.g.,* droughts, floods, competition, variations in prey abundance, toxic spills), which may further depress recruitment.

Gross et al (2002) modeled population growth rates for three species of sturgeon that varied in life history traits such as size, lifespan, age to maturity, and migration. All three sturgeons showed similar elasticity profiles, and thus the authors concluded that general interpretation could be applied to sturgeon across species. In contrast to other elasticity profiles for long-lived species, elasticity in sturgeon was highest in individual young-of-the-year and juvenile age classes, dropped at the onset of maturity, and continued to decline for each successive adult age class. Fecundity had relatively low elasticity, as the effects of changes in fecundity are shared among all adult age classes of these long-lived species, and the value of changes to egg numbers is lessened by the high mortality of the young-of-the-year age class. The authors concluded that population growth rate will show little response to improvements in fecundity, but greater responses in survival at either the young-of-the-year or juvenile age classes. However, since survival of the juvenile and adult age classes is naturally high, improvements at these stages will have smaller effects to improving population growth rate than increases to survival of young-of-the-year, when natural mortality is greater. The authors note that among biologists and managers involved in sturgeon conservation, habitat improvement was regarded as the most important conservation undertaking for sturgeon. Results from this study indicate that restoration efforts should target the survival of age classes with high elasticity, young-of-year and juvenile.

Paragamian and Hansen (2008) drew similar conclusions in modeling effects on population growth of the Kootenai River white sturgeon. The authors found that subadult and adult survival (>90%) was much higher than that of juveniles (40% in the first year), and recovery was most dependant on increasing first-year survival. The authors suggested that to have the largest effect on recovery, the managers should increase the current targeted recruitment rate.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Alabama sturgeon reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survival of young fish through the first year. Alabama sturgeon may also experience effects on growth, swimming performance, condition, and development. While sturgeon have developed a life history that allows them to cope with low survivorship to maturity and occasional hits to recruitment, these adaptation are not designed to withstand a constant reduction in both fecundity and early life stage survival. The reductions we estimate in survival of young fish through the first year in particular would significantly decrease recovery of this species. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. We would anticipate a consequent reduction in numbers of Alabama sturgeon. An effected Alabama sturgeon population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Alabama sturgeon are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the Alabama sturgeon.

Formal Draft Biological Opinion.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Alabama sturgeon include a flow regime and river system which allows for all life stages and processes of the species, a river channel with a stable sand and gravel bottom, as well as a rock wall and associated mussel beds, limestone outcrops and cut limestone banks, riverline spawning sites with substance suitable for embryo deposition and development, large sections of free flowing water for spawning migrations and the development of young, water of 32 degrees Celsius (90 degrees Fahrenheit) or less, dissolved oxygen at 5 milligrams per liter or more, and pH from 6.0 to 8.5.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Alabama sturgeon critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Alabama sturgeon, and cause Alabama sturgeon to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of Alabama sturgeon. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Alabama sturgeon’s extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Alabama sturgeon.

### *Amblyopsidae*

## OZARK CAVEFISH
*Amblyopsis rosae*

Ozark cavefish exposed to cyanide at the criterion continuous concentration (CCC) are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Ozark cavefish exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Ozark cavefish exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the first-year fish that do survive could experience reduced growth rates that would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, such as Ozark cavefish, than on species with greater adult survival. Ozark cavefish can live up to 7-10 years, but do not reach adulthood until 4 or more years of age. Furthermore, only about 20% of the female population breeds each year, producing 20-25 eggs. We anticipate the effects of cyanide on fecundity and juvenile survival could have a substantial population-level effect on Ozark cavefish. Individual Ozark cavefish females reproduce infrequently and produce few offspring. A reduction in potential fecundity and juvenile survival in the wild at the magnitude we estimate will add to the challenges to recovering the Ozark cavefish.

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Ozark cavefish 's potential recruitment substantially. The Ozark cavefish 's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction and survival could be exacerbated further if reduced growth rates diminish survival through to adulthood.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Ozark cavefish's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Ozark cavefish may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. These potential effect could add substantially to the difficulties in reintroducing reproductively successful populations and attaining recovery. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Ozark cavefish are not likely to overcome the effects to their recruitment in waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could limit the reproduction, numbers, and distribution of the Ozark cavefish .

## ALABAMA CAVEFISH

*Speoplatyrhinus poulsoni*

Alabama cavefish exposed to cyanide at the criterion continuous concentration (CCC) are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Alabama cavefish exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Alabama cavefish exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the first-year fish that do survive could experience reduced growth rates that would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Alabama cavefish may live 5-10 years. Cavefish do not reproduce every year, numbers of reproductive females are few, and those that do spawn lay very few eggs. We anticipate the potential effects of cyanide on fecundity and juvenile survival would have a substantial population-level effect on Alabama cavefish. A reduction in potential fecundity and juvenile survival in the wild at the magnitude we estimate will add to the significant challenges to recovering the Alabama cavefish.

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Alabama cavefish's potential recruitment substantially. The Alabama cavefish 's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival beyond the early life stages analyzed.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Alabama cavefish's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Alabama cavefish may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be inadequate to avoid population decline. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Alabama cavefish are not likely to overcome the effects to their recruitment where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could limit the reproduction, numbers, and distribution of the Alabama cavefish.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Alabama cavefish include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of Alabama cavefish, and cause fish to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 48% and the reduction in the survival of young fish as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of Alabama cavefish. Approval of the CCC would adversely affect the quality of water to the degree that normal individual and population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Alabama cavefish.

### *Atherinidae*

## WACCAMAW SILVERSIDE

*Menidia extensa*

Waccamaw silverside exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were

available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Waccamaw silversides exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Waccamaw silversides exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than species with greater adult survival. The Waccamaw silverside has a 1-year life cycle and depends upon successful reproduction each year for its survival. Almost all adults die soon after spawning. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on freshwater fish with similar longevity and reproductive lifespans to the Waccamaw silverside found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the Waccamaw silverside's reproductive performance would be reduced substantially. The Waccamaw silverside's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Waccamaw silverside's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Waccamaw silversides may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Waccamaw silverside population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Waccamaw silversides are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Waccamaw silverside.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Waccamaw silverside include high quality water which is clear, open, and has a neutral pH, and a clean sand substrate. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Waccamaw silverside critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Waccamaw silversides, and cause Waccamaw silversides to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of Waccamaw silverside. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Waccamaw silverside's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Waccamaw silverside.

### *Catostomidae*

### MODOC SUCKER
*Catostomus microps*

Modoc suckers exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Modoc suckers exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%. We estimate that Modoc suckers exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Modoc suckers typically lives up to 5 years, achieving sexual maturation in its third year, resulting in a reproductive life span of only one or two years. A comparison of 88 freshwater fish species found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on suckers with similar life-history traits to the Modoc sucker found population growth rate to be highly susceptible to perturbations in juvenile survival, accounting for about 60% of the total elasticity (Vélez-Espino et al., 2006). In combination, juvenile survival and fecundity in these species accounted for about 80% of the total elasticity of the population growth rate. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the Modoc sucker's reproductive performance would be reduced substantially. The reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Modoc sucker's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Modoc suckers may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Modoc sucker's population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Modoc suckers are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Modoc sucker.

**Critical Habitat**: The physical and biological features of critical habitat essential to the conservation of the Modoc sucker include intermittent and permanent-water creeks, and adjacent land areas that provide vegetation for cover and protection from soil erosion. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Modoc sucker critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Modoc suckers, and cause Modoc sucker to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 39% and the reduction in the survival of young fish through the first year as high as 344%. These effects are estimated to be of a magnitude great enough to reduce numbers of Modoc sucker. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Modoc sucker's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Modoc sucker.

## SANTA ANA SUCKER

*Catostomus santaanae*

Santa Ana suckers exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Santa Ana suckers to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%. We estimate that Santa Ana suckers exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival, such as sturgeon. Santa Ana sucker females are short-lived (up to 3 years), mature early, and may have prolonged spawning periods. Fecundity is considered to be exceptionally high for a species of its small size. In a comparison of 88 freshwater fish species, longevity and age at maturity were found to be the best predictors of elasticity patterns Vélez-Espino et al., 2006). Elasticity analyses performed on freshwater fish with similar longevity, age at maturity, and reproductive lifespan as the Santa Ana sucker found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the Santa Ana sucker's reproductive performance would be reduced substantially. The Santa Ana sucker's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Santa Ana sucker's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Santa Ana suckers may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Santa Ana sucker population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Santa Ana suckers are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Santa Ana sucker.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Santa Ana sucker include the following: A functioning hydrological system that experiences peaks and ebbs in the water volume that reflects seasonal variation in precipitation throughout the year; A mosaic of loose sand, gravel, cobble, and boulder substrates in a series of riffles, runs, pools, and shallow sandy stream margins; Water depths greater than 3 cm (1.2 in) and bottom water velocities greater than 0.03 meter per second (0.01 feet per second); Non-turbid water or only seasonally turbid water; Water temperatures less than 30 [deg]C (86 [deg]F); and stream habitat that includes algae, aquatic emergent vegetation, macroinvertebrates, and riparian vegetation.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Santa Ana sucker critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Santa Ana suckers, and cause Santa Ana suckers to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 39% and the reduction in the survival of young fish through the first year as high as 44%. These effects are estimated to be of a magnitude great enough to reduce numbers of Santa Ana sucker. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Santa Ana sucker's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Santa Ana sucker.

## WARNER SUCKER

*Catostomus warnerensis*

Warner suckers exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Warner suckers exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%. We estimate that Warner suckers exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through

the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). Warner suckers have been documented to live up to 17 years, though individuals residing in lakes are presumed to live longer than those found in rivers. Sexual maturity occurs at an age of 3 to 4 years. Spawning usually occurs in April and May in streams, in silt-free, gravel-bottomed, slow flowing sections of creeks. In years when access to stream spawning is limited by low flow or by physical in-stream blockages (such as beaver dams or diversion structures), suckers may attempt to spawn on gravel beds along the lake shorelines.

For species like the Warner sucker that exhibit intermediate longevity, moderately delayed reproduction, and multiple opportunities to spawn, population growth rate tends to be most sensitive to perturbation in juvenile survival (Vélez-Espino et al 2006). Elasticity analyses performed on suckers within the genus *Catostomus* that exhibit similar life-history traits to the Warner sucker found this pattern to hold true, with juvenile survival accounting for 50-60% of the total elasticity (Vélez-Espino et al., 2006). In combination, juvenile survival and fecundity accounted for 70 to 80% of the total elasticity of the population growth rate. A comparison of elasticities among closely related species found that elasticity values are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the Warner sucker's reproductive performance would be reduced substantially. The Warner sucker's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Warner sucker's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Warner suckers may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Warner sucker population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Warner suckers are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Warner sucker.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Warner sucker include the following: Streams should have clean, unpolluted flowing water and a stable riparian zone. The streams should support a variety of aquatic insects, crustaceans, and other small invertebrates for food.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Warner sucker critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Warner suckers, and cause Warner suckers to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 39% and the reduction in the survival of young fish through the first year as high as 44%. These effects are estimated to be of a magnitude great enough to reduce numbers of Warner sucker. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Warner sucker's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Warner sucker.

## SHORTNOSE SUCKER
*Chasmistes brevirostris*

Shortnose sucker exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate shortnose suckers exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%.   We estimate that shortnose suckers exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole.  Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). The shortnose sucker can live more than 30 years, though a 2007 study from the Upper Klamath Lake indicated that the average life expectancy after entering the spawning population was only 3.6 years, suggesting that in some populations, adults may be dying before reproducing often enough for population replacement.  Sexual maturity for shortnose suckers appears to occur between the ages of 4 and 6 years.  Freshwater fish species with longevity and reproductive lifespan characteristics similar to those of the shortnose sucker were found to have population growth rates that were particularly susceptible to perturbations in juvenile survival, accounting for 40% to 50% of the total elasticity (Vélez-Espino et al., 2006).  Recruitment for this species is historically low and continues to be below levels to sustain population growth.  The 2007 5-Year Review for the shortnose sucker identified recruitment of young fish to the breeding population as a high priority goal for recovery.

We anticipate the shortnose sucker's reproductive performance and recruitment would be reduced substantially.  The shortnose sucker's reproductive performance rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival.  The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive.  These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the shortnose sucker's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year.  Shortnose suckers may also experience effects on growth, swimming performance, condition, and development.  Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed.  An effected shortnose sucker population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated.  Based upon the magnitude of effect we anticipate could occur, we conclude ultimately shortnose suckers are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the shortnose sucker.

## CUI-UI

*Chasmistes cujus*

Cui-ui exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate cui-uis exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%. We estimate that cui-uis exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). The cui-ui can live more than 40 years, reaching sexual maturity in 6 to 12 years. Females can produce from 25,000 to 186,000 eggs and larval survival is presumed to be extremely small. Freshwater fish species with longevity and reproductive lifespan characteristics similar to those of the Cui-ui were found to have population growth rates that were particularly susceptible to perturbations in juvenile survival, accounting for 40% to 50% of the total elasticity (Vélez-Espino et al., 2006). The Recovery Plan for the cui-ui stresses the need for increased survival, and consequent recruitment, of young.

We anticipate the cui-ui's reproductive performance and recruitment would be reduced substantially. The cui-ui's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the cui-ui's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Cui-uis may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected cui-ui population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately cui-uis are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the cui-ui.

## JUNE SUCKER
*Chasmistes lioris*

June suckers exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate June suckers exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%. We estimate that June suckers exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

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The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). Freshwater fish species with longevity and reproductive lifespan characteristics similar to those of the June sucker were found to have population growth rates that were particularly susceptible to perturbations in juvenile survival, accounting for 40% to 50% of the total elasticity (Vélez-Espino et al., 2006). June suckers can live to be over 40 years old and reach sexual maturity at age 9. Survival rates for Red Butte refuge population were estimated as follows: age 1, 0.4225; age 2, 0.4625; age 3 0.8020, and adults, 0.9576 (Billman and Crowl, 2007). However, the wild population endemic to Utah Lake has continued to experience wide recruitment failure leaving it dominated by adult fish (Billman and Crowl, 2007). The June sucker Recovery Plan identifies the minimization of factors limiting recruitment as a priority in the recovery of the species.

We anticipate the June sucker's reproductive performance and subsequent recruitment would be further reduced. The June sucker's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the June sucker's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. June suckers may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected June sucker population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately June suckers are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the June sucker.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the June sucker include the following: One to three feet of high quality water constantly flowing over a clean, unsilted gravel substrate. Larval June suckers require shallow areas with low velocities connected to the main channel of the river.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect June sucker critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of June suckers, and cause June suckers to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 39% and the reduction in the survival of young fish through the first year as high as 44%. These effects are estimated to be of a magnitude great enough to reduce numbers of June sucker. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the June sucker's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the June sucker.

## LOST RIVER SUCKER
*Deltistes luxatus*

Lost River sucker exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Lost River suckers exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%. We estimate that Lost River suckers exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). The Lost River sucker can live more than 40 years, though a 2007 study from the Upper Klamath Lake indicated that the average life expectancy after entering the spawning population was only 9 years, suggesting that in some populations, adults may be dying before reproducing often enough for population replacement. Sexual maturity for Lost River suckers occurs between the ages of 6 to 14 years, with most maturing at age 9. Freshwater fish species with longevity and reproductive lifespan characteristics similar to those of the Lost River sucker were found to have population growth rates that were particularly susceptible to perturbations in juvenile survival, accounting for 40% to 50% of the total elasticity (Vélez-Espino et al., 2006). A lack of recruitment has been reported for the river spawing population of the Lost River sucker, and the 2007 5-Year Review identified recruitment of young fish to the breeding population as a whole as a high priority goal for recovery due to large-scale die-offs of adults in the mid-1990's.

We anticipate the Lost River sucker's reproductive performance and recruitment would be reduced substantially. The Lost River sucker's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Lost River sucker's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Lost River suckers may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Lost River sucker population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Lost River suckers are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Lost River sucker.

## RAZORBACK SUCKER
*Xyrauchen texanus*

Razorback suckers exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were

available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate razorback suckers exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 39%. We estimate that razorback suckers exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 44%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). The razorback sucker can live more than 40 years, and begin spawning at 3 to 4 years of age. Freshwater fish species with longevity and reproductive lifespan characteristics similar to those of the razorback sucker were found to have population growth rates that were particularly susceptible to perturbations in juvenile survival, accounting for 40% to 50% of the total elasticity (Vélez-Espino et al., 2006). Though the razorback sucker is a long-lived species, current populations have continued to experience persistent recruitment failure, resulting in the depletion and extirpation of numerous populations. Although razorback sucker in certain locations number in the thousands, low reproductive success, low survival of young, and little or no recruitment have contributed to high demographic uncertainty. Wild razorback sucker populations in many locations of the Colorado River Basin have become aged, senile, and perished from inadequate recruitment.

We anticipate the razorback sucker’s reproductive performance and subsequent recruitment would be further reduced substantially. The razorback sucker’s reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the razorback sucker’s reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Razorback suckers may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected razorback sucker population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately razorback suckers are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the razorback sucker.

### *Cotidae*

### PYGMY SCULPIN
*Cottus paulus*

Pygmy sculpins exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate pygmy sculpins exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that pygmy sculpins exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Pygmy sculpin females are probably short-lived (1-3 years), mature early, and spawn year-round with a peaks in August and late winter. A comparison of 88 freshwater fish species found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the pygmy sculpin found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the pygmy sculpin's reproductive performance would be reduced substantially. The pygmy sculpin's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the pygmy sculpin's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Pygmy sculpins may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected pygmy sculpin population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately pygmy sculpins are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the pygmy sculpin.

## *Cyprinidae*

### BLUE SHINER
*Cyprinella caerulea*

Blue shiner exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were

available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table13). Compared to control populations, we estimate blue shiners exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that blue shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Blue shiner females are short-lived (about 3 years). Most spawning adults are aged 2 years and spawn from early May through late August.

We anticipate the blue shiner's reproductive performance would be reduced substantially. The blue shiner's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the blue shiner's reproductive performance by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Blue shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected blue shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately blue shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the blue shiner.

**BEAUTIFUL SHINER**
*Cyprinella formosa*

Beautiful shiner exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated beautiful shiner exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that beautiful shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like beautiful shiners, than on species with greater adult survival, such as sturgeon. Beautiful shiner females are short-lived (less than 3 years).

We anticipate the beautiful shiner's fertility would be reduced substantially. The beautiful shiner's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

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In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the beautiful shiner's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Beautiful shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected beautiful shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately beautiful shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the beautiful shiner.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the beautiful shiner include permanently flowing small streams with riffles, or intermittent creeks with pools and riffles in the Rio Yaqui drainage. The waters must be clear and unpolluted, and free of exotic fishes. The beautiful shiner needs water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience adequate individual and population growth.

Approval of the CCC in State and Tribal water quality standards would authorized States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect beautiful shiner critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of delta smelt, and cause delta smelt to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of delta smelt. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the beautiful shiner's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the beautiful shiner.

## DEVILS RIVER MINNOW

*Dionda diaboli*

Devils River minnow exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Devils River minnow exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Devils River minnows exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Devils River minnow females are short-lived (1-2 years), mature early, and probably spawn from January through August.

The Devils River minnow probably has a reproductive strategy similar to the smalleye shiner (*Notropis buccula*). Durham and Wilde (2009) studied the population dynamics of the smalleye shiner in the Brazos River, Texas. Smalleye shiner are members of a reproductive guild of cyprinids that broadcast spawn multiple batches of nonadhesive, semibuoyant ova throughout an extended reproductive season, and experience extremely high post-spawning mortality. Elasticity analysis and sensitivity simulations of the projection matrix indicated that age-0 survival and age-1 fecundity were the most influential parameters in the population dynamics of smalleye shiners. In combination, these two stages accounted for the majority (70%) of the total elasticity of the population growth rate.

We anticipate the Devils River minnow's reproductive performance would be reduced substantially. The Devils River minnow's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Devils River minnow's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Devils River minnows may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Devils River minnow population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Devils River minnows are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Devils River minnow.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Devils River minnow include the following: streams with slow to moderate water velocities (10 to 40 cm per second or 4 to 16 inches per second) and shallow to moderate water depths (10 cm to 1.5 m or 4 inches to 4.9 feet). These streams must be near vegetative structure, such as emergent or submerged vegetation or stream bank riparian vegetation that overhangs the water column. Gravel and cobble substrates with low or moderate amounts of fine sediment and low or moderate amounts of substrate embeddedness. Pool, riffle, run, and backwater components free of structures that would prevent fish movement up or downstream. High quality water from groundwater springs and seeps which is: between 17 and 29 degrees Celsius, has dissolved oxygen levels greater than 5 mg/l, a pH between 7 and 8.2, has less than 0.7 mS/cm conductivity and salinity of less than 1 part per thousand, has ammonia levels of less than 0.4 mg/l, and finally has no (or minimal amounts of) pollutants such as copper, arsenic, mercury, cadmium, human and animal waste, pesticides, fertilizers, suspended sediments, petroleum compounds, gasoline, and diesel fuel. Finally, the habitat must possess an adequate algae food base, and no nonnative aquatic species. By high quality water, we mean water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth sufficient for the critical habitat to serve its intended conservation function.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Devils River minnow critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Devils River minnows, and cause Devils River minnows to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Devils River minnow. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Devils River minnow's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Devils River minnow.

## SPOTFIN CHUB
*Cyprinella monarcha*

Spotfin chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate spotfin chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 68%. We estimate that spotfin chub exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 76%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like spotfin chub, than on species with greater adult survival, such as sturgeon. Most spotfin chub start spawning after their second year and may not live beyond the age of three. Counts of mature ova numbered 157-791, but may greatly underestimate fecundity if this species spawns fractionally.

The spotfin chub shares a relatively similar life history with darters. Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to conduct population viability analyses for the two species.

Average slackwater darter fecundity was estimated as 92 and 197, respectively, for one-batch and two-batch fecundity. Based on estimates of adult survival, Hartup (2005) calculated the adult fertility rate would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the spotfin chub's potential recruitment substantially. The spotfin chub's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the spotfin chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Spotfin chub may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected spotfin chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately spotfin chub are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the spotfin chub.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the spotfin chub include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of spotfin chubs, and cause chubs to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 68% and the reduction in the survival of young fish through the first year as high as 76%. These effects are estimated to be of a magnitude great enough to reduce numbers of spotfin chub. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the spotfin chub.

## SLENDER CHUB
*Erimystax cahni*

Slender chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated slender chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that slender chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Slender chub females are short-lived (up to 3 years). A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the slender chub found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the slender chub's reproductive performance would be reduced substantially. The slender chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the slender chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Slender chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected slender chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately slender chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the slender chub.

**Critical Habitat:** Continued approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect slender chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of slender chub, and cause slender chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of slender chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the slender chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the slender chub.

## MOJAVE TUI CHUB

*Gila bicolor mohavensisi*

Mojave tui chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Mojave tui chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Mojave tui chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Mojave tui chub females are short-lived (1-4 years) and highly fecund. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the slender chub found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the Mojave tui chub's reproductive performance would be reduced substantially. The Mojave tui chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Mojave tui chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Mojave tui chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Mojave tui chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Mojave tui chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Mojave tui chub.

## OWENS TUI CHUB

*Gila bicolor snyderi*

Owens tui chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Owens tui chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Owens tui chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

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The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the Owens tui chub found that population growth rates for these species were most susceptible to perturbations in juvenile survival (Vélez-Espino et al., 2006). Juvenile survival accounted for about 58% of the total elasticity of the population growth rate in all species with these life-history traits. A comparison of elasticities among closely related species found that elasticity values are highly conserved among genera within the same taxonomic family. Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*).

We anticipate the Owens tui chub's reproductive performance would be reduced substantially. The Owens tui chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Owens tui chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Owens tui chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Owens tui chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Owens tui chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Owens tui chub.

**Critical Habitat:** Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Owens tui chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Owens tui chub, and cause Owens tui chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Owens tui chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Owens tui chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Owens tui chub.

## BORAX LAKE CHUB
*Gila boraxobius.*

Borax Lake chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Borax Lake chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Borax Lake chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Borax Lake chub females are short-lived (mostly 1-2 years), mature early, and probably spawn twice per year. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the Moapa dace found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn. A comparison of elasticities among closely related species found that elasticity values are highly conserved among genera within the same taxonomic family. Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*).

We anticipate the Borax Lake chub's reproductive performance would be reduced substantially. The Borax Lake chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Borax Lake chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Borax Lake chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Borax Lake chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Borax Lake chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Borax Lake chub.

**Critical Habitat:** Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Borax Lake chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Borax Lake chub, and cause Borax Lake chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Borax Lake chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Borax Lake chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Borax Lake chub.

## HUMPBACK CHUB

*Gila boraxobius.*

Humpback chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated humpback chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that humpback chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Little information is available for humpback chub. A comparison of 88 freshwater fish species, found that elasticities among closely related species are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*) and fecundity accounted for another 19% (Vélez-Espino et al., 2006).

We anticipate the humpback chub's reproductive performance would be reduced substantially. The humpback chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the humpback chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Humpback chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected humpback chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately humpback chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the humpback chub.

**Critical Habitat:** Continued approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect humpback chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of humpback chub, and cause humpback chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of humpback chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the humpback chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the humpback chub.

## SONORA CHUB
*Gila ditaenia.*

Sonora chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Sonora chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Sonora chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Little information is available for Sonora chub. A comparison of 88 freshwater fish species, found that elasticities among closely related species are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*) and fecundity accounted for another 19% (Vélez-Espino et al., 2006).

We anticipate the Sonora chub’s reproductive performance would be reduced substantially. The Sonora chub’s reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Sonora chub’s reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Sonora chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Sonora chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Sonora chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Sonora chub.

**Critical Habitat:** Continued approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Sonora chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Sonora chub, and cause Sonora chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Sonora chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Sonora chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Sonora chub.

## BONYTAIL CHUB
*Gila elegans*

Bonytail chub exposed to cyanide at the criterion continuous concentration (CCC) are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate bonytail chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 57%. We estimate that bonytail chub exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 66%.

As noted previously, the first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival, such as bonytail. Bonytail may live for as many as 50 years and are very fecund. In hatcheries, females produce between 1,000 and 17,000 eggs and the survival rate of juveniles is 17-38%. Despite this reproductive potential, we anticipate the effects of cyanide on fecundity and juvenile survival would have a substantial population-level effect on bonytail. Although bonytail spawn many eggs, they do not guard their eggs and predation by non-native fish in the Colorado River basin is substantial, probably contributing to reproductive failure in the wild. A reduction in potential fecundity and juvenile survival in the wild at the magnitude we estimate will add to the significant challenges to recovering the bonytail.

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the bonytail chub's potential recruitment substantially. The bonytail chub's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the bonytail chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Bonytail chub may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. These potential effects could add substantially to the difficulties in reintroducing reproductively successful populations and attaining recovery. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately bonytail chub are not likely to overcome the effects to their recruitment in waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could limit the reproduction, numbers, and distribution of the bonytail chub.

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**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the bonytail chub include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of bonytail chubs, and cause chubs to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 57% and the reduction in the survival of young fish through the first year as high as 66%. These effects are estimated to be of a magnitude great enough to reduce numbers of bonytail chubs. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the bonytail chub.

## GILA CHUB
*Gila intermedia*

Gila chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Gila chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Gila chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Little information is available for Gila chub. A comparison of 88 freshwater fish species, found that elasticities among closely related species are highly conserved among genera within the same taxonomic family (Vélez -Espino et al., 2006). Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*) and fecundity accounted for another 19% (Vélez -Espino et al., 2006).

We anticipate the Gila chub's reproductive performance would be reduced substantially. The Gila chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Gila chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Gila chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Gila chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Gila chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Gila chub.

**Critical Habitat:** Continued approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Gila chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Gila chub, and cause Gila chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Gila chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Gila chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Gila chub.

## YAQUI CHUB
*Gila purpurea.*

Yaqui chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Yaqui chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Yaqui chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Little information is available for Yaqui chub. A comparison of 88 freshwater fish species, found that elasticities among closely related species are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*) and fecundity accounted for another 19% (Vélez-Espino et al., 2006).

We anticipate the Yaqui chub's reproductive performance would be reduced substantially. The Yaqui chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Yaqui chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Yaqui chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Yaqui chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Yaqui chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Yaqui chub.

**Critical Habitat:** Continued approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Yaqui chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Yaqui chub, and cause Yaqui chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Yaqui chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Yaqui chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Yaqui chub.

## PAHRANAGAT ROUNDTAIL CHUB
*Gila robusta jordani.*

Pahranagat roundtail chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Pahranagat roundtail chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Pahranagat roundtail chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Little information is available for Pahranagat roundtail chub. A comparison of 88 freshwater fish species, found that elasticities among closely related species are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*) and fecundity accounted for another 19% (Vélez-Espino et al., 2006).

We anticipate the Pahranagat roundtail chub's reproductive performance would be reduced substantially. The Pahranagat roundtail chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Pahranagat roundtail chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Pahranagat roundtail chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected

Pahranagat roundtail chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Pahranagat roundtail chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Pahranagat roundtail chub.

**Critical Habitat:** Continued approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Pahranagat roundtail chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Pahranagat roundtail chub, and cause Pahranagat roundtail chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Pahranagat roundtail chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Pahranagat roundtail chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Pahranagat roundtail chub.

## VIRGIN RIVER CHUB

*Gila robusta seminuda*

Virgin River chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Virgin River chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Virgin River chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Little information is available for Virgin River chub. A comparison of 88 freshwater fish species, found that elasticities among closely related species are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). Juvenile survival accounted for about 59% of the total elasticity of the population growth rate for the Utah chub (*Gila atraria*) and fecundity accounted for another 19% (Vélez-Espino et al., 2006).

We anticipate the Virgin River chub's reproductive performance would be reduced substantially. The Virgin River chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Virgin River chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Virgin River chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Virgin River chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Virgin River chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Virgin River chub.

**Critical Habitat:** Continued approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Virgin River chub critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Virgin River chub, and cause Virgin River chub to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Virgin River chub. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Virgin River chub's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Virgin River chub.

## RIO GRANDE SILVERY MINNOW

*Hybognathus amarus*

Rio Grande silvery minnow exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Rio Grande silvery minnow exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Rio Grande silvery minnows exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Rio Grande silvery minnow females are short-lived (less than

2 years). Norris et al. (2008) conducted a population viability analysis for the Rio Grande silvery minnow. The simulations indicated that reproductive output of those individuals just shy of one year old entering their first spawning season – hereafter referred to as Age 0 fish – is a primary factor that determines the extent of population growth from year to year. This reproductive rate, as defined in our analysis, includes both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the reproductive output of Age 0 fish would be reduced substantially. The Rio Grande silvery minnow's reproductive output would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Rio Grande silvery minnow's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Rio Grande silvery minnows may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Rio Grande silvery minnow population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Rio Grande silvery minnows are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Rio Grande silvery minnow.

The physical and biological features of critical habitat essential to the conservation of the Rio Grande silvery minnow include a hydrologic regime which provides sufficient flowing water with low to moderate currents that form the aquatic habitats which the minnow prefers, such as backwaters, shallow side channels, pools, eddies, and runs of varying depth and velocity. The minnow also requires substrates of sand or silt, and sufficient water quality (water of proper temperatures and conditions such as dissolved oxygen content and pH).

**Critical Habitat:** Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Rio Grande silvery minnow critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Rio Grande silvery minnows, and cause Rio Grande silvery minnows to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Rio Grande silvery minnows. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Rio Grande silvery minnow's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Rio Grande silvery minnows.

## BIG SPRING SPINEDACE

*Lepidomeda mollispinis pratensis*

Big Spring spinedace exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 31). Compared to control populations, we estimated Big Spring spinedace exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Big Spring spinedaces exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Big Spring spinedace are sexually mature after about a year and live up to 4 years. Most spawning of the Big Spring spinedace occurs from March through June, with some continuing sporadically into July. The big spring spinedace is believed to broadcast spawn over gravel substrate. Females generally produce from 380-640 eggs during spawning, with older females sometimes producing two complements of ova in one breeding season. (BRRC 2001).

A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the Big Spring spinedace found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the Big Spring spinedace's reproductive performance would be reduced substantially. The Big Spring spinedace's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Big Spring spinedace's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Big Spring spinedaces may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Big Spring spinedace population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Big Spring spinedaces are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Big Spring spinedace.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Big Spring spinedace include the following: Clean, permanent, flowing, spring-fed stream habitat with deep pool areas and shallow marshy areas along the shore; and the absence of nonnative fishes.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Big Spring spinedace critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Big Spring spinedaces, and cause Big Spring spinedaces to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Big Spring spinedace. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Big Spring spinedace's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Big Spring spinedace.

## LITTLE COLORADO SPINEDACE
*Lepidomeda vitatta*

Little Colorado spinedace exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Little Colorado spinedace exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Little Colorado spinedaces exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Little Colorado spinedace are believed to live 3 to 4 years and mature early. Spinedace are late-spring to early-summer spawners although some females have been found to contain mature eggs as late as October. Spawning occurs in slow current over cobbles. Females may spawn more than once per year, and fecundity estimates range from 650-1000 total eggs per female.

A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the Little Colorado spinedace found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the Little Colorado spinedace's reproductive performance would be reduced substantially. The Little Colorado spinedace's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Little Colorado spinedace's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Little Colorado spinedaces may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Little Colorado spinedace population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Little Colorado spinedaces are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Little Colorado spinedace.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Little Colorado spinedace include clean, permanently flowing water with pools and a fine gravel or silt-mud substrate. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Little Colorado spinedace critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Little Colorado spinedace, and cause Little Colorado spinedace to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Little Colorado spinedace. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Little Colorado spinedace's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Little Colorado spinedace.

## SPIKEDACE
*Meda fulgida*

Spikedace exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated spikedace exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that spikedace exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Spikedace females are short-lived (1-2 years) and mature early. Spawning occurs in the spring between April and June and seems to be triggered by a combination of stream discharge and water temperature. Females may be fractional spawners, with elapsed periods of days to weeks between spawning. Fecundity is correlated to age and length, and has been found to be between 90 to 250 ova.

A comparison of 88 freshwater fish species, found that life history parameters such as age at maturity, reproductive lifespan, fecundity, juvenile survivorship, and longevity could be predictors of elasticity patterns (Vélez-Espino et al., 2006). The spikedace has a reproductive strategy similar to the smalleye shiner (*Notropis buccula*), a member of a reproductive guild of cyprinids that broadcast spawn multiple batches of nonadhesive, semibuoyant ova throughout an extended reproductive season, and experience extremely high post-spawning mortality. Durham and Wilde (2009) studied the population dynamics of the smalleye shiner in the Brazos River, Texas. Elasticity analysis and sensitivity simulations of the projection matrix indicated that age-0 survival and age-1 fecundity were the most influential parameters in the population dynamics of smalleye shiners. In combination, these two stages accounted for the majority (70%) of the total elasticity of the population growth rate. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the spikedace's reproductive performance would be reduced substantially. The spikedace's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the spikedace's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Spikedace may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected spikedace population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately spikedace are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the spikedace.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the spikedace include the following: living areas for each stage of the spikedace, a proper food base, a lack of non-native species, and high quality water, with the proper amount of dissolved oxygen, and no or minimal pollutant levels for pollutants such as copper, arsenic, mercury, and cadmium; human and animal waste products; pesticides; suspended sediments; and gasoline or diesel fuels.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect spikedace critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of spikedace, and cause spikedace to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of spikedace. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the spikedace's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the spikedace.

## MOAPA DACE
*Moapa coriacea*

Moapa dace exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Moapa dace exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Moapa daces exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Moapa dace reach sexual maturity at 1 year of age and can live up to 4 years. Moapa dace apparently reproduce year-round, peaking in the spring, in water temperatures ranging from 28°C-32°C. Fecundity is related to fish size and egg counts have range from 60 to 772 depending on size. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the Moapa dace found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the Moapa dace's reproductive performance would be reduced substantially. The Moapa dace's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Moapa dace's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Moapa daces may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Moapa dace population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Moapa daces are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Moapa dace.

## PALEZONE SHINER
*Notropis albizonatus*

Palezone shiner exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Formal Draft Biological Opinion.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated palezone shiner exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that palezone shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Palezone shiner females are short-lived (3-4 years).

We anticipate the palezone shiner's reproductive performance would be reduced substantially. The palezone shiner's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the palezone shiner's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Palezone shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected palezone shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately palezone shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration.

Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the palezone shiner.

## CAHABA SHINER
*Notropis cahabae*

Cahaba shiner exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Cahaba shiner exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Cahaba shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Cahaba shiner females are short-lived (probably 3- 4 years), mature early, and spawn from late May through June.

We anticipate the Cahaba shiner's reproductive performance would be reduced substantially. The Cahaba shiner's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Cahaba shiner's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Cahaba shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Cahaba shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Cahaba shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Cahaba shiner.

## ARKANSAS RIVER SHINER

*Notropis girardi*

Arkansas River shiner exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Arkansas River shiner exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Arkansas River shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Arkansas River shiner females are short-lived (less than 3 years) and appear to experience extremely high mortality after spawning.

The Arkansas River minnow has a reproductive strategy similar to the smalleye shiner (*Notropis buccula*). Durham and Wilde (2009) studied the population dynamics of the smalleye shiner in the Brazos River, Texas. Smalleye shiner are members of a reproductive guild of cyprinids that broadcast spawn multiple batches of nonadhesive, semibuoyant ova throughout an extended reproductive season, and post-spawning mortality is extremely high. Elasticity analysis and sensitivity simulations of the projection matrix indicated that age-0 survival and age-1 fecundity were the most influential parameters in the population dynamics of smalleye shiners. In combination, these two stages accounted for the majority (70%) of the total elasticity of the population growth rate.

We anticipate the Arkansas River shiner's reproductive performance would be reduced substantially. The Arkansas River shiner's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Arkansas River shiner's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Arkansas River shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Arkansas River shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Arkansas River shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Arkansas River shiner.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Arkansas River shiner include the following: —(i) a natural, unregulated hydrologic regime complete with episodes of flood and drought or, if flows are modified or regulated, a hydrologic regime characterized by the duration, magnitude, and frequency of flow events capable of forming and maintaining channel and instream habitat necessary for particular Arkansas River shiner life-stages in appropriate seasons; (ii) a complex, braided channel with pool, riffle (shallow area in a streambed causing ripples), run, and backwater components that provide a suitable variety of depths and current velocities in appropriate seasons; (iii) a suitable unimpounded stretch of flowing water of sufficient length to allow hatching and development of the larvae; (iv) a river bed of predominantly sand, with some patches of gravel and cobble; (v) water quality characterized by low concentrations of contaminants and natural, daily and seasonally variable temperature, turbidity, conductivity, dissolved oxygen, and pH; (vi) suitable reaches of aquatic habitat, as defined by primary constituent elements (i) through (v) above, and adjacent riparian habitat sufficient to support an abundant terrestrial, semiaquatic, and aquatic invertebrate food base; and (vii) few or no predatory or competitive non-native fish species present.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Arkansas River shiner critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Arkansas River shiners, and cause Arkansas River shiners to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Arkansas River shiner. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Arkansas River shiner's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Arkansas River shiner.

## CAPE FEAR SHINER

*Notropis mekistocholas*

Cape Fear shiners exposed to cyanide at the criterion continuous concentration (CCC) are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Cape Fear shiners exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 59%. We estimate that Cape Fear shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 68%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.
The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Cape Fear shiners, than on species with greater adult survival, such as sturgeon. Cape Fear shiners live about two to three years. No information is presently available about this species' reproductive characteristics.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on Cape Fear shiner, we infer the Cape Fear shiner's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Cape Fear shiner's fertility substantially. The Cape Fear shiner's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

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In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Cape Fear shiner's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Cape Fear shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Cape Fear shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Cape Fear shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Cape Fear shiner.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Cape Fear shiner include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of Cape Fear shiners, and cause shiners to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 59% and the reduction in the survival of young fish through the first year as high as 68%. These effects are estimated to be of a magnitude great enough to reduce numbers of Cape Fear shiners. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Cape Fear shiner.

## PECOS BLUNTNOSE SHINER

*Notropis simus pecosensis*

Pecos bluntnose shiner exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Pecos bluntnose shiner exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Pecos bluntnose shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Pecos bluntnose shiner females are short-lived (up to 3 years).

The Pecos bluntnose shiner has a reproductive strategy similar to the smalleye shiner (*Notropis buccula*). Durham and Wilde (2009) studied the population dynamics of the smalleye shiner in the Brazos River, Texas. Smalleye shiner are members of a reproductive guild of cyprinids that broadcast spawn multiple batches of nonadhesive, semibuoyant ova throughout an extended reproductive season, and post-spawning mortality is extremely high. Elasticity analysis and sensitivity simulations of the projection matrix indicated that age-0 survival and age-1 fecundity were the most influential parameters in the population dynamics of smalleye shiners. In combination, these two stages accounted for the majority (70%) of the total elasticity of the population growth rate.
We anticipate the Pecos bluntnose shiner's reproductive performance would be reduced substantially. The Pecos bluntnose shiner's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Pecos bluntnose shiner's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Pecos bluntnose shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Pecos bluntnose shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Pecos bluntnose shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Pecos bluntnose shiner.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Pecos bluntnose shiner include clean permanent water, a main river channel habitat with a sandy substrate, and a low velocity flow. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Pecos bluntnose shiner critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Pecos bluntnose shiners, and cause Pecos bluntnose shiners to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Pecos bluntnose shiner. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Pecos bluntnose shiner's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Pecos bluntnose shiner.

## TOPEKA SHINER
*Notropis Topeka*

Topeka shiner exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Topeka shiner exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Topeka shiners exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Topeka shiners, than on species with greater adult survival, such as sturgeon. Topeka shiner females are short-lived (1-2 years), mature early, and spawn year-round with two peaks in August and late winter.

The Topeka shiner has a reproductive strategy similar to the smalleye shiner (*Notropis buccula*). Durham and Wilde (2009) studied the population dynamics of the smalleye shiner in the Brazos River, Texas. Smalleye shiner are members of a reproductive guild of cyprinids that broadcast spawn multiple batches of nonadhesive, semibuoyant ova throughout an extended reproductive season, and post-spawning mortality is extremely high. Elasticity analysis and sensitivity simulations of the projection matrix indicated that age-0 survival and age-1 fecundity were the most influential parameters in the population dynamics of smalleye shiners. In combination, these two stages accounted for the majority (70%) of the total elasticity of the population growth rate. Although Topeka shiners utilized a different spawning strategy than smalleye shiners, they are similar in that survival of post-spawning adults is extremely low. Consequently, we anticipate that age-0 survival and age-1 fecundity will similarly be most important to the Topeka shiner's growth rate.

We anticipate the Topeka shiner's reproductive performance would be reduced substantially. The Topeka shiner's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Topeka shiner's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Topeka shiners may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Topeka shiner population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Topeka shiners are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Topeka shiner.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Topeka shiner include the following: stream most often with permanent flow, but which become dry during intermittent periods. Side channel pools and oxbows which are either seasonally connected to a stream or maintained by groundwater inputs are also required. Streams and side channel pools with water quality (temperature, turbidity, conductivity, salinity, dissolved oxygen, pH, chemical contaminants, and other chemical characteristics) necessary for unimpaired behavior, growth, and viability of all life stages is essential. Living and spawning areas for adults are required, which must have water velocities of less than a half meter per second, and depths from 0.1 to 2 meters. Similarly, living areas for juveniles are required. These must have the same flow rate as areas for adults, but require greatly decreased depths, that of less than a quarter of a meter. Proper substrate for the Topeka shiner is sand, gravel, cobble, or silt, with proper amounts of fine sediment and substrate embeddedness. A proper food base of terrestrial, semiaquatic and aquatic invertebrates is necessary. A hydrologic regime capable of forming, maintaining, or restoring flow periodicity, channel morphology, fish community composition, off channel habitats, and habitat components essential for the fish. Finally, this fish requires few or no nonnative predatory or competitive species to be present in their habitat.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Topeka shiner critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Topeka shiners, and cause Topeka shiners to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Topeka shiner. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Topeka shiner's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Topeka shiner.

## OREGON CHUB
*Oregonichthys crameri*

Formal Draft Biological Opinion.

Oregon chub exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Oregon chub exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Oregon chubs exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Oregon chub females exhibit an intermediate life span, mature in their second year, and spawn from mid-May through August with peak activity in July. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan as the Oregon chub found that population growth rates for these species were highly susceptible to perturbations in juvenile survival. Juvenile survival accounted for about 58% of the total elasticity of the population growth rate in all species with these life-history traits. A comparison of elasticities among closely related species found that elasticity values are highly conserved among genera within the same taxonomic family.

We anticipate the Oregon chub's reproductive performance would be reduced substantially. The Oregon chub's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Oregon chub's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Oregon chubs may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Oregon chub population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Oregon chubs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Oregon chub.

## BLACKSIDE DACE
*Phoxinus cumberlandensis*

Blackside dace exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated blackside dace exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that blackside daces exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Formal Draft Biological Opinion.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival. Blackside dace generally have a lifespan of 3 years, reach sexual maturity at age 1, and are broadcast spawners, spawning over clean, gravel-sized substrate from April to July. Average fecundity is about 1500. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns, and elasticity values are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). Sensitivity and elasticity analyses were performed for the southern redbelly dace, (*Phoxinus erythrogaster*), a closely-rated species with similar longevity, age at maturity, and reproductive lifespan (Stasiak 2007). Results indicated that population growth rate was most sensitive (>96%) to changes in first-year survival and most elastic to changes in first-year reproduction, followed by first-year survival. Overall fertility accounted for 70% of total elasticity. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the blackside dace's fertility would be reduced substantially. The blackside dace's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the blackside dace's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Blackside daces may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected blackside dace population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately blackside daces are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the blackside dace.

## WOUNDFIN

*Plagopterus argentissimus*

Woundfin exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Woundfin exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Woundfin exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, than on species with greater adult survival. The generation time for the woundfin is predominately limited to 1 year, with most individuals reaching sexual maturity in the second year of a 3-year lifespan. Individuals must achieve sufficient growth prior to the spring spawning period in order to contribute to the next generation. In a comparison of 88 freshwater fish species, longevity and age at maturity were found to be the best predictors of elasticity patterns Vélez-Espino et al., 2006). Elasticity analyses performed on cyprinids with similar longevity, age at maturity, and reproductive lifespan to the woundfin found that population growth rates were most susceptible to perturbations in juvenile survival, followed by fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the woundfin's reproductive performance fertility would be reduced substantially. The woundfin's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the woundfin's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Woundfins may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected woundfin population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately woundfins are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the woundfin.

**Critical Habitat:** Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect woundfin critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of woundfins, and cause woundfins to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of woundfin. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the woundfin's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the woundfin.

## COLORADO PIKEMINNOW

*Ptychocheilus lucius*

Colorado pikeminnows exposed to cyanide at the criterion continuous concentration (CCC) are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were

available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Colorado pikeminnows exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 63%. We estimate that Colorado pikeminnows exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 71%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

The Colorado pikeminnow is a long-lived fish (40+ years) that evolved in a variable system, with high adaptability to natural environmental variability and resilience to natural catastrophes. This evolution has become manifest as pulsed recruitment from periodic strong year classes, great longevity of adults, and low vulnerability of adults to environmental influences. Great longevity and stability of adults provides a "storage effect" for populations, into which periodic recruitment from strong year classes allows fish to become stored (Gilpin 1993). This is seen as a way that Colorado pikeminnow maintain long-term population viability and stability under environmental variation.

As stated in the recovery plan for this species, a critical aspect of recovery is increased frequency of strong year classes. The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Colorado pikeminnow's potential recruitment substantially. The Colorado pikeminnow's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter. The reductions in the Colorado pikeminnow's recruitment could diminish the frequency and could even preclude the occurrence of strong year classes.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Colorado pikeminnow's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Colorado pikeminnows may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Colorado pikeminnow population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Colorado pikeminnows are likely to become extirpated from waters where they are exposed to cyanide toxicity at the CCC. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Colorado pikeminnow.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Colorado pikeminnow include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of Colorado pikeminnow, and cause pikeminnows to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 683 and the reduction in the survival of young fish through the first year as high as 71%. These effects are estimated to be of a magnitude great enough to reduce numbers of Colorado pikeminnows. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Colorado pikeminnow.

## ASH MEADOWS SPECKLED DACE

*Rhinichthys osculus spp*

Ash Meadows speckled dace exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Ash Meadows speckled dace exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Ash Meadows speckled daces exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, than on species with greater adult survival. Ash Meadows speckled dace females reach sexual maturity at 2 years, spawn primarily over the spring and summer, and are believed to live up to 4 years. Elasticity analyses performed on cyprinids with similar life-history traits, including the blacknose dace**,** *Rhinichthys atratulus***,** found that population growth rates for these species were most susceptible to perturbations in juvenile survival, followed by fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate. A comparison of elasticities among closely related species found that elasticity values are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the Ash Meadows speckled dace's fertility would be reduced substantially. The Ash Meadows speckled dace's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Ash Meadows speckled dace's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Ash Meadows speckled daces may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Ash Meadows speckled dace population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Ash Meadows speckled daces are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Ash Meadows speckled dace.

**Critical Habitat:** Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect Ash Meadows speckled dace critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Ash Meadows speckled daces, and cause Ash Meadows speckled daces to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of Ash Meadows speckled dace. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Ash Meadows speckled dace's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the Ash Meadows speckled dace.

## KENDALL WARM SPRINGS DACE

*Rhinichthys osculus thermalis*

Kendall Warm Springs dace exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Kendall Warm Springs dace exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Kendall Warm Springs daces exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn. Kendall Warm Springs dace females reach sexual maturity at 2 years, are thought to spawn several times a year, and are likely to live between 3 and 5 years. Elasticity analyses performed on cyprinids with similar life-history traits, including the blacknose dace**,** *Rhinichthys atratulus***,** found that population growth rates for these species were most susceptible to perturbations in juvenile survival, followed by fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate. A comparison of elasticities among closely related species found that elasticity values are highly conserved among genera within the same taxonomic family (Vélez-Espino et al., 2006). These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn

We anticipate the Kendall Warm Springs dace's reproductive performance would be reduced substantially. The Kendall Warm Springs dace's reproductive performance rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Kendall Warm Springs dace's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Kendall Warm Springs daces may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Kendall Warm Springs dace population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Kendall Warm Springs daces are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Kendall Warm Springs dace.

## LOACH MINNOW
*Tiaroga cobitis*

Loach minnow exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated Loach minnow exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 31%. We estimate that Loach minnows exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 33%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than species with greater adult survival. Loach minnow females are short-lived (1-2 years), and mature early. Elasticity analyses performed on cyprinids with similar life-history traits found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the Loach minnow's reproductive performance would be reduced substantially. The Loach minnow's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Loach minnow's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Loach minnows may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Loach minnow population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Loach minnows are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Loach minnow.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the loach minnow include the following: permanent flowing water with no or low levels of pollutants, which must include living areas for adult loach minnows with moderate to swift flow velocities (9 to 32 inches per second) in shallow water between approximately 1 and 30 inches (3 to 75 cm) in depth. The substrate of these areas should consist of gravel, cobble, or rubble. Furthermore, there must be living areas for juveniles as well. To support juveniles, areas must have moderate to swift flow velocities, from 1 to 34 inches per second (3 and 85 cm/second). Juveniles require the same depth and substrates as adults. It is the larval loach minnow which requires greatly different habitat. Loach minnow larvae require slow to moderate flow velocities, from 3 to 20 inches per second (9 to 50 cm/second) and shallow water with sand, gravel, and cobble substrates.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect loach minnow critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of loach minnows, and cause loach minnows to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. These effects are estimated to be of a magnitude great enough to reduce numbers of loach minnow. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the loach minnow's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the loach minnow.

## ***Gasterosteidae***

## UNARMORED THREESPINE STICKLEBACK
***Gasterosteus aculeatus williamsoni***

Unarmored threespine stickleback exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated unarmored threespine stickleback exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Unarmored threespine stickleback s exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than species with greater adult survival. Unarmored threespine stickleback females are short-lived (about 1 year), mature early, and lay 50-300 eggs into nests guarded by males. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on freshwater fish with similar longevity and reproductive lifespans to the Unarmored threespine stickleback found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the unarmored threespine stickleback's reproductive performance would be reduced substantially. The unarmored threespine stickleback's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the unarmored threespine stickleback's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Unarmored threespine stickleback s may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected unarmored threespine stickleback population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately unarmored threespine sticklebacks are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the unarmored threespine stickleback .

### *Gobiidae*

#### TIDEWATER GOBY
*Eucyclogobius newberryi*

Tidewater goby exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimated tidewater goby exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 36%. We estimate that tidewater gobies exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through

the first year and that reduction could be as much as, but is not likely to be greater than, 40%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than species with greater adult survival. Tidewater goby females are short-lived (about 1 year), mature early, and can Reproduce all year. A comparison of 88 freshwater fish species, found that longevity and age at maturity were the best predictors of elasticity patterns (Vélez-Espino et al., 2006). Elasticity analyses performed on freshwater fish with similar longevity and reproductive lifespans to the tidewater goby found that population growth rates for these species were highly susceptible to perturbations in juvenile survival and fecundity (Vélez-Espino et al., 2006). In combination, these two stages accounted for over 90% of the total elasticity of the population growth rate in all species with these life-history traits. These reproductive parameters, as defined in our analysis, include both egg production and larval / juvenile survival rate to the next year's spawn.

We anticipate the tidewater goby's reproductive performance would be reduced substantially. The tidewater goby's reproductive performance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the tidewater goby's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Tidewater gobies may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected tidewater goby population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately tidewater gobies are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration.

Formal Draft Biological Opinion.

Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the tidewater goby.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the tidewater goby include the following: Persistent, shallow (in the range of about 0.1 to 2 m), still-to-slow-moving aquatic habitat most commonly ranging in salinity from less than 0.5 ppt to about 10 to 12 ppt; Substrates (e.g., sand, silt, mud) suitable for the construction of burrows for reproduction; Submerged and emergent aquatic vegetation, such as Potamogeton pectinatus and Ruppia maritima, that provides protection from predators; and Presence of a sandbar(s) across the mouth of a lagoon or estuary during the late spring, summer, and fall that closes or partially closes the lagoon or estuary, thereby providing relatively stable water levels and salinity.

Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect tidewater goby critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of tidewater gobies, and cause tidewater goby to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 36% and the reduction in the survival of young fish through the first year as high as 40%. These effects are estimated to be of a magnitude great enough to reduce numbers of tidewater goby. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the tidewater goby's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the tidewater goby.

### ***Goodeidae***

### WHITE RIVER SPRINGFISH (ASH SPRING)

*Crenichthys baileyi baileyi*

White River springfish exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate White River springfish exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that White River springfish exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Springfish (genus *Crenichthys*) are uniquely adapted to survive in high temperatures (e.g. 84 to 97 F for Railroad Valley springfish, U.S. Fish and Wildlife Service 1996) and only occur in hot spring environments in Southern Nevada. Their thermal tolerance has enabled them to evolve in habitats that are otherwise uninhabitable for other fish species that are native to the region. Thus, historic populations have benefited from the lack of competition and predation by other fishes. Although life history information is limited, Springfish are believed to be relatively short lived (3 to 4 years), spawn infrequently (average two spawnings per year), and deposit few eggs per spawn (10 to 17; Kopec 1949). Such a reproductive strategy is unlike any other that was previously described in the *Population Response* section and would appear to be highly vulnerable to stressors which reduce fecundity or survival of larvae, juveniles or adults. The vulnerability of this strategy was evidenced by the decline in the Springfish populations which occurred following the introduction of nonnative fish species (Tuttle et al. 1990, U.S. Fish and Wildlife Service 1996 & 1998). These reductions were attributed in large part to predation by nonnative fishes on springfish larvae (U.S. Fish and Wildlife Service 1998). The effects of cyanide on fecundity, egg hatchability, and larval/juvenile survival could result in similar impacts on the springfish populations, as well as, worsen effects caused by nonnative fish predation.

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the White River springfish's potential recruitment substantially. The White River springfish's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the White River springfish's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. White River springfish may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected White River springfish population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately White River springfish are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the White River springfish.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the White River springfish include the following: warm water springs and their outflows and surrounding areas that provide vegetation for cover and habitat for insects and other invertebrates on which the species feeds. By warm water springs and their outflows, we mean water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth sufficient for the critical habitat to serve its intended conservation function.

Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to the criterion concentration. This approval could adversely affect White River springfish critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of White River springfish, and cause White River springfish to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of White River springfish. Continued approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the White River springfish's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the White River springfish.

## HIKO WHITE RIVER SPRINGFISH

*Crenichthys baileyi grandis*

Hiko White River springfish exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Hiko White River springfish exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Hiko White River springfish exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Springfish (genus *Crenichthys*) are uniquely adapted to survive in high temperatures (e.g. 84 to 97 F for Railroad Valley springfish, U.S. Fish and Wildlife Service 1996) and only occur in hot spring environments in Southern Nevada. Their thermal tolerance has enabled them to evolve in habitats that are otherwise uninhabitable for other fish species that are native to the region. Thus, historic populations have benefited from the lack of competition and predation by other fishes. Although life history information is limited, Springfish are believed to be relatively short lived (3 to 4 years), spawn infrequently (average two spawnings per year), and deposit few eggs per spawn (10 to 17; Kopec 1949). Such a reproductive strategy is unlike any other that was previously described in the *Population Response* section and would appear to be highly vulnerable to stressors which reduce fecundity or survival of larvae, juveniles or adults. The vulnerability of this strategy was evidenced by the decline in the Springfish populations which occurred following the introduction of nonnative fish species (Tuttle et al. 1990, U.S. Fish and Wildlife Service 1996 & 1998). These reductions were attributed in large part to predation by nonnative fishes on springfish larvae (U.S. Fish and Wildlife Service 1998). The effects of cyanide on fecundity, egg hatchability, and larval/juvenile survival could result in similar impacts on the springfish populations, as well as, worsen effects caused by nonnative fish predation.

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Hiko White River springfish's potential recruitment substantially. The Hiko White River springfish's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Hiko White River springfish's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Hiko White River springfish may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Hiko White River springfish population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Hiko White River springfish are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Hiko White River springfish.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Hiko White River springfish include the following: warm water springs and their outflows and surrounding areas that provide vegetation for cover and habitat for insects and other invertebrates on which the species feeds. By warm water springs and their outflows, we mean water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth sufficient for the critical habitat to serve its intended conservation function.

Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to the criterion concentration. This approval could adversely affect Hiko White River springfish critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Hiko White River springfish, and cause Hiko White River springfish to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of Hiko White River springfish. Continued approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Hiko White River springfish's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Hiko White River springfish.

## RAILROAD VALLEY SPRINGFISH
*Crenichthys nevadae*

Railroad Valley springfish exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Formal Draft Biological Opinion.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Railroad Valley springfish exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Railroad Valley springfish exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Springfish (genus *Crenichthys*) are uniquely adapted to survive in high temperatures (e.g. 84 to 97 F for Railroad Valley springfish, U.S. Fish and Wildlife Service 1996) and only occur in hot spring environments in Southern Nevada. Their thermal tolerance has enabled them to evolve in habitats that are otherwise uninhabitable for other fish species that are native to the region. Thus, historic populations have benefited from the lack of competition and predation by other fishes. Although life history information is limited, Springfish are believed to be relatively short lived (3 to 4 years), spawn infrequently (average two spawnings per year), and deposit few eggs per spawn (10 to 17; Kopec 1949). Such a reproductive strategy is unlike any other that was previously described in the *Population Response* section and would appear to be highly vulnerable to stressors which reduce fecundity or survival of larvae, juveniles or adults. The vulnerability of this strategy was evidenced by the decline in the Springfish populations which occurred following the introduction of nonnative fish species (Tuttle et al. 1990, U.S. Fish and Wildlife Service 1996 & 1998). These reductions were attributed in large part to predation by nonnative fishes on springfish larvae (U.S. Fish and Wildlife Service 1998). The effects of cyanide on fecundity, egg hatchability, and larval/juvenile survival could result in similar impacts on the springfish populations, as well as, worsen effects caused by nonnative fish predation.

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Railroad Valley springfish's potential recruitment substantially. The Railroad Valley springfish's potential recruitment would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

Formal Draft Biological Opinion.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Railroad Valley springfish's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Railroad Valley springfish may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Railroad Valley springfish population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Railroad Valley springfish are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Railroad Valley springfish.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Railroad Valley springfish include the following: clear, unpolluted thermal spring waters ranging in temperature from 84 to 97 F in pools, flowing channels, and marshy areas with aquatic plants, insects and mollusks. By clear, unpolluted thermal spring waters, we mean water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth sufficient for the critical habitat to serve its intended conservation function.

Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to the criterion concentration. This approval could adversely affect Railroad Valley springfish critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of Railroad Valley springfish, and cause Railroad Valley springfish to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of Railroad Valley springfish. Continued approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the Railroad Valley springfish's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Railroad Valley springfish.

### ***Osmeridae***

### DELTA SMELT
*Hypomesus transpacificus*

Delta smelt exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate delta smelt exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that delta smelt exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like delta smelt, than on species with greater adult survival, such as sturgeon. Delta smelt live only 1 year and die after spawning. Female smelt lay between 1,600 and 2,600 eggs in the spring. The number and hatchability of eggs together with the survival of larval and juvenile smelt are what drives annual population abundance in this annual fish species.

We anticipate the delta smelt's annual abundance would be reduced substantially. The delta smelt's annual abundance would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year to spawn, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the delta smelt's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish. Delta smelt may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected delta smelt population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur and the delta smelt's current population abundance, we conclude ultimately delta smelt are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the delta smelt.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the delta smelt include water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience adequate individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorized States and Tribes to continue to manage cyanide in waters to the criteria concentration. This approval could adversely affect delta smelt critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of delta smelt, and cause delta smelt to experience adverse effects to growth swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of delta smelt. Continued approval of the CCC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the delta smelt's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for the conservation of the delta smelt.

## ***Percidae***

### SLACKWATER DARTER
*Etheostoma boschungi*

Slackwater darters exposed to cyanide at the CCC are likely to experience reduced survival, reproduction, growth, swimming performance, condition, and exhibit developmental abnormalities, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Few data are available to estimate the magnitude of effects that could occur. We developed a quantitative estimate of the effects on fecundity and juvenile survival from exposure to cyanide at the CCC (Table 13). Compared to control populations, we estimate slackwater darters exposed to cyanide at the CCC could experience a significant reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that slackwater darters exposed to cyanide at the CCC could experience a significant reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the juveniles that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and increase the interval between spawning events.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on shorter-lived fish, like darters, than on longer-lived species, such as sturgeon. Slackwater darter females are short-lived (up to 4 years) and reproduce in no more than three years. The slackwater darter has the potential to spawn multiple times per season, but its spawning period is short, about 1 month, and it is not known how many clutches are spawned (Hartup 2005).

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to conduct population viability analyses for the two species. Average slackwater darter fecundity was estimated as 92 and 197 eggs, respectively, for one-batch and two-batch fecundity. Based on estimates of adult survival, Hartup (2005) calculated the adult fertility rate would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable (the fertility rate was defined as the number of female offspring per female aged *i* per year). Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce Hartup's (2005) estimated slackwater darter fertility (0.896) substantially. Incorporating our estimates, the revised fertility rates would be reduced substantially, as follows: based upon (a) reductions in numbers of hatched eggs (0.31), (b) reductions in young fish surviving through the first year (0.23), and (c) the additive effect of reduced fecundity and survival (0.07). These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the slackwater darter's reproduction by reducing the number of eggs females spawn, reducing the hatchability of spawned eggs, and reducing the survival of young fish through the first year. Slackwater darters may also experience effects on growth, swimming performance, condition, and development. Based upon our estimates of the magnitude of effects, we would not expect a reduction in density-dependence, if any, to compensate for the reductions in fecundity and juvenile survivorship. We would anticipate a consequent reduction in numbers of slackwater darters. An effected slackwater darter population's growth rate could stabilize at a reduced absolute population number, or the population could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude slackwater darters could become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. The proposed chronic criterion could reduce the reproduction, numbers, and distribution of the slackwater darter.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the slackwater darter include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of slackwater darters, and cause darters to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 65% and the reduction in the survival of young fish through the first year as high as 74%. These effects are estimated to be of a magnitude great enough to reduce numbers of slackwater darters. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the slackwater darter.

## VERMILION DARTER
*Etheostoma chermocki*

Vermilion darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate vermilion darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that vermilion darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon. Comprehensive studies of vermilion darter life history have not been completed, but it is believed that its life history attributes are similar to those of other snubnose darters, which reach sexual maturity at 1 year of age. Most darters are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on vermilion darters, we infer the vermilion darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the vermilion darter's fertility substantially. The vermilion darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the vermilion darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Vermilion darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected vermilion darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately vermilion darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the vermilion darter.

## RELICT DARTER

*Etheostoma chienense*

Relict darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate relict darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that relict darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon. We have no information on the relict darter's lifespan, survival, or fecundity. We assume the relict darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on relict darters, we infer the relict darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the relict darter's fertility substantially. The relict darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the relict darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Relict darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected relict darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately relict darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the relict darter.

## ETOWAH DARTER
*Etheostoma etowahae*

Etowah darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Etowah darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that Etowah darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon. We have no information on the Etowah darter's lifespan, survival, or fecundity. We assume the Etowah darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on Etowah darters, we infer the Etowah darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Etowah darter's fertility substantially. The Etowah darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Etowah darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Etowah darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Etowah darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Etowah darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Etowah darter.

## FOUNTAIN DARTER

Formal Draft Biological Opinion.

*Etheostoma fonticola*

Fountain darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column. Exposure of fountain darters to cyanide at the criterion maximum concentration (CMC) is likely to reduce survival.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate fountain darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 81%. We estimate that fountain darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 86%. In addition, we estimate that fountain darters exposed to cyanide at the CMC are likely to experience a substantial reduction in survival and that reduction could be as much as, but is not likely to be greater than, 58.2%. Effects of cyanide at the CMC on the survival of other fountain darter life stages is expected to be of lesser magnitude.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like fountain darters, than on species with greater adult survival, such as sturgeon. Fountain darter females are short-lived (1-2 years), mature early, and spawn year-round with two peaks in August and late winter. In the San Marcos River, mature ova were collected from individuals about 3.5 months old (Linam et al. 1993). Bonner et al. (1998) reported mean egg production (± 1 SD) is 760 (±310) per two breeding pairs during a 33-day period at 23 degrees C, under laboratory conditions. Annual fecundity is probably substantial.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The fountain darter is shorter-lived but more fecund than the slackwater darter and holiday darter. Nevertheless, we anticipate the fountain darter's fertility would be reduced substantially. The fountain darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the fountain darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Fountain darters may also experience effects on growth, swimming performance, condition, and development. Exposure to cyanide at the acute criterion could also substantially reduce the survivorship of juvenile fountain darters. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected fountain darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately fountain darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the acute and chronic criteria could reduce the reproduction, numbers, and distribution of the fountain darter.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the fountain darter include adequate flows, undisturbed substrate, aquatic vegetation including filamentous green algae, and water quality. Approval of the CCC and CMC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect fountain darter critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of fountain darters, and cause fountain darters to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high as 31% and the reduction in the survival of young fish through the first year as high as 33%. Reduction in survivorship of juvenile fish exposed at the CMC could be as high as 58.2% These effects are estimated to be of a magnitude great enough to reduce numbers of fountain darter. Continued approval of the CCC and CMC could adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the fountain darter's extirpation from critical habitat containing cyanide at the CCC and CMC. Impacts to water quality resulting from management of cyanide to the CCC and CMC would diminish the ability of critical habitat to provide for the conservation of the fountain darter.

## NIANGUA DARTER
*Etheostoma nianguae*

Niangua darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Niangua darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that Niangua darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Niangua darters, than on species with greater adult survival, such as sturgeon. The Niangua darter reaches sexual maturity at 1 year of age and can live up to 4 years. Estimates of fecundity are based upon counts of mature ova from collected fish: the number of mature ova averaged 189.8 for four females of age-group I, 387.5 for two females of age-group II. A female of age-group IV had 748 mature eggs.

Niangua darter females are short-lived (up to 4 years) and reproduce in no more than three years. The Niangua darter spawns from March to June.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The Niangua darter's fecundity is somewhat greater than the slackwater darter's and holiday darter's fecundity and their lifespans are similar. The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Niangua darter's fertility substantially. The Niangua darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Niangua darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Niangua darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Niangua darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Niangua darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Niangua darter.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Niangua darter include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of Niangua darters, and cause darters to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 65% and the reduction in the survival of young fish through the first year as high as 74%. These effects are estimated to be of a magnitude great enough to reduce numbers of Niangua darters. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Niangua darter.

## WATERCRESS DARTER
*Etheostoma nuchale*

Watercress darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate watercress darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that watercress darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon. We have no information on the watercress darter's lifespan, survival, or fecundity. We assume the relict darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on watercress darters, we infer the watercress darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the watercress darter's fertility substantially. The watercress darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the watercress darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Watercress darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected watercress darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately watercress darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the watercress darter.

## OKALOOSA DARTER
*Etheostoma okaloosae*

Okaloosa darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Okaloosa darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that Okaloosa darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon. Fecundity of Okaloosa darters is very low - mean mature ova were only 29 (Ogilvie 1980).

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity 1analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The Okaloosa darter's fecundity is much less than the slackwater darter's and holiday darter's fecundity. The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Okaloosa darter's fertility substantially. The Okaloosa darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Okaloosa darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Okaloosa darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Okaloosa darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Okaloosa darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Okaloosa darter.

## DUSKYTAIL DARTER

*Etheostoma percnurum*

Duskytail darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate duskytail darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that duskytail darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like duskytail darters, than on species with greater adult survival, such as sturgeon. Duskytail darter adult survival is very low. A very small percentage of the population survives to age 3. Duskytail darters can spawn as 1-year-olds. Spawning frequency ranged from 5-7 clutches per year, and fecundity from 135-189 (Layman 1991)

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The duskytail darter's fecundity and survival approximates or is somewhat less than the slackwater darter's and holiday darter's fecundity and survival. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the duskytail darter's fertility substantially. The duskytail darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the duskytail darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Duskytail darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected duskytail darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately duskytail darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the duskytail darter.

## BAYOU DARTER

*Etheostoma rubrum*

Bayou darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate bayou darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that bayou darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like bayou darters, than on species with greater adult survival, such as sturgeon. Most Bayou darters start spawning after their first year and do not live beyond the age of three. Depending on the size of the female, clutches can range from 20-75 eggs. A single female likely spawns at least twice per reproductive season based on the size classes of ova.

Bayou darter females are short-lived (up to 3 years) and reproduce in no more than three years. The bayou darter spawns from March to June. Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The bayou darter's fecundity approximates or is slightly less than the slackwater and holiday darter's. The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the bayou darter's fertility substantially. The bayou darter's fertility would be diminished because of (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the bayou darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Bayou darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected bayou darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately bayou darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the bayou darter.

## CHEROKEE DARTER
*Etheostoma scotti*

Cherokee darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Cherokee darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that Cherokee darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration.

Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Cherokee darters, than on species with greater adult survival, such as sturgeon. We have no information on the Cherokee darter's lifespan, survival, or fecundity. We assume the Cherokee darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on Cherokee darters, we infer the Cherokee darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Cherokee darter's fertility substantially. The Cherokee darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Cherokee darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Cherokee darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Cherokee darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Cherokee darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Cherokee darter.

## MARYLAND DARTER
*Etheostoma sellare*

Maryland darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Maryland darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that Maryland darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon. We have no information on the Maryland darter's lifespan, survival, or fecundity. We assume the Maryland darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on Maryland darters, we infer the Maryland darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Maryland darter's fertility substantially. The Maryland darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Maryland darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Maryland darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Maryland darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Maryland darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Maryland darter.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Maryland darter include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of Maryland darters, and cause darters to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 65% and the reduction in the survival of young fish through the first year as high as 74%. These effects are estimated to be of a magnitude great enough to reduce numbers of Maryland darters. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Maryland darter.

## BLUEMASK DARTER
*Etheostoma sp.*

Bluemask darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate bluemask darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 65%. We estimate that bluemask darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon. We have no information on the bluemask darter’s lifespan, survival, or fecundity. We assume the bluemask darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on bluemask darters, we infer the bluemask darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the bluemask darter's fertility substantially. The bluemask darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the bluemask darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Bluemask darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected bluemask darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately bluemask darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the bluemask darter.

## BOULDER DARTER

Formal Draft Biological Opinion.

*Etheostoma wapiti*

Boulder darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate boulder darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 59%. We estimate that boulder darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 74%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like boulder darters, than on species with greater adult survival, such as sturgeon. No life history studies have been conducted on this species. We assume the boulder darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years, and spawning occurs in spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

Since there are no data on boulder darters, we infer the boulder darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the boulder darter's fertility substantially. The boulder darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the boulder darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Boulder darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected boulder darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately boulder darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the boulder darter.

## AMBER DARTER
*Percina antesella*

Amber darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate amber darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 63%. We estimate that amber darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 72%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species, like amber darters, that have fewer opportunities to spawn than on species with greater adult survival, such as sturgeon. Amber darter females are short-lived (approximately 3 years) and probably reproduce in only two years. The amber darter spawns from late fall to early spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

There are no data on the fecundity and survival of amber darters. We infer the amber darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the amber darter's fertility substantially. The amber darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the amber darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Amber darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected amber darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately amber darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the amber darter.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the amber darter include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of amber darters, and cause darters to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 63% and the reduction in the survival of young fish through the first year as high as 72%. These effects are estimated to be of a magnitude great enough to reduce numbers of amber darters. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the amber darter.

## GOLDLINE DARTER
*Percina aurolineata*

Goldline darters exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate goldline darters exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 63%. We estimate that goldline darters exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 72%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like goldline darters, than on species with greater adult survival, such as sturgeon. The life history of the goldline darter is unknown. We assume the goldline darter is like other darters: females are short-lived (up to 4 years) and reproduce in no more than three years.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

There are no data on the fecundity and survival of goldline darters. We assume the goldline darter's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the amber darter's fertility substantially. The goldline darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the goldline darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Goldline darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected goldline darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately goldline darters are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the goldline darter.

## CONASAUGA LOGPERCH

*Percina jenkinsi*

Conasauga logperch exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Conasauga logperch exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 63%. We estimate that Conasauga logperch exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 72%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Conasauga logperch, than on species with greater adult survival, such as sturgeon. Conasauga logperch females are short-lived (probably up to 4 years) and probably reproduce in no more than three years. The Conasauga logperch's spawning is probably limited to spring.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005). There are no data on the fecundity and survival of Conasauga logperch. We infer the Conasauga logperch's fecundity and survival are no greater than those estimated for the slackwater darter and holiday darter. Consequently, the reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the Conasauga logperch's fertility substantially. The Conasauga logperch's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Conasauga logperch's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Conasauga logperch may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Conasauga logperch population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Conasauga logperch are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Conasauga logperch.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Conasauga logperch include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of Conasauga logperch, and cause logperch to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 63% and the reduction in the survival of young fish through the first year as high as 72%. These effects are estimated to be of a magnitude great enough to reduce numbers of Conasauga logperch. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Conasauga logperch.

## LEOPARD DARTER
*Percina pantherina*

Leopard darter exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate leopard darter exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 63%. We estimate that leopard darter exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through

the first year and that reduction could be as much as, but is not likely to be greater than, 72%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like leopard darter, than on species with greater adult survival, such as sturgeon. Leopard darter females are short-lived (up to 4 years) and reproduce in no more than three years. The leopard darter spawns from March to April.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

There is limited data on the fecundity of leopard darters and no information on survival. Robison's (1978) and Hartup's (2005) estimates of fecundity are not directly comparable because their methods differed. Nevertheless, the leopard darter's life history is similar to that of other darters, including the slackwater and holiday darter. The reductions we estimate in hatched eggs and survival of young fish through the first year would reduce the leopard darter's fertility substantially. The leopard darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the leopard darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Leopard darters may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected leopard darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately leopard darter are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the leopard darter.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the leopard darter include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of leopard darter, and cause darters to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 63% and the reduction in the survival of young fish through the first year as high as 72%. These effects are estimated to be of a magnitude great enough to reduce numbers of leopard darter. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the leopard darter.

## ROANOKE LOGPERCH
*Percina rex*

Roanoke logperch exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were

available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Roanoke logperch exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 63%. We estimate that Roanoke logperch exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 72%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like most darters, than on species with greater adult survival, such as sturgeon. Roanoke logperch females live 5-6 years.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The Roanoke logperch is longer-lived and more fecund than the slackwater darter and holiday darter. Nevertheless, we anticipate the Roanoke logperch's fertility would be reduced substantially. The Roanoke logperch's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Roanoke logperch's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Roanoke logperch may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Roanoke logperch population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Roanoke logperch are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Roanoke logperch.

## SNAIL DARTER
*Percina tanasi*

Snail darter exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate snail darter exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 63%. We estimate that snail darter exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 72%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration.

Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like snail darter, than on species with greater adult survival, such as sturgeon. Snail darter females are short-lived (up to 4 years) and reproduce in no more than three years. The snail darter spawns in early February through April and average fecundity is about 600 eggs.

Hartup (2005) conducted field research on the slackwater darter and holiday darter (*E. brevirostrum*) and developed species-specific estimates for fecundity, adult survival, and population sizes, then used these data to perform population viability analyses for the two species. Average slackwater darter annual fecundity was estimated as 92 and 197 eggs spawned, respectively, for one-batch and two-batch fecundity. Two-batch fecundity for the holiday darter was averaged 175 eggs spawned. Based on estimates of adult survival, Hartup (2005) estimated the adult fertility rate (number of female offspring surviving to year 1) would need to average 0.896 for the slackwater darter population to be stable and 0.818 for the holiday darter population to be stable. Assuming the slackwater darter population was stable, Hartup estimated the average proportion of spawned eggs needing to survive through the first year as 0.019 or 0.009, for one- and two-batch fecundity, respectively. For the holiday darter, the two-batch estimate was also 0.009. An elasticity analysis for the slackwater darter and holiday darter (*E. brevirostrum*) identified fertilities (defined as fecundity plus juvenile survival) as having the largest relative contribution to population growth rates for each species, as opposed to adult survival (Hartup 2005).

The snail darter has a similar lifespan but is more fecund than the slackwater darter and holiday darter. We anticipate the snail darter's fertility would be reduced substantially, but less so than the slackwater or holiday darter's fertility. The snail darter's fertility rates would be reduced based upon (a) reductions in numbers of hatched eggs, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the snail darter's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. snail darter may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected snail darter population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately snail darter are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the snail darter.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the snail darter include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of snail darter, and cause darters to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 63% and the reduction in the survival of young fish through the first year as high as 72%. These effects are estimated to be of a magnitude great enough to reduce numbers of snail darter. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the snail darter.

### ***Poeciliidae***

## BIG BEND GAMBUSIA
*Gambusia gaigei*

Big Bend gambusia exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity and survival of young first-year fish (Table 13). Compared to control populations, we estimate Big Bend gambusia exposed to cyanide at the CCC could experience a substantial reduction in the number of young produced and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Big Bend gambusia exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Big Bend gambusia is a live-bearing fish, that is short lived (life expectancy 2 years; U.S. Fish and Wildlife Service 2009), has a prolonged reproductive period (5 to 8 months), and females may produce 50 or more young in the peak season. Such a life history strategy would appear to be highly sensitive to reductions in fecundity and survival of young fish. An elasticity analysis for other short-lived fish species indicates that population growth is, in fact, most influenced by changes in fecundity and juvenile survival (Vélez-Espino et al. 2006). Although gambusia are viviparous, we believe that our estimates of effects on Big Bend gambusia fecundity, which were based on studies with oviparous fish, are applicable. Koyo et al. (2000) described the dynamics of oocyte (egg) and embryonic development in *Gambusia affinis*, the mosquitofish. The process of oocyte development, fertilization, and "hatching" in viviparous and oviparous fishes are comparable, except that fertilization and "hatching" occur internally in viviparous species. Thus, the effects of cyanide on oocyte development described in the *Chronic Toxicity to Fish* section are likely to occur in Gambusia, with similar outcomes. In studies with oviparous fish, only the largest and most mature oocytes were spawned (ovulation) and capable of being fertilized. Similarly, in the viviparous gambusia only the largest and most mature oocytes are capable of being fertilized in the follicle. Thus reductions in the number of mature oocytes, by cyanide, would reduce the number of eggs capable of being fertilized, the number of embryos that are "hatched" (internally), and the number of live young that are "born". We would also expect predicted effects on juvenile survival to be the same for live-bearing species.

The reductions we estimate in the number of young produced and survival of young fish through the first year would reduce the Big Bend gambusia's potential recruitment substantially. The Big Bend gambusia's potential recruitment would be diminished because of (a) reductions in numbers of young produced, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the young that are produced in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Big Bend gambusia's reproduction by reducing the number of young produced by females, and reducing the survivorship of young fish in their first year. Big Bend gambusia may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Big Bend gambusia population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Big Bend gambusia are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Big Bend gambusia.

## SAN MARCOS GAMBUSIA
*Gambusia georgei*

San Marcos gambusia exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity and survival of young first-year fish (Table 13). Compared to control populations, we estimate San Marcos gambusia exposed to cyanide at the CCC could experience a substantial reduction in the number of young produced and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that San Marcos gambusia exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Little is known about the reproductive capabilities of San Marcos gambusia except that it is a live-bearing fish and two individuals kept in laboratory aquaria produced 12, 30, and 60 young although the largest clutch appeared to have been aborted and did not survive (Edwards et al. 1980). However, more is known about the closely related Big Bend gambusia (*Gambusia gaigei*) which we will use to provide insight into the reproductive traits of San Marcos gambusia. Big Bend gambusia is short lived (life expectancy 2 years; U.S. Fish and Wildlife Service 2009), has a prolonged reproductive period (5 to 8 months), and females may produce 50 or more young in the peak season. Such a life history strategy would appear to be highly sensitive to reductions in fecundity and survival of young fish. An elasticity analysis for other short-lived fish species indicates that population growth is, in fact, most influenced by changes in fecundity and juvenile survival (Vélez-Espino et al. 2006). Although gambusia are viviparous, we believe that our estimates of effects on San Marcos gambusia fecundity, which were based on studies with oviparous fish, are applicable. Koyo et al. (2000) described the dynamics of oocyte (egg) and embryonic development in *Gambusia affinis*, the mosquitofish. The process of oocyte development, fertilization, and “hatching” in viviparous and oviparous fishes are comparable, except that fertilization and “hatching” occur internally in viviparous species. Thus, the effects of cyanide on oocyte development described in the *Chronic Toxicity to Fish* section are likely to occur in Gambusia, with similar outcomes. In studies with oviparous fish, only the largest and most mature oocytes were spawned (ovulation) and capable of being fertilized. Similarly, in the viviparous gambusia only the largest and most mature oocytes are capable of being fertilized in the follicle. Thus reductions in the number of mature oocytes, by cyanide, would reduce the number of eggs capable of being fertilized, the number of embryos that are "hatched" (internally), and the number of live young that are "born". We would also expect predicted effects on juvenile survival to be the same for live-bearing species.

The reductions we estimate in the number of young produced and survival of young fish through the first year would reduce the San Marcos gambusia's potential recruitment substantially. The San Marcos gambusia's potential recruitment would be diminished because of (a) reductions in numbers of young produced, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the young that are produced in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the San Marcos gambusia's reproduction by reducing the number of young produced by females, and reducing the survivorship of young fish in their first year. San Marcos gambusia may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected San Marcos gambusia population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately San Marcos gambusia are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the San Marcos gambusia.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the San Marcos gambusia were not identified in the final rule designating critical habitat. We describe them here to include water of sufficient quality for the species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth sufficient for the critical habitat to serve its intended conservation function.

Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to the criterion concentration. This approval could adversely affect San Marcos gambusia critical habitat by diminishing the quality of water to the degree that it would impair individual reproduction and survival of San Marcos gambusia, and cause San Marcos gambusia to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of young produced could be as high 48% and the reduction in the survival of young fish through the first year as high as 56%. These effects are estimated to be of a magnitude great enough to reduce numbers of San Marcos gambusia. Continued approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in the San Marcos gambusia's extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the San Marcos gambusia.

## CLEAR CREEK GAMBUSIA

*Gambusia heterochir*

Clear Creek gambusia exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity and survival of young first-year fish (Table 13). Compared to control populations, we estimate Clear Creek gambusia exposed to cyanide at the CCC could experience a substantial reduction in the number of young produced and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Clear Creek gambusia exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The Clear Creek gambusia is viviparous and females can produce several broods per year. In Clear Creek, females are reproductive for 7 months (March-September) and all stream reaches inhabited by Clear Creek gambusia have pregnant females during the midsummer reproductive period. In the lab (at 25 C), females produced up to 50 young every 42 days. However, in the cooler waters of Clear Creek (20 C) the estimated interbrood interval is 60 days. It's not clear how long Clear Creek gambusia live, but the closely related Big Bend gambusia (*Gambusia gaigei*) has a life expectancy of 2 years (U.S. Fish and Wildlife Service 2009). If the Clear Creek gambusia has similar longevity, their life history strategy (short-lived with extended reproductive periods and relatively low numbers of young per brood) would appear to be highly sensitive to reductions in fecundity and survival of young fish. An elasticity analysis for other short-lived fish species indicates that population growth is, in fact, most influenced by changes in fecundity and juvenile survival (Vélez-Espino et al. 2006). Although gambusia are viviparous, we believe that our estimates of effects on Clear Creek gambusia fecundity, which were based on studies with oviparous fish, are applicable. Koyo et al. (2000) described the dynamics of oocyte (egg) and embryonic development in *Gambusia affinis*, the mosquitofish. The process of oocyte development, fertilization, and "hatching" in viviparous and oviparous fishes are comparable, except that fertilization and "hatching" occur internally in viviparous species. Thus, the effects of cyanide on oocyte development described in the *Chronic Toxicity to Fish* section are likely to occur in Gambusia, with similar outcomes. In studies with oviparous fish, only the largest and most mature oocytes were spawned (ovulation) and capable of being fertilized. Similarly, in the viviparous gambusia only the largest and most mature oocytes are capable of being fertilized in the follicle. Thus reductions in the number of mature oocytes, by cyanide, would reduce the number of eggs capable of being fertilized, the number of embryos that are "hatched" (internally), and the number of live young that are "born". We would also expect predicted effects on juvenile survival to be the same for live-bearing species.

The reductions we estimate in the number of young produced and survival of young fish through the first year would reduce the Clear Creek gambusia's potential recruitment substantially. The Clear Creek gambusia's potential recruitment would be diminished because of (a) reductions in numbers of young produced, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the young that are produced in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Clear Creek gambusia's reproduction by reducing the number of young produced by females, and reducing the survivorship of young fish in their first year. Clear Creek gambusia may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Clear Creek gambusia population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Clear Creek gambusia are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Clear Creek gambusia.

## PECOS GAMBUSIA
*Gambusia nobilis*

Pecos gambusia exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity and survival of young first-year fish (Table 13). Compared to control populations, we estimate Pecos gambusia exposed to cyanide at the CCC could experience a substantial reduction in the number of young produced and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Pecos gambusia exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Pecos gambusia bear live young. Females produce up to 40 young per brood during spring and summer. The interbrood interval for Pecos gambusia is unknown, but other species in the Poeciliidae family typically bear young every 1-2 months while reproductive. It's not clear how long Pecos gambusia live, but the closely related Big Bend gambusia (*Gambusia gaigei*) has a life expectancy of 2 years (U.S. Fish and Wildlife Service 2009). If the Pecos gambusia has similar longevity, their life history strategy (short-lived with extended reproductive periods and relatively low numbers of young per brood) would appear to be highly sensitive to reductions in fecundity and survival of young fish. An elasticity analysis for other short-lived fish species indicates that population growth is, in fact, most influenced by changes in fecundity and juvenile survival (Vélez-Espino et al. 2006). Although gambusia are viviparous, we believe that our estimates of effects on Pecos gambusia fecundity, which were based on studies with oviparous fish, are applicable. Koyo et al. (2000) described the dynamics of oocyte (egg) and embryonic development in *Gambusia affinis*, the mosquitofish. The process of oocyte development, fertilization, and "hatching" in viviparous and oviparous fishes are comparable, except that fertilization and "hatching" occur internally in viviparous species. Thus, the effects of cyanide on oocyte development described in the *Chronic Toxicity to Fish* section are likely to occur in Gambusia, with similar outcomes. In studies with oviparous fish, only the largest and most mature oocytes were spawned (ovulation) and capable of being fertilized. Similarly, in the viviparous gambusia only the largest and most mature oocytes are capable of being fertilized in the follicle. Thus reductions in the number of mature oocytes, by cyanide, would reduce the number of eggs capable of being fertilized, the number of embryos that are "hatched" (internally), and the number of live young that are "born". We would also expect predicted effects on juvenile survival to be the same for live-bearing species.

The reductions we estimate in the number of young produced and survival of young fish through the first year would reduce the Pecos gambusia's potential recruitment substantially. The Pecos gambusia's potential recruitment would be diminished because of (a) reductions in numbers of young produced, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the young that are produced in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Pecos gambusia's reproduction by reducing the number of young produced by females, and reducing the survivorship of young fish in their first year. Pecos gambusia may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Pecos gambusia population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Pecos gambusia are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Pecos gambusia.

## GILA TOPMINNOW (including YAQUI TOPMINNOW)

*Poeciliopsis occidentalis*

Gila topminnow exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity and survival of young first-year fish (Table 13). Compared to control populations, we estimate Gila topminnow exposed to cyanide at the CCC could experience a substantial reduction in the number of young produced and that reduction could be as much as, but is not likely to be greater than, 48%. We estimate that Gila topminnow exposed to cyanide at the CCC could experience a substantial reduction in the survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 56%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

Topminnow bear live young and two broods are carried simultaneously (one further along in development than the other). Brood size is from 1 to 20, and gestation time is 24-28 days. Breeding season is from January to August, with some populations capable of breeding all year if temperatures and food availability are suitable. Life span is approximately 1 year; however, this varies with season of birth and fluctuations in environmental conditions in the habitat. Such a life history strategy (short-lived with extended reproductive periods and relatively low numbers of young per brood) would appear to be highly sensitive to reductions in fecundity and survival of young fish. An elasticity analysis for other short-lived fish species indicates that population growth is, in fact, most influenced by changes in fecundity and juvenile survival (Vélez-Espino et al. 2006). Although topminnows are viviparous, we believe that our estimates of effects on Gila topminnow fecundity, which were based on studies with oviparous fish, are applicable. Koyo et al. (2000) described the dynamics of oocyte (egg) and embryonic development in another live-baring species in the Poeciliidae family, *Gambusia affinis* (the mosquitofish). The process of oocyte development, fertilization, and "hatching" in viviparous and oviparous fishes are comparable, except that fertilization and "hatching" occur internally in viviparous species. Thus, the effects of cyanide on oocyte development described in the *Chronic Toxicity to Fish* section are likely to occur in Gambusia, with similar outcomes. In studies with oviparous fish, only the largest and most mature oocytes were spawned (ovulation) and capable of being fertilized. Similarly, in the viviparous gambusia only the largest and most mature oocytes are capable of being fertilized in the follicle. Thus reductions in the number of mature oocytes, by cyanide, would reduce the number of eggs capable of being fertilized, the number of embryos that are "hatched" (internally), and the number of live young that are "born". We would also expect predicted effects on juvenile survival to be the same for live-bearing species.

The reductions we estimate in the number of young produced and survival of young fish through the first year would reduce the Gila topminnow's potential recruitment substantially. The Gila topminnow's potential recruitment would be diminished because of (a) reductions in numbers of young produced, (b) reductions in young fish surviving through the first year, and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the young that are produced in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rates diminish survival through the first winter. These reductions in reproduction could be exacerbated further if reduced growth rates delay maturity or lengthen the time needed between batched spawning events.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Gila topminnow's reproduction by reducing the number of young produced by females, and reducing the survivorship of young fish in their first year. Gila topminnow may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Gila topminnow population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Gila topminnow are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Gila topminnow.

### *Salmonidae*

### BULL TROUT

*Salvelinus confluentus*

Bull trout exposed to cyanide at the criterion continuous concentration (CCC) are likely to experience reduced survival and reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the *Overview* section. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column. Exposure of bull trout to cyanide at the criterion maximum concentration (CMC) is likely to reduce survival.

Relatively few studies are available for estimating the magnitude of effects to the bull trout that could occur following exposure to cyanide at criterion concentrations. However, data are available to develop quantitative estimates of the effects of such exposure on bull trout fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate that bull trout exposed to cyanide at the CCC are likely to experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 87%. We estimate that bull trout exposed to cyanide at the CCC are likely to experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 90%. In addition, we estimate that juvenile bull trout exposed to cyanide at the CMC are likely to experience a substantial reduction in survival and that reduction could be as much as, but is not likely to be greater than, 99.9%. Effects of cyanide at the CMC on the survival of other bull trout life stages is expected to be of lesser magnitude.

Although these effects are based on modeled estimates, direct toxicity tests with other chemicals indicate that bull trout can be very sensitive to the adverse effects of water pollutants. Several studies have compared the sensitivity of bull trout to rainbow trout. Among the species considered in the cyanide criteria document (and in the BE), the rainbow trout was the most sensitive and, thus, was the species on which the acute and chronic cyanide criteria were based. Bull trout have been found to be less sensitive than rainbow trout to some metals (cadmium and zinc; Hansen et al. 2002a), but as sensitive to other metals (copper; Hansen et al. 2002b) and as sensitive or more sensitive to some herbicides (Fairchild et al. 2006). Dioxin, like cyanide, is a potent reproductive toxin. Cook et al. (2000) reported that, among fish that have been tested, bull trout was the most sensitive to dioxin. The bull trout was three times more sensitive than the lake trout (the next most sensitive species) and more sensitive than the brook trout or the rainbow trout.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

Rieman and McIntyre (1993) characterized the demographic and habitat requirements needed for bull trout conservation. As part of their evaluation they considered the consequences of habitat disturbances in terms of their potential impacts on reproduction, survival and extinction risk. Using a population model, they simulated four types of bull trout populations that differ in growth and maturation rates in order to capture the variation exhibited in natural populations, e.g. resident versus migratory life forms. For each simulated population they varied the survival from egg to age one to determine the level of mortality each population could sustain without collapsing. They next calculated the level of survival from egg to fry stage needed to attain those minimum, egg-to-age-one, survival rates. The required egg-to-fry survival rates were highest for slow growth/late maturity populations (0.25 – 0.49) and lowest for the fast growth populations (0.03 - 0.05). Rieman and McIntyre (1993) reported that egg-to-fry survival rates of 0.25 to 0.50 may approach the highest values possible in many streams. For slow growth populations, the egg-to-fry survival rates would have to be at or near the highest possible rates in order to sustain the population. Thus, reductions in egg-to-fry survival caused by exposure to cyanide would result in an egg-to-age-one survival rate that would not sustain the population. For fast growth populations, the egg-to-fry survival rates are lower than the highest possible rates, so these populations may be able to absorb additional mortality at this life stage. However, the large reduction in egg-to-fry survival caused by cyanide (90%), coupled with reductions in the number of eggs spawned, would likely reduce the egg-to-age-one survival rate below the minimum required to sustain the population.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the bull trout's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year by 87 to 90 percent. Bull trout may also experience effects on growth, swimming performance, condition, and development. Exposure to cyanide at the acute criterion could also substantially reduce the survivorship of juvenile bull trout. Because of the high magnitude of these effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An affected bull trout population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect that is likely to occur at the chronic criterion level, we conclude that ultimately bull trout are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the acute and chronic criteria at the rangewide scale is likely to reduce the reproduction, numbers, and distribution of the bull trout at the rangewide scale.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the bull trout include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC and CMC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of the bull trout, and cause adverse effects to bull trout growth, swimming performance, condition, and development. We estimate that implementation of the proposed action could reduce water quality conditions in critical habitat to the extent that a reduction in the number of hatched eggs could be as high 87% and a reduction in the survival of young fish through the first year could be as high as 90%. Reduction in survivorship of juvenile fish exposed at the CMC could be as high as 99.9%. These effects are estimated to be of a magnitude great enough to reduce numbers of bull trout using areas of critical habitat that support breeding and rearing areas for the bull trout. Approval of the CCC and CMC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation of affected populations from critical habitat containing cyanide at the CCC and CMC. Impacts to water quality resulting from management of cyanide to the CCC and CMC would diminish the ability of critical habitat to provide for conservation of the bull trout.

## LITTLE KERN GOLDEN TROUT
*Oncorhynchus aquabonita whitei*

Little Kern golden trout exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Little Kern golden trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 60%. We estimate that Little Kern golden trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 69%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like darters, than on species with greater adult survival, such as sturgeon.

Little Kern golden trout appear to be intermediate. Females reach reproductive maturity in 3-4 years and live up to 9 years. While their fecundity appears relatively low, females contain between 41 and 65 eggs per year, they appear to reproduce each year after reaching maturity. Nevertheless, we expect that reductions in fecundity and survival of young fish through the first year would have a substantial population-level effect on Little Kern golden trout.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Little Kern golden trout's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Little Kern golden trout may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Little Kern golden trout population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Little Kern golden trout are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the Little Kern golden trout.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the Little Kern golden trout include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of the Little Kern golden trout, and cause trout to experience adverse effects to growth, swimming performance, condition, and development. We estimate the reduction in the number of hatched eggs could be as high 60% and the reduction in the survival of young fish through the first year as high as 69%. These effects are estimated to be of a magnitude great enough to reduce numbers of Little Kern golden trout. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the Little Kern golden trout.

## APACHE TROUT
*Oncorhynchus apache*

Apache trout exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column. Exposure of Apache trout to cyanide at the criterion maximum concentration (CMC) is likely to reduce survival.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Apache trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 87%. We estimate that Apache trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 90%. In addition, we estimate that juvenile Apache trout exposed to cyanide at the CMC are likely to experience a substantial reduction in survival and that reduction could be as much as, but is not likely to be greater than, >99.9%. Effects of cyanide at the CMC on the survival of other Apache trout life stages is expected to be of lesser magnitude.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Apache trout, than on species with greater adult survival, such as sturgeon.

We expect that reductions in fecundity and survival of young fish through the first year would have a greater population-level effect on Apache trout than on longer-lived species, because the majority of Apache trout females spawn only twice in their lifetime. It is estimated that female Apache trout produce between 72 and 1,083 eggs per female, depending on size class. If we reduce the number of hatched eggs by 87% and reduce the survival of fish in their first year by 90% then few fish will survive to the adult stage. If we consider the effects additive, then only 0.2 fish will survive to the adult stage. The combined effect could be less than additive (i.e. between 0.2 and 0.6) if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. The reductions we estimate in hatched eggs and juvenile survivorship could reduce, by an order of magnitude, the number of individuals surviving to adulthood. These reductions in reproduction could be exacerbated further if reductions in juvenile growth rates result in smaller females reproducing in a given year. As we noted in the Status/Baseline section, the number of eggs female Gila trout produce is proportional to their mass.

Apache trout are considered a subspecies of Gila trout and share a very similar life history and the same threats. Brown et al. (2001) performed a population viability analysis for the Gila trout to explore potential management strategies. A base model was constructed to be used as a benchmark for comparison of the effects of different management strategies. Fecundity was estimated from the overall mean count of ova from field-stripped fish. Among their findings, the model was sensitive to large changes in fecundity. Halving fecundity significantly increased the probability of extinction as compared to the base model. We would anticipate a similar response pattern for Apache trout. Exposure to cyanide at the chronic criterion affects not only fecundity, but also hatchability, and juvenile survival of Apache trout. The combined effects could be much greater than what was analyzed by Brown et al. (2001).

Brown et al. (2001) identified catastrophic events as having a much larger influence on the viability of Gila trout than population size, fecundity, or population structure. This conclusion applied to the relative importance of each variable separately and not in combination. Any combination of variables would pose a greater risk to continued viability than any one risk alone. The reductions in reproduction and survival we estimate could occur would diminish the ability of the Apache trout's populations to recover from a catastrophic event, such as a forest fire or drought, and would increase the risk of extirpation of an effected population. Cyanide toxicity could result in the extant population persisting at a reduced population size prior to a catastrophic event. If the local population initially survives the catastrophic event, reductions in reproduction and survival could lengthen the time to recover and leave the population more vulnerable to extirpation.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Apache trout's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Apache trout may also experience effects on growth, swimming performance, condition, and development. Exposure to cyanide at the acute criterion could also substantially reduce the survivorship of juvenile Apache trout. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Apache trout population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Apache trout are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the acute and chronic criteria could reduce the reproduction, numbers, and distribution of the Apache trout.

## LAHONTAN CUTTHROAT TROUT

*Oncorhynchus clarkii henshawi*

Lahontan cutthroat trout exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the *Overview* section. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column. Exposure of Lahontan cutthroat trout to cyanide at the criterion maximum concentration (CMC) is likely to reduce survival.

Relatively few studies are available for estimating the magnitude of effects that could occur following exposure of this species to cyanide at criterion concentrations. However, data are available to develop quantitative estimates of the effects of such exposure on Lahontan cutthroat trout fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate that Lahontan cutthroat trout exposed to cyanide at the CCC are likely to experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 80%. We estimate that Lahontan cutthroat trout exposed to cyanide at the CCC are likely to experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 85%. In addition, we estimate that juvenile Lahontan cutthroat trout exposed to cyanide at the CMC are likely to experience a substantial reduction in survival and that reduction could be as much as, but is not likely to be greater than, 43%. Effects of cyanide at the CMC on the survival of other life stages of Lahontan cutthroat trout is expected to be of lesser magnitude.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also reduce productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section above on *Population Responses to Reductions in Fecundity and Juvenile Survival*, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Lahontan cutthroat trout where the majority of females die after their first spawning, than on species with greater adult survival, such as sturgeon.

In summary, exposure to cyanide concentrations at the chronic criterion are likely to substantially reduce the Lahontan cutthroat trout's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Lahontan cutthroat trout may also experience adverse effects on growth, swimming performance, condition, and development. Exposure to cyanide at the acute criterion could also substantially reduce the survivorship of juvenile Lahontan cutthroat trout. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An affected Lahontan cutthroat trout population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect that is likely to occur at the criterion concentration, we conclude that ultimately Lahontan cutthroat trout are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the acute and chronic criteria at the rangewide scale is likely to reduce the reproduction, numbers, and distribution of the Lahontan cutthroat trout at the rangewide scale.

## PAIUTE CUTTHROAT TROUT

*Oncorhynchus clarkii seleniris*

Paiute cutthroat trout exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Paiute cutthroat trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 60%. We estimate that Paiute cutthroat trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 69%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Paiute cutthroat trout, than on species with greater adult survival, such as sturgeon.

We expect that reductions in fecundity and survival of young fish through the first year would have a greater population-level effect on Paiute cutthroat trout than on longer-lived species, because female Paiute cutthroat trout spawn few times in their lifetime.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Paiute cutthroat trout's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Paiute cutthroat trout may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Paiute cutthroat trout population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Paiute cutthroat trout are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the Paiute cutthroat trout.

## GREENBACK CUTTHROAT MOUNTAIN TROUT
*Oncorhynchus clarki stomias*

Greenback cutthroat trout exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Formal Draft Biological Opinion.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Greenback cutthroat trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 60%. We estimate that Greenback cutthroat trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 69%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn than on species with greater adult survival, such as sturgeon.

The reductions we estimate in hatched eggs and survival of young fish through the first year could reduce substantially the number of individuals surviving to adulthood. These reductions in reproduction could be exacerbated further if reductions in juvenile growth rates result in smaller females reproducing in a given year. The number of eggs female greenback trout produce is proportional to their mass.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Greenback cutthroat trout's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Greenback cutthroat trout may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Greenback cutthroat trout population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Greenback cutthroat trout are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the Greenback cutthroat trout.

## GILA TROUT

Formal Draft Biological Opinion.

*Oncorhynchus gilae*

Gila trout exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Gila trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 60%. We estimate that Gila trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 69%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Gila trout, than on species with greater adult survival, such as sturgeon.

We expect that reductions in fecundity and survival of young fish through the first year would have a greater population-level effect on Gila trout than on longer-lived species, because the majority of Gila trout females spawn only twice in their lifetime. Brown et al. (2001) estimated that female fish produced between 62 and 197 eggs per spawn, depending on size class. In the Gila Trout Recovery Plan (Service 2003) the Service estimated that for every 100 eggs that hatch about half will survive to the juvenile life stage. Of those approximately 50 fish, only about 6 will survive to the subadult stage and of those 6 subadults, only 2 will survive to the adult life stage. This estimate provides a good reference for characterizing the effects of chronic cyanide toxicity. If we simply reduce the number of hatched eggs by 60% and assume survivorship remains the same for other transitions, then only 0.7 fish will survive to the adult stage. If we reduce the survival of fish in their first year by 69% and assume survivorship remains the same for other transitions, then only 0.6 fish will survive to the adult stage. If we consider the effects additive, then only 0.2 fish will survive to the adult stage. The combined effect could be less than additive (i.e. between 0.2 and 0.6) if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. The reductions we estimate in hatched eggs and juvenile survivorship could reduce, by an order of magnitude, the number of individuals surviving to adulthood. These reductions in reproduction could be exacerbated further if reductions in juvenile growth rates result in smaller females reproducing in a given year. As we noted in the Status/Baseline section, the number of eggs female Gila trout produce is proportional to their mass.

Brown et al. (2001) performed a population viability analysis for the Gila trout to explore potential management strategies. A base model was constructed to be used as a benchmark for comparison of the effects of different management strategies. Fecundity was estimated from the overall mean count of ova from field-stripped fish. Among their findings, the model was sensitive to large changes in fecundity. Halving fecundity significantly increased the probability of extinction as compared to the base model. Exposure to cyanide at the chronic criterion affects not only fecundity, but also hatchability, and juvenile survival. The combined effects could be much greater than what was analyzed by Brown et al. (2001).

Brown et al. (2001) identified catastrophic events as having a much larger influence on the viability of Gila trout than population size, fecundity, or population structure. This conclusion applied to the relative importance of each variable separately and not in combination. Any combination of variables would pose a greater risk to continued viability than any one risk alone. The reductions in reproduction and survival we estimate could occur would diminish the ability of the Gila trout's populations to recover from a catastrophic event, such as a forest fire or drought, and would increase the risk of extirpation of an effected population. Cyanide toxicity could result in the extant population persisting at a reduced population size prior to a catastrophic event. If the local population initially survives the catastrophic event, reductions in reproduction and survival could lengthen the time to recover and leave the population more vulnerable to extirpation. Brown et al. (2001) found the Gila trout's extinction risk was sensitive to the number of populations. For example, increasing the number of Gila trout populations in the model from 10 to 16 significantly reduced risk of extinction by 15%.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Gila trout's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Gila trout may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Gila trout population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Gila trout are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the Gila trout.

**ATLANTIC SALMON**
*Salmo salar*
Gulf of Maine Distinct Population Segment

Atlantic salmon exposed to cyanide at the chronic criterion concentration are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above in the Effects Overview. The effects of chronic toxicity could be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. However, data were available to develop quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 13). Compared to control populations, we estimate Atlantic salmon exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 36%. We estimate that Atlantic salmon exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than, 41%.

As noted previously, the young first-year fish that do survive could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. It could also delay reproductive maturity and productivity.

The reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section on Population Level Effects, we noted that reductions in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to spawn, like Atlantic salmon, than on species with greater adult survival, such as sturgeon.

Atlantic salmon typically live about 5 years, spending 1-3 years as freshwater smolts prior to migrating to the ocean as adults where they will develop in about 2-3 years into mature salmon and then return to their natal freshwater rivers to spawn. We expect that reductions in fecundity and survival of young fish through the first year would have a greater population-level effect on Atlantic salmon than on longer-lived species, because the majority of Atlantic salmon females spawn only once in their lifetime. Females deposit 7,000-8,000 eggs per spawn. Studies in Maine indicate less than 10% of the eggs spawned will survive to emerge as feeding fry. The Gulf of Maine DPS has declined to critically low levels. Adult returns, juvenile abundance estimates and survival have continued to decline since the listing. In 2004, total adult returns to the eight rivers still supporting wild Atlantic salmon populations within the DPS were estimated to range from 60 to 113 individuals. No adults were documented in three of the eight rivers. Declining smolt production has also been documented in recent years, despite fry stocking. For example, from 1996 through 1999, annual smolt production in the Narraguagus River was estimated to average about 3,000 fish. Smolt production declined significantly in 2000 and for the past three years has averaged only about 1,500 fish per year. Overwinter survival in the Narraguagus River since 1997 has only averaged about 12%, approximately half of the survival rate of previous years and significantly less than the 30% previously accepted for the region. These estimates provide a good reference for characterizing the effects of chronic cyanide toxicity.

Based on a minimum of 7000 eggs per spawn, if we simply reduce the number of hatched eggs by 36% assuming overwinter survival is at the previously accepted level of 30%, then only 134 smolts (as opposed to 210) will survive through the first year. If we reduce the smolt survivorship by 41% then only 124 fish will survive through the first year. If we consider the effects additive then only 79 smolts will survive through the first year. The combined effect could be less than additive (i.e. between 79 and 134) if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. If we consider that in some rivers survival is as low as 12%, the effects of cyanide at the CCC could result in as little as 32 smolts per spawn surviving through the first year.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Atlantic salmon's reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survivorship of young fish in their first year. Atlantic salmon may also experience effects on growth, swimming performance, condition, and development. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Atlantic salmon population's decline could stabilize at a reduced absolute population number or could continue to decline until it is extirpated. Based upon the magnitude of effect we anticipate could occur, we conclude ultimately Atlantic salmon are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion likely reduces the reproduction, numbers, and distribution of the Atlantic salmon.

### 7.2 Effects to Invertebrates

Direct effects to invertebrates are anticipated for two species within the genus Gammaras, the Illinois cave amphipod (*Gammarus acherondytes*), and Noel's amphipod (*Gammarus desperatus*). No direct effects are anticipated for other invertebrate species (see Appendix B for screening methodology).

**ILLINOIS CAVE AMPHIPOD**
*Gammarus acherondytes*

## NOEL'S AMPHIPOD

*Gammarus desperatus*

$EC_A$ values for the Illinois cave amphipod and Noel's amphipod were derived using EPA's Interspecies Correlation Estimate (ICE) model for the genus *Gammaras* with *Daphnia magna* as the surrogate species (EPA 2003b). The lower 95% confidence interval of the $LC_{50}$ value was divided by the cyanide-specific value of 1.21 to derive the acute $EC_A$ (24.5 ug/L) and by the invertebrate ACR to derive the chronic $EC_A$ (3.92 ug/L). The proposed chronic criterion (5.2 ug/L) exceeds the chronic $EC_A$ for these species.. On that basis, exposure of these amphipods to cyanide concentrations at these criteria is likely to cause adverse effects.

Two, acute 96-hour $LC_{50}$ values were found in the literature for species within the genus *Gammarus*: *G. fasciatus* ($LC_{50}$ value = 903 ug/L) and *G. pseudolimnaeus* (142.9 ug/L) (Ewell et al 1986, Smith et al.1978). Smith et al. (1978) also derived a lethal threshold concentration at which the first signs of lethality appear of 74 ug/L for *G. pseudolimnaeus*. Although the measured $LC_{50}$ values are an order of magnitude greater than that derived using the lower 95% confidence interval of the ICE estimate (29.6 ug/L), it is not unusual for species that are closely related taxonomically to have widely variable sensitivities to particular toxicants. For cyanide, numerous $LC_{50}$ values for species within the family Cyprinidae were measured and the results varied by a factor of 18 between the most and least sensitive species. The measured $LC_{50}$ values for *Gammarus* represent the only two such estimates known for this group, a large genus comprised of over 200 species.

In the only study in which chronic toxicity was tested in freshwater amphipods (Smith et al.1979, Oseid and Smith 1979), *G. pseudolimnaeus* was found to be 6 to 25 times more sensitive to cyanide than the isopod species *Asellus communis* for acute and chronic endpoints. When *G. pseudolimnaeus* was housed alone, overall mass (free individuals plus eggs and young) in tanks containing cyanide concentrations of 16 ug/L and 32 ug/L and above was significantly lower than controls. Smith et al. (1979) and Oseid and Smith (1979) did not believe that reductions in the 16 ug/L tank were due to cyanide exposure, but provided no alternate explanation for the decline. Effects of cyanide exposure were heightened for *G. pseudolimnaeus* when exposed concurrently with *A. communis* in the same test tank. While the aggressive and competitive *G. pseudolimnaeus* greatly out-competed *A. communis* in tanks containing the control or 4 ug/L treatment, a significant shift took place in tanks with concentrations of 9 ug/L or greater cyanide. When exposed to cyanide in tanks containing *A. communis*, total mass of *G. pseudolimnaeus* was reduced 63% compared to controls at 9 ug/L and 97% at 21 ug/L, the next highest concentration tested. This impact occurred at concentrations lower than when *G. pseudolimnaeus* was exposed alone. Smith et al. (1979) and Oseid and Smith (1979) speculated that decreased survival of *G. pseudolimnaeus* may have been caused by predation by *A. communis.* Thus, cyanide not only exerts direct effects on *G. pseudolimnaeus*, but can shift the competitive advantage to more tolerant species in mixed communities. Smith et al. (1979) and Oseid and Smith (1979) concluded that where a mixed community exists, Gammarus is likely to be excluded in the presence of cyanide pollution. No effects were seen at cyanide concentrations of 4 ug/L and below. The differential sensitivity reported by Smith et al. (1979) and Oseid and Smith (1979) was supported by Ewell et al. (1986), who did not look at chronic effects, but found that acute sensitivity of *G. fasciatus* was nearly twice that of *A. intermedius.*

These results may be applicable to the listed amphipods, as interspecific interactions with other amphipod and isopod species are believed to be important factors in the biology of Gammerids. The Illinois cave amphipod was regularly recorded in caves containing several species of amphipod and isopod species, as well as other invertebrates and fish (Webb et al. 1998). Noel's amphipod is part of a complex of Gammerus species that co-occur together in the Pecos River Basin of New Mexico and Texas. Resource partitioning according to substrate and water depth has been examined for these species, but is not fully understood. While fungi and detritus have been suggested as important food sources for amphipods and isopods, it is also recognized that these species can be predators on other species and even exhibit cannibalistic behaviors in high density situations.

Both the chronic $EC_A$ and the chronic data available indicate that cyanide at criteria concentrations may result in the loss of individuals of the Illinois cave amphipod and Noel's amphipod. Effects of cyanide appear to be exasperated when amphipods are exposed in situations of resource competition or possibly predation, when a population may experiences a loss of mass that is greater than 0% and less than 63%. The Illinois cave amphipod and Noel's amphipod are known to occur in systems that contain other invertebrate species that may occupy a similar niche and exhibit more tolerance to cyanide, making these two species more susceptible to the effects of cyanide under these conditions. Because both of these species exist in populations that are geographically isolated from one another, the ability of these amphipods to recolonize habitat is limited. For these reasons, exposure of these two species to cyanide concentrations at or near the proposed chronic criterion value is likely to contribute to the elimination of a population unit. The loss of a population unit for either amphipod species would substantially reduce its reproduction, numbers, or distribution at the rangewide scale.

### 7.3 Effects to Mussels

Listed mussels are not anticipated to experience direct effects of cyanide at criteria concentrations (see Appendix B for screening metholodoly and discussion of direct effects to museels), but were screened in based on indirect effects to host fish species For most mussel species, transformation on a host fish is a required element of their life cycle that cannot be bypassed. Host fish availability and density have been found to be significant factors influencing mussel persistence in particular habitats (Haag and Warren 1998). Contact between glochidia and suitable host fishes is a low-probability event even in healthy populations (Neves et al. 1997), and despite the large number of glochidia produced by an individual mussel, infestation rates tend to be very low (Haag and Warren 1999, Layzer et al. 2003). Host fish generalists, whose glochidia can transform on multiple species, release glochidia in large mucous webs that entangle fish indiscriminately. Host fish specialists, relying on one or few species to transform, tend to employ specialized lures that mimic prey items of host fish (Strayer et al. 2004). This strategy likely reduces the probability of infestation in unsuitable fish hosts, which can result in immune system incompatibility, and discharge of glochidia. Glochidia that fail to infest a suitable host or that have been sloughed off by an unsuitable host will not transform to the adult stage and will survive only as long as their energy reserves last, from a few days to up to two weeks. The odds of an individual glochidium infesting a host and completing transformation have been estimated at 4 in 100,000. Given those odds, any reduction of host fish species populations caused by exposure to cyanide at either the acute or chronic criterion levels is likely to cause an adverse effect to the listed mussel species.

In addition, mussel glochidia infested on fish are completely parasitic and are dependent on the host for oxygen, nutrition, and overall survival. Once contact is made with a suitable host, successful glochidia encyst on the gills, fins, or skin for a period of several weeks. Any mortality to host fish during this period will necessarily result in mortality to infested glochidia. Thus, the host fish become surrogates for the listed mussels during the parasitic stage.

*Estimating Acute and Chronic $EC_A$ Values for Host Fish Species*

In estimating risks to glochidia host species, the Service derived acute and chronic assessment effects concentrations ($EC_A$) estimates according to the prioritization described in Figures 2 and 4 of the 2004 Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria (Table 15). Although host fish species identified for listed mussels were themselves not listed species, the host fish species is an obligate part of the mussel lifecycle to which it has a parasitic dependence while attached, and is treated with the same conservatism as a listed species.

1. When acute toxicity data were available for a host species, the species mean acute value from Table 1 of the BE was used to derive the acute $EC_A$, and, when appropriate, the chronic $EC_A$. However, when only one $LC_{50}$ was available for a species, the lower 95% confidence interval of the $LC_{50}$, when available, was used to derive the $EC_A$.
2. For yellow perch (*Perca flavescens*), the $LC_{50}$ reported in Table 1 of the BE could not be reproduced from the original source, so an acute $EC_A$ value was calculated from the mean of all 96-hour $LC_{50}$ values (converted to free CN) reported in Smith et al. (1978).
3. For fish with no species-specific data, ICE models were derived at the lowest taxonomic level with adequate data (EPA 2003b). The lower 95% confidence interval of the predicted $LC_{50}$ was used to determine $EC_A$ values.
4. For species for which no ICE models were available, the 5$^{th}$ percentile $LC_{50}$ from the appropriate SSD was used in lieu of the mean $LC_{50}$

All acute $EC_A$ values were calculated by division of the $LC_{50}$ by the cyanide-specific 1.21 factor derived by the Service.

**Table 15. Surrogate taxa used to estimate $LC_{50}$ values for evaluation of to host fish effects**

| Surrogate taxa used to estimate host fish $LC_{50}$ | $LC_{50}$ (ug CN/L) | Acute $EC_A$ (ug/L) | Chronic $EC_A$ (ug/L) |
|---|---|---|---|
| Actinopterygii (class) | 66.5[1] | 54.93 | 2.86 |
| Cypriniformes (order) | 84.5[1] | 69.88 | 3.64 |
| Cyprinidae (family) | 101.7[2] | 84.07 | 4.38 |
| *Pimephales promelas* (species) | 138.4[3] | 114.38 | 5.96 |
| Cyprinodontiformes | | | |
| *Poecilia reticulate* (species) | 187.8[3] | 155.21 | 8.09 |
| Perciformes (order) | 90.8[1] | 75.04 | 3.91 |
| Centrarchidae (family) | 73.2[2] | 60.46 | 3.15 |
| *Lepomis* (genus) | 89.5[2] | 73.99 | 3.85 |
| *Lepomis cyanellus* (species) | 126.2[2] | 104.03 | 5.43 |
| *Lepomis macrochirus* (species) | 126.1[3] | 104.21 | 5.43 |
| *Micropterus (genus)* | | | |
| *Micropterus. salmoides* (species) | 95.7[4] | 79.09 | 4.12 |
| *Pomoxis (genus)* | | | |
| *Pomoxis nigromaculatus* (species) | 84.5[4] | 70.66 | 3.64 |
| Percidae (family) | 42.3[2] | 34.97 | 1.82 |
| *Etheostoma* (genus) | 40.0[2] | 33.07 | 1.72 |
| *Perca* (genus) | | | |
| *Perca flavescens* (species) | 93.3[5] | 77.11 | 4.02 |
| Salmoniformes | | | |
| *Salmo salar* (species) | 90[3] | 74.38 | 3.88 |
| *Salmo trutta* (species) | 54.9[2] | 45.38 | 2.36 |
| Siluriformes (order) | | | |
| Ictaluridae (family) | 182.8[2] | 151.06 | 7.87 |
| *Ictalurus punctatus* (species) | 190.3[2] | 83.83 | 8.20 |

[1] $LC_{50}$ based on 5$^{th}$ percentile estimate from species sensitivity distribution (SSD), Table 2 – Cyanide BE.
[2] $LC_{50}$ estimate based on lower bound of the 95% CI from ICE model (Appendix D)
[3] $LC_{50}$ based on measured value from the Cyanide BE (Table 1);
[4] $LC_{50}$ based on lower bound of the 95% CI of the measured value from the Cyanide BE (Table 1);
[5] $LC_{50}$ based on mean measured value from Smith et al 1978

*Effects to Listed Mussel Species*

Table 16 identifies host fish species for all currently listed mussel species, with estimates of their acute and chronic $EC_A$ values, and predicted maximum effects to fecundity and juvenile survival. We used listed species accounts, NatureServe, recent FWS Recovery Plans and 5-Year Reviews, the open literature, and consultation with species experts to update and expand the list of known hosts for these species.

Two of the listed mussels, the fat pocketbook (*Potamilus capax)* and the scaleshell mussel (*Leptodea leptodon)*, have an identified obligate relationship with a single host, the freshwater drum (*Aplodinotus grunniens)*. The remaining listed mussels either lack a known obligate relationship with a host fish or host fish relationships are unknown. Of the 300 identified North American mussel species, there have been no host fish species identified for a significant percentage, and it is likely that the complement of host fish remains incomplete for many other species. Individual mussel species have been found to have up to 25 fish species known to serve as suitable hosts, though host fish specificity to one species or a group of species related by taxonomy or food guild can be common in mussels. To assess effects to these species, we looked at the range of all fish species that have been identified as hosts for listed mussels (Table 17). We then totaled the number of species identified as hosts in each family, as well as the frequency that each family was represented as a host fish for a listed mussel species, as identified in Table 15. Fish within families that may be sensitive to cyanide at concentrations below acute or chronic criteria, accounted for 96% of the species diversity (Table 18), and 98% of all fish identified as hosts for listed mussels (Table 19). Extrapolating these results to fish with no known hosts or a potentially incomplete list of hosts, we can assume that any mussel is likely to have at least one host fish for which adverse effects cannot be ruled out at exposure to cyanide at criteria concentrations Therefore, for mussels lacking a known obligate relationship with a host fish, in the absence of information to the contrary, it was assumed that the species was likely to have at least one host fish that is sensitive to cyanide at the criteria concentrations, whether one is currently identified or not. On that basis, these mussel species are likely to be adversely affected at cyanide criteria concentrations due to potential reductions in host fish abundance.

## Table 16. Cyanide sensitivity of fish species that serve as hosts for glochidia of listed mussels.

| Listed Mussel | Host Fish | Acute $EC_A$ (ug/L) | Chronic $EC_A$ (ug/L) | Source/surrogate taxa for $LC_{50}$ values (Table 15) | Estimated reduction in fecundity and larvae/juvenile survival of host fish due based on surrogate species data sets (Appndix F) | | |
|---|---|---|---|---|---|---|---|
| | | | | | Fathead Minnow (Reduction in the mean number of eggs spawned) | Brook Trout (Reduction in the mean number of eggs spawned) | Bluegill (reduction in larvae/juvenile survival) |
| **MUSSLES WITH OBLIGATE HOST FISH** | | | | | | | |
| Fat Pocketbook *Potamilus capax* | Freshwater drum | 75.04 | 3.91 | Perciformes (order) | 36% (24%, 47%) | 24% (0%, 53%) | 40% (0%, 79%) |
| Scaleshell Mussel *Leptodea leptodon* | Freshwater drum | 75.04 | 3.91 | Perciformes (order) | 36% (24%, 47%) | 24% (0%, 53%) | 40% (0%, 79%) |
| **MUSSLES WITH NO KNOWN OBLIGATE HOST FISH** | | | | | | | |
| Cumberland Elktoe *Alasmidonta atropurpurea* | Rainbow darter<br>Redline darter<br>Fantail darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| | Banded sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| | Rock bass | 60.46 | 3.15 | *Centrarchidae* (family) | 44% (35%, 53%) | 28% (0%, 54%) | 52% (0%, 81%) |
| | Northern hogsucker | 69.88 | 3.64 | Cypriniformes (order) | 39% (28%, 49%) | 26% (0%, 54%) | 44% (0%, 80%) |
| | Longear sunfish | 73.99 | 3.85 | *Lepomis* (genus) | 36% (24%, 47%) | 24% (0%, 54%) | 41% (0%, 79%) |
| | Whitetail shiner | 84.07 | 4.38 | Cyprinidae (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| Dwarf Wedgemussel *Alasmidonta heterodon* | Tesselated darter<br>Johnny darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| | Roanoke darter | 34.97 | 1.82 | Percidae (family) | 63% (58%, 68%) | 39% (18%, 57%) | 72% (43%, 87%) |
| | Mottled sculpin<br>Slimy sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| | Atlantic salmon | 74.38 | 3.88 | *Salmo salar* (species) | 36% (24%, 47%) | 24% (0%, 54%) | 41% (0%, 79%) |
| Appalachian Elktoe *Alasmidonta raveneliana* | Banded sculpin<br>Mottled sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| Fat Threeridge *Amblema neislerii* | Blackbanded darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| | Redear sunfish | 73.99 | 3.85 | *Lepomis* (genus) | 36% (24%, 47%) | 24% (0%, 54%) | 41% (0%, 79%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Largemouth bass | 79.09 | 4.12 | *Micropterus salmoides* (species) | 34% (21%, 45%) | 23% (0%, 53%) | 38% (0%, 79%) |
| | Weed shiner | 84.04 | 4.38 | Cyprinidae (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| | Bluegill | 104.21 | 5.43 | *Lepomis macrochirus* (species) | --- | --- | --- |
| Ouachita Rock Pocketbook *Arkansia wheeleri* | White crappie | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| | Warmouth Smallmouth bass | 60.46 | 3.15 | Centrarchidae (family) | 44% (35%, 53%) | 28% (0%, 54%) | 52% (0%, 81%) |
| | Black crappie | 70.66 | 3.64 | *Pomoxis nigromaculatus* (species) | 39% (24%, 49%) | 26% (0%, 54%) | 44% (0%, 80%) |
| | River carpsucker | 69.88 | 3.64 | Cypriniformes (order) | 39% (28%, 49%) | 26% (0%, 54%) | 44% (0%, 80%) |
| | Longear sunfish Orangespotted sunfish | 73.99 | 3.85 | *Lepomis* (genus) | 36% (24%, 47%) | 24% (0%, 54%) | 41% (0%, 79%) |
| | Freshwater drum | 75.04 | 3.91 | Perciformes (order) | 36% (24%, 47%) | 24% (0%, 53%) | 40% (0%, 79%) |
| | Largemouth bass | 79.67 | 4.12 | *Micropterus salmoides* (species) | 34% (21%, 45%) | 23% (0%, 53%) | 38% (0%, 79%) |
| | Dusky shiner Bleeding shiner Golden shiner Emerald shiner | 84.07 | 4.38 | Cyprinidae (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| | Green sunfish | 104.03 | 5.43 | *Lepomis cyanellus* (species) | --- | --- | --- |
| | Bluegill | 104.21 | 5.43 | *Lepomis macrochirus* (species) | --- | --- | --- |
| Birdwing Pearlymussel *Conradilla caelata* | Greenside darter Tennessee snubnose darter Banded darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| Fanshell *Cyprogenia stegaria* | Banded darter Greenside darter Tennessee snubnose darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| | Blotchside logperch Logperch Tangerine darter | 34.97 | 1.82 | Percidae (family) | 63% (58%, 68%) | 39% (18%, 57%) | 72% (43%, 87%) |
| | Mottled sculpin Banded sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| Dromedary perlymussel *Dromus dromas* | Fantail darter Banded darter Tangerine darter Greenside darter Tennessee snubnose darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Gilt darter<br>Channel darter<br>Logperch<br>Blotchside logperch | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Black sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| Chipola Slabshell<br>*Elliptio chipolaensis* | Bluegill | 104.21 | 5.43 | *Lepomis macrochirus* (species) | --- | --- | --- |
| Tar River Spinymussel<br>*Elliptio steinstansana* | Bluehead chub<br>Satinfin shiner<br>White shiner<br>Pinewoods shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| Purple Bankclimber<br>*Elliptoideus sloatianus* | Blackbanded darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Eastern mosquitofish | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| | Greater Jumprock | 69.88 | 3.64 | Cypriniformes (order) | 39%<br>(28%, 49%) | 26%<br>(0%, 54%) | 44%<br>(0%, 80%) |
| | Guppy | 155.21 | 8.09 | *Poecilia reticulate* (species) | --- | --- | --- |
| Cumberlandian Combshell<br>*Epioblasma brevidens* | Wounded darter<br>Redline darter<br>Bluebreast darter<br>Snubnose darter<br>Greenside darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Logperch | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Banded sculpin<br>Mottled sculpin<br>Black sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| Oyster Mussel<br>*Epioblasma capsaeformis* | Wounded darter<br>Redline darter<br>Bluebreast darter<br>Greenside darter<br>Fantail darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Dusky darter | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Banded sculpin<br>Black sculpin<br>Mottled sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| Curtis Pearly Mussel | Rainbow darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| *Epioblasma florentina curtisii* | Banded sculpin<br>Mottled sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| Yellow Blossom (Pearlymussel)<br>*Epioblasma florentina florentina* | Not known | | | | | | |
| Tan Riffleshell<br>*Epioblasma florentina walkeri* | Fantail darter<br>Greenside darter<br>Redline darter<br>Snubnose darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Banded sculpin<br>Mottled sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| Upland Combshell<br>*Epioblasma metastriata* | Not known | | | | | | |
| Catspaw (Purple Cat's Paw Pearlymussel)<br>*Epioblasma obliquata obliquata* | Blackside darter<br>Logperch | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Mottled sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| | Rock bass | 60.46 | 3.15 | Centrarchidae (family) | 44%<br>(35%, 53%) | 28%<br>(0%, 54%) | 52%<br>(0%, 81%) |
| | Stonecat | 151.06 | 7.87 | Ictaluridae (family) | --- | --- | --- |
| White Catspaw Pearlymussel<br>*Epioblasma obliquata perobliqua* | Not known | | | | | | |
| Southern Acornshell<br>*Epioblasma othcaloogensis* | Not known | | | | | | |
| Southern Combshell<br>*Epioblasma penita* | Not known | | | | | | |
| Tubercled Blossom<br>*Epioblasma torulosa torulosa* | Not known | | | | | | |
| Turgid Blossom<br>*Epioblasma turgidula* | Not known | | | | | | |
| Green Blossom<br>*Epioblasma torulosa gubernaculums* | Not known | | | | | | |
| Northern Riffleshell<br>*Epioblasma turulosa rangiana* | Banded darter<br>Bluebreast darter<br>Johhny darter<br>Iowa darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Brown trout | 45.38 | 2.36 | *Salmo trutta* (species) | 55%<br>(24%, 61%) | 34%<br>(9%, 56%) | 63%<br>(23%, 84%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Banded sculpin<br>Mottled sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| Shiny Pigtoe<br>*Fusconaia cor* | Whitetail shiner<br>Common shiner<br>Warpaint shiner<br>Telescope shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| Fine-rayed Pigtoe<br>*Fusconaia cuneolus* | Mottled sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| | River chub<br>Central stoneroller<br>Telescope shiner<br>Tennessee shiner<br>Whitetail shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| | Fathead minnow | 114.38 | 5.96 | *Pimephales promelas* (species) | --- | --- | --- |
| Cracking Pearlymussel<br>*Hemistena lata* | Not known | | | | | | |
| Pink Mucket<br>*Lampsilis abrupta* | Walleye<br>Sauger | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Spotted bass<br>Smallmouth bass | 60.46 | 3.15 | Centrarchidae (family) | 44%<br>(35%, 53%) | 28%<br>(0%, 54%) | 52%<br>(0%, 81%) |
| | Freshwater drum | 75.04 | 3.91 | Perciformes (order) | 36%<br>(24%, 47%) | 24%<br>(0%, 53%) | 40%<br>(0%, 79%) |
| | Largemouth bass | 79.67 | 4.12 | *Micropterus salmoides* (species) | 34%<br>(21%, 45%) | 23%<br>(0%, 53%) | 38%<br>(0%, 79%) |
| Fine-lined Pocketbook<br>*Lampsilis altilis* | Redeye bass<br>Spotted bass | 60.46 | 3.15 | Centrarchidae (family) | 44%<br>(35%, 53%) | 28%<br>(0%, 54%) | 52%<br>(0%, 81%) |
| | Largemouth bass | 79.09 | 4.12 | *Micropterus salmoides* (species) | 34%<br>(21%, 45%) | 23%<br>(0%, 53%) | 38%<br>(0%, 79%) |
| | Green sunfish | 104.03 | 5.43 | *Lepomis cyanellus* (species) | --- | --- | --- |
| Higgins Eye<br>*Lampsilis higginsii* | Sauger<br>Walleye | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Smallmouth bass | 60.46 | 3.15 | Centrarchidae (family) | 44%<br>(35%, 53%) | 28%<br>(0%, 54%) | 52%<br>(0%, 81%) |
| | Black crappie | 70.66 | 3.64 | *Pomoxis nigromaculatus* (species) | 39%<br>(24%, 49%) | 26%<br>(0%, 54%) | 44%<br>(0%, 80%) |
| | Freshwater drum | 75.04 | 3.91 | Perciformes (order) | 36%<br>(24%, 47%) | 24%<br>(0%, 53%) | 40%<br>(0%, 79%) |
| | Largemouth bass | 79.09 | 4.12 | *Micropterus salmoides* | 34%<br>(21%, 45%) | 23%<br>(0%, 53%) | 38%<br>(0%, 79%) |
| | Yellow perch | | 4.02 | *Perca flavescens* (species) | 35%<br>(24%, 46%) | 23%<br>(0%, 53%) | 38%<br>(0%, 79%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Orangenacre Mucket *Lampsilis perovalis* | Chain pickerel | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| | Redeye bass<br>Spotted bass | 60.46 | 3.15 | Centrarchidae (family) | 44% (35%, 53%) | 28% (0%, 54%) | 52% (0%, 81%) |
| | Largemouth bass | 79.67 | 4.12 | *Micropterus salmoides* (species) | 34% (21%, 45%) | 23% (0%, 53%) | 38% (0%, 79%) |
| Arkansas Fatmucket *Lampsilis powelli* | Spotted bass | 60.46 | 3.15 | Centrarchidae (family) | 44% (35%, 53%) | 28% (0%, 54%) | 52% (0%, 81%) |
| | Largemouth bass | 79.09 | 4.12 | *Micropterus salmoides* (species) | 34% (21%, 45%) | 23% (0%, 53%) | 38% (0%, 79%) |
| Speckled Pocketbook *Lampsilis streckeri* | Warmouth<br>Spotted bass<br>Smallmouth bass<br>Shadow bass | 60.46 | 3.15 | Centrarchidae (family) | 44% (35%, 53%) | 28% (0%, 54%) | 52% (0%, 81%) |
| | Longear sunfish | 73.99 | 3.85 | *Lepomis* (genus) | 36% (24%, 47%) | 24% (0%, 54%) | 41% (0%, 79%) |
| | Green sunfish | 104.03 | 5.43 | *Lepomis cyanellus* (species) | --- | --- | --- |
| | Bluegill | 104.21 | 5.43 | *Lepomis macrochirus* (species) | --- | --- | --- |
| Shinyrayed pocketbook *Lampsilis subangulata* | Eastern mosquitofish | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| | Spotted bass | 60.46 | 3.15 | Centrarchidae (family) | 44% (35%, 53%) | 28% (0%, 54%) | 52% (0%, 81%) |
| | Largemouth bass | 79.09 | 4.12 | *Micropterus salmoides* (species) | 34% (21%, 45%) | 23% (0%, 53%) | 38% (0%, 79%) |
| | Bluegill | 104.21 | 5.43 | *Lepomis macrochirus* (species) | --- | --- | --- |
| | Guppy | 155.21 | 8.09 | *Poecilia reticulate* (species) | --- | --- | --- |
| Alabama Lampmussel *Lampsilis virescens* | Not known | | | | | | |
| Carolina Heelsplitter *Lasmigon decorate* | Not known | | | | | | |
| Louisiana Pearlshell *Margaritifera hembeli* | Brown madtom | 151.06 | 7.87 | Ictaluridae (family) | --- | --- | --- |
| | Blackspotted topminnow | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| | Striped shiner<br>Redfin shiner<br>Golden shiner | 84.07 | 4.38 | Cyprinidae (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| Alabama Moccasinshell *Medionidus acutissimus* | Tuskaloosa darter<br>Redfin darter<br>Blackbanded darter<br>Southern sand darter<br>Johnny darter<br>Speckled darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Saddleback darter<br>Naked sand darter<br>Logperch | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Blackspotted topminnow | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| Coosa Moccasinshell<br>*Medionidus parvulus* | Blackbanded darter | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| Gulf Moccasinshell<br>*Medionidus penicillatus* | Blackbanded darter<br>Brown darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Eastern mosquitofish | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| | Guppy | 155.21 | 8.09 | *Poecilia reticulate* (species) | --- | --- | --- |
| Ochlockonee Moccasinshell<br>*Medionidus simpsonianus* | Not known | | | | | | |
| Ring Pink<br>*Obovaria retusa* | Not known | | | | | | |
| Littlewing Pearlymussel<br>*Pegis fibula* | Redline darter<br>Greenside darter<br>Emerald darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Banded sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| White Wartyback<br>*Plethobasus cicatricosus* | Not known | | | | | | |
| Orangefoot Pimpleback<br>*Plethobasus cooperianus* | Not known | | | | | | |
| Clubshell<br>*Pleurobema clava* | Blackside darter<br>Logperch | 34.97 | 1.82 | Percidae (family) | 63%<br>(58%, 68%) | 39%<br>(18%, 57%) | 72%<br>(43%, 87%) |
| | Striped shiner<br>Central stoneroller | 84.07 | 4.38 | *Cyprinidae* (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| James Spinymussel<br>*Pleurobema collina* | Fantail darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65%<br>(60%, 70%) | 40%<br>(20%, 58%) | 74%<br>(46%, 88%) |
| | Pumpkinseed | 73.99 | 3.85 | *Lepomis* (genus) | 36%<br>(24%, 47%) | 24%<br>(0%, 54%) | 41%<br>(0%, 79%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | Bluehead chub<br>Rosyside dace<br>Satinfin shiner<br>Rosefin shiner<br>Blacknose dace<br>Central stoneroller<br>Mountain redbelly dace<br>Swallowtail shiner<br>Common shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| Black Clubshell<br>*Pleurobema curtum* | Not known | | | | | | |
| Southern Clubshell<br>*Pleurobema decisum* | Blacktail shiner<br>Alabama shiner<br>Tricolor shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| Dark Pigtoe<br>*Pleurobema furvum* | Blackspotted topminnow | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| | Largescale stoneroller<br>Alabama shiner<br>Blacktail shiner<br>Creek chub | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| Southern Pigtoe<br>*Pleurobema georgianum* | Alabama shiner<br>Blacktail shiner<br>Tricolor shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| Cumberland Pigtoe<br>*Pleurobema gibberum* | Telescope shiner<br>Striped shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| Flat Pigtoe<br>*Pleurobema marshalli* | Not known | | | | | | |
| Ovate Clubshell<br>*Pleurobema perovatum* | Not known | | | | | | |
| Rough Pigtoe<br>*Pleurobema plenum* | Not known | | | | | | |
| Oval pigtoe<br>*Pleurobema pyriforme* | Eastern mosquitofish | 54.93 | 2.86 | Actinopterygii (class) | 48%<br>(39%, 56%) | 30%<br>(1%, 55%) | 56%<br>(3%, 82%) |
| | Sailfin shiner | 84.07 | 4.38 | Cyprinidae (family) | 31%<br>(18%, 44%) | 22%<br>(0%, 53%) | 33%<br>(0%, 78%) |
| | Guppy | 155.21 | 8.09 | *Poecilia reticulate* (species) | --- | --- | --- |
| Heavy Pigtoe<br>*Pleurobema taitianum* | Not known | | | | | | |
| Alabama Heelsplitter<br>*Potamilus inflatus* | Freshwater drum | 75.04 | 3.91 | Perciformes (order) | 36%<br>(24%, 47%) | 24%<br>(0%, 53%) | 40%<br>(0%, 79%) |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Triangular Kidneyshell *Ptychobranchus greeni* | Warrior darter Tuskaloosa darter Blackbanded darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| | Logperch | 34.97 | 1.82 | Percidae (family) | 63% (58%, 68%) | 39% (18%, 57%) | 72% (43%, 87%) |
| Rough rabbitsfoot *Quadrula cylindrical strigillata* | Whitetail shiner Spotfin shiner Bigeye chub | 84.07 | 4.38 | Cyprinidae (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| Winged Mapleleaf *Quadrula fragosa* | Blue catfish | 151.06 | 7.87 | Ictaluridae (family) | --- | --- | --- |
| | Channel catfish | 83.83 | 8.20 | *Ictalurus punctatus* (species) | --- | --- | --- |
| Cumberland Monkeyface *Quadrula intermedia* | Streamline chub Blotched chub | 84.07 | 4.38 | *Cyprinidae* (family) | 31% (18%, 44%) | 22% (0%, 53%) | 33% (0%, 78%) |
| Appalachian Monkeyface *Quadrula sparsa* | Not known | | | | | | |
| Stirrupshell *Quadrula stapes* | Not known | | | | | | |
| Pale Lilliput *Toxolasma cylindrellus* | Not known | | | | | | |
| Purple Bean *Villosa perpurpurea* | Fantail darter Greenside darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |
| | Black sculpin Mottled sculpin Banded sculpin | 54.93 | 2.86 | Actinopterygii (class) | 48% (39%, 56%) | 30% (1%, 55%) | 56% (3%, 82%) |
| Cumberland Bean *Villosa trabalis* | Fantail darter Striped darter | 33.07 | 1.72 | *Etheostoma* (genus) | 65% (60%, 70%) | 40% (20%, 58%) | 74% (46%, 88%) |

**Table 17. Fish species identified as hosts of listed mussels.**

| **Family** | **Common Name** | **Genus species** |
|---|---|---|
| | | |
| Catostomidae | | |
| | Northern hogsucker | *Hypentelium nigricans* |
| | Greater jumprock | *Moxostoma lachneri* |
| Centrarchidae | | |
| | Shadow bass | *Ambloplites ariommus* |
| | Rock bass | *Ambloplites rupestris* |
| | Warmouth | *Chaenobryttus gulosus* |
| | Green sunfish | *Lepomis cyanellus* |
| | Bluegill | *Lepomis macrochirus* |
| | Longear sunfish | *Lepomis megalotis* |
| | Redear sunfish | *Lepomis microlophus* |
| | Redeye bass | *Micropterus coosae* |
| | Smallmouth bass | *Micropterus dolomieu* |
| | Spotted bass | *Micropterus punctulatus* |
| | Largemouth bass | *Micropterus salmoides* |
| | Black crappie | *Pomoxis nigromaculatus* |
| Cottidae | | |
| | Black sculpin | *Cottus baileyi* |
| | Mottled sculpin | *Cottus bairdii* |
| | Banded sculpin | *Cottus carolinae* |
| | Slimy sculpin | *Cottus cognatus* |
| Cyprinidae | | |
| | Central stoneroller | *Campostoma anomalum* |
| | Largescale stoneroller | *Campostoma oligolepis* |
| | Rosyside dace | *Clinostomus funduloides* |
| | Satinfin shiner | *Cyprinella analostana* |
| | Alabama shiner | *Cyprinella callistia* |
| | Whitetail shiner | *Cyprinella galactura* |
| | Spotfin shiner | *Cyprinella spiloptera* |
| | Tricolor shiner | *Cyprinella trichroistia* |
| | Blacktail shiner | *Cyprinella venusta* |
| | Streamline chub | *Erimystax dissimilis* |
| | Blotched chub | *Erimystax insignis* |
| | Bigeye chub | *Hybopsis amblops* |
| | Striped shiner | *Luxilus chrysocephalus* |
| | Common shiner | *Luxilus cornutus* |
| | Rosefin shiner | *Lythrurus ardens* |
| | Pinewoods shiner | *Lythrurus matutinus* |
| | Redfin shiner | *Lythrurus umbratilis* |
| | Bluehead chub | *Nocomis leptocephalus* |
| | River chub | *Nocomis micropogon* |

| | | |
|---|---|---|
| | Golden shiner | *Notemigonus crysoleucas* |
| | White shiner | *Notropis albeolus* |
| | Warpaint shiner | *Notropis coccogenis* |
| | Tennessee shiner | *Notropis leuciodus* |
| | Swallowtail shiner | *Notropis procne* |
| | Telescope shiner | *Notropis telescopus* |
| | Weed shiner | *Notropis texanus* |
| | Mountain redbelly | *Phoxinus oreas* |
| | Sailfin shiner | *Pteronotropis hypselopterus* |
| | Blacknose dace | *Rhinichthys atratulus* |
| | Creek chub | *Semotilus atromaculatus* |
| Esocidae | | |
| | Chain pickerel | *Esox niger* |
| Fundulidae | | |
| | Blackspotted topminnow | *Fundulus olivaceus* |
| Ictaluridae | | |
| | Blue catfish | *Ictalurus furcatus* |
| | Channel catfish | *Ictalurus punctatus* |
| | Stonecat | *Noturus flavus* |
| | Brown madtom | *Noturus phaeus* |
| Percidae | | |
| | Naked sand darter | *Ammocrypta beanii* |
| | Emerald darter | *Etheostoma baileyi* |
| | Warrior darter | *Etheostoma bellator* |
| | Greenside darter | *Etheostoma blennioides* |
| | Rainbow darter | *Etheostoma caeruleum* |
| | Bluebreast darter | *Etheostoma camurum* |
| | Tuskaloosa darter | *Etheostoma douglasi* |
| | Brown darter | *Etheostoma edwini* |
| | Fantail darter | *Etheostoma flabellare* |
| | Southern sand darter | *Etheostoma meridianum* |
| | Johnny darter | *Etheostoma nigrum* |
| | Tesselated darter | *Etheostoma olmstedi* |
| | Redline darter | *Etheostoma rufilineatum* |
| | Tennessee snubnose darter | *Etheostoma simoterum* |
| | Snubnose darter | *Etheostoma simoterum* |
| | Speckled darter | *Etheostoma stigmaeum* |
| | Striped darter | *Etheostoma virgatum* |
| | Wounded darter | *Etheostoma vulneratum* |
| | Redfin darter | *Etheostoma whipplei* |
| | Banded darter | *Etheostoma zonale* |
| | Yellow perch | *Perca flavescens* |
| | Tangerine darter | *Percina aurantiaca* |
| | Blotchside logperch | *Percina burtoni* |
| | Logperch | *Percina caprodes* |

| | | |
|---|---|---|
| | Channel darter | *Percina copelandi* |
| | Gilt dater | *Percina evides* |
| | Blackside darter | *Percina maculata* |
| | Blackbanded darter | *Percina nigrofasciata* |
| | Roanoke darter | *Percina roanoka* |
| | Dusky darter | *Percina sciera* |
| | Saddleback darter | *Percina vigil* |
| | Fathead minnow | *Pimephales promelas* |
| | Sauger | *Sander canadensis* |
| | Walleye | *Sander vitreus* |
| Poecilidae | | |
| | Eastern mosquitofish | *Gambusia holbrooki* |
| | Guppy | *Poecilia reticulata* |
| Salmonidae | | |
| | Atlantic salmon | *Salmo salar* |
| | Brown trout | *Salmo trutta* |
| Sciaenidae | | |
| | Freshwater drum | *Aplodinotus grunniens* |

**Table 18. Species diversity of host fish for glochidia of listed mussels.**

| Host fish by species | | |
|---|---|---|
| **Family** | **# species** | **% all species** |
| Percidae* (darters, perch) | 35 | 34 |
| Cyprinidae* (shiners, chub, dace, stonerollers) | 34 | 33 |
| Centrarchidae* (bass, bluegill, sunfish) | 15 | 15 |
| Cottidae* (sculpin) | 4 | 4 |
| Ictaluridae (catfish) | 4 | 4 |
| Catostomidae* (suckers) | 3 | 3 |
| Poecilidae* (mosquitofish, guppies) | 2 | 2 |
| Salmonidae* (salmon, trout) | 2 | 2 |
| Sciaenidae* (drum) | 1 | 1 |
| Esocidae* (pickerel) | 1 | 1 |
| Fundulidae* (topminnows) | 1 | 1 |
| **TOTAL** | **102** | **100** |

*Sensitive to cyanide concentrations below EPA's proposed Aquatic Life Criteria.

**Table 19. Frequency of occurrence of host fish for glochidia of listed mussels.**

| **Host fish by frequency of occurrence** | | |
|---|---|---|
| **Family** | **# occurrences** | **% all occurrences** |
| Percidae* (darters, perch) | 82 | 36% |
| Cyprinidae* (shiners, chub, dace, stonerollers) | 51 | 22% |
| Centrarchidae* (bass, bluegill, sunfish) | 42 | 18% |
| Cottidae* (sculpin) | 26 | 11% |
| Poecilidae* (mosquitofish, guppies) | 8 | 4% |
| Sciaenidae* (drum) | 6 | 3% |
| Ictaluridae (catfish) | 4 | 2% |
| Catostomidae* (suckers) | 3 | 1% |
| Fundulidae* (topminnows) | 3 | 1% |
| Salmonidae* (salmon, trout) | 2 | 1% |
| Esocidae* (pickerel) | 1 | <1% |
| **TOTAL** | **228** | **100%** |

*Sensitive to cyanide concentrations below EPA's proposed Aquatic Life Criteria

*Effects to Mussel Populations*

There are no host fish for listed mussel species currently identified for which acute effects are anticipated at the criteria values. For species where hosts have not yet been identified, the probability of acute effects to host fish appears to be small. The greatest effect to mussel populations due to host fish effects is anticipated to result from declines in host fish abundance due to adverse effects to their fecundity and juvenile survival caused by exposure to cyanide at chronic criterion levels.

For effects to host fish caused by exposure to cyanide at chronic criterion levels, estimates were derived using methodologies for listed fish (Appendix F). Based on those calculations, host fish species for which adverse effects are anticipated at the chronic criterion may experience a reduction in eggs hatched that could be as much as, but not likely greater than, 31% to 63% compared to unexposed control populations (Table 16). Additionally, reductions in juvenile survival are expected at magnitudes that could be as high as 33% to 72% of control populations. Given the importance of fecundity and juvenile survival on overall effects to population abundance, this magnitude of adverse effect to the reproduction of host fish species is likely to translate into a decreased abundance of affected host fish species.

For listed mussels, any perturbation that limits fertilization rates and survivability of the glochidia, decreases host fish abundance, or decreases host fish community composition is detrimental to mussel population viability and, ultimately, the species as a whole (Downing et al. 1993, Neves 1993, Neves et al. 1997). Densities of host-specialist mussels, particularly those lacking elaborate host-attracting mechanisms, have been correlated to densities of host fish present in two drainage basins of Alabama (Haag and Warren 1998). No correlation was found for host-generalists or host-specialists with attractant lures. These data suggest that mussel species that showed a positive correlation may exhibit a density-dependence with host fish, limited by their abundance. It has been hypothesized that a gradual underlying decline of host fish abundance may play a major role in the steady decline of endangered mussel populations.

Although adult survival is typically the most influential life stage in population growth models of long-lived species like mussels, modeled effects to changes in reproduction can also have a significant influence on population growth. In simulation modeling performed for the three-ridge mussel (*Amblema plicata)*, deterministic methods using life history tables estimated that a 20% drop in the average number of young produced by females would negatively impact the population growth rate, resulting in a yearly population decline of 4.3% (Hart et al. 2004). Simulations incorporating environmental stochasticity predicted a 98% percent decline in the number of individuals after 100 years.

To determine the effects of exposure to chronic criteria concentrations of cyanide on listed mussel populations, species were grouped into one of the following four categories according to the number and diversity of fish species that have been identified as suitable hosts, and the estimated effects on those host species at criteria concentrations:

Category 1. Host-fish obligates, where the host fish is sensitive to cyanide exposure at criteria concentrations;

Category 2. Host-fish generalists or specialists, where most or all of known hosts are sensitive to cyanide exposure at criteria concentrations;

Category 3. No host fish identified; and

Category 4. Few fish hosts identified. While those which are currently known may be either sensitive or insensitive to cyanide at criteria concentrations, there are no data to indicate whether these fish species represent obligate hosts, a significant portion of the species assemblage of host fish, or a minor portion.

Category 1:

The mussel species listed below are host fish obligates. Using the methods described herein, the obligate host fish for these species, the freshwater drum, may experience a reduction in eggs hatched that could be as much as, but not likely greater than, 36% compared to unexposed control populations (Table 16). Additionally, reductions in juvenile survival are expected at magnitudes that could be as high as 40% of control populations. Given the importance of fecundity and juvenile survival on overall effects to population abundance, this magnitude of adverse effect to the reproduction of host fish species is likely to translate into a decreased abundance of the freshwater drum. If the abundance of an obligate host fish species decreases as a result of cyanide exposure, increased glochidia mortality is likely to occur as a result of their inability to attach to a suitable host. Since attachment of glochidia to a suitable host is a rare and necessary event in the mussel reproductive cycle, reductions in host fish abundance are likely to cause adverse impacts to mussel reproduction and population numbers. Since the oblighate host fish identified for the following mussel species are likely to exhibit population declines at cyanide criteria concentrations, these mussel species are likely to be subject to reduced reproduction, numbers, and distribution:

| | |
|---|---|
| Scaleshell Mussel | *Leptodea leptodon* |
| Fat Pocketbook | *Potamilus capax* |

Category 2

The mussel species listed below are either host fish specialists or generalists. Using the methods described herein, most of the identified host fish species for these mussels are likely to be subject to reduced levels of reproduction and juvenile survival as a result of exposure to cyanide at criteria concentrations (Table 16). Some host fish species identified for these mussels may not experience adverse effects to reproduction and juvenile survival at these concentrations (Table 16). There may also be host fish for these mussels which have yet to be identified that will be either be sensitive to or tolerant of cyanide exposure at criteria concentrations. If the abundance of any host fish species decreases as a result of cyanide exposure, increased glochidia mortality is likely to occur as a result of their inability to attach to a suitable host or the host dies during infestation. Since attachment of glochidia to a suitable host is a rare and necessary event in the mussel reproductive cycle, reductions in host fish abundance or the death of individual host fish that are infested by glochidia are likely to cause adverse impacts to mussel reproduction and population numbers. Since the majority of fish hosts identified for the following mussel species are likely to exhibit population declines at cyanide criteria concentrations, these mussel species are likely to be subject to reduced reproduction, numbers, and distribution:

| | |
|---|---|
| Cumberland Elktoe | *Alasmidonta atropurpurea* |
| Dwarf Wedgemussel | *Alasmidonta heterodon* |
| Fat Threeridge | *Amblema neislerii* |
| Ouachita Rock Pocketbook | *Arkansia wheeleri* |
| Birdwing Pearlymussel | *Conradilla caelata* |
| Fanshell | *Cyprogenia stegaria* |
| Dromedary Pearlymussel | *Dromus dromas* |
| Tar Spinymussel | *Elliptio steinstansana* |
| Purple Bankclimber | *Elliptoideus sloatianus* |
| Cumberland Combshell | *Epioblasma brevidens* |
| Oyster Mussel | *Epioblasma capsaeformis* |
| Curtis Pearlymussel | *Epioblasma florentina curtisii* |
| Tan Riffleshell | *Epioblasma florentina walkeri* |
| Catspaw (Purple cat's paw pearlymussel) | *Epioblasma obliquata obliquata* |
| Northern Riffleshell | *Epioblasma turulosa rangiana* |
| Shiny Pigtoe | *Fusconaia cor* |
| Fine-rayed Pigtoe | *Fusconaia cuneolus* |
| Pink Mucket | *Lampsilis abrupta* |
| Fine-lined Pocketbook | *Lampsilis altilis* |
| Higgins Eye | *Lampsilis higginsii* |
| Orangenacre Mucket | *Lampsilis perovalis* |

| | |
|---|---|
| Speckled Pocketbook | *Lampsilis streckeri* |
| Shinyrayed pocketbook | *Lampsilis subangulata* |
| Louisiana Pearlshell | *Margaritifera hembeli* |
| Alabama Moccasinshell | *Medionidus acutissimus* |
| Gulf Moccasinshell | *Medionidus penicillatus* |
| Littlewing Pearlymussel | *Pegis fibula* |
| Clubshell | *Pleurobema clava* |
| James Spinymussel | *Pleurobema collina* |
| Southern Clubshell | *Pleurobema decisum* |
| Dark Pigtoe | *Pleurobema furvum* |
| Southern Pigtoe | *Pleurobema georgianum* |
| Oval Pigtoe | *Pleurobema pyriforme* |
| Triangular Kidneyshell | *Ptychobranchus greeni* |
| Rough Rabbitsfoot | *Quadrula cylindrical strigillata* |
| Purple Bean | *Villosa perpurpurea* |

Category 3

No fish hosts have been identified for 23 of the listed mussel species considered in this biological opinion. While host fish are a necessary factor in the reproduction of freshwater mussels, there has not been adequate study to identify which fish can serve for hosts for a significant portion of North American mussels, including those which are threatened and endanagered. In cases of uncertainty, it is Service policy to error on the side of listed species. Therefore, for purposes of this analysis, we are assuming that the mussel species in this category are either host fish specialists or generalists but that all of the host fish species are sensitive to cyanide exposure at criteria concentrations to an extent that their populations are likely to be reduced. This effect is likely to cause increased glochidia mortality due to their inability to attach to a suitable host or death of the host. Since attachment of glochidia to a suitable host is a rare and necessary event in the mussel reproductive cycle, reductions in host fish abundance or death of individual host fish that are infested by glochidia are likely to cause declines in the reproduction, numbers, and distribution of the following mussel species:

| | |
|---|---|
| Yellow Blossom | *Epioblasma florentina florentina* |
| Upland Combshell | *Epioblasma metastriata* |
| White Catspaw Pearlymussel | *Epioblasma obliquata perobliqua* |
| Southern Acornshell | *Epioblasma othcaloogensis* |
| Southern Combshell | *Epioblasma penita* |
| Tubercled Blossom | *Epioblasma torulosa torulosa* |
| Turgid Blossom | *Epioblasma turgidula* |
| Green Blossom | *Epioblasma torulosa gubernaculums* |
| Cracking Pearlymussel | *Hemistena lata* |
| Alabama Lampmussel | *Lampsilis virescens* |
| Carolina Heelsplitter | *Lasmigon decorate* |
| Ochlockonee Moccasinshell | *Medionidus simpsonianus* |
| Ring Pink | *Obovaria retusa* |
| White Wartyback | *Plethobasus cicatricosus* |

| | |
|---|---|
| Orangefoot Pimpleback | *Plethobasus cooperianus* |
| Black Clubshell | *Pleurobema curtum* |
| Flat Pigtoe | *Pleurobema marshalli* |
| Ovate Clubshell | *Pleurobema perovatum* |
| Rough Pigtoe | *Pleurobema plenum* |
| Heavy Pigtoe | *Pleurobema taitianum* |
| Appalachian Monkeyface | *Quadrula sparsa* |
| Stirrupshell | *Quadrula stapes* |
| Pale Lilliput | *Toxolasma cylindrellus* |

Category 4:

For several species, few host fish have been identified. While those which are currently known may be either sensitive or insensitive to cyanide at criteria concentrations, there are no data to indicate whether these fish represent obligate hosts, a significant portion of the species assemblage of host fish, or a minor portion. As noted above, in cases of uncertainty, it is Service policy to error on the side of listed species. Therefore, for purposes of this analysis, we are assuming that the mussel species in this category are either host fish specialists or generalists but that many or all of the host fish species are sensitive to cyanide exposure at criteria concentrations to an extent that their populations are likely to be reduced. This effect is likely to cause increased glochidia mortality due to their inability to attach to a suitable host or death of the host. Since attachment of glochidia to a suitable host is a rare and necessary event in the mussel reproductive cycle, reductions in host fish abundance or death of individual host fish that are infested by glochidia are likely to cause declines in the reproduction, numbers, and distribution of the following mussel species:

| | |
|---|---|
| Appalachian Elktoe | *Alasmidonta raveneliana* |
| Chipola Slabshell | *Elliptio chipolaensis* |
| Arkanasas Fatmucket | *Lampsilis powelli* |
| Coosa Moccasinshell | *Medionidus parvulus* |
| Cumberland Pigtoe | *Pleurobema gibberum* |
| Alabama Heelsplitter | *Potamilus inflatus* |
| Winged Mapleleaf | *Quadrula fragosa* |
| Cumberland Monkeyface | *Quadrula intermedia* |
| Cumberland Bean | *Villosa trabalis* |

*Effects to Critical Habitat for Listed Mussels*

Category 1 Species:
No critical habitat has been designated for species in this category.

Category 2 Species:
The physical and biological features of critical habitat essential to the conservation of mussels include water of sufficient quality for normal behavior, growth, and survival of all life stages of the mussel and its host fish. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This

approval is likely to adversely affect the quality of water within critical habitat for the following mussel species to the degree that it would impair individual reproduction and survival of host fish, and cause host fish to experience adverse effects to growth, swimming performance, condition, and development (Table 16). For those reasons, we conclude that management of cyanide to the CCC level in water within critical habitat for the following mussel species is likely to degrade or preclude the proper function of the primary constituent elements of critical habitat that support water quality for normal behavior, growth, and survival of all life stages of the mussel and its host fish:

| | |
|---|---|
| Cumberland Elktoe | *Alasmidonta atropurpurea* |
| Fat Threeridge | *Amblema neislerii* |
| Purple Bankclimber | *Elliptoideus sloatianus* |
| Cumberlandian Combshell | *Epioblasma brevidens* |
| Oyster Mussel | *Epioblasma capsaeformis* |
| Fine-lined Pocketbook | *Lampsilis altilis* |
| Orangenacre Mucket | *Lampsilis perovalis* |
| Shinyrayed pocketbook | *Lampsilis subangulata* |
| Alabama Moccasinshell | *Medionidus acutissimus* |
| Gulf Moccasinshell | *Medionidus penicillatus* |
| Southern Clubshell | *Pleurobema decisum* |
| Dark Pigtoe | *Pleurobema furvum* |
| Southern Pigtoe | *Pleurobema georgianum* |
| Oval Pigtoe | *Pleurobema pyriforme* |
| Triangular Kidneyshell | *Ptychobranchus greeni* |
| Rough Rabbitsfoot | *Quadrula cylindrical strigillata* |
| Purple Bean | *Villosa perpurpurea* |

Category 3 Species:
The physical and biological features of critical habitat essential to the conservation of mussels include water of sufficient quality for normal behavior, growth, and survival of all life stages of the mussel and its host fish. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval is likely to adversely affect the quality of water within critical habitat for the following mussel species to the degree that it would impair individual reproduction and survival of potential host fish, and cause host fish to experience adverse effects to growth, swimming performance, condition, and development (Table 16). For those reasons, we conclude that management of cyanide to the CCC level in water within critical habitat for the following mussel species is likely to degrade or preclude the proper function of the primary constituent elements of critical habitat that support water quality for normal behavior, growth, and survival of all life stages of the mussel and its host fish:

| | |
|---|---|
| Upland Combshell | *Epioblasma metastriata* |
| Southern Acornshell | *Epioblasma othcaloogensis* |
| Carolina Heelsplitter | *Lasmigon decorate* |
| Ochlockonee Moccasinshell | *Medionidus simpsonianus* |
| Ovate Clubshell | *Pleurobema perovatum* |

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<u>Category 4 Species</u>:
The physical and biological features of critical habitat essential to the conservation of mussels include water of sufficient quality for normal behavior, growth, and survival of all life stages of the mussel and its host fish. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval is likely to adversely affect the quality of water within critical habitat for the following mussel species to the degree that it would impair individual reproduction and survival of known and/or potential host fish, and cause host fish to experience adverse effects to growth, swimming performance, condition, and development (Table 16). For those reasons, we conclude that management of cyanide to the CCC level in water within critical habitat for the following mussel species is likely to degrade or preclude the proper function of the primary constituent elements of critical habitat that support water quality for normal behavior, growth, and survival of all life stages of the mussel and its host fish:

| | |
|---|---|
| Appalachian Elktoe | *Alasmidonta raveneliana* |
| Chipola Slabshell | *Elliptio chipolaensis* |
| Coosa Moccasinshell | *Medionidus parvulus* |

### 7.4 Effects to Amphibians

Our assessment of the sensitivity of listed amphibian species to cyanide was based on multiple lines of evidence. First, we evaluated the available information on cyanide-induced effects on amphibians. We then reviewed the approach EPA used in their Biological Evaluation to assess the sensitivity of listed amphibians to cyanide and the protectiveness of the cyanide criteria. Next, we examined additional toxicity information for amphibians, not used by EPA, and constructed regression models for predicting the acute sensitivities of amphibian genera to cyanide. Finally, we compared the predicted sensitivity of amphibians with that of rainbow trout; the most sensitive freshwater species (based on measured cyanide $LC_{50}$s) and the species that was used to set the acute and chronic cyanide criteria. Taken together, these data provided the basis for our effects determination.

The scientific literature for cyanide toxicity to amphibians is limited and somewhat dated. Early investigators studied the effects of cyanide on amphibian development. These experiments were generally focused on early embryogenesis including oviposited and fertilized egg morphogenesis and post gastrulation development. Repressive effects of cyanide on embryonic respiration and development were documented by several authors (Spiegelman and Moog, 1943, Lovtrup and Pigon, 1958, Nakatsuji, 1974). Others used sub-lethal exposure concentrations of cyanide as a mechanism to arrest or retard development in order to test various hypotheses regarding metabolism or physiology (Spiegelman and Steinbach, 1945; Ornstein and Gregg, 1952). Although these historical studies are important for understanding the physiological actions of cyanide on amphibians, they do not provide the traditional quantitative measures of acute and chronic toxicity (i.e. $LC_{50}$s, NOECs, $EC_x$s) that have been used in water quality criteria development.

Because cyanide-specific toxicity data ($LC_{50}$s) for amphibians were not available, EPA based their effects analysis on the relative sensitivity of amphibians to other pollutants (EPA 2007). They examined the rank order of amphibian $LC_{50}$s for seven water pollutants using data sets from ambient water quality criteria documents (Table 20). The 7 data sets included $LC_{50}$s for 9 amphibian species (in total), although 4 of the data sets contained $LC_{50}$s for only 1 amphibian species and the other 3 data sets contained data for 2 species. So among these seven criteria documents, the amphibian class was represented by no more than one or two species at a time. With so few species used to characterize the sensitivity of an entire class there is considerable uncertainty as to whether the most sensitive amphibian species are adequately represented, especially considering the large interspecies variability in cyanide toxicity observed for other taxa (see acute effects section of BO). It seems highly unlikely that the amphibians species included in these data sets were among the most sensitive amphibians. Nevertheless, for two of the seven pollutants the single amphibian species in the data set ranked among the most sensitive species/genera in the multi-taxa data sets used to develop criteria. For the remaining five pollutants the GMAVs for amphibians ranged from the 26$^{th}$ percentile to the 100$^{th}$ percentile. Considering the low number of species used to represent amphibians in the analysis and the fact that amphibians were among the most sensitive species/genera for 28% of the pollutants examined we believe that there is a more than a discountable chance that some amphibian species may be highly sensitive to cyanide. Therefore, we do not believe these results alone support EPAs determination that the listed amphibian species are not likely to be adversely affected by cyanide at criteria concentrations.

To better understand how to interpret the results from EPAs analysis we extended our evaluation to include rainbow trout; a species frequently included in criteria development and often among the more sensitive species tested (Table 20). Using data for the same seven pollutants we found that the over all pattern of rankings for rainbow trout were much like those for amphibians, i.e. most near or above the median and two or three falling among the most sensitive species. However we know that in terms of cyanide, rainbow trout is the most sensitive freshwater species that has been tested, more sensitive than the 5$^{th}$ percentile estimated species (EPA 1985). (That is, rainbow trout fell in the "sensitive tail" of the species sensitivity distribution.) So, there is at least one example where the "ranking profile" (for these 7 pollutants) shared by amphibians and rainbow trout was associated with a species that was highly sensitive to cyanide. In addition, we found that for these seven pollutants amphibian species were more sensitive than rainbow trout 43% of the time (3 of 7). To further investigate the relative sensitivity of amphibians to other taxa we reviewed other references on amphibian toxicology.

**Table 20. Rank and corresponding percentile of GMAVs (genus mean acute values) for amphibians and rainbow trout versus all aquatic taxa and chordates (fishes) only. Data for amphibians are from Appendix D of EPAs Cyanide Biological Evaluation (EPA 2007). Data for rainbow trout are from criteria documents (see footnotes).**

| Chemical | Amphibian Species | Amphibian GMAV Rank Vs. Other Taxa | Rainbow (GMAV) Rank vs. Other Taxa | Percentile (Amphibians) | Percentile (Rainbow trout) | Amphibians more (+) or less (-) sensitive than Rainbow trout |
|---|---|---|---|---|---|---|
| Atrazine | Bufo americanus | 11 of 19 | 4 of 19[1] | 0.58 | 0.21 | - |
| Atrazine | Rana sp. | 14 of 19 | | 0.74 | | |
| Cadmium | Ambystoma gracile | 29 of 57 | 4 of 57[2] | 0.51 | 0.07 | - |
| Cadmium | Xenopus laevis | 33 of 57 | | 0.58 | | |
| Diazinon | Rana clamitans | 8 of 21 | 12 of 21[3] | 0.26 | 0.57 | + |
| Lindane | Pseudacris triseriata | 22 of 23 | 10 of 23[4] | 0.96 | 0.43 | - |
| Lindane | Bufo woodhousei | 23 of 23 | | 1.00 | | |
| Nonylphenol | Bufo boreas | 2 of 15 | 8 of 15[5] | 0.13 | 0.53 | + |
| Parathion | Pseudacris triseriata | 23 of 31 | 25 of 31[5] | 0.74 | 0.81 | + |
| Pentachlorophenol | Rana satesbeiana | 4[6] of 32 | 3 of 32[5] | 0.13 | 0.09 | - |

[1] Draft aquatic life ambient water quality criteria for atrazine (EPA 2003)
[2] 2001 update of the aquatic life ambient water quality criteria for cadmium (EPA 2001)
[3] Aquatic life ambient water quality criteria for diazinon (EPA 2005)
[4] 1995 updates: water quality criteria documents for the protection of aquatic life in ambient water (EPA 1996)
[5] Aquatic life ambient water quality criteria for nonylphenol (EPA 2005)
[6] Rank was changed from 5 to 4 based on GMAV ranks for pentachlorophenol (EPA 1996)

Birge at al. (2003) performed a comparative toxicity analysis for 29 amphibian species in contrast to various species of fish. Amphibian testing included seven salamander species (family Ambystomidae) and 22 frog species (families Microhylidae, Hylidae, Ranidae, and Bufonidae). Though no toxicity testing was performed for cyanide, sufficient data was produced to generate comparisons between amphibians and fish for 34 inorganic compounds and 27 organic compounds. Comparisons include all amphibian test species for 50 of these 61 compounds. Although exposure times varied among species due to differences in hatching times, comparable stages of development (eggs, embryos, and early larvae) were included in all tests. Fish species included in this study for which sensitivity to cyanide is known are the rainbow trout ($LC_{50}$ = 59.22ug/g), largemouth bass (101.7 ug/g) and fathead minnow (138.4 ug/g).

When compared to rainbow trout, $LC_{50}$ values for amphibians were more sensitive 52% of the time for metals (N=203), 36% for organics (N=44), and 49% for all compounds combined (N=247). For largemouth bass, amphibians were more sensitive 83% of the time (N=182), 60% for organics (N=15), and 81% for all compounds (N=197). For fathead minnow, amphibians were more sensitive 89% of the time for metals (N=18), 63% for organics (N=24), and 74% for all compounds (N=42). The generally more sensitive species of Microhylidae and Hylidae were not available for toxicity testing for several organic compounds. For the 15 most sensitive amphibian species, $LC_{50}$ values were below fish values (including species used above, plus channel catfish and goldfish) 74% of the time.

Bridges et al (2002), performed toxicity testing for five compounds on southern leopard frog (*Rana sphenocephala*) tadpoles and compared results with published values for the boreal toad (*Bufo boreas*), rainbow trout, fathead minnow, and bluegill. The two amphibian species showed the highest correlation of $LC_{50}$ values for the rainbow trout. Correlations for the fathead minnow and bluegill were much weaker. The authors suggest that rainbow trout thus may be the most appropriate species for assessing toxicity to anuran tadpoles. However, the authors also argue that since amphibians are very tolerant to some chemicals, and very sensitive to others, individual toxicity testing is suggested rather than relying on surrogate species.

The comparative toxicity data sets from Birge et al. (2003) provided an opportunity to construct ICE-like regression models that could be used to estimate cyanide $LC_{50}$s for amphibians (Table 21; Figures 5 and 6). Following EPA guidelines (EPA 2003b), regression models were developed to estimate the sensitivity of two amphibian genera (Rana and Ambystoma) using rainbow trout as the surrogate species.

**Table 21.** Estimated cyanide $LC_{50}$s for two amphibian genera (*Rana* and *Ambystoma*) using rainbow trout as a surrogate species

| Predicted Taxon | Surrogate Species | LCI $LC_{50}$ (ug/L) | MLE $LC_{50}$ (ug/L) | UCI $LC_{50}$ (ug/L) | Corr. Coeff. (r) | MSE | log-log a | log-log b | p | n | Chem. |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Rana (genus) | Rainbow Trout | 30.82 | 54.25 | 95.51 | 0.789 | 0.648 | 0.259 | 0.832 | <0.001 | 84 | 32 |
| Ambystoma (genus) | Rainbow Trout | 60.56 | 120.61 | 639.63 | 0.638 | 0.415 | 0.966 | 0.629 | <0.002 | 23 | 4 |

**Figure 6. Rainbow Trout Interspecies Correlation Plot for the Genus Ambystoma**

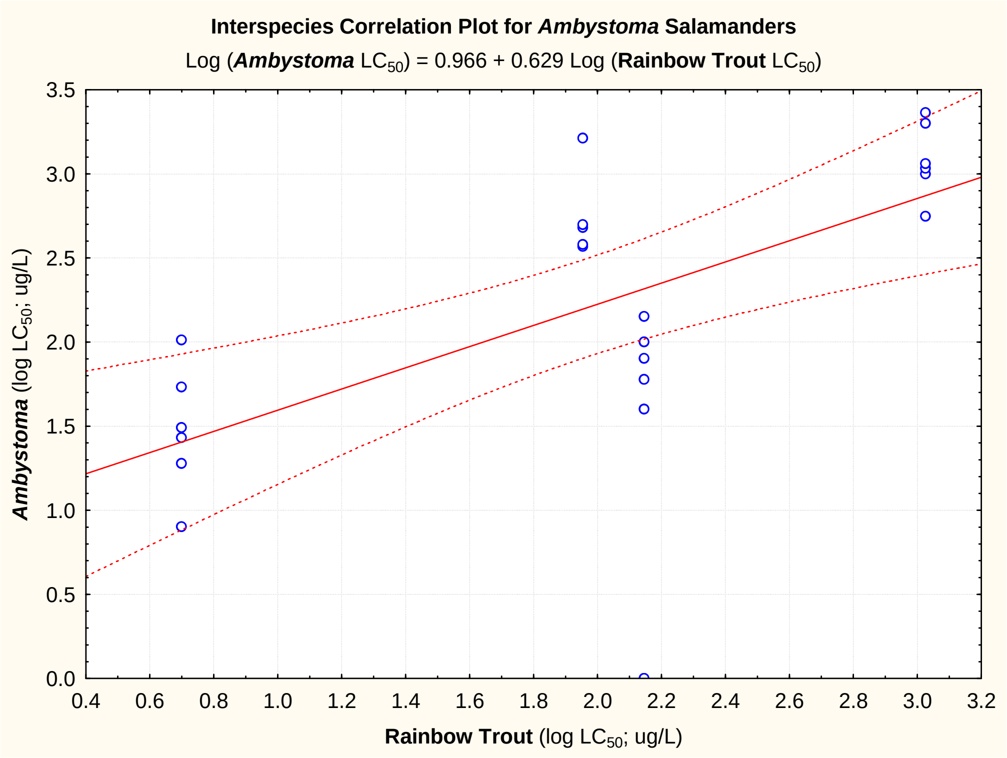
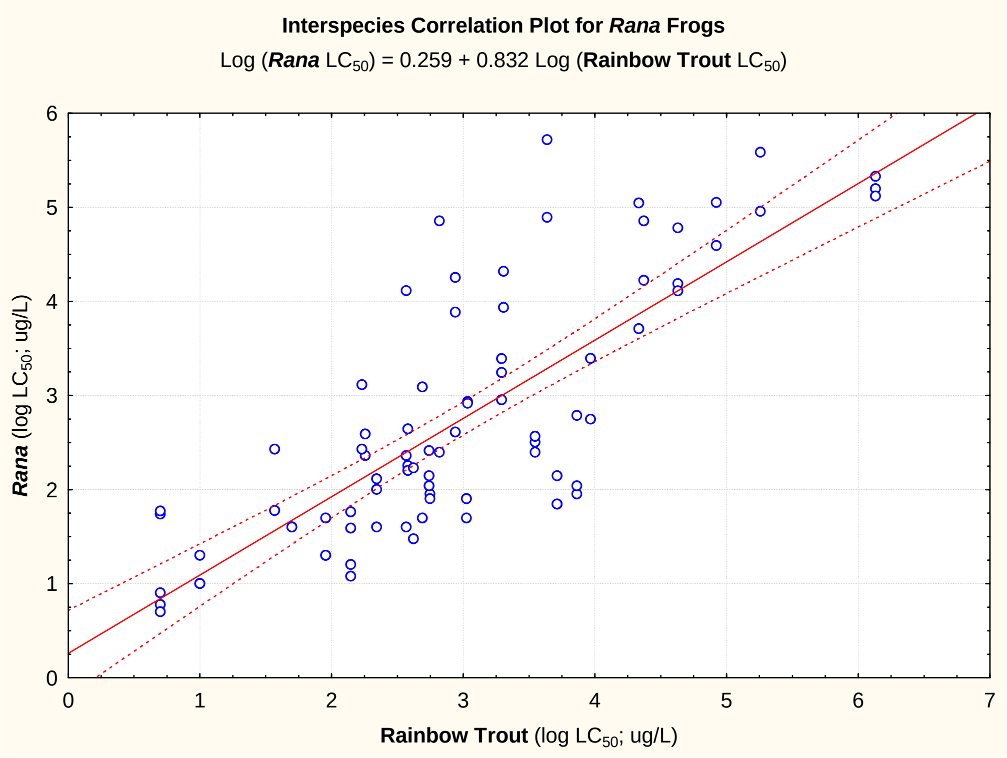



**Figure 7. Rainbow Trout Interspecies Correlation Plot for the Genus Rana**



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Estimated $LC_{50}$s for the two amphibian genera, Ambystoma ($LC_{50}$ 60.56 ug CN/L) and Rana ($LC_{50}$ 30.82 ug CN/L), are approximately equal to or less than the $LC_{50}$ for rainbow trout (59 ug CN/L).

As previously mentioned, rainbow trout had the lowest measured cyanide $LC_{50}$ of all fish species considered in the cyanide criteria document as well as the cyanide BE. Based on the method described in the *Fish* section of Appendix B, the chronic $EC_A$ for rainbow trout would be 2.54 ug CN/L (i.e. 59 ug CN/L / 23.22) and the acute $EC_A$ would be 51.75 ug CN/L (i.e. 59 ug CN/L / 1.14). Because the chronic $EC_A$ is below 5.2 ug CN/L rainbow trout would likely be adversely affected by exposure to cyanide at the CCC. Thus, amphibian species are estimated to be as or more sensitive to cyanide than rainbow trout and thus likely to be adversely affected by exposure to cyanide at the chronic criterion.

*Individual Species and Critical Habitat Accounts*

### *Ambystomatidae*

## RETICULATED FLATWOODS SALAMANDER
*Ambystoma bishop*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that reticulated flatwoods salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, reticulated flatwoods salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The life history of the reticulated flatwoods salamander can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Reticulated flatwoods salamanders breed in relatively small, isolated ephemeral ponds where the larvae develop until metamorphosis. Post-metamorphic salamanders migrate out of the ponds and into the uplands where they live until they move back to ponds to breed as adults. The flatwoods salamander reproduces at 1 year of age for males and two years of age for females. Males and females court before the breeding sites flood. Females then lay their eggs, either singly or in clumps, beneath leaf litter, under logs, sphagnum moss mats, small trees, bushes or clumps of grass at dry locations in seasonal wetlands. If rainfall is insufficient to result in adequate pond flooding, breeding may not occur or, if larvae do develop, they may die before metamorphosis. Egg development from deposition to hatching occurs in approximately 2 weeks, but eggs do not hatch until they are inundated. Depending on when they are inundated, the larvae metamorphose 11 to 18 weeks after hatching. Exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, and in the case of the reticulated flatwoods salamander such reduced growth rates could preclude emergence prior to pond drying.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the reticulated flatwoods salamander's fertility rates substantially. The salamander's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). Taylor et al. (2006) constructed a model, based on extensive population data available for the marbled salamander (*Ambystoma opacum*), to look at how many years of reproductive failure would be required to result in local extinction of pond-breeding salamanders (with varying lifespans) and found that even without total reproductive failure, populations required moderate to high upland post-metamorphic survival to persist. Catastrophic reproductive failure in this study created fluctuations in the population, raised the threshold of survival required to achieve persistence, and imposed the possibility of extinction even under otherwise favorable environmental conditions. Even in populations with multiple breeding ponds, amphibian populations may be unable to recolonize areas after local extirpations due to their physiological constraints, relatively low mobility, and site fidelity. In the case of the reticulated flatwoods salamander, only 20 populations are known and 14 (70 percent) of these populations are supported by only one breeding site. For those populations with only one breeding pond, habitat destruction associated with cyanide at CCC levels may adversely effect flatwoods salamander reproduction and survival resulting in extirpation of the population supported by that breeding pond. For populations with more than one breeding pond, habitat destruction associated with cyanide at CCC levels may result in a reduction of available breeding sites leading to a reduction in population size and range and an increased vulnerability to catastrophic events that may adversely affect breeding and survival.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the reticulated flatwoods salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. The reticulated flatwoods salamander has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected flatwoods salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. The majority of extant populations are supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, reticulated flatwoods salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the reticulated salamander.

**Critical Habitat:** Critical habitat for the reticulated flatwoods salamander has been designated in: Calhoun, Holmes, Jackson, Santa Rosa, Walton, and Washington Counties Florida; Baker and Miller Counties Georgia. The physical and biological features of critical habitat essential to the conservation of the reticulated flatwoods salamander includes breeding habitats consisting of small, acidic depressional standing bodies of water that are seasonally flooded by rainfall in late fall or early winter and dry in late spring or early summer; and are geographically isolated from other water bodies. In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of reticulated flatwoods salamanders and cause salamanders to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of reticulated flatwoods salamanders and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## FROSTED FLATWOODS SALAMANDER
*Ambystoma cingulatum*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that frosted flatwoods salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, frosted flatwoods salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The life history of the frosted flatwoods salamander can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Frosted flatwoods salamanders breed in relatively small, isolated ephemeral ponds where the larvae develop until metamorphosis. Post-metamorphic salamanders migrate out of the ponds and into the uplands where they live until they move back to ponds to breed as adults. The flatwoods salamander reproduces at 1 year of age for males and two years of age for females. Males and females court before the breeding sites flood. Females then lay their eggs, either singly or in clumps, beneath leaf litter, under logs, sphagnum moss mats, small trees, bushes or clumps of grass at dry locations in seasonal wetlands. If rainfall is insufficient to result in adequate pond flooding, breeding may not occur or, if larvae do develop, they may die before metamorphosis. Egg development from deposition to hatching occurs in approximately 2 weeks, but eggs do not hatch until they are inundated. Depending on when they are inundated, the larvae metamorphose 11 to 18 weeks after hatching. Exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, and in the case of the frosted flatwoods salamander such reduced growth rates could preclude emergence prior to pond drying.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the frosted flatwoods salamander's fertility rates substantially. The salamander's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). Taylor et al. (2006) constructed a model, based on extensive population data available for the marbled salamander (*Ambystoma opacum*), to look at how many years of reproductive failure would be required to result in local extinction of pond-breeding salamanders (with varying lifespans) and found that even without total reproductive failure, populations required moderate to high upland post-metamorphic survival to persist. Catastrophic reproductive failure in this study created fluctuations in the population, raised the threshold of survival required to achieve persistence, and imposed the possibility of extinction even under otherwise favorable environmental conditions. Even in populations with multiple breeding ponds, amphibian populations may be unable to recolonize areas after local extirpations due to their physiological constraints, relatively low mobility, and site fidelity. Surveys indicate there are 25 populations of the frosted flatwoods salamander, some of which have been inferred from the capture of a single individual. Twenty-two (88 percent) of the known frosted flatwoods salamander populations occur primarily on public land. Sixteen of the populations (64 percent of total populations of the species) on public land represent metapopulations supported by more than one breeding site. For populations with only one breeding pond, if the habitat at that site is destroyed, recolonization would be impossible and the population supported by that breeding pond would be extirpated. For those populations with only one breeding pond, habitat destruction associated with cyanide at CCC levels may adversely effect flatwoods salamander reproduction and survival resulting in extirpation of the population supported by that breeding pond. For populations with more than one breeding pond, habitat destruction associated with cyanide at CCC levels may result in a reduction of available breeding sites leading to a reduction in population size and range and an increased vulnerability to catastrophic events that may adversely affect breeding and survival.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the frosted flatwoods salamander’s reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. The frosted flatwoods salamander has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander’s life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected flatwoods salamander population’s decline could stabilize at a reduced absolute population number or could continue to decline until it’s extirpated. The majority of extant populations are supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, frosted flatwoods salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the frosted flatwoods salamander.

**Critical Habitat:** Critical habitat has been designated for the frosted flatwoods salamander in: Baker, Franklin, Jefferson, Liberty, and Wakulla Counties, Florida; and in Berkeley, Charleston, and Jasper Counties, South Carolina. The physical and biological features of critical habitat essential to the conservation of the frosted flatwoods salamander includes breeding habitats consisting of small, acidic depressional standing bodies of water that are seasonally flooded by rainfall in late fall or early winter and dry in late spring or early summer; and are geographically isolated from other water bodies. In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of frosted flatwoods salamanders and cause salamanders to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of frosted flatwoods salamanders and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## CALIFORNIA TIGER SALAMANDER

*Ambystoma californiese*

Central California population

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that California tiger salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, California tiger salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The life history of the California tiger salamander can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Adult California tiger salamanders mate in vernal pools and similar water bodies, and the females lay their eggs in the water. Females attach their eggs singly or, in rare circumstances, in groups of two to four, to twigs, grass stems, vegetation, or debris. In ponds with little or no vegetation, females may attach eggs to objects, such as rocks and boards on the bottom. After breeding, adults leave the pool and return to small mammal burrows in surrounding uplands, although they may continue to come out nightly for approximately the next two weeks to feed. In drought years, the seasonal pools may not form and the adults may not breed. The eggs hatch in 10 to 14 days with newly hatched salamanders (larvae) ranging in size from 11.5 to 14.2 mm (0.5 to 0.6 in) in total length. The larval stage of the California tiger salamander usually lasts three to six months, because most seasonal ponds and pools dry up during the summer, although some larvae in Contra Costa and Alameda Counties may remain in their breeding sites over the summer. Amphibian larvae must grow to a critical minimum body size before they can metamorphose (change into a different physical form) to the terrestrial stage. One study found larvae metamorphosed and left the breeding pools 60 to 94 days after the eggs had been laid, with larvae developing faster in smaller, more rapidly drying pools. The longer the inundation period, the larger the larvae and metamorphosed juveniles are able to grow, and the more likely they are to survive and reproduce. The larvae perish if a site dries before they complete metamorphosis. There was a strong positive correlation between inundation period and total number of metamorphosing juvenile amphibians, including tiger salamanders. Size at metamorphosis is positively correlated with stored body fat and survival of juvenile amphibians, and negatively correlated with age at first reproduction

Exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the California tiger salamander could preclude emergence prior to pond drying, resulting in larval mortalities.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the California tiger salamander’s fertility rates substantially. The salamander’s fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section of Population Level Effects, we noted that reduction in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to reproduce, like California tiger salamanders. Lifetime reproductive success for California and other tiger salamanders is low. One study found the average female bred 1.4 times and produced 8.5 young that survived to metamorphosis per reproductive effort. This resulted in roughly 11 metamorphic offspring over the lifetime of a female. Most California tiger salamanders in this study did not reach sexual maturity until four or five years old. While individuals may survive for more than 10 years, many breed only once, and one study estimated that less than five percent of metamorphic juveniles survive to become breeding adults. The mechanisms for recruitment are clearly dependent on a number of factors such as migration, terrestrial survival, and population turnover, whose interaction is not well understood.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the California tiger salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. The California tiger salamander has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected California tiger salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, California tiger salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the California Tiger salamander.

**Critical Habitat:** The exact locations of the critical habitat are depicted on maps in the federal register. The physical and biological features of critical habitat essential to the conservation of the California tiger salamander includes standing bodies of fresh water (including natural and manmade (e.g., stock)) ponds, vernal pools, and other ephemeral or permanent water bodies which typically support inundation during winter rains and hold water for a minimum of 12 weeks in a year of average rainfall. In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of California tiger salamanders and cause salamanders to experience adverse effects to growth, locomotion, condition, and development. Based on data for the genus *Oncorhyncus*, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of California tiger salamanders and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## CALIFORNIA TIGER SALAMANDER

*Ambystoma californiese*

Santa Barbara County population

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that California tiger salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, California tiger salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects *Overview*. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

This population is restricted to Santa Barbara County, California. The life history of the California tiger salamander can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Adult California tiger salamanders mate in vernal pools and similar water bodies, and the females lay their eggs in the water. Females attach their eggs singly or, in rare circumstances, in groups of two to four, to twigs, grass stems, vegetation, or debris. In ponds with little or no vegetation, females may attach eggs to objects, such as rocks and boards on the bottom. After breeding, adults leave the pool and return to small mammal burrows in surrounding uplands, although they may continue to come out nightly for approximately the next two weeks to feed. In drought years, the seasonal pools may not form and the adults may not breed. The eggs hatch in 10 to 14 days with newly hatched salamanders (larvae) ranging in size from 11.5 to 14.2 mm (0.5 to 0.6 in) in total length. The larval stage of the California tiger salamander usually lasts three to six months, because most seasonal ponds and pools dry up during the summer, although some larvae in Contra Costa and Alameda Counties may remain in their breeding sites over the summer. Amphibian larvae must grow to a critical minimum body size before they can metamorphose (change into a different physical form) to the terrestrial stage. One study found larvae metamorphosed and left the breeding pools 60 to 94 days after the eggs had been laid, with larvae developing faster in smaller, more rapidly drying pools. The longer the inundation period, the larger the larvae and metamorphosed juveniles are able to grow, and the more likely they are to survive and reproduce. The larvae perish if a site dries before they complete metamorphosis. There was a strong positive correlation between inundation period and total number of metamorphosing juvenile amphibians, including tiger salamanders. Size at metamorphosis is positively correlated with stored body fat and survival of juvenile amphibians, and negatively correlated with age at first reproduction

Exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the California tiger salamander could preclude emergence prior to pond drying, resulting in larval mortalities.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the California tiger salamander's fertility rates substantially. The salamander's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section of Population Level Effects, we noted that reduction in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to reproduce, like California tiger salamanders. Lifetime reproductive success for California and other tiger salamanders is low. One study found the average female bred 1.4 times and produced 8.5 young that survived to metamorphosis per reproductive effort. This resulted in roughly 11 metamorphic offspring over the lifetime of a female. Most California tiger salamanders in this study did not reach sexual maturity until four or five years old. While individuals may survive for more than 10 years, many breed only once, and one study estimated that less than five percent of metamorphic juveniles survive to become breeding adults. The mechanisms for recruitment are clearly dependent on a number of factors such as migration, terrestrial survival, and population turnover, whose interaction is not well understood.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the California tiger salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. The California tiger salamander has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected California tiger salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, California tiger salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the California Tiger salamander.

**Critical Habitat:** Critical habitat has been established in Santa Barbara County for this population segment of the California tiger salamander in six separate locations east of Vandenberg Air Force Base. The physical and biological features of critical habitat essential to the conservation of the California tiger salamander includes standing bodies of fresh water (ponds, vernal pools, dune ponds, or other ephemeral or permanent water bodies) sufficient for the aquatic portion of the salamander's life cycle. In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of California tiger salamanders and cause salamanders to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of California tiger salamanders and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## CALIFORNIA TIGER SALAMANDER

*Ambystoma californiese*

Sonoma County population

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that California tiger salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, California tiger salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The Sonoma population appears to have been geographically isolated from the remainder of the California tiger salamander population by distance, mountains and major waterway barriers for more than 700,000 years. It occurs only in association with vernal pool ecosystems and stock ponds remaining on the Santa Rosa Plain of Sonoma County, California. There are 8 known breeding sites within the Santa Rosa Plain. The life history of the California tiger salamander can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Adult California tiger salamanders mate in vernal pools and similar water bodies, and the females lay their eggs in the water. Females attach their eggs singly or, in rare circumstances, in groups of two to four, to twigs, grass stems, vegetation, or debris. In ponds with little or no vegetation, females may attach eggs to objects, such as rocks and boards on the bottom. After breeding, adults leave the pool and return to small mammal burrows in surrounding uplands, although they may continue to come out nightly for approximately the next two weeks to feed. In drought years, the seasonal pools may not form and the adults may not breed. The eggs hatch in 10 to 14 days with newly hatched salamanders (larvae) ranging in size from 11.5 to 14.2 mm (0.5 to 0.6 in) in total length. The larval stage of the California tiger salamander usually lasts three to six months, because most seasonal ponds and pools dry up during the summer, although some larvae in Contra Costa and Alameda Counties may remain in their breeding sites over the summer. Amphibian larvae must grow to a critical minimum body size before they can metamorphose (change into a different physical form) to the terrestrial stage. One study found larvae metamorphosed and left the breeding pools 60 to 94 days after the eggs had been laid, with larvae developing faster in smaller, more rapidly drying pools. The longer the inundation period, the larger the larvae and metamorphosed juveniles are able to grow, and the more likely they are to survive and reproduce. The larvae perish if a site dries before they complete metamorphosis. There was a strong positive correlation between inundation period and total number of metamorphosing juvenile amphibians, including tiger salamanders. Size at metamorphosis is positively correlated with stored body fat and survival of juvenile amphibians, and negatively correlated with age at first reproduction

Exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the California tiger salamander could preclude emergence prior to pond drying, resulting in larval mortalities.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the California tiger salamander's fertility rates substantially. The salamander's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Formal Draft Biological Opinion.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). In the section of Population Level Effects, we noted that reduction in fecundity and juvenile survival would have a greater population-level effect on species that have fewer opportunities to reproduce, like California tiger salamanders. Lifetime reproductive success for California and other tiger salamanders is low. One study found the average female bred 1.4 times and produced 8.5 young that survived to metamorphosis per reproductive effort. This resulted in roughly 11 metamorphic offspring over the lifetime of a female. Most California tiger salamanders in this study did not reach sexual maturity until four or five years old. While individuals may survive for more than 10 years, many breed only once, and one study estimated that less than five percent of metamorphic juveniles survive to become breeding adults. The mechanisms for recruitment are clearly dependent on a number of factors such as migration, terrestrial survival, and population turnover, whose interaction is not well understood.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the California tiger salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. The California tiger salamander has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected California tiger salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, California tiger salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the California Tiger salamander.

## SONORA TIGER SALAMANDER

*Ambystoma tigrinum stebbinsi*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that Sonora tiger salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, Sonora tiger salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

This species probably once inhabited springs, streams, backwaters, and cienegas that held permanent or nearly permanent water sources in the San Rafael Valley, Arizona, and Sonora, Mexico. Cattle ponds or tanks are now the primary habitat for Sonora tiger salamanders. Terrestrial salamanders likely spend much of the year in rodent burrows, rotted logs, and other moist cover sites. The life history of the Sonora tiger salamander can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that may be fully aquatic or primarily terrestrial. Sonora tiger salamanders begin their life as jelly-coated eggs laid in water. They hatch and grow as aquatic larvae with gills, and then either mature as gilled aquatic adults called branchiate adults, neotenes, or paedomorphs, or metamorphose into terrestrial salamanders without gills. Sonora tiger salamanders begin breeding as early as January, and eggs can be found in ponds as late as early May. Breeding after monsoon rains in July and August is rare. Courtship takes place under water, and after fertilization, female tiger salamanders lay 200 to 2000 eggs attaching them to aquatic vegetation, sticks, rocks, or substrate individually or in clumps of up to 50. Eggs take from 2-4 weeks to hatch; the colder the water, the longer the eggs take to develop. Following hatching, Sonora tiger salamander larvae can develop to the minimum size necessary to metamorphose in as little as two months. However, because many sites with Sonora salamanders hold water all year, larvae often remain in the water longer before metamorphosing, or develop into branchiate adults instead of metamorphosing. The proportion of larvae that metamorphose depends heavily on pond permanence. In ponds that dry, all larvae that are large enough metamorphose. In ponds that do not dry, approximately 17 percent of larvae that are large enough metamorphose (Collins *et al.* 1988).

Exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the Sonora tiger salamander could preclude emergence prior to pond drying, resulting in larval mortalities.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the Sonora salamander's fertility rates substantially. The salamander's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

In permanent water bodies, approximately 83% of the Sonora salamander larvae develop into branchiate adults instead of metamorphosing. In these cases, individuals could experience increased exposure to cyanide at the CCC due to their continued use of aquatic habitats throughout all stages of their life cycle. Whereas metamorphosed terrestrial adults may experience reduced adult growth, survival and fitness due to the long-term effects of cyanide exposure during egg and larval development, fully aquatic adults will be at greater risk of such adverse and compounded affects due to continued exposure to cyanide at CCC as adults.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). Metamorphs are the only life history stage that can disperse from pond to pond and establish new populations. Data suggest that only a small proportion of salamanders in a pond are likely to have dispersed from another pond, so salamanders in each pond are referred to as a population. For those populations consisting primarily of branchiate adults, habitat destruction associated with cyanide at CCC levels may adversely effect Sonora tiger salamander reproduction and survival to the extent that the population supported by that breeding pond becomes extirpated. Due to the low occurrence or dispersal amongst terrestrial adults, habitat destruction associated with cyanide at CCC levels may also result in population extirpation.

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Sonora tiger salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. The Sonora tiger salamander has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are either wholly aquatic or primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e.

breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Sonora tiger salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, Sonora tiger salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Sonora tiger salamander.

## SANTA CRUZ LONG-TOED SALAMANDER

*Ambystoma macrodactylum croceum*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that Santa Cruz long-toed salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, Santa Cruz long-toed salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The life history of the Santa Cruz long-toed salamander can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that may be fully aquatic or primarily terrestrial. Santa Cruz long-toed salamanders spend most of their lives underground in small mammal burrows and along the root systems of plants in upland chaparral and woodland areas of coast live oak (*Quercus agrifolia*) or Monterey pine (*Pinus radiata*) as well as riparian strips of arroyo willows (*Salix lasiolepis*) and other species. Ideal breeding locations appear to be shallow, temporary, freshwater ponds that lack fishes and hold water at least through the spring months. Eggs are laid singly on submerged stalks of spike rush (*Eleocharis* spp.) or other vegetation about two to three centimeters apart. Free floating, unattached, and clustered eggs have also been observed. Each female lays about 300 (range 215 to 411) eggs per year. Eggs usually hatch in 15 to 30 days. The larvae remain in the pond environment for 90 to 145 days. Larvae metamorphose when they reach a minimum size of about 32 mm snout to vent length. Metamorphosis can be accelerated by adverse pond conditions, such as reduction in food resources, water pollution, increased temperatures, and drying of the pond environment. Metamorphosed salamanders leave the pond. These salamanders become sexually mature in 3-4 years and do not return to the pond except to breed.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the Santa Cruz long-toed salamander could preclude emergence prior to pond drying, resulting in larval mortalities.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the Santa Cruz long-toed salamander’s fertility rates substantially. The salamander’s fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). Santa Cruz long-toed salamanders are known from 3 metapopulations, each comprised of one or more subpopulations, located in Santa Cruz and Monterey Counties. Habitat destruction associated with cyanide at CCC levels may result in a reduction of available breeding sites and the loss of subpopulations that could lead to a reduction in population size and range and an increased vulnerability to catastrophic events that may adversely affect breeding and survival. Pollution, siltation, and the degradation of water quality in breeding ponds resulting from nearby development and agriculture was cited as one of the primary threats to the Santa Cruz long-toed salamander at the time of listing and is continues to be a threat to species survival

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Santa Cruz long-toed salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. The Santa Cruz long-toed salamander has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Santa Cruz long-toed salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, Santa Cruz long-toed salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Santa Cruz long-toed salamander.

### *Plethodontidae*

### SAN MARCOS SALAMANDER
*Eurycea nana*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that San Marcos salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, San Marcos salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The San Marcos salamander is aquatic throughout its life cycle. Information on the reproduction of these salamanders is limited. It is found in rocky spring openings and rocky areas downstream from the dams at Spring Lake in San Marcos, TX, as well as in shallow spring areas on the northernmost portion of Spring Lake on a limestone shelf. Six essential elements are required for the San Marcos salamander: thermally constant water, flowing water, clean and clear water, sand, gravel, and rock substrates with little mud or detritus, vegetation for cover, and finally an adequate food supply. Males and females are sexually mature at 19 to 23.5 and 21 mm, respectively. No eggs have been found in nature, but the presence of gravid females and small larvae throughout the year suggests year round breeding. Artificial habitat studies have indicated that the average clutch size is 20, and that eggs are laid in standing pools with thick vegetation. Larvae emerge from the jelly covered eggs after 24 days.

The recovery plan lists a number of major threats to the San Marcos salamander, including decreases in water quantity and quality (including dissolved ions, trace elements, pH, nutrients, dissolved oxygen, and organic contaminants). As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and larval survival. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because the San Marcos salamander is aquatic throughout its entire life cycle, adults could also be at risk of increased adverse and compounded affects due to continued exposure to cyanide at CCC.

Little is known about the salamander's reproduction or longevity in the wild. We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the San Marcos Salamander's fertility rates substantially. The salamander's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

The San Marcos Salamander consists of a single population that is found only in Spring Lake and downstream in the San Marcos River below Spring Lake for 150 m. The San Marcos salamander population is estimated to exceed 53,000 individuals. Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the San Marcos salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected San Marcos salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, San Marcos salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the San Marcos salamander.

**Critical Habitat:** Critical habitat has been established for the San Marcos salamander in Hays County, Texas. It includes Spring Lake and its outflow and the San Marcos River downstream for 50 meters from the Spring Lake Dam. Primary constituent elements were not identified in the final rule designating critical habitat, but in order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of San Marcos salamanders and cause salamanders to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of San Marcos salamanders and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

Formal Draft Biological Opinion.

## BARTON SPRINGS SALAMANDER
*Eurycea sosorum*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that Barton Springs salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, Barton Springs salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The Barton Springs salamander has been found only at the four spring outlets that make up Barton Springs within the City of Austin’s Zilker Park in Travis County, Texas. The salamander requires stable aquatic environmental conditions, including perennially flowing spring water which is clear, clean, near neutral, varies very little in temperature (annual average must be 21 to 22 degrees Celsius) and proper flowing conditions to maintain dissolved oxygen content. The salamander also prefers clean, loose gravel substrates. The Barton Springs salamanders retain their larval gills throughout their lives, becoming mature and reproducing underwater. Known longevity for Barton Springs salamanders in captivity is at least 10 Years. Gravid females, eggs, and larvae are typically found throughout the year in the Barton Springs, which indicates that the salamander can reproduce year-round. Captive salamanders indicate that females are sexually mature at 11 to 17 months. During courtship the male deposits a spermatophore, which becomes attached to a plant, rock, or other substrate. Females can store the spermatophore in a specialized portion of the cloaca, known as the spermatheca for a month or longer. Females of some salamander species may store spermatophores for up to 2.5 years before ovulation and fertilization occur. In most salamanders, fertilization is internal and occurs during egg-laying whereby sperm are released onto eggs as they pass through the female’s cloaca. Clutch sizes range from 5 to 39 eggs with an average of 22 eggs. Hatching of eggs in captivity occurred within 16 to 39 days after the eggs were laid, and the first three months following hatching were a critical period for juvenile survival.

Both the listing notice and the recovery plan for the Barton Springs salamander cite diminished water quality as a critical threat to the species. Analysis has shown that the water quality at Barton Springs has decreased. Dissolved oxygen has decreased (16 percent over 25 years) and conductivity (which has shown levels nearing the rate at which 100% mortality can be expected in 24 hours), sulfates, turbidity, nitrate-nitrogen, and total organic carbon have all increased. While data is limited concerning the Barton Springs salamander's vulnerability to contaminants, its semipermeable skin and reproductive processes suggest that it may be similar to other amphibians. As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and larval survival. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because the Barton Springs salamander is aquatic throughout its entire life cycle, adults could also be at risk of increased adverse and compounded affects due to continued exposure to cyanide at CCC.

Little is known about the salamander's reproduction or longevity in the wild. We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the Barton Springs Salamander's fertility rates substantially. The salamander's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

The Barton Springs Salamander consists of a single population that is found only within Barton Springs in Austin, TX. Total population estimates for the entire species are difficult because of the challenge posed by population surveys. Previous surveys would indicate a number in the hundreds at most, especially as some surveys have found no salamanders even in years with sufficient water flow. Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation). .

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Barton Springs salamander's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Barton Springs salamander population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, Barton Springs salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Barton Springs salamander.

## TEXAS BLIND SALAMANDER
*Typhlomolge rathbuni*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that Texas blind salamanders exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, Texas blind salamanders exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of larval salamanders.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The Texas blind salamander lives only underground in the Edwards Aquifer in Hays County, Texas. It is neotenic (non-transforming) and aquatic throughout its life. It lives in water filled caverns in the aquifer, and is well adapted to the environment, but is believed to be sensitive to temperature changes, as the aquifer has a near constant temperature of 21 degrees Celsius. It is likely that the Texas blind salamander is sexually active year round, which results from the very little seasonal change in the conditions of the aquifer. Gravid females have been found each month of the year. This species reproduced in captivity at the Cincinnati Zoo. In two months, three spawning events occurred. Clutch sizes ranged from 8 to 21 eggs. The unpigmented eggs were attached in ones, twos, and threes to pieces of gravel. Temperatures of close to 21 degrees Celsius are required for proper egg development.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and larval survival. Larval salamanders that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Reduced growth rates could also delay reproductive maturity and productivity. Because the Texas blind salamander is aquatic throughout its entire life cycle, adults could also be at risk of increased adverse and compounded affects due to continued exposure to cyanide at CCC.

The Texas blind salamander has been found only in Hays County, Texas and the total distribution of the species may be as small as 10 square kilometers. Population estimates have not been established. Little is known about the salamander’s reproduction or longevity in the wild. We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the Texas blind salamander’s fertility rates substantially. The salamander’s fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Texas blind salamander’s reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of larval salamanders. Salamanders may also experience effects on growth, locomotion, condition, and development. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the salamander’s life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Texas blind salamander population’s decline could stabilize at a reduced absolute population number or could continue to decline until it’s extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, Texas blind salamanders are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Texas blind salamander.

### *Bufonidae*

### WYOMING TOAD
*Bufo baxteri*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that Wyoming toads exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, Wyoming toads exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of tadpoles.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The life history of the Wyoming can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Wyoming toads inhabit floodplains, ponds, and the margins of small seepage lakes in the shortgrass communities of the Laramie Basin. The toad is known to exist in an area between 10 and 20 miles west of Laramie, Wyoming. Sightings since 1987 have come only from a 2 square mile area around Mortenson Lake and its associated meadows. Breeding occurs in shallow water typically less than six inches deep. Vegetated margins and bays of lakes, ponds, and irrigated meadows are preferred breeding areas. Breeding sites are often dry by late summer. Adult toads appear at breeding sites in May after daytime temperatures reach 70 degrees Fahrenheit. Males appear first and attract females with their calls. Breeding congregations are not large, usually consisting of half a dozen males and a few females gathering at a pond or lake margin. Breeding takes place from mid-May to mid-June depending upon weather conditions in any given year. Eggs are deposited in gelatinous strings containing 2,000 to 5,000 eggs and strands are often intertwined among vegetation. Eggs hatched in less than 1 week in water temperatures ranging between 77 degrees Fahrenheit during the day and 50 degrees at night. Tadpoles transformed into toadlets by 4 to 6 weeks following egg deposition.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Tadpoles that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the Wyoming toad could delay reproductive maturity and productivity.

No long-term studies have been performed to determine the maximum longevity of Wyoming toads in the wild. Corn (1993a) observed that few adult toads lived >2 yr at Mortenson Lake, but that was in a population afflicted with chytrid fungus. In captivity, one female toad with an estimated birth date in 1989 lived in captivity from 1994 until its death in 1997 (Callaway, 1998) and produced large numbers of healthy young from 1994 - 96. Little else is known about the toad's reproduction or longevity in the wild. We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the Wyoming toad's fertility rates substantially. The toad's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Wyoming toad's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of tadpoles. Toads may also experience effects on growth, locomotion, condition, and development. The Wyoming toad has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the toad's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Wyoming toad population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, Wyoming toads are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Wyoming toad.

## ARROYO TOAD
*Bufo californicus*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that arroyo toads exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, arroyo toads exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of tadpoles.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The arroyo toad occurs principally along coastal drainages, but it has been recorded at several locations on the desert slopes of the of the Transverse and Peninsular Mountain ranges south of the Santa Clara River, Los Angeles County, California. The life history of the Wyoming toad can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. The arroyo toad requires shallow, slow-moving stream habitats, and riparian habitats that are disturbed naturally on a regular basis, primarily by flooding. In the northern portion of their range, arroyo toads are found in foothill canyons and intermountain valleys where medium- to large-sized rivers are bordered closely by low hills, riverbed gradients are low, and the surface stream flows frequently pool or are intermittent for at least a few months of the year. In southern California (central portion of the arroyo toad's range), they also occur on the coastal plain and on a few desert slopes. For breeding, adult arroyo toads use open sites such as overflow pools, and old flood channels which are less than 30 cm deep with clear water. Breeding sites usually have flow rates less than 5 cm per second. The breeding period lasts from late January or February to early July, although weather can extend the period. If conditions are unsuitable, females may not obtain sufficient resources for egg production and will forgo breeding during that year. Anywhere from 2000-10,000 eggs are laid in two parallel gelatinous strings on substrates of sand, gravel, cobble, or mud generally located away from vegetation in the shallow margins of the pool. Embryos usually hatch in 4 to 6 days at water temperatures of 12 to 16 degrees Celsius (54 to 59 degrees Fahrenheit). The larval period for arroyo toads lasts about 65 to 85 days, depending on water Temperatures. Newly metamorphosed juveniles remain on sparsely vegetated sand and gravel bars bordering the natal pool for 3–5 wk (Sweet, 1992). Male arroyo toads can reach sexual maturity in 1 year, if conditions are favorable, but females require 2 or 3 years. Mark–recapture studies suggest that few arroyo toads survive into their fifth year, and that these are predominantly females (Sweet, 1993). In the absence of American bullfrogs, adult arroyo toads have a high survivorship during the active season, but suffer 55–80% mortality as they overwinter (Sweet, 1993).

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Tadpoles that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the arroyo toad could delay reproductive maturity and productivity.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the arroyo toad's fertility rates substantially. The toad's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the arroyo toad's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of tadpoles. Toads may also experience effects on growth, locomotion, condition, and development. The arroyo toad has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the toad's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected arroyo toad population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, arroyo toads are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the arroyo toad.

**Critical Habitat:** Critical habitat consists of 11,695 acres in five counties in California: Riverside, San Bernardino, Los Angeles, Ventura, Santa Barbara. The physical and biological features of critical habitat essential to the conservation of the arroyo toad include rivers or streams with hydrologic regimes that supply water to provide space, food, and cover needed to sustain eggs, tadpoles, metamorphosing juveniles, and adult breeding toads. In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of arroyo toads and cause toads to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of arroyo toads and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## HOUSTON TOAD
*Bufo houstonensis*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that Houston toads exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, Houston toads exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of tadpoles.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Houston toads are associated with forest ecosystems and sandy soils. The life history of the Houston toad can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Adult Houston toads respond to cold or summer heat by burrowing into moist sand or hiding under rocks, leaf litter, logs, or in abandoned animal burrows. Breeding occurs in ephemeral, rain-fed pools, flooded fields, and permanent ponds from late January to June with a peak from February to March. Reported egg-laying dates range from February 18 through June 26 and clutch sizes range from 512 to 6,199. Depending on environmental conditions, eggs may hatch within a week and tadpoles develop into toadlets within 40-80 days. Mortality rates are high, and only 1% of eggs laid are believed to survive to adulthood. Captive raised males are sexually mature at 1 year and females at 1 to 2 years.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Tadpoles that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the Houston toad could delay reproductive maturity and productivity. For tadpoles developing in ephemeral ponds, reduced growth rates resulting in a prolonged time to metamorphosis could preclude full development prior to pond drying, resulting in larval mortalities.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the Houston toad's fertility rates substantially. The toad's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Houston toad's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of tadpoles. Toads may also experience effects on growth, locomotion, condition, and development. The Houston toad has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the toad's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Houston toad population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, Houston toads are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Houston toad.

**Critical Habitat:** Critical habitat has been established in Bastrop, Burleson and Harris County, Texas. Although not described when critical habitat was designated, physical and biological features essential to the conservation of the Houston toad include seasonally-flooded breeding ponds, deep sandy soils, and forest or woodlands. In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of Houston toads and cause toads to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of Houston toads and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## *Eleutherodactylidae*

### GUAJON
*Eleutherodactylus cooki*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that guajons exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, guajons exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of tadpoles.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The guajon is associated with the granitic rocks found in the Cuchilla de Panduras mountain range in southeastern Puerto Rico where it inhabits caves formed by large boulders of granite rock known as “guajonales,” but can also be found in associated streams with patches of rock without caves systems. In streams, the guajón has been found only in patches of rock in the streambed. The streams can be perennial, or ephemeral formed during heavy rain and are surrounded by secondary forest. Rocks in the streambed form crevices and grottoes. The guajón deposits eggs on humid boulders within grottoes and on flat surfaces. Eggs are guarded by males. The mean clutch size of the guajón is 17.35 eggs, the developmental time of eggs is 20 to 29 days, and parental care contributes to hatching success. Hatching success of this species is 85 percent, with hatchlings remaining together as a group in the nest for several days before dispersing.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Tadpoles that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the guajon could delay reproductive maturity and productivity. For tadpoles developing in ephemeral ponds, reduced growth rates resulting in a prolonged time to metamorphosis could preclude full development prior to pond drying, resulting in larval mortalities.

Little else is known about the frog's reproduction or longevity in the wild. We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the guajon's fertility rates substantially. The frog's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the guajon's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of tadpoles. Frogs may also experience effects on growth, locomotion, condition, and development. The guajon has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the frog's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected guajon population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, guajons are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the guajon.

**Critical Habitat:** Critical habitat units were established for Humacao, Las Piedras, Maunabo, Patillas, and Yabucoa, Puerto Rico. The physical and biological features of critical habitat essential to the conservation of the guajon includes plutonic, granitic, or sedimentary rocks/boulders that form caves, crevices, and grottoes (interstitial spaces) in a streambed, and that are in proximity, or connected, to a permanent, ephemeral, or subterranean clear-water stream or water source. The interstitial spaces between or underneath rocks provide microenvironments characterized by generally higher humidity and cooler temperatures than outside the rock formations. In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of guajon and cause these frogs to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of California red-legged frogs and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

### *Ranidae*

### CALIFORNIA RED-LEGGED FROG
*Rana aurora draytonii*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that California red-legged frogs exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, California red-legged frogs exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of tadpoles.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.
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California red-legged frogs have been documented in 46 counties in California, but now remain in only 238 streams or drainages in 31 counties. The life history of the California red-legged frog can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily aquatic but use a variety of aquatic, riparian, and upland habitats. The California red-legged frog's larvae, tadpoles, and metamorphs can be found in streams, deep pools, backwaters, creeks, ponds, marshes, sag ponds, dune ponds, and lagoons. Breeding adults are commonly associated with water deeper than 0.7 m (2 feet) which is slow moving and choked by shrubby riparian or emergent vegetation. California red-legged frogs breed from November to April. Males appear at breeding sites 2-4 weeks before females. Once a pair of frogs has moved into the breeding position, they move to where the eggs are laid and fertilized. The 2,000 to 5,000 eggs float near the surface, attached to emergent vegetation, roots, or twigs. Eggs hatch within 6 to 14 days depending on water temperatures and require approximately 20 days to develop into tadpoles. Tadpoles in turn require anywhere between 11 to 20 weeks to develop into terrestrial frogs. At some locations, larvae may overwinter before metamorphosing. Water bodies suitable for tadpole rearing must remain watered at least until the tadpoles metamorphose into adults, typically between July and September. Adult California red-legged frogs can survive in moist upland areas after breeding habitat has dried, and can live several years to make new breeding attempts. Therefore, aquatic breeding habitat need not be available every year, but it must be available often enough and for appropriate hydroperiods to maintain a California red-legged frog population during most years.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Tadpoles that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the California red-legged frog could delay reproductive maturity and productivity. For tadpoles developing in ephemeral ponds, reduced growth rates resulting in a prolonged time to metamorphosis could preclude full development prior to pond drying, resulting in larval mortalities.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the California red-legged frog's fertility rates substantially. The frog's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the California red-legged frog's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of tadpoles. Frogs may also experience effects on growth, locomotion, condition, and development. The California red-legged frog has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the frog's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected California red-legged frog population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, California red-legged frogs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the California red-legged frog.

**Critical Habitat:** California red-legged frog critical habitat has been established in 34 locations in Alameda, Butte, Contra Costa, El Dorado, Kern, Los Angeles, Marin, Merced, Monterey, Napa, Nevada, San Benito, San Luis Obispo, San Mateo, Santa Barbara, Santa Clara, Santa Cruz, Solano, Ventura and Yuba Counties, California, and includes aquatic breeding and non-breeding habitats that provide the physical and biological features of critical habitat essential to the conservation of the California red-legged frog. Aquatic breeding habitat is essential for providing space, food, and cover necessary to sustain the early life history stages of larval and juvenile California red-legged frogs. It consists of low-gradient fresh water bodies, including natural and manmade (e.g., stock) ponds, backwaters within streams and creeks, marshes, lagoons, and dune ponds. It does not include deep lacustrine water habitat (e.g., deep lakes and reservoirs 50 ac (20 ha) or larger in size). To be considered essential breeding habitat, the aquatic feature must have the capability to hold water for a minimum of 20 weeks in all but the driest of years.

Nonbreeding aquatic habitat consists of those aquatic elements identified above, and also includes, but is not limited to, other wetland habitats such as intermittent creeks, seeps, and springs. California red-legged frogs can use large cracks in the bottom of dried ponds as refugia to maintain moisture and avoid heat and solar exposure. Without these non-breeding aquatic features, California red-legged frogs would not be able to survive drought periods, or be able to disperse to other breeding habitat.

In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of California red-legged frogs and cause frogs to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of California red-legged frogs and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## CHIRICAHUA LEOPARD FROG
*Rana chiricahuensis*

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that Chiricahua leopard frogs exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, Chiricahua leopard frogs exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of tadpoles.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

The range of the Chiricahua leopard frog is split into two disjunct parts - northern populations along the Mogollon Rim in Arizona east into the mountains of west-central New Mexico, and southern populations in southeastern Arizona, southwestern New Mexico, and Mexico The life history of the Chiricahua leopard frog can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily aquatic. It is an inhabitant of montane and river valley cienegas, springs, pools, cattle tanks, lakes, reservoirs, streams, and rivers. It is a habitat generalist that historically was found in a variety of aquatic habitat types, but is now limited to the comparatively few aquatic systems that support few or no non-native predators (e.g. American bullfrogs, fishes, and crayfishes). The species also requires permanent or semi-permanent pools for breeding, water characterized by low levels of contaminants and moderate pH, and may be excluded or exhibit periodic die-offs where a pathogenic chytridiomycete fungus is present. Egg masses of Chiricahua leopard frogs have been reported in all months except January, November, and December, but reports of oviposition in June are uncommon. Hatching time of egg masses in the wild has not been studied in detail. Eggs of the Ramsey Canyon leopard frog hatch in approximately 14 days depending on temperature, and hatching time may be as short as eight days in geothermally influenced springs. Tadpoles metamorphose in three to nine months (Jennings 1988, 1990), and may overwinter.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Tadpoles that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the Chiricahua leopard frog could delay reproductive maturity and productivity. For tadpoles developing in ephemeral ponds, reduced growth rates resulting in a prolonged time to metamorphosis could preclude full development prior to pond drying, resulting in larval mortalities.

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Little is known about age and size at reproductive maturity or the longevity of the Chiricahua leopard frog. We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the Chiricahua leopard frog's fertility rates substantially. The frog's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the Chiricahua leopard frog's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of tadpoles. Frogs may also experience effects on growth, locomotion, condition, and development. The Chiricahua leopard frog has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily aquatic. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the frog's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected Chiricahua leopard frog population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, Chiricahua leopard frogs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the Chiricahua leopard frog.

## MOUNTAIN YELLOW-LEGGED FROG

*Rana muscosa*

Southern California Population

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Based on multiple lines of evidence, including the relative sensitivity of amphibians to other pollutants, their relative sensitivity to rainbow trout and the sensitivity of rainbow trout to cyanide we conclude that mountain yellow-legged frogs exposed to cyanide at the CCC could experience substantial reductions in recruitment due to reductions in fecundity, hatching success, and larval development and survival. Compared to control populations, we estimate rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than 52% (Appendix E). We estimate that rainbow trout exposed to cyanide at the CCC could experience a substantial reduction in survival of young fish through the first year and that reduction could be as much as, but is not likely to be greater than 61% (Appendix E). Based on the relative sensitivity of amphibians as compared to rainbow trout we estimate the effects of cyanide at the CCC for amphibians could be of similar or greater magnitude than effects estimated for rainbow trout. As such, mountain yellow-legged frogs exposed to cyanide at the CCC could experience a substantial reduced egg hatching and decreased survival to metamorphosis of tadpoles.

Although no comparable data are available for amphibians, cyanide may also adversely effect growth, locomotion, condition, and development as described above for fish in the Effects Overview. The effects of chronic toxicity could also be influenced by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column.

Currently the mountain yellow-legged frog is known from only seven locations in southern California in portions of the San Gabriel, San Bernardino, and San Jacinto Mountains. The life history of the mountain yellow-legged frog can be characterized as a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily aquatic. Water depth, persistence, and configuration (*i.e.* gently sloping shorelines and margins) appear to be important for mountain yellow-legged frogs, allowing for shelter from predators along shores or in deeper waters, and habitat for breeding, foraging, egg-laying, thermoregulation (to regulate the body temperature through behavior), and overwintering. Breeding activity typically begins in April at lower elevations, to June or July at upper elevations and continues for approximately a month. Egg masses vary in size from as few as 15 eggs to 350 eggs per mass, which is considered low, relative to a range of several hundred to several thousand for other true frogs. Egg masses are normally deposited in shallow waters where they may be attached to rocks, gravel, vegetation, or similar substrates. As larvae develop, they tend to gravitate towards warmer waters to elevate body temperatures which may facilitate larval and metamorphic development by allowing for a higher metabolic rate. Even with this behavior, larvae apparently must overwinter at least two times for 6 to 9 month intervals before attaining metamorphosis because the active season is short and the aquatic habitat maintains warm temperatures for only brief intervals. Time to develop from fertilization to metamorphosis appears to be variable, ranging up to 3.5 years, with reproductive maturity reached from 3 to 4 years following metamorphosis. Little is known about adult longevity, but the species is presumed to be long-lived due to adult survivorship.

As noted above, exposure to cyanide at the CCC could result in a substantial reduction in egg hatching and survival to metamorphosis. Tadpoles that do survive could experience reduced growth rates and reduced swimming abilities which would increase their vulnerability to a host of potential stressors including, temperature, flow, predation, and inter- and intraspecific competition for food and cover. Because amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage, reduced growth rates of the mountain yellow-legged frog could delay reproductive maturity and productivity.

We anticipate that the estimated reductions in hatched eggs and larval survival would reduce the mountain yellow-legged frog's fertility rates substantially. The frog's fertility rates would be reduced based upon (a) reductions in numbers of eggs hatched, (b) reductions in larvae surviving through the first year and (c) the additive effect of reduced fecundity and survival. The combined effect could be less than additive if the eggs that hatch in the presence of cyanide represent individuals that are less sensitive. These reductions in reproduction could be exacerbated further if reduced growth rated delay maturity or lengthen the time between mating.

Reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the CCC. Changes in the fitness of the individuals in a population will affect the population as a whole. Effects to populations can be measured in terms of changes in population growth rates and changes in risk of extinction (or extirpation).

In summary, exposure to cyanide concentrations at the chronic criterion could substantially reduce the mountain yellow-legged frog's reproduction by reducing the number of eggs laid by females, the hatchability of laid eggs, and a reduction of survivorship of tadpoles. Frogs may also experience effects on growth, locomotion, condition, and development. The mountain yellow-legged frog has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily terrestrial. Although exposure to cyanide concentrations at the chronic criterion may be isolated to the aquatic phases of the frog's life cycle (i.e. breeding activities and egg and larval development), the long-term effects of exposure could result in reduced adult growth, survival and fitness. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed. An effected mountain yellow-legged frog population's decline could stabilize at a reduced absolute population number or could continue to decline until it's extirpated. For extant populations supported by a single breeding pond, reproductive failure at that site could result in the extirpation of the population and even those populations with multiple breeding sites may have difficulty recolonizing after local extirpations due to physiological, morphological, and behavioral constraints. Based upon the magnitude of effect we anticipate could occur, we conclude that ultimately, mountain yellow-legged frogs are likely to become extirpated from waters where they are exposed to cyanide toxicity at the chronic criterion concentration. Continued approval of the chronic criterion could reduce the reproduction, numbers, and distribution of the mountain yellow-legged frog.

**Critical Habitat:** The physical and biological features of critical habitat essential to the conservation of the mountain yellow-legged frog include (1) Water source(s) found between 1,214 to 7,546 feet (370 to 2,300 meter) in elevation that are permanent. Water sources include, but are not limited to, streams, rivers, perennial creeks (or permanent plunge pools within intermittent creeks), pools (i.e., a body of impounded water that is contained above a natural dam) and other forms of aquatic habitat. Aquatic habitats that are used by mountain yellow-legged frog for breeding purposes must maintain water during the entire tadpole growth phase, which can last for up to 2 years. During periods of drought, or less than average rainfall, these breeding sites may not hold water long enough for individuals to complete metamorphosis, but they would still be considered essential breeding habitat in wetter years. Further, the aquatic includes: a.) Bank and pool substrates consisting of varying percentages of soil or silt, sand, gravel cobble, rock, and boulders; b.) Open gravel banks and rocks projecting above or just beneath the surface of the water for sunning posts; c.) Aquatic refugia, including pools with bank overhangs, downfall logs or branches, and/or rocks to provide cover from predators; and d.) Streams or stream reaches between known occupied sites that can function as corridors for movement between aquatic habitats used as breeding and/or foraging

In order for these aquatic areas to provide the necessary conditions for successful breeding and survival they must include water of sufficient quality to allow the species to carry out normal feeding, movement, sheltering, and reproductive behaviors and to experience individual and population growth. Approval of the CCC in State and Tribal water quality standards would authorize States and Tribes to continue to manage cyanide in waters to the criteria concentrations. This approval could adversely affect the quality of water to the degree that it would impair individual reproduction and survival of California red-legged frogs and cause frogs to experience adverse effects to growth, locomotion, condition, and development. Based on data for the rainbow trout, our surrogate species for assessing impacts to amphibians, we estimate the reduction in number of hatched eggs could be as much as, but is not likely to be greater than 52% and the reduction in larval survival to metamorphosis could be as much as, but is not likely to be greater than 61%. These effects are estimated to of a magnitude great enough to reduce numbers of California red-legged frogs and result in the extirpation of local populations. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth is likely to be impacted and could result in extirpation from critical habitat containing cyanide at the CCC. Impacts to water quality resulting from management of cyanide to the CCC would diminish the ability of critical habitat to provide for conservation of the species.

## 8.0 CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, tribal, local or private actions that are reasonably certain to occur in the action area considered in this biological opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

As noted in Section 5.0 above, the action area for this consultation consists of all waters of the United States, including territorial seas, which extend seaward a distance of three miles from the coast (CWA section 502), where federally listed endangered, threatened, and proposed species reside. The action area includes such waters within and surrounding Indian country, the 50 States, and all U.S. territories. Given the size of the action area, it is not practical to specifically evaluate cumulative effects in this biological opinion.

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In general, the threatened and endangered aquatic species and designated critical habitats considered in this biological opinion are likely to be adversely affected by non-federal activities that affect the quantity, quality, and hydrographic patterns of water, waterways, and habitats important to these species and critical habitats. These activities could include changes in land and water use and management patterns in ways that increase erosion and sedimentation, increase introduction of pollutants into waterways, and result in introductions and spread of non-native invasive species that directly or indirectly affect listed species and critical habitats. These species and their critical habitats could also be affected by illegal harvest. States or private entities may also engage in activities to restore, enhance, and improve water quality and quantity and restore more natural hydrographic patterns that benefit listed species and their habitats. All of the species and critical habitats considered in this document are likely to be subject to these types of activities in the future to varying extents. The final listing and designation rules, recovery plans, and 5-year reviews for these species and critical habitats are good sources of information, in part, on the threats and benefits associated with these types of activities. These documents are cited in Appendix A.

## 9.0 CONCLUSION

After reviewing the current status of the following listed species, the environmental baseline for the action area, the effects of EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide, and cumulative effects, it is the Service's biological opinion that the action, as proposed, is likely to jeopardize the continued existence of the following species:

Gulf sturgeon, Kootenai River white sturgeon, Pallid sturgeon, Alabama sturgeon, Ozark cavefish, Alabama cavefish, Waccamaw silverside, Modoc sucker, Santa Ana sucker, Warner sucker, Shortnose sucker, Cui-ui, June sucker, Lost River sucker, Razorback sucker, Pygmy sculpin, Blue shiner, Beautiful shiner, Devils River minnow, Spotfin chub, Slender chub, Mojave tui chub, Owens tui chub, Borax Lake chub, Humpback chub, Sonora chub, Bonytail chub, Gila chub, Yaqui chub, Pahranagat roundtail chub, Virgin River Chub, Rio Grande silvery minnow, Big Spring spinedace, Little Colorado spinedace, Spikedace, Moapa dace, Palezone shiner, Cahaba shiner, Arkansas River shiner, Cape Fear shiner, Pecos bluntnose shiner, Topeka shiner, Oregon chub, Blackside dace, Woundfin, Colorado pikeminnow (=squawfish), Ash Meadows speckled dace, Kendall Warm Springs dace, Loach minnow, Unarmored threespine stickleback, Tidewater goby, White River springfish, Hiko White River springfish, Railroad Valley springfish, Delta smelt, Slackwater darter, Vermilion darter, Relict darter, Etowah darter, Fountain darter, Niangua darter**,** Watercress darter, Okaloosa darter, Duskytail darter, Bayou darter, Cherokee darter, Maryland darter, Bluemask darter, Boulder darter, Amber darter, Goldline darter, Conasauga logperch, Leopard darter, Roanoke logperch, Snail darter, Big Bend gambusia, San Marcos gambusia, Clear Creek gambusia, Pecos gambusia, Gila topminnow, Bull trout, Little Kern Golden trout, Apache trout, Lahontan Cutthroat trout, Paiute Cutthroat trout, Greenback Cutthroat Mountain trout, Gila trout, Atlantic salmon, Illinois cave amphipod,

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Noel's amphipod, Cumberland elktoe, Dwarf wedgemussel, Appalachian elktoe, Fat three-ridge, Ouachita rock pocketbook, Birdwing pearlymussel, Fanshell, Dromedary pearlymussel, Chipola slabshell, Tar River spinymussel, Purple bankclimber, Cumberlandian combshell, Oyster mussel, Curtis pearlymussel, Yellow blossom, Tan riffleshell, Upland combshell, Catspaw, White catspaw, Southern acornshell, Southern combshell, Green blossom, Northern riffleshell, Tubercled blossom, Turgid blossom, Shiny pigtoe, Finerayed pigtoe, Cracking pearlymussel, Pink mucket, Fine-lined pocketbook, Higgins eye, Orangenacre mucket, Arkansas fatmucket, Speckled pocketbook, Shinyrayed pocketbook, Alabama lampmussel, Carolina heelsplitter, Scaleshell mussel, Louisiana pearlshell, Alabama moccasinshell, Coosa moccasinshell, Gulf moccasinshell, Ochlockonee moccasinshell, Ring pink, Littlewing pearlymussel, White wartyback pearlymussel, Orangefoot pimpleback, Clubshell, James spinymussel, Black clubshell, Southern clubshell, Dark pigtoe, Southern pigtoe, Cumberland pigtoe, Flat pigtoe, Ovate clubshell, Rough pigtoe, Oval pigtoe, Heavy pigtoe, Fat pocketbook, Alabama heelsplitter, Triangular kidneyshell, Rough rabbitsfoot, Winged mapleleaf, Cumberland monkeyface, Appalachian monkeyface, Stirrupshell, Pale Lilliput, Purple bean, Cumberland bean, reticulated flatwoods salamander, frosted flatwoods salamander California tiger salamander (central California DPS), California tiger salamander (Santa Barbara County DPS), California tiger salamander (Sonoma DPS), Santa Cruz long-toed salamander, Sonora Tiger salamander, San Marcos salamander, Barton Springs salamander, Texas blind salamander, Wyoming toad, arroyo toad, Houston toad, guajon, California red-legged frog, Chiricahua leopard frog, mountain yellow-legged frog

Exposure of the above listed fish species to cyanide at the proposed chronic criterion concentration is likely to substantially reduce their reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and by reducing the survivorship of young fish in their first year. These fish may also experience effects on growth, swimming performance, condition, and development. In addition, Fountain darters, Bull trout, Apache trout, and Lahontan cutthroat trout exposed to cyanide at the acute criterion are likely to experience substantial reductions in survival. Based upon the magnitude of adverse effects caused by the exposure of these listed species to cyanide at the proposed acute and/or chronic criteria concentrations, these fish species are likely to become extirpated from waters where they are exposed to cyanide toxicity at the CMC and/or CCC. Continued approval of the acute and/or chronic criteria at the rangewide scale of these listed species is likely to reduce their reproduction, numbers, and distribution.

Exposure of the Illinois cave amphipod and Noel's amphipod to cyanide at the chronic criterion concentration is likely to result in the loss of individuals, especially in situations when these amphipods are subject to interspecific resource competition or predation. Because both amphipod species exist in populations that are geographically isolated from one another, the ability of amphipods to recolonize perturbed habitats is limited. Thus, cyanide exposure may result in the elimination of a population unit that cannot rebound. The loss of a population unit for either amphipod species would substantially reduce the reproduction, numbers, or distribution of these species.

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For the above mentioned mussel species, exposure of their host fish to cyanide at criterion concentrations is likely to reduce the abundance of fish hosts for glochidia, thereby decreasing the likelihood that glochidia will survive because they will be unable to attach to suitable host. Since attachment of glochidia to a suitable host is a rare and necessary event in the mussel reproductive cycle, reductions in host fish abundance are likely to negatively impact mussel reproductive output and ultimately population numbers. Thus, host fish abundance for these species is anticipated to substantially decline to levels that will reduce reproduction, numbers, or distribution of these mussel species.

After reviewing the current status of critical habitat, the environmental baseline for the action area, the effects of EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide, and cumulative effects, it is the Service's biological opinion that the action, as proposed, is likely to result in the destruction or adverse modification of critical habitat that has been designated for the following species:

Gulf sturgeon, Kootenai River white sturgeon, Alabama sturgeon, Alabama cavefish, Waccamaw silverside, Modoc sucker, Santa Ana sucker, Warner sucker, June sucker, Razorback sucker, Beautiful shiner, Devils River minnow, Spotfin chub, Slender chub, Owens tui chub, Borax Lake chub, Humpback chub, Sonora chub, Bonytail chub, Gila chub, Yaqui chub, Virgin River Chub, Rio Grande silvery minnow, Big Spring spinedace, Little Colorado spinedace, Spikedace, Arkansas River shiner, Cape Fear shiner, Pecos bluntnose shiner, Topeka shiner, Woundfin, Colorado pikeminnow (=squawfish), Loach minnow, Tidewater goby, White River springfish, Hiko White River springfish, Railroad Valley springfish, Delta smelt, Slackwater darter, Fountain darter, Niangua darter**,** Maryland darter, Amber darter, Conasauga logperch, Leopard darter, Snail darter, San Marcos gambusia, Bull trout, Little Kern Golden trout, Cumberland elktoe, Appalachian elktoe, Fat three-ridge, Chipola slabshell, Purple bankclimber, Cumberlandian combshell, Oyster mussel, Upland combshell, Southern acornshell, Fine-lined pocketbook, Orangenacre mucket, Shinyrayed pocketbook, Carolina heelsplitter, Alabama moccasinshell, Coosa moccasinshell, Gulf moccasinshell, Ochlockonee moccasinshell, Southern clubshell, Dark pigtoe, Southern pigtoe, Ovate clubshell, Oval pigtoe, Triangular kidneyshell, Rough rabbitsfoot, Purple bean, reticulated flatwoods salamander, frosted flatwoods salamander, California tiger salamander (central California DPS), California tiger salamander (Santa Barbara County DPS), San Marcos salamander, arroyo toad, Houston toad, guajon, California red-legged frog, mountain yellow-legged frog

The physical and biological features of critical habitat essential to the conservation of these listed species include water of sufficient quality for species to carry out normal feeding, movement, sheltering, and reproductive behaviors, and to experience individual and population growth. Approval of the CCC in state water quality standards would authorize states to manage cyanide in waters to these levels. This approval and cyanide in waters to these levels is likely to adversely affect the quality of water to the degree that it would impair individual reproduction and survival of these listed fish species as well as fish species that are hosts for the above mentioned listed mussels. Approval of the CCC

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and cyanide in waters to these levels is likely to adversely affect the quality of water to the degree that it would impair normal population growth and likely cause the extirpation of these listed fish from their critical habitat containing cyanide at the CCC. In addition, the majority of fish hosts identified for the above mentioned mussel species are likely to exhibit population declines at cyanide criteria concentrations. For these reasons, impacts to water quality resulting from cyanide in waters to the level of the CCC would diminish the intended conservation function of critical habitat for these listed fishes and mussels.

## 10.0 REASONABLE AND PRUDENT ALTERNATIVES

The regulations (50 CFR 402.02) implementing section 7 of the ESA define reasonable and prudent alternatives (RPAs) as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) would, the Service believes, avoid the likelihood of jeopardizing the continued existence of listed species or the destruction or adverse modification of critical habitat.

The Service has developed the following RPAs to the EPA's proposed action:

1. By December 1, 2012, EPA shall, subject to Service approval, review the geographic ranges of the listed species and designated critical habitats addressed in this biological opinion and insure that the water bodies or water body segments within those ranges include: a) a designated use for which aquatic life criteria apply; b) aquatic life cyanide criteria at least as stringent as described below; and, c) appropriate language within any EPA-approved general policies requiring State coordination with local Service field offices on implementation of general policies. The Service recommends the EPA consider the following criteria for water bodies and water body segments:

   Fish: The acute and chronic cyanide criteria for protection of listed fish species are shown in Table 21. These values are based on the acute and chronic Assessment Effects Concentrations ($EC_A$) which represent *the highest* concentration of cyanide where the effects on listed species are expected to be insignificant (see Appendix B for details).

**Table 21. Freshwater acute and chronic cyanide criteria for the protection of listed fish species (NC, no change).**

| Listed Species | | Order/Family | Surrogate Taxa | Recommended Acute Criteria [Acute $EC_A$ (ug CN/L)] | Recommended Chronic Criteria [Chronic $EC_A$ (ug CN/L)] |
|---|---|---|---|---|---|
| Gulf sturgeon | *Acipenser oxyrinchus desotoi* | Acipenseriformes Acipenseridae (sturgeon) | Actinopterygii (class) | NC | 2.86 |
| Kootenai River white sturgeon | *Acipenser transmontanus* | | | | |
| Pallid sturgeon | *Scaphirhynchus albus* | | | | |

| | | | | | |
|---|---|---|---|---|---|
| Alabama sturgeon | *Scaphirhynchus suttkusi* | | | | |
| Waccamaw silverside | *Menidia extensa* | Atheriniformes Atherinopsidae | | | |
| Modoc sucker | *Catostomus micorps* | Cypriniformes Catosdomidae (suckers) | Cypriniformes (order) | NC | 3.64 |
| Santa Anna sucker | *Catostomus santaanae* | | | | |
| Warner sucker | *Catostomus warnerensis* | | | | |
| Shortnose sucker | *Chasmistes brevirostris* | | | | |
| Cui ui | *Chasmistes cujus* | | | | |
| June sucker | *Chasmistes liorus* | | | | |
| Lost River sucker | *Deltistes luxatus* | | | | |
| Razorback sucker | *Xyrauchen texanus* | | *Xyrauchen texanus* | NC | 3.61 |
| Spotfin chub | *Cyprinella monacha* | Cypriniformes Cyprinidae | *Cyprinella monacha* | NC | 1.58 |
| Blue shiner | *Cyprinella caerulea* | | Cyprinidae (family) | NC | 4.38 |
| Beautiful shiner | *Cyprinella formosa* | | | | |
| Devils River minnow | *Dionda diaboli* | | | | |
| Slender chub | *Erimystax cahni* | | | | |
| Mohave tui chub | *Gila bicolor mohavensis* | | | | |
| Owens tui chub | *Gila bicolor snyderi* | | | | |
| Hutton tui chub | *Gila bicolor* ssp. | | | | |
| Borax Lake chub | *Gila boraxobius* | | | | |
| Humpback chub | *Gila cypha* | | | | |
| Sonora chub | *Gila ditaenia* | | | | |
| Gila chub | *Gila intermedia* | | | | |
| Yaqui chub | *Gila purpurea* | | | | |
| Pahranagat roundtail chub | *Gila robusta jordani* | | | | |
| Virgin River chub | *Gila robusta seminuda* | | | | |
| Rio Grand silvery minnow | *Hybognathus amarus* | | | | |
| Big Spring spinedace | *Lepidomeda mollispinis pratensis* | | | | |
| Little Colorado spinedace | *Lepidomeda vittata* | | | | |
| Spikedace | *Meda fulgida* | | | | |
| Moapa dace | *Moapa coriacea* | | | | |
| Palezone shiner | *Notropis albizonatus* | | | | |
| Cahaba shiner | *Notropis cahabae* | | | | |
| Arkansas River shiner | *Notropis girardi* | | | | |
| Pecos bluntnose shiner | *Notropis simus pecosensis* | | | | |
| Topeka shiner | *Notropis Topeka* | | | | |
| Oregon chub | *Oregonichthys crameri* | | | | |
| Blackside dace | *Phoxinus cumberlandensis* | | | | |
| Woundfn | *Plagopterus agrentissimus* | | | | |
| Ash Meadows speckled dace | *Rhinichthys osculus nevadensis* | | | | |
| Kendall Warm Springs dace | *Rhinichthys osculus thermalis* | | | | |
| Foskett speckled dace | *Rhinichthys osculus* ssp. | | | | |
| Loach minnow | *Tiaroga cobitis* | | | | |
| Bonytail chub | *Gila elegans* | | *Gila elegans* | NC | 2.19 |

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|---|---|---|---|---|---|
| Cape Fear shiner | *Notropis mekistocholas* | | *Notropis mekistocholas* | NC | 2.09 |
| Colorado pikeminnow | *Ptychocheilus lucis* | | *Ptychocheilus lucis* | NC | 1.87 |
| White River springfish | *Crenichthys baileyi baileyi* | Cyprinodontiformes Goodeidae | Actinopterygii (class**)** | NC | 2.86 |
| Hiko White River springfish | *Crenichthys baileyi grandis* | | | | |
| Railroad Valley springfish | *Crenichthys nevadae* | | | | |
| Big Bend gambusia | *Gambusia gaigei* | Cyprinodontiformes Poeciliidae | | | |
| San Marcos gambusia | *Gambusia georgei* | | | | |
| Clear Creek gambusia | *Gambusia heterochir* | | | | |
| Pecos gambusia | *Gambusia nobilis* | | | | |
| Gila topminnow | *Poeciliopsis occidentalis occidentalis* | | | | |
| Yaqui topminnow | *Poeciliopsis occidentalis sonoriensis* | | | | |
| Unarmoned threespine stickleback | *Gasterosteus aculeatus williamsoni* | Gasterosteiformes Gasterosteidae | | | |
| Delta smelt | *Hypomesus transpacificus* | Osmeriformes Osmeridae | | | |
| Tidewater goby | *Eucyclogobius newberryi* | Perciformes Gobiidae | Perciformes (order) | NC | 3.91 |
| Slackwater darter | *Etheostoma boschungi* | Perciformes Percidae | *Etheostoma* (genus) | NC | 1.72 |
| Vermilion darter | *Etheostoma chermocki* | | | | |
| Relict darter | *Etheostoma chienense* | | | | |
| Etowah darter | *Etheostoma etowahae* | | | | |
| Niangua darter | *Etheostoma nianguae* | | | | |
| Watercress darter | *Etheostoma nuchale* | | | | |
| Okaloosa darter | *Etheostoma okaloosae* | | | | |
| Duskytail darter | *Etheostoma percnurum* | | | | |
| Bayou darter | *Etheostoma rubrum* | | | | |
| Cherokee darter | *Etheostoma scotti* | | | | |
| Maryland darter | *Etheostoma sellare* | | | | |
| Bluemask darter | *Etheostoma sp.* | | | | |
| Boulder darter | *Etheostoma wapiti* | | | | |
| Fountain darter | *Etheostoma fonticola* | | *Etheostoma fonticola* (species) | **17.2** | 0.93 |
| Amber darter | *Percina antesella* | | Percidae (family) | NC | 1.82 |
| Goldline darter | *Percina aurolineata* | | | | |
| Conasauga logperch | *Percina jenkinsi* | | | | |
| Leopard darter | *Percina pantherina* | | | | |
| Roanoke logperch | *Percina rex* | | | | |
| Snail darter | *Percina tanasi* | | | | |
| Ozark cavefish | *Amblyopsis rosae* | Percopsiformes Amblyopsidae | Actinopterygii (class**)** | NC | 2.86 |
| Alabama cavefish | *Speoplatyrhinus poulsoni* | | | | |
| Little Kern golden trout | *Oncorhynchus aguabonita whitei* | Salmoniformes Salmonidae | *Oncorhynchus* (genus) | NC | 2.02 |
| Paiute cutthroat trout | *Oncorhynchus clarki seleniris* | | | | |

| | | | | | |
|---|---|---|---|---|---|
| Greenback cutthroat trout | *Oncorhynchus clarki stomias* | | | | |
| Gila trout | *Oncorhynchus gilae* | | | | |
| Apache trout | *Oncorhynchus apache* | | *Oncorhynchus apache* (species) | **14.47** | 0.71 |
| Lahontan cutthroat trout | *Oncorhynchus clarki henshawi* | | *Oncorhynchus clarki henshawi* (species) | **20.00** | 0.98 |
| Atlantic salmon | *Salmo salar* | | *Salmo salar* (species) | NC | 3.87 |
| Bull trout | *Salvelinus confluentus* | | *Salvelinus* (genus) | **13.77** | 0.68 |
| Pygmy sculpin | *Cottus paulus* | Scorpaeniformes Cottidae | Actinopterygii (class) | NC | 2.86 |

Freshwater mussels: The acute and chronic criteria for listed mussels are based on the protection of their host fish. For mussels with obligate host fish, the RPA is based on the acute and chronic Assessment Effects Concentrations ($EC_A$) which represent *the highest* concentration of cyanide where the effects on host fish species are expected to be insignificant (Table 22, see Appendix B for details).

Table 22. Freshwater acute and chronic cyanide criteria for the protection of listed mussels with known obligate host fish species (NC, no change).

| **Listed Species** | **Host Fish** | **Surrogate Taxa** | **Recommended Acute Criteria [Acute $EC_A$ (ug CN/L)]** | **Recommended Chronic Criteria [Chronic $EC_A$ (ug CN/L)]** |
|---|---|---|---|---|
| Fat Pocketbook *Potamilus capax* | Freshwater drum | Perciformes (order) | NC | 2.86 |
| Scaleshell Mussel *Leptodea leptodon* | Freshwater drum | Perciformes (order) | NC | 2.86 |

For mussels with multiple host fish (non-obligates), or for which host fish are unknown, the recommended acute and chronic criteria for listed mussels are based on the protection of fish from the genus *Etheostoma*, family Pericade (Table 23). Percids make up approximately one-third of all known host fish species for listed mussels. For non-obligate listed mussels for which percids have not been identified as hosts, there is a reasonable possibility that these species serve as hosts where they occur. Protection of fish from the genus *Etheostoma* is expected to protect for the majority of species in this family.

For listed mussels either occurring in areas that do not support percids, or are known not to transform on these species, we recommend acute and chronic criteria for listed mussels based on the protection of fish from the class Actinopterygii (Table 23).

Table 23. Freshwater acute and chronic cyanide criteria for the protection of listed mussels with multiple host fish or for which host fish are unknown (NC, no change).

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| Listed Species | Host Fish | Surrogate Taxa | Recommended Acute Criteria [Acute $EC_A$ (ug CN/L)] | Recommended Chronic Criteria [Chronic $EC_A$ (ug CN/L)] |
|---|---|---|---|---|
| Mussels no known obligate host fish | Multiple or unknown | *Etheostoma* (genus) | NC | 1.72 |
| Mussels with non-percid hosts[1] | Non-percid hosts | Actinopterygii (class) | NC | 2.86 |

[1] Habitat is known to not support percid species or mussels species are known to not transform on percids.

Amphipods: The acute and chronic cyanide criteria for protection of listed amphipod species are shown in Table 24. These values are based on the acute and chronic Assessment Effects Concentrations ($EC_A$) which represent the *the highest* concentration of cyanide where the effects on listed species are expected to be insignificant (see Appendix B for details).

Table 24. Freshwater acute and chronic cyanide criteria for the protection of listed amphipod species (NC, no change).

| Listed Species | | Order/Family | Surrogate Taxa | Recommended Acute Criteria [Acute $EC_A$ (ug CN/L)] | Recommended Chronic Criteria [Chronic $EC_A$ (ug CN/L)] |
|---|---|---|---|---|---|
| Illinois cave amphipod | *Gammarus acherondytes* | Amphipoda Cambaridae | *Gammarus* (genus) | NC | 3.33 |
| Noel's Amphipod | *Gammarus desperatus* | | | NC | 3.33 |

Amphibians: EPA shall implement RPA Alternative #2 for Amphibians.

and/or,

2. In place of RPA 1(b), the EPA shall, subject to the Service's approval, develop and implement the research necessary to replace modeled estimates of species sensitivities to cyanide with direct evidence, using listed species or more closely related surrogates, as the basis for defining cyanide criteria to insure an appropriate level of protection is afforded to listed species and critical habitats addressed by this RPA. This RPA shall be implemented for all amphibians addressed in this biological opinion, and is optional for all other taxa. This task shall be completed by December 1, 2012.

Because this biological opinion has found jeopardy and adverse modification, the EPA is required to notify the Service of its final decision on implementation of the reasonable and prudent alternatives.

## 11.0 INCIDENTAL TAKE STATEMENT

The Service has developed the following Incidental Take Statement based on the premise that the RPA will be implemented.

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Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without special exemption. Take is defined as harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harass is defined by the Service as an intentional or negligent act or omission which creates the likelihood of injury to a listed species by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding or sheltering. Harm is defined by the Service to include significant habitat modification or degradation that results in death or injury to listed species by impairing behavioral patterns including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be a prohibited taking under the ESA, provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

The Service anticipates that with implementation of the RPA, incidental take of the listed fish, mussel, amphipod, and amphibian species considered in this biological opinion is not likely to occur from exposure to cyanide at revised criteria concentrations. However, other elements of water quality standards could allow for exceedance of criteria concentrations and may contribute to incidental take. The other elements of water quality standards will be the focus of subsequent tiered consultations on individual State and Tribal water quality standards. Therefore, no incidental take exemptions are provided in this biological opinion.

## 12.0 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. We recommend that EPA implement the following actions:

1. In consultation with the Service, develop a conservation program for threatened and endangered species and, in collaboration with States and Tribes, develop conservation plans that specifically addresses threats to listed species and how implementation of Clean Water Act programs can ameliorate those threats;
2. Work with the Service and the National Marine Fisheries Service to reinvigorate implementation of the 2001 MOA on ESA and the CWA, especially to address local and regional water quality concerns, research needs, and revisions to the criteria derivation process.

In order for the Service to be kept informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, the Service requests notification of the implementation of any conservation recommendations.

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## 13.0 REINITIATION STATEMENT

This concludes formal consultation on EPA's continuing programmatic approval of cyanide criteria in state and tribal water quality standards. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

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# United States Department of the Interior

FISH AND WILDLIFE SERVICE
911 NE 11th Avenue
Portland, Oregon 97232-4181

In Reply Refer To:
FWS/R1/AES

JUN 25 2015

Dan Opalski, Director
Office of Water and Watersheds
U.S. Environmental Protection Agency
1200 Sixth Avenue
Seattle, Washington 98101

Dear Mr. Opalski:

Enclosed are the U.S. Fish and Wildlife Service's (Service) Biological Opinion (Opinion) and concurrence determinations on the Idaho Water Quality Standards for Numeric Water Quality Criteria for Toxic Pollutants (proposed action). The Opinion addresses the effects of the proposed action on the following listed species and critical habitats: the endangered Snake River physa snail (*Physa natricina*), threatened Bliss Rapids snail (*Taylorconcha serpenticola*), endangered Banbury Springs lanx (*Lanx* sp.; undescribed), the endangered Bruneau hot springsnail (*Pyrgulopsis bruneauensis*), the threatened bull trout (*Salvelinus confluentus*) and its critical habitat, and the endangered Kootenai River white sturgeon (*Acipenser transmontanus*) and its critical habitat.

The concurrence determinations address the following listed species: the threatened grizzly bear (*Ursus arctos horribilis*), endangered Southern Selkirk Mountains woodland caribou (*Rangifer tarandus caribou*), threatened Canada lynx (*Lynx canadensis*), threatened northern Idaho ground squirrel (*Spermophilus brunneus brunneus*), threatened MacFarlane's four-o'clock (*Mirabilis macfarlanei*), threatened water howellia (*Howellia aquatilis*), threatened Ute ladies'-tresses (*Spiranthes diluvialis)*, threatened Spalding's catchfly (*Silene spaldingii*), and the proposed threatened slickspot peppergrass (*Lepidium papilliferum*).

The Opinion concludes that the proposed action is likely to jeopardize the continued existence of the Snake River physa snail, Bliss Rapids snail, Banbury Springs lanx, Bruneau hot springsnail, bull trout, and the Kootenai River white sturgeon. The Opinion also concludes that the proposed action is likely to adversely modify bull trout critical habitat and Kootenai River white sturgeon critical habitat for the reasons discussed in the enclosed Opinion.

In accordance with regulation and in collaboration with your staff, the enclosed Opinion includes reasonable and prudent alternatives (RPAs) to avoid jeopardizing the continued existence of listed species and destroying or adversely modifying critical habitat. The RPAs reflect two components: (1) an interim alternative (while new criteria are being developed); and (2) a final alternative (involving the development of new protective criteria). Both components of the RPA are consistent with meeting section 7(a)(2) requirements, but may vary in their level of protectiveness, the effort needed to implement them, and subsequent Endangered Species Act compliance processes. As identified in section 2.8.10 of the enclosed Opinion, the Service requests that you positively affirm your acceptance of the RPAs and indicate if you intend to adopt the interim alternative as final or whether you intend to establish new criteria within the identified time frames set forth in the RPAs.

The enclosed Opinion was prepared in accordance with section 7 of the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 et seq.) and is based on information provided in the Environmental Protection Agency's 1999 Biological Assessment (Assessment), as amended in 2000 and 2014, and other sources of information cited in the Opinion. A complete decision record of this consultation is on file at the Service's Idaho Fish and Wildlife Office in Boise, Idaho.

If you have any questions regarding this matter, please contact Mr. Michael Carrier, State Supervisor of our Idaho Fish and Wildlife Office, at (208) 378-5243.

Sincerely,

Regional Director

Enclosure

cc:

| | |
|---|---|
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# BIOLOGICAL OPINION
# FOR THE
# IDAHO WATER QUALITY STANDARDS FOR NUMERIC WATER QUALITY CRITERIA FOR TOXIC POLLUTANTS

**01EIFW00-2014-F-0233**


**U.S. FISH AND WILDLIFE SERVICE**
**IDAHO FISH AND WILDLIFE OFFICE**
**BOISE, IDAHO**


**Supervisor** ______________________

**Date** JUN 25 2015

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# 1. INTRODUCTION AND BACKGROUND INFORMATION

## 1.1 Introduction

This document transmits the U.S. Fish and Wildlife Service's (Service) Biological Opinion (Opinion) regarding the effects of the U.S. Environmental Protection Agency's (EPA) approval of the Idaho Water Quality Standards for Numeric Water Quality Criteria for Toxic Pollutants on the following listed species and critical habitats: the endangered Snake River physa snail (*Physa natricina*), threatened Bliss Rapids Snail (*Taylorconcha serpenticola*), endangered Banbury Springs lanx (*Lanx* n sp.; undescribed), endangered Bruneau hot springsnail (*Pyrgulopsis bruneauensis*), threatened bull trout (*Salvelinus confluentus*), bull trout critical habitat, endangered Kootenai River white sturgeon, (*Acipenser transmontanus*), and Kootenai River white sturgeon critical habitat. This Opinion was prepared in accordance with section 7 of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.). Your December 20, 1999, request for formal consultation was received on December 22, 1999.

Please note that this Opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 CFR 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat.

This Opinion is based on information provided in the EPA's Biological Assessment (Assessment) (EPA 1999a, 2000), as amended, and other sources of information cited herein. A complete decision record of this consultation is on file at the Service's Idaho Fish and Wildlife Office in Boise, Idaho.

## 1.2 Consultation History

ESA consultation on Idaho water quality standards began over two decades ago in 1993. A very complex consultation history followed the initiation of this process. From mid-1993 through 1999, the consultation involved many discussions and correspondences between the EPA and the Service that are part of the service's administrative record for this action. These discussions culminated in an EPA letter, dated December 20, 1999, that was received by the Service on December 22, 1999, in which the EPA requested formal consultation on the effects of EPA's proposed approval of Idaho's water quality standards on the bull trout and the Kootenai River white sturgeon. Due to missing information in the EPA's biological assessment, formal consultation on this action did not begin until that information was transmitted by the EPA to the Service on August 9, 2000.

From 2000 to 2005, the Service and the EPA attempted to work through several issues regarding the consultation, which included the agencies agreeing to work collaboratively through an alternative dispute resolution (ADR) process. Ultimately, this process was unsuccessful, and on September 3, 2005, the Service received a final report on the ADR process from the EPA, that concluded "the interagency group never reached agreement on a set of recommended action for completing the consultation." At this point, the consultation stalled.

On December 3, 2012 the Service, NMFS National Marine Fisheries Service (NMFS), and the EPA received a Notice of Intent to Sue from Northwest Environmental Advocates (NWEA) for failure to complete consultation on the Idaho Water Quality Standards for Toxic Pollutants. On September 24, 2013, these same agencies received a complaint filed by NWEA and the Idaho Conservation League (Plaintiffs) alleging unreasonable delay of the ESA section 7 consultation. Subsequently, on November 22, 2013, the EPA sent a letter to the Service revising the proposed action and requesting formal consultation. Although critical habitat for the bull trout and the Kootenai River white sturgeon were not addressed in the EPA's final revised Assessment, the EPA requested the Service to address impacts to those critical habitats in its November 22, 2013, letter revising the proposed action.

The Service issued the draft Opinion to EPA for their review and comment on February 27, 2015 and received their comments back on May 12 and June 3, 2015. The Service also specifically discussed the draft RPAs with EPA and the Idaho Department of Environmental Quality (IDEQ) on April 13, May 5, and May 21, 2015, and received final comments on the RPAs from EPA and IDEQ on May 27, 2015. The Pacific Regional Office provided the final signed Opinion to EPA as of the date identified on the cover letter and Opinion title page.

# 1.3 Informal Consultation

The Service concurs with EPA's determination that the proposed action is not likely to adversely affect the threatened grizzly bear (*Ursus arctos horribilis*), endangered Southern Selkirk Mountains woodland caribou (*Rangifer tarandus caribou*), threatened Canada lynx (*Lynx canadensis*), threatened northern Idaho ground squirrel (*Spermophilus brunneus brunneus*), threatened MacFarlane's four-o'clock (*Mirabilis macfarlanei*), threatened water howellia (*Howellia aquatilis*), threatened Ute ladies'-tresses (*Spiranthes diluvialis)*, and threatened Spalding's catchfly (*Silene spaldingii*). The Service has also concluded that the proposed action is not likely to adversely affect the proposed threatened slickspot peppergrass (*Lepidium papilliferum*). The rationale for the Service's concurrence determinations is presented below.

For the reasons presented in the *Effects of the Action* section of this Opinion, we do not agree with the EPA's other "not likely to adversely affect" determinations for the following species: the endangered Snake River physa snail (*Physa natricina*), threatened Bliss Rapids snail (*Taylorconcha serpenticola*), endangered Banbury Springs lanx (*Lanx* n sp.; undescribed), endangered Bruneau hot springsnail (*Pyrgulopsis bruneauensis*), threatened bull trout (*Salvelinus confluentus*), bull trout critical habitat, endangered Kootenai River white sturgeon (*Acipenser transmontanus*), and Kootenai River white sturgeon critical habitat.

### Grizzly Bear

Given the isolated areas where grizzly bears are known to occur in the action area and given their diet is comprised largely of vegetation and terrestrial insects, it is unlikely that bears would be adversely affected through contact with surface waters or consumption of food items contaminated through waterborne toxins. In many instances, dietary concentrations of metals documented to cause adverse effects in mammals would require an animal to consume a 100 percent fish diet of highly contaminated fish. The Service concludes that such a scenario is unlikely.

**Southern Selkirk Mountains Woodland Caribou**

Woodland caribou are known to occur in isolated areas, and their diet is comprised largely of lichens and other vegetation. For these reasons, the proposed action is unlikely to cause adverse effects through contact with surface waters or consumption of food items contaminated through waterborne toxins.

**Canada Lynx**

Canada lynx occur in isolated areas, and their diet is comprised largely of snowshoe hare. For these reasons, the proposed action is unlikely to adversely affect the lynx through contact with surface waters or consumption of food items contaminated through waterborne toxins. In many instances, dietary concentrations of metals documented to cause adverse effects in mammals would require an animal to consume a 100 percent fish diet of highly contaminated fish.

**Northern Idaho Ground Squirrel**

The proposed action is not likely to adversely affect the northern Idaho ground squirrel because it would rarely, if ever, consume aquatic insects, drink from surface waters or live in flood-contaminated soils.

**MacFarlane's Four-o'clock**

MacFarlane's four-o'clock is a terrestrial plant species that occurs on well-drained soils. Most individual plants of this species occur in uplands that would never or very rarely be exposed to flood waters for at most, extremely brief durations. Therefore, exposure to waterborne toxins would be limited, be extremely infrequent, and short in duration. On that basis, the Service concludes that effects to the MacFarlane’s four-o’clock caused by the proposed action are likely to be insignificant and discountable.

**Water Howellia**

Water howellia is an annual, aquatic plant endemic to the Pacific Northwest region of the United States. Listed as a threatened species in 1994, its current known distribution includes the states of California, Idaho, Montana, Oregon and Washington. Water howellia typically inhabit small, vernal freshwater wetlands and ponds with an annual cycle of filling with water in spring and drying up in summer or autumn (USFWS 1996, p. 14). As of 2012, six occurrences of howellia have been documented in Idaho, all in Latah County, in oxbow ponds in the floodplain of the Palouse River. Given that these ponds are isolated from the river and dry up annually, it is highly unlikely that these populations would be impacted by any of the waterborne toxins addressed in this Opinion. On that basis, the Service concurs that effects to the water howellia from the proposed action are likely to be insignificant and discountable.

**Ute Ladies'-tresses**

Ute ladies'-tresses is a perennial, terrestrial orchid endemic to mesic or wet meadows and riparian/wetland habitats near springs, seeps, lakes, or perennial streams. Soils may be inundated early in the growing season, normally becoming drier but retaining subsurface moisture through the season. Grazing and recreational use appear to be the most likely activities affecting the plant. Any exposure of this plant to waterborne toxins caused by the proposed action is expected to be limited in duration and frequency. On that basis, the Service concurs that effects of the proposed action to the Ute ladies'-tresses are likely to be insignificant and discountable.

**Spalding's Catchfly**

Spalding's catchfly is a terrestrial plant species that occurs on open grasslands and deepsoiled valley/foothill areas. This species occurs in uplands that would never or very rarely be exposed to flood waters and water borne contaminants. The Service therefore concludes that effects to Spalding's catchfly from the proposed action will be insignificant and discountable.

**Slickspot Peppergrass**

The slickspot peppergrass occurs in semi-arid sagebrush-steppe habitats on the Snake River Plan, Owyhee Plateau, and adjacent foothills in southern Idaho. The peppergrass is restricted to small depositional microsites similar to vernal pools generally known as slickspots, mini-playas, or natric sites within communities dominated by other plants. These sparsely vegetated microsites are very distinct from the surrounding shrubland vegetation, and are characterized by relatively high concentrations of clay and salt. This is a species that occurs in sagebrush- steppe habitat and is not located in or near waterbodies, and is not anticipated to be exposed to waterborne pollutants. For those reasons, the Service concurs that the effects of the proposed action to the slickspot peppergrass are likely to be insignificant and discountable.

# 2. BIOLOGICAL OPINION

## 2.1 Description of the Proposed Action

This section describes the proposed Federal action, including any measures that may avoid, minimize, or mitigate adverse effects to listed species or critical habitat, and the extent of the geographic area affected by the action (i.e., the action area). The term "action" is defined in the implementing regulations for section 7 as "all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies in the United States or upon the high seas." The term "action area" is defined in the regulations as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action."

### 2.1.1 Action Area

The proposed action applies to all waters in the state of Idaho, defined as all accumulations of water, natural and artificial, public and private, or parts thereof which are wholly or partially within, which flow through, or border upon the State. In addition, as many Idaho streams/water bodies do not terminate within the borders of Idaho, the action area also extends downstream of (or to interconnected areas, as is the case with lakes and reservoirs) interstate/international waters. Effects in these downstream waters, however, are difficult to differentiate as water quality standards in adjacent states are similar to those proposed through this effort.

## 2.1.2 Proposed Action

Pursuant to Section 303(c) of the Clean Water Act (CWA), States are required to adopt water quality standards to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. A water quality standard defines the water quality goals of a waterbody by designating the use or uses to be made of the water, by setting criteria necessary to protect the uses, and by preventing degradation of water quality through antidegradation processes. States have primary responsibility for developing appropriate designated uses, and also setting criteria that will provide for a level of water quality such that the designated uses can be attained and protected. Numeric criteria are expressed as concentrations of chemicals or pollutants in water representing a quality of water that supports a particular use (50 CFR §131.3).

Typically, two levels are derived for each numeric criterion, both of which include an averaging period and a frequency of allowed exceedance. The following definitions are taken from the Idaho Water Quality Standards (IDAPA 16.01.02.003).

The higher level, or **acute criteria**, is the maximum instantaneous or one hour average concentration of a pollutant which ensures adequate protection of sensitive species of aquatic organisms from acute toxicity resulting from exposure to the pollutant. **Acute toxicity** is defined as the existence of mortality or injury to aquatic organisms resulting from a single or short-term (i.e., 96 hours or less) exposure to a substance. Acute criteria will adequately protect the designated aquatic life use if not exceeded more than once every three years.

The lower level, or **chronic criteria**, is the four-day average concentration of a toxic substance or effluent which ensures adequate protection of sensitive species of aquatic organisms from chronic toxicity resulting from exposure to the toxic substance. **Chronic toxicity** is defined as the existence of mortality, injury, reduced growth, impaired reproduction, or any other adverse effect on aquatic organisms resulting from a long-term (i.e., one-tenth or more of the organism's life span) exposure to a substance. Chronic criteria will adequately protect the designated aquatic life use if not exceeded more than once every three years.

The proposed action is EPA's approval of Idaho's Water Quality Standards pertaining to the aquatic life numeric criteria for toxic pollutants. The EPA has approved, subject to completion of this consultation, Idaho's aquatic life criteria for 11 organic chemicals and replacement of existing aquatic life criteria for 11 metals. The proposed aquatic life criteria would apply to all waters in the state that are protected for aquatic life beneficial uses. It is also important to note that "the analyses for the protectiveness of numeric criteria assume that the organisms are exposed to concentrations of pollutants at the water quality criteria levels, not the conditions which currently exist in Idaho's waters" (EPA 2000, p. 6; also see Section 2.5.1.3, *Assumptions in Effects Analyses*, section of this Opinion).[1]

The following additional information on the proposed action is adapted from the biological opinion issued by NMFS (2014a) on the proposed action.

The proposed numeric criteria are ambient water quality criteria, which are concentrations of each pollutant measured in the water column. Under EPA policy, States may choose to adopt metals criteria measured as either dissolved metal or total recoverable metal. Idaho's aquatic life criteria for metals were based on total recoverable metal (dissolved + suspended). The proposed action would change the aquatic life criteria to concentrations based on dissolved metals only, using a conversion factor (CF) to account for the suspended fraction. With the use of dissolved criteria, water samples are filtered to remove suspended solids before analysis.

The proposed water quality standards will apply to actions that require National Pollutant Discharge Elimination System (NPDES) permits[2], to development of total maximum daily loads (TMDLs) in streams with impaired water quality, and in situations where remedial actions are required to clean up spills or contaminated sites. When a TMDL is needed to regulate discharges into an impaired water body, the dissolved metals criteria must be converted or translated back to a total recoverable value so that the TMDL calculations can be performed. The translator can simply be the CF (i.e., divide the dissolved criterion by the CF to get back to the total criterion), or a dissolved-to-total ratio based on site-specific total/dissolved metal concentrations in the receiving water.

For some of the pollutants subject to this consultation, Idaho has also adopted criteria to protect human health from risk from exposure to the substances through eating fish or shellfish or ingestion of water through recreating on water. Although EPA is not consulting on the human health-based criteria, on a practical level, permitted discharges to a given water body would be constrained by the most stringent applicable criteria. In other words, when they are more stringent than the aquatic life criteria, the human health criteria will constrain discharge levels. During the pendency of this consultation, Idaho has further revised some of the criteria under consultation. The EPA has updated its action to reflect these revisions and they are being consulted on as shown Table 1.

The application of water quality criteria is based on the principle of designated beneficial uses of

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[1] This approach to the effects analysis differs from that described in the Oregon Water Quality Criteria for Toxics Biological and Conference Opinions (USFWS 2012a, pp. 13-14). While acknowledging that the criteria apply to the waters of the state of Oregon designated for fish and aquatic life use, in the Oregon Opinion the Service states that for the purposes of determining the effects of the proposed action on listed species or critical habitat, the EPA and the Service considered the likelihood of exposure to water pollutants based on the ability to identify all current and future point and nonpoint source discharges; the biological and conference opinions only assessed the effects of the proposed water quality standards on listed species and critical habitat in those areas where there are likely to be point or nonpoint source discharges subject to these standards.

[2] In Idaho, the NPDES program is administered by EPA, which means EPA is responsible for issuing and enforcing all NPDES permits in Idaho (https://www.deq.idaho.gov/permitting/water-quality-permitting/npdes.aspx) (Accessed February 9, 2015).

water. Together, ambient water quality criteria and use designations are used to meet the primary objective of the CWA – to "restore and maintain the chemical, physical and biological integrity of the Nation's waters." A further goal of the CWA is that wherever attainable, an interim goal of water quality is to provide "for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water." (CWA, §101(a)).

The water quality criteria that are the subject of this consultation are summarized in Table 1. These criteria are currently in effect and are applicable to all waters in the State of Idaho pursuant to Section 16 of the Idaho Administrative Procedures Act, Title 01, Chapter 02 (IDAPA 16.01.02). All EPA approval actions on the criteria will be made subject to successful completion of this consultation (i.e., the proposed action is not likely to violate ESA section 7(a)(2)).

**Table 1. Ambient Water Quality Criteria for toxic pollutants submitted for consultation in EPA's 1999 Assessment. Also shown are AWQC that have subsequently been revised by the State of Idaho (Idaho Department of Environmental Quality 2011). The following Table is presented in two parts, inorganic and organic substances (adapted from NMFS 2014a).**

**Part 1. Criteria for metals and other inorganic substances**

| Substance | Proposed Aquatic Life Criteria in the 1999 Assessment (EPA 1999a) (µg/L) | | Idaho Revised Criteria included in EPA's Updated Action (November 2013) (µg/L) | | Human Health Criteria (Recreation) (µg/L) | Conversion Factor[a] | |
|---|---|---|---|---|---|---|---|
| | Acute | Chronic | Acute | Chronic | | Acute | Chronic |
| Arsenic (As) | 360 | 190 | 340 | 150 | 10 | 1.000 | 1.000 |
| Cadmium (Cd) | Consultation completed in 2011 | | | | | | |
| Copper (Cu) | 17[b] | 11[b] | 17[b] | 11[b] | | 0.960 | 0.960 |
| Cyanide (CN, weak acid dissociable) | 22[e] | 5.2[e] | 22[e] | 5.2[e] | | N/A | N/A |
| Lead (Pb) | 65[b] | 2.5[b] | 65[b] | 2.5[b] | | 0.791[d] | 0.791[d] |
| Mercury (Hg) | 2.1 | 0.012 (unfiltered) | 2.1[g] | 0.012[g] | 0.3 mg/kg in fish tissue, fresh weight | 0.85 | N/A |
| Selenium (Se) | 20 | 5.0 (unfiltered) | 20 | 5.0 (unfiltered) | | N/A | N/A |
| Zinc (Zn) [b] | 114[b] | 105[b] | 120[b] | 120[b] | | 0.978 | 0.986 |
| Chromium (Cr) ((III)) [b] | 550[c] | 180[c] | 570[c] | 74[c] | | 0.316 | 0.860 |
| Chromium (Cr) (VI) | 15 | 10 | 16 | 11 | | 0.982 | 0.962 |
| Nickel (Ni) | 1,400[b] | 160[b] | 470[b] | 52[b] | | 0.998 | 0.997 |
| Silver (Ag) | 3.4[b] | N/A | 3.4[b] | N/A | | 0.85 | N/A |

(µg/L: micrograms per liter; Metals criteria are shown for a water hardness of 100 mg/L).

| **Part 2. Criteria for organic toxic substances** | | | | | | |
|---|---|---|---|---|---|---|
| Substance | Proposed Aquatic Life Criteria (µg/L) | | Human Health Criteria (Recreation) (µg/L)[a] | Idaho Human Health Criteria Revised after 1999 Assessment (µg/L)[h] | Conversion Factor | |
| | Acute | Chronic | | | Acute | Chronic |
| Endosulfan (α and β) | 0.22 | 0.056 | 2.0 | 89.0 | N/A | N/A |
| Aldrin | 3 | – | 0.00014 | 0.000050 | N/A | N/A |
| Chlordane | 2.4 | 0.0043 | 0.00057 | 0.00081 | N/A | N/A |
| 4,4'-DDT | 1.1 | 0.001 | 0.00059 | 0.00022 | N/A | N/A |
| Dieldrin | 2.5 | 0.0019 | 0.00014 | 0.000054 | N/A | N/A |
| Endrin | 0.18 | 0.0023 | 0.81 | 0.060 | N/A | N/A |
| Heptachlor | 0.52 | 0.0038 | 0.00021 | 0.000079 | N/A | N/A |
| Lindane (gamma-BHC) | 2 | 0.08 | 0.063 | 1.8 | N/A | N/A |
| Polychlorinated biphenyls (PCBs) | N/A | 0.014 | 0.000045 | 0.000064 | N/A | N/A |
| Pentachlorophenol (PCP) | 20[e] | 13[e] | 6.2 | 3.0 | N/A | N/A |
| Toxaphene | 0.73 | 0.0002 | 0.00075 | 0.00028 | N/A | N/A |

- N/A - no applicable criteria

a. Conversion factors for translating between dissolved and total recoverable criteria.

b. Criteria for these metals are expressed as a function of total hardness (mg/L as CaCo3), and the following formula:

Acute Criteria = WER exp{mA[ln(hardness)]+bA} x Acute Conversion Factor
Chronic Criteria = WER exp{mC[ln(hardness)]+bC} x Chronic Conversion Factor
where:

| Metal | $m_A$[f] | $b_A$[f] | $m_C$[f] | $b_C$[f] |
|---|---|---|---|---|
| Chromium (III) | 0.8190 | 3.688 | 0.8190 | 1.561 |
| Copper | 0.9422 | -1.464 | 0.8545 | -1.465 |
| Lead | 1.273 | -1.460 | 1.273 | -4.705 |
| Nickel | 0.8460 | 3.3612 | 0.8460 | 1.1645 |
| Silver | 1.72 | -6.52 | N/A | N/A |
| Zinc | 0.8473 | 0.8604 | 0.8473 | 0.7614 |

The term "exp" represents the base e exponential function.

c. For comparison purposes, the values displayed in this table correspond to a total hardness of 100 mg/l $CaCO_3$

and a Water Effects Ratio (WER) of 1.0.

d. The conversion factor for lead is hardness dependent. The values shown in the table correspond to a hardness of 100 mg/L $CaCO_3$. Conversion factors for lead: Acute and Chronic- CF=1.46203-[(ln(hardness))x(0.145712)].

e. Criteria expressed as Weak Acid Dissociable.

f. $m_A$ and $m_c$ are the slopes of the relationship for hardness, while $b_A$ and $b_C$ are the Y-intercepts for these relationships.

g. Criteria for pentachlorophenol increase as pH increases and are calculated as follows:
Acute Criterion = exp(1.005 (pH) - 4.830)
Chronic Criterion = exp(1.005 (pH) - 5.290) Values shown in the table are for pH 7.8.

h. The state of Idaho repealed the water column aquatic life criteria for mercury in 2006, based upon IDEQ's (2005) analysis that concluded the available science no longer supported EPA's (1985g) aquatic life criteria, and that a fish tissue based human-health criteria would be better supported by the science, be adequate to protect aquatic life, and would be more stringent than the 1985 chronic aquatic life criterion of 0.012 µg/L. EPA disapproved Idaho's repeal of its water column acute and chronic mercury criteria on policy grounds that, an exception for California notwithstanding, water column based aquatic criteria were required for Idaho, Idaho's criteria did not include a sufficiently detailed implementation for translating the human health tissue criterion to a protective aquatic life criteria that could be used with effluent limits (Gearheard 2008). The disapproval addressed policy interpretations and was silent on IDEQ's arguments that the EPA (1985g) mercury chronic was outdated and that a 0.3 mg/kg fish tissue criterion was more protective. Gearheard (2008) considered the 0.012 µg/L chronic criterion to be effective for NPDES discharge permits and TMDLs issued by EPA, although the criterion remains repealed under state law and nowhere appears in Idaho administrative rules.

i. Although Idaho's revised human health criteria are considerably more stringent than the previous human health criteria, EPA has not approved these revised criteria and EPA does not consider the more stringent criteria to be effective for Clean Water Act purposes.

Per EPA's guidance, States, when adopting criteria for metals, may adopt criteria measured as either dissolved or total recoverable metal. The Idaho metals criteria under consultation are expressed as dissolved metals, meaning that water samples are filtered to remove suspended solids before analysis.

Metals and inorganic toxic substances addressed in this consultation include: arsenic, copper, cyanide, lead, mercury, selenium, zinc, chromium (III), chromium (VI), nickel, and silver. For several of these chemicals, the water quality criteria are equation-based, meaning the criteria applicable to a particular site vary based on site-specific conditions. The equation-based metals are chromium (III), chromium (VI), copper, lead, mercury, nickel, silver, and zinc. To determine criteria for these metals for a given water body, site-specific data must be obtained, input to an equation, and numeric criteria computed. There are three types of site-specific data that may be necessary to determine and/or modify the criteria for these metals at a site: (1) water hardness; (2) CF and translators; and (3) water effect ratios; refer to the Assessment for more details (EPA 1999a).

# 2.2 Analytical Framework for the Jeopardy and Adverse Modification Determinations

## 2.2.1 Jeopardy Determination

In accordance with policy and regulation, the jeopardy analysis in this Opinion relies on four components:

1. The *Status of the Species*, which evaluates the species rangewide condition, the factors responsible for that condition, and its survival and recovery needs.
2. The *Environmental Baseline*, which evaluates the condition of the species in the action area, the factors responsible for that condition, and the relationship of the action area to the survival and recovery of the species.
3. The *Effects of the Action*, which determines the direct and indirect impacts of the proposed Federal action and the effects of any interrelated or interdependent activities on the species.
4. *Cumulative Effects*, which evaluates the effects of future, non-Federal activities in the action area on the species.

In accordance with policy and regulation, the jeopardy determination is made by evaluating the effects of the proposed Federal action in the context of the species current status, taking into account any cumulative effects, to determine if implementation of the proposed action is likely to cause an appreciable reduction in the likelihood of both the survival and recovery of the species in the wild.

The jeopardy analysis in this Opinion places an emphasis on consideration of the rangewide survival and recovery needs of the species and the role of the action area in the survival and recovery of the species as the context for evaluating the significance of the effects of the proposed Federal action, taken together with cumulative effects, for purposes of making the jeopardy determination.

In the case of the bull trout, interim recovery units (formerly recognized as Distinct Population Segments, DPS) have been designated for the bull trout for purposes of recovery planning and application of the jeopardy standard (see *Status of the Species* section). Per Service national policy (USFWS 2006a, entire), it is important to recognize that the establishment of recovery units does not create a new listed entity. Jeopardy analyses must always consider the impacts of a proposed action on the survival and recovery of the species that is listed. While a proposed Federal action may have significant adverse consequences to one or more recovery units, this would only result in a jeopardy determination if these adverse consequences reduce appreciably the likelihood of both the survival and recovery of the listed entity; in this case, the coterminous U.S. population of the bull trout.

The joint Service and National Marine Fisheries Service (NMFS) *Endangered Species Consultation Handbook* (USFWS and NMFS 1998, p. 4-38),which represents national policy of both agencies, further clarifies the use of recovery units in the jeopardy analysis:

> When an action appreciably impairs or precludes the capacity of a recovery unit from providing both the survival and recovery function assigned to it, that action may represent jeopardy to the species. When using this type of analysis, include in the biological

opinion a description of how the action affects not only the recovery unit's capability, but the relationship of the recovery unit to both the survival and recovery of the listed species as a whole.

The jeopardy analysis in this Opinion conforms to the above analytical framework.

### 2.2.2 Adverse Modification Determination

As noted above, this Opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 CFR §402.02. Instead, we have relied upon the statutory provisions of the Act to complete the following analysis with respect to critical habitat.

In accordance with policy, the adverse modification analysis in this Opinion relies on four components:

1. The *Status of Critical Habitat*, which evaluates the rangewide condition of designated critical habitat for the species in terms of primary constituent elements (PCEs), the factors responsible for that condition, and the intended recovery function of the critical habitat overall.
2. The *Environmental Baseline*, which evaluates the condition of the critical habitat in the action area, the factors responsible for that condition, and the recovery role of the critical habitat in the action area.
3. The *Effects of the Action*, which determines the direct and indirect impacts of the proposed Federal action and the effects of any interrelated or interdependent activities on the PCEs and how that will influence the recovery role of affected critical habitat units.
4. *Cumulative Effects*, which evaluates the effects of future, non-Federal activities in the action area on the PCEs and how that will influence the recovery role of affected critical habitat units.

For purposes of the adverse modification determination, the effects of the proposed Federal action on species critical habitat are evaluated in the context of the rangewide condition of the critical habitat, taking into account any cumulative effects, to determine if the critical habitat rangewide would remain functional (or would retain the current ability for the PCEs to be functionally established in areas of currently unsuitable but capable habitat) to serve its intended recovery role for the species.

The analysis in this Opinion places an emphasis on using the intended rangewide recovery function of species critical habitat and the role of the action area relative to that intended function as the context for evaluating the significance of the effects of the proposed Federal action, taken together with cumulative effects, for purposes of making the adverse modification determination.

# 2.3 Status of the Species and Critical Habitat

## 2.3.1 Snake River Physa Snail

### 2.3.1.1 Listing Status

The Service listed the Snake River physa as endangered effective January 13, 1993 (57 FR 59244). No critical habitat has been designated for this species. A recovery plan for the Snake River physa was published by the Service as part of the Snake River Aquatic Species Recovery Plan (USFWS 1995, entire). The target recovery area for this species is from river kilometer (RKM) 890 to 1,086 (river mile (RM) 553 to 675) (USFWS 1995, p. ii).

### 2.3.1.2 Species Description

The Snake River physa (or Physa) was formally described by Taylor (1988, pp. 67-74; Taylor 2003, pp. 147-148), from which the following characteristics are taken. The shells of adult Snake River physa may reach 7 mm (0.28 inches (in)) in length with 3 to 3.5 whorls, and are amber to brown in color and ovoid in overall shape. The aperture whorl is inflated compared to other Physidae in the Snake River, the aperture whorl being ≥ 1/2 of the entire shell width. The growth rings are oblique to the axis of coil at about 40° and relatively course, appearing as raised threads. The soft tissues have been described from limited specimens and greater variation in these characteristics may be present upon detailed inspection of more specimens. The body is nearly colorless, but tentacles have a dense black core of melanin in the distal half. Penal complex lacks pigmentation although the penal sheath may be opaque. Tip of the penis is simple (not ornamented). The preputial gland is nearly as long as the penal sheath.

The Snake River physa is a pulmonate species, in the family Physidae, order Basommatophora (Taylor 2003, pp. 147-148). The rarity of Snake River physa collections, combined with difficulties associated with distinguishing this species from other physids, has resulted in some uncertainties over its status as a separate species. Taylor (2003, pp. 135-137) presented a systematic and taxonomic review of the family, with Snake River physa being recognized as a distinct species (*Haitia* (*Physa*) *natricina*) based on morphological characters he originally used to differentiate the species in 1988. Later authors concluded that the characters described by Taylor (1988) were within the range of variability observed in the widely distributed *Physa acuta*, and placed Snake River physa as a junior synonym of *P. acuta* (Rogers and Wethington 2007, entire). Genetic material from early Snake River physa collections was not available when Rogers and Wethington published and their work included no analysis or discussion on the species' genetics.

More recent collections of specimens resembling Taylor's (2003, pp. 147-148) descriptions of Snake River physa have been used to assess morphological, anatomical, and molecular uniqueness. Live snails resembling Snake River physa collected by the Bureau of Reclamation (BOR) below Minidoka Dam as part of monitoring recommended in a 2005 Biological Opinion (USFWS 2005a, pp. 162-163) began to be recovered in numbers sufficient to provide specimens for morphological review and genetic analysis. Burch (2008, *in litt.*) and Gates and Kerans (2010, pp. 41-61) identified snails collected by BOR as Snake River physa using Taylor's (2003, pp. 147-148) shell and soft tissue characters. Their genetic analysis also found these specimens to be a species distinct from *P. acuta*.

Gates and Kerans (2011, pp. 6-7) also performed similar analyses on 15 of 51 live-when-collected specimens recently identified as Snake River physa (Keebaugh 2009, pp. 102-121), and collected by the Idaho Power Company (IPC) between 1998 and 2003 in the Snake River from Bliss Dam RKM 901 (RM 560) downstream to near Ontario, Oregon RKM 592 (RM 368). Gates and Kerans (2011, pp. 9-11) found that these specimens were not genetically distinct from Snake River physa collected below Minidoka Dam (but were genetically distinct from *P. acuta*), and provided additional support that Taylor's (1988) shell description of Snake River physa is diagnostic (Gates and Kerans 2011, p. 6).

# 2.3.1.3 Life History

**Biology**

Freshwater pulmonate snail species such as Snake River physa do not have gills, but absorb oxygen across the inner surface of the mantle (outer wall of the mollusk's body that encloses the internal organs) (Dillon 2006, p. 252). The walls of the mantle are heavily vascularized (filled with blood vessels), and air is drawn into the mantle cavity via expansion and contraction of the mantle muscles (Vaughn et al. 2008, entire). Freshwater pulmonates usually carry an air bubble within the mantle as a source of oxygen, replenished via occasional trips to the surface; the bubble is manipulated to adjust buoyancy and allow transportation to the surface (Dillon 2006, p. 252). However, some freshwater pulmonate species do not carry air bubbles; oxygen instead diffuses from the water directly into their tissues across the surface of the mantle (Dillon 2006, p. 252), the likely mode of respiration for Snake River physa. Since they live in moderately swift current, individuals that would release from substrates to replenish air at the surface would likely be transported some distance downstream away from their colony and habitat of choice, possibly into unsuitable habitat.

The protean physa (*Physella virgate*) has been observed to move and remain out of the water for up to 2 hours in reaction to chemical cues given off by crayfish foraging on other nearby protean physa (Alexander and Covich 1991, p. 435). The Snake River physa may have the same capability for out-of-water survival, though the fact that the species has rarely been collected in shallow water (less than 0.30 meters (m) (0.98 feet (ft))) and has been found in greatest abundance at depths greater than or equal to 1.5 m (4.9 ft) (Gates and Kerans 2010, p. 23), indicates that the Snake River physa does not routinely occur in shallow water or spend extended periods out of water.

Snake River physa have not yet been cultured and studied in the laboratory, and the species' reproductive biology has not been studied under natural conditions. Another Physa species, *Physa acuta,* reach sexual maturity at between 6 to 8 weeks at 22-24 degrees Celsius (°C) (71.6-75.2 degrees Fahrenheit (°F)) in laboratory conditions (Escobar et al. 2009, p. 2792); additionally Dillon et al. (2004, p. 65) reported mean fecundity of 39 hatchlings per pair per week for *P. acuta*. It is not known whether the Snake River physa exhibits similar reproductive output as *Physa acuta*.

All freshwater pulmonates are reported to be able to reproduce successfully by self-fertilization (Dillon 2000, p. 83). While self-fertilization (selfing) in pulmonates can be forced under laboratory conditions by isolating individual snails, there is considerable variation within and among pulmonate genera and species in the degree of selfing that occurs in natural populations. Of the many Physa species in North America and world-wide, studies of self-fertilization effects on population genetics seem to have been conducted only on *Physa acuta*. Selfing and its implications for genetic variation and fitness are unknown for Snake River physa.

Water temperature requirements of Snake River physa have not been identified. Gates and Kerans (2010, p. 21) reported a mean water temperature of 22.6°C (72.7°F) for sites occupied by the species at the time of sampling (in August and October), but it is not known if this represents an optimal range. Snake River physa were collected in the Bruneau arm of C.J. Strike Reservoir and in the Snake River when water temperatures were averaging 23.4°C (74.1°F) (Winslow 2013, *in litt.*). The maximum temperature for cold water aquatic life in Idaho is 22°C (71.6°F). Based on available information, Snake River physa appear to be able to tolerate water temperatures slightly above the cold water standard of 22°C (71.6°F), although their upper temperature limit has not been identified. Conversely, water temperatures below 10.0°C (50°F) are known to inhibit reproduction in the tadpole physa (DeWitt 1955, p. 43). Springs originating from the Eastern Snake River Plain Aquifer (ESPA) flow at temperatures from 14 to 16°C (57.2 to 60.8°F) year around. Extensive monitoring and surveys in these cold-water springs for the Bliss Rapids snail (*Taylorconcha serpenticola*) and Banbury Springs lanx (*Lanx* n sp.) have never found the Snake River physa, indicating these habitats are not preferred by the Snake River physa. Average dissolved oxygen (DO) measured in occupied Snake River physa habitat has been reported to range from 8.35 to 9.99 milligrams per liter (mg/L) in studies by Gates and Kerans (2010, p. 21) and the USBOR (2013, p. 22).

Onset of egg-laying by physid species appears to be a function of water temperature. McMahon (1975, entire) summarized a range of water temperatures at which egg laying occurred for two species that occur in North America (acute bladder snail and tadpole physa), and one European (common bladder snail [*Physa fontinalis*]) physid species as between 7-13°C (44.6-55.4°F) in northern temperate climates. Dillon (2000, pp. 156-170) noted a commonly reported temperature for the onset of gastropod egg-laying (including physid species) as being 10°C (50°C). Although the acute bladder snail and tadpole physa are known to occur in the Snake River, neither species is common in habitats preferred by Snake River physa.

In summary, the Snake River physa likely diffuses oxygen from the water directly into its tissues across the surface of the mantle. The Snake River physa is likely able to reproduce both sexually and asexually, though implications of selfing on genetic variation and fitness are unknown. Snake River physa have been found in water temperatures above 22°C (71.6°F) and have not been found in the cool-water springs that flow into the Snake River.

## Habitat

Based on the most recent findings (Gates and Kerans 2010, entire) of the Snake River physa's distribution and habitat preferences, the conservation needs of the species includes instream conditions that produce or sustain beds of pebble to gravel, and possibly cobble to gravel, that are largely free of substrates finer than gravel which can fill in the interstitial spaces between gravel. Given the lack of fine substrates within their preferred habitat, these preferred habitat areas are also largely free of macrophytes (USFWS 2012b, Appendix A). Macrophyte beds can reduce water velocity, causing fines such as sand, silt, and clay to fall out of the water column, potentially embedding or covering Snake River physa habitat (USFWS 2012b, p. 68)[3].

In general, the locations of live, confirmed specimens of Snake River physa have been most frequently recorded from the free-flowing reaches of the Snake River downstream of the following dams: Minidoka Dam, Lower Salmon Falls Dam, Bliss Dam, C.J. Strike Dam, and Swan Falls Dam. Free-flowing reaches are defined here as areas of the Snake River where water velocities generally keep gravel and pebble beds free of fine sediments and subsequent macrophyte growth, and habitats at the range of depths (0.5 m to 3 m) where Snake River physa has been found. Maintaining these areas of suitable habitat for the Snake River physa in these free-flowing reaches of river is reliant on maintaining suitable water quality conditions, particularly temperature, fine sediments, and nutrient load, to minimize macrophyte growth (USFWS 2012b, p. 68).

Gates and Kerans' detailed study (2010, entire), which sampled for Snake River physa across sections of the Snake River's profile directly below Minidoka Dam, characterized Snake River physa habitat as run, glide, and pool habitats with a moderate mean water velocity (0.57 m/second (1.87 ft/ second)). The mean depth of samples containing live Snake River physa was 1.74 m (5.71 ft), with most found at depths of 1.5 to 2.5 m (4.9 to 8.2 ft). Depths in which all Snake River physa were found ranged from less than 0.5 m (1.6 ft) to over 3.0 m (9.8 ft), and the highest density (12 or more) collected per $m^2$ were at depths greater than 1.5 m (4.9 ft). Eighty percent of samples containing live Snake River physa were located generally in the middle of the river (Gates and Kerans 2010, p. 20); most typically in deeper water habitats.

In an effort to clarify habitat use for describing Snake River physa distribution, the Service, in coordination with IPC biologists, conducted an analysis (USFWS 2012b, Appendix A) of substrate selection in areas where the species has been found in relatively large numbers. This analysis also looked at substrate composition and distribution in the Snake River, including the type locality. This analysis identified that Snake River physa were found to strongly select for substrates ranging in size from gravel to pebble, and possibly from gravel to cobble. This substrate selection is somewhat different than Taylor's (1982a, p. 2) description of boulder to gravel substrates, with his specimens being collected from boulders. This preference for gravel to pebble, and possibly gravel to cobble, however, are consistent in both the C.J. Strike (RKM 795 (RM 494)) to Weiser (RKM 592 (RM 368)) reach and the Minidoka reach (RKM 1086-1068 (RM 663.5 – 675)), two sections of the Snake River occupied by the Snake River physa which are separated by over 322 river km (200 river mi) (USFWS 2012b, Appendix A, p. 64).

[3] For example, experiments conducted by Li et al. (2012, p. 81) found that the macrophyste *Vallisneria natans* "grown in silt and clay substrates had greater height, more ramets and leaves, as well as greater biomass accumulation" compated to *V. natans* grown on pebble and gravel substrates.

Gravel and pebble were the most common substrates reported by Gates and Kerans (2010, p. 23) in the Minidoka reach (USFWS 2012b, Appendix A, p. 63). This suggests that the existence of relatively large, contiguous areas of this habitat type in this reach may be one factor contributing to the comparatively high densities and abundance of Snake River physa which occur there. Densities were generally less than or equal to 32 individuals per $m^2$ (approximately 3 individuals per $ft^2$), but 3 samples had up to 40 to 64 individuals per $m^2$ (3.7 to 6.0 individuals per $ft^2$). Although Gates and Kerans (2010, p. 37) documented relatively high densities of Snake River physa in their study area, they also concluded that Snake River physa occurred in a diffusely distributed population, and suggested that the species rarely exhibits high density colony behavior.

Dams can act as sediment traps, reducing fine sediment loading in rivers downstream of the dam (Poff et al. 1997, pp. 772-774). The American Falls Dam (RKM 1149 (RM 714)) and Minidoka Dam (RKM 1068 (RM 675)), which are both upstream of the largest known population of Snake River physa, likely act as effective sediment traps (Newman 2011, *in litt.*). In addition, Lake Walcott (reservoir behind Minidoka Dam) is largely operated as run of river (operates based on available streamflow with limited storage capability), meaning that bottom sediments at the dam's face are typically not mobilized. Water leaving the power plant and passing through the spillway gates is relatively free of fine sediment and provides little or no sediments that could embed cobble substrates and support macrophytes.

In addition, Minidoka Dam is normally operated so that the Snake River downstream somewhat mimics a natural hydrograph of a lowland western river, with flows increasing in spring, peaking during summer, and tapering off through the fall; with the primary departure from a natural hydrograph being that high flows are maintained downstream of Minidoka Dam well into September (USFWS 2012b, p. 15). The effect of this high and prolonged summer flow regime is to keep the pebble and gravel beds relatively free of fine sediment during the period of highest insolation and summer temperatures, resulting in reduced presence of macrophyte growth throughout the Minidoka reach where Snake River physa can be encountered (USFWS 2012b, p. 15). Flow operations at Swan Falls Dam are inverse from those at Minidoka Dam, with flows highest in winter, and lowest in summer (usually July and August) during the period when macrophyte production and growth would be the highest (USFWS 2012b, p. 15). Proliferation of macrophytes on cobble/ gravel beds downstream of Swan Falls Dam have been attributed to nutrient loading and high sediment loads passing Swan Falls Dam (Groves and Chandler 2005, pp. 479-480). Compared to the number of Snake River physa found by Gates and Kearns (2010) downstream of Minidoka Dam, the IPC collected far fewer Snake River physa downstream of Swan Falls Dam per sampling effort[4], which may be in part attributable to low summer flows, higher sediment load combined with high nutrient loads, and therefore a higher percentage of macrophytes downstream of Swan Falls Dam (USFWS 2012b, p. 16).

The section of the Snake River between Lower Salmon Falls Dam (RKM 922 (RM 573)) and C.J. Strike Reservoir (RKM 795 (RM 494)), which includes the type locality, does not appear to

[4] 4.8 times more Snake River physa were collected downstream of Minidoka Dam for approximately double the sampling effort, compared to what was collected downstream of Swan Falls Dam (USFWS 2012a, pp. 61-62).

contain large areas of preferred habitat (pebble to gravel to cobble) for the Snake River physa (USFWS 2012b, p. 14). Even though sampling for Snake River physa has not been extensive throughout this reach, its history of low detections in this section suggests that under the current habitat conditions, the probability of encountering Snake River physa within this reach will likely remain low into the future (USFWS 2012b, p. 14).

Between C.J. Strike Dam (RKM 795 (RM 494)) and Swan Falls Dam (RKM 736.6 (RM 457.7)), there were 12 live-when-collected specimens of Snake River physa collected in 2001 and 2002 (IPC 2012, *in litt.* ). C.J. Strike Dam is operated in a load-following mode in response to electricity demand (USFWS 2004a, p. 20). While we have limited information regarding Snake River physa habitat conditions downstream of C.J. Strike Dam, given existing dam operations (load-following versus irrigation water release) we anticipate Snake River physa habitat conditions to be more similar to the habitat conditions downstream of Swan Falls Dam (sediment, extensive macrophytes) as opposed to downstream of Minidoka Dam (pebble to cobble, limited macrophytes).

Data collected to date indicate the conditions of sites where Snake River physa have been collected are characterized by swift current, where the river transitions from lotic (free-flowing) to more lentic (standing water) environments. In contrast, the two specimens of Snake River physa found in the reservoir pool of the Bruneau River arm of C.J. Strike Reservoir is in an area usually characterized by very slow moving lentic conditions. Little is known of the species' distribution or habitat in the Bruneau River arm of C.J. Strike Reservoir, compared to habitat conditions where it has been found elsewhere in the Snake River.

In summary, Snake River physa are generally found in free-flowing Snake River reaches characterized by gravel to pebble-sized and possibly cobble-sized substrates, where these substrate types stay relatively free of fines and macrophyte growth. The species is rare in Snake River reaches with widely scattered, low proportions of cobble to gravel substrates, as in the reach between C.J. Strike Reservoir (RKM 795 (RM 494)) and Lower Salmon Falls Dam (RKM 922 (RM 573)). Snake River physa is patchily distributed in the free-flowing reaches from C. J. Strike Dam downstream to near Ontario, Oregon, but it is found at higher densities downstream of Minidoka Dam.

**Diet**

The diet preferences of Snake River physa are not known. Species within the family Physidae live in a wide variety of habitats and exhibit a variety of dietary preferences. Physidae from numerous studies consumed materials as diverse as aquatic macrophytes, benthic diatoms (diatom films that primarily grow on rock surfaces, also called periphyton), bacterial films, and detritus (Dillon 2000, pp. 66-70). The tadpole physa (*Physa gyrina*), which co-occurs with Snake River physa in the Snake River, consumes dead and decaying vegetation, algae, water molds, and detritus (DeWitt 1955, p. 43; Dillon 2000, p. 67). The Snake River physa likely has feeding patterns similar to the tadpole physa.

## 2.3.1.4 Status and Distribution

Existing populations of the Snake River physa are known only from the Snake River in central and south-southwest Idaho (and a small portion of Oregon), with the exception of two (live-when-collected) specimens recovered in 2002 from the Bruneau River arm of C.J. Strike

Reservoir (Keebaugh 2009, p. 123). Fossil evidence indicates this species existed in the Pleistocene-Holocene lakes and rivers of northern Utah and southeastern Idaho, and as such, is a relict species from Lake Bonneville, Lake Thatcher, the Bear River, and other lakes and watersheds that were once connected to these water bodies (Frest et al. 1991, p. 8, Link et al. 1999, pp. 251-253).

In the Snake River Species Aquatic Recovery Plan, the Service (USFWS 1995, p. 8) reported that the "modern" range of the Snake River physa extended within the Snake River from Grandview (RKM 784 (RM 487)) to the Hagerman reach (RKM 922 (RM 573)), with a possible colony downstream of Minidoka Dam (RKM 1086 (RM 675)). The first known collection of Snake River physa in the Snake River since listing was in 2006, when live specimens were collected by USBOR in the Minidoka reach (RKM 1086-1068 (RM 675-663.5)). Surveys conducted by the USBOR from 2006 through 2012 (Gates and Kerans 2010, entire; Gates et al. 2013, entire; USBOR 2013, p. 18), and subsequent analysis in 2009 of collections by the IPC between 1995 and 2003 (Keebaugh 2009, entire) have established the Snake River physa's current distribution to be from RKM 592 (RM 368) near Ontario, Oregon, upstream to Minidoka Dam RKM 1086 (RM 675)). The site near Ontario, Oregon is approximately 206 kilometers (km) (128 miles (mi)) downstream from the species previously recognized downstream-most extent of distribution. The additional site in the Bruneau River arm of C.J. Strike Reservoir was identified by Gates and Kerans (2011, p. 10) when they confirmed that shell morphology, diagnostic of Snake River physa, matched that of specimens with similar morphology also confirmed as Snake River physa by DNA analysis. Within this range, live Snake River physa have been collected in two general areas: (1) the reach below of Lower Salmon Falls Dam (RKM 922 (RM 573)) downstream to approximately Ontario, Oregon (RKM 592 (RM 368)), and (2) in the Minidoka reach (RKM 1086-1068 (RM 675-663.5)). Within this 494 km (307 mi) range, the species remains rare with only 385 confirmed live-when-collected specimens taken over a 53-year period between 1959 and 2012.

It is important to note that while live Snake River physa have been collected from the same survey transects in successive years (2006-2012) downstream of Minidoka Dam (Gates and Kerans 2010, p. 24; USBOR 2013, p. 24), the species has not been regularly or reliably located throughout the rest of its range. Snake River physa have not been found in the reaches between Lower Salmon Falls Dam and the Minidoka reach (RKM 922-1068 (RM 573- 663.5)), although surveys in this area have been limited. Snake River physa have not been collected in the area of the type locality (RKM 916-917 (RM 569-570)) described by Taylor since 1988. Taylor's 1959, 1988 (1982a entire; 1988, pp. 67-74), and Frest and others' (1991, p. 8) 1988 collections are the only known live, confirmed collections from the type locality. The Snake River physa were first documented downstream of C.J. Strike Reservoir during the 2009 inspection of samples collected by IPC from 1995 - 2003 (Keebaugh 2009, entire). In his review of over 19,000 physids collected from IPC's 917 collection events, Keebaugh (2009, p. 4) identified 52 live-when-captured individuals in 34 collection events matching the morphological characteristics of Snake River physa (Gates and Kerans 2011, p. 10). A subset (15 individual snails) was confirmed to be Snake River physa through genetic analysis (Gates and Kerans 2011, p. 4; Gates et al. 2013, p. 163).

At this time the Service considers the colonies downstream of Minidoka Dam and spillway as the upstream-most extent of the species' current range. Previous identification of Snake River physa from surveys upstream of Minidoka Dam by Pentec Environmental Incoporated (PEI) (1991) and Frest (1991, p. 8) at RKMs 1191 and 1205 (RMs 740 and 749) had not been confirmed through genetic analysis. In addition, 2011 surveys conducted by the USBOR upstream of Minidoka Dam, and downstream of American Falls Dam (approximately RKM 1135 - 1144 (RM 705 - 711)) have failed to yield any live Snake River physa or its shells (Newman 2012, *in litt.*).

In summary, the currently confirmed range of the Snake River physa is from RKM 1086 (RM 675) at Minidoka Dam downstream to RKM 592 (RM 368) near Ontario, Oregon. Within this 494 km (307 mi) range the species is generally rare and occurs in patchy distribution, with only 385 confirmed live-when-collected specimens taken over a 53-year period between 1959 and 2012. The species highest abundance and densities are currently found in the 18.5 km (11.5 mi) river segment downstream of Minidoka Dam where the population size and status of the Snake River physa appears to be relatively robust as well as stable. Conversely, the Snake River physa has not been found in the remainder of its range from Lower Salmon Falls Dam to Ontario, Oregon, since 2003, though survey efforts have been limited. Since the Snake River physa is rarely found at high densities, survey efforts may have been inadequate to detect the species in the Lower Salmon Falls Dam to Ontario, Oregon reach.

While Gates and Kerans (2010, p. 37) helped identify the spatial extent and distribution of Snake River physa downstream of Minidoka Dam their study design did not allow for an estimate of the population's size. Limited survey data from 2006 through 2012 indicates the Snake River physa occurs at relatively low densities (generally less than or equal to 32 individuals per sq. m, except in the Minidoka reach, referenced above, where up to 64 snails per sq. m were documented. The Service is not aware of any studies that would allow us to estimate, with any degree of confidence, current abundance estimates or long-term demographic trends for the Snake River physa.

## 2.3.1.5 Conservation Needs

Survival and recovery of the Snake River physa is considered contingent on "conserving and restoring essential mainstem Snake River and cold-water spring tributary habitats (USFWS 1995, p. 27)." The primary conservation actions outlined for this species are to "Ensure State water quality standards for cold-water biota ... " (USFWS 1995, p. 31). For more information on threats to the Snake River physa see section 2.4.1.2, *Factors Affecting Snake River Physa Snails in the Action Area.*

Priority 1 tasks consist of:

- Securing, restoring, and maintaining free-flowing mainstem habitats between the C.J. Strike Reservoir and American Falls Dam; and securing, restoring, and maintaining existing cold-water spring habitats.
- Rehabilitating, restoring, and maintaining watershed conditions.
- Monitoring populations and habitat to further define life history, population dynamics, and habitat requirements (USFWS 1995, pp. 27-28).

Priority 2 tasks consist of:

- Monitoring populations and habitat to further define life history, population dynamics, and habitat requirements.

- Updating and revising recovery plan criteria and objectives as more information becomes available, recovery tasks are completed, or as environmental conditions change (USFWS 1995, p. 28).

While substantial new information has been obtained on the species' distribution and habitat preferences since 1995, specifics on its water quality requirements or preferences are lacking, making effective planning difficult. In addition, the Snake River physa is only known to occur in the Snake River, a highly managed system with multiple anthropogenic influences. For this reason, conservation efforts may be restricted to implementation of clean water laws, water quality targets (e.g., TMDLs), maximizing minimum flows, and eliminating or minimizing impacts from extractive uses of waters and habitats within the Snake River. Maintaining or enhancing the habitat conditions currently existing in the Minidoka Reach of the Snake River is currently the most important factor to ensure the continuing existence of the Snake River physa. The existing river gradient and flows currently found below Minidoka Dam help ensure that the existing gravels, pebbles, and cobbles, that comprise most of the benthic habitat, remain free of fine sediments and excessive macrophyte growth. Human-caused or natural factors that reduce water quality or quantity in this reach can be expected to have adverse effects on the resident population of Snake River physa. The species does not occur with any degree of certainty elsewhere within its documented range, so conservation actions outside of the Minidoka Reach might not have direct beneficial effect on the species unless the limiting habitat factors found in these areas, such as sediment, nutrients, and inadequate flows, are addressed.

Recently, the Service’s 5-year status review (USFWS 2014a) recommended the following actions for Snake River physa conservation.

1. Gather, through research and surveys, additional information regarding basic biology and known range. Much remains unknown regarding the basic biology of the Snake River physa, including reproduction and life history traits, and diet preferences. In addition, surveys for presence within their current range have been limited in extent, especially outside of the Minidoka reach. Additional survey effort is needed in areas where they have been recently collected, particularly downstream of C.J. Strike and Swan Falls Dams, and within the Bruneau arm of C.J. Strike Reservoir.
2. Given the existing monitoring of Snake River physa below Minidoka Dam is a 5-year effort that was initiated in 2012, we recommend continued monitoring of that population, beyond the present effort, to further track population trends. In addition, if the Snake River physa can be reliably collected outside of the Minidoka reach, a monitoring program should be established in those areas to obtain population trends at a larger, rangewide scale.
3. Revise the Snake River Aquatic Species Recovery Plan with objectives and measurable criteria that are specific to the Snake River physa.

4. Additional work is needed to address factors that have led to the degradation of the Snake River physa's habitat. Actions may include decreasing nutrients, such as TP, and suspended sediment inputs to the Snake River from certain land uses within its range, while reducing existing substrate embeddedness and excessive macrophyte growth by modifying dam operations to enhance seasonal flows (i.e. increasing river flows during the summer months) in certain areas of their range.

## 2.3.2 Bliss Rapids Snail

### 2.3.2.1 Listing Status

The Bliss Rapids snail was listed as a threatened species on December 14, 1992 (57 FR 59244). Critical habitat for this species has not been designated. The recovery area for this species includes the Snake River and tributary cold-water spring complexes between RKM 880 to 942 (RM 547 to 585) (USFWS 1995, p. ii).

On December 26, 2006, the state of Idaho and the IPC petitioned the Service to delist the Bliss Rapids snail from the Federal list of threatened and endangered species, based on new information that the species was more widespread and abundant than determined at the time of its listing. The Service reviewed the information provided in the petition and initiated a 12-month review of the species' status. After compilation and review of new information, the Service hosted an expert panel of scientists and a panel of Service managers to reevaluate the species' status. On September 16, 2009, based on the findings of these expert panels, the Service posted a notice in the Federal Register stating the Bliss Rapids snail still warranted protection as a threatened species given its restricted range and the persistence of threats (USFWS 2008a, pp. 19-37).

### 2.3.2.2 Species Description

The shells of adult Bliss Rapids snails are 2.0 to 4.1 mm (0.08 to 0.16 in) long with 3.5 to 4.5 whorls, and are clear to white when empty (Hershler et al. 1994, p. 235). The species can occur in two different color morphs, the white or pale form, or the red form (Hershler et al. 1994, p. 240). It is not known what controls these color forms, but some populations do contain more than one color form.

### 2.3.2.3 Life History

The Bliss Rapids snail is dioecious (has separate sexes). Fertilization is internal and eggs are laid within capsules on rock or other hard substrates (Hershler et al. 1994, p. 239). Individual, life-time fecundity is not known, but deposition of 5 to 12 eggs per cluster have been observed in laboratory conditions (Richards et al. 2009c, p. 26). Reproductive phenology probably differs between habitats and has not been rigorously studied in the wild. Hershler et al. (1994, p. 239) stated that reproduction occurred from December through March. However, a more thorough investigation by Richards (2004, p. 135) suggested a bimodal phenology with spring and fall reproductive peaks, but with some recruitment occurring throughout the year.

The seasonal and inter-annual population densities of Bliss Rapids snails can be highly variable. The greatest abundance values for Bliss Rapids snails are in spring habitats, where they frequently reach localized densities in the tens to thousands per square meter (Richard 2004, p.

129; Richards and Arrington 2009, Figures 1-6, pp. 23-24). This is most likely due to the stable environmental conditions of these aquifer springs, which provide steady flows of consistent temperatures and relatively good water quality throughout the year. Despite the high densities reached within springs, Bliss Rapids snails may be absent from springs or absent from portions of springs with otherwise uniform water quality conditions. The reasons for this patchy distribution are uncertain but may be attributable to factors such as habitat quality (USFWS 2008a, pp. 11-13), competition from species such as the New Zealand mudsnail (Richards 2004, pp. 89-91), elevated water velocity, or historical events that had eliminated Bliss Rapids snails in the past (e.g., construction of fish farms at spring sources, spring diversion, etc.).

By contrast, river-dwelling populations are subjected to highly variable river dynamics where flows and temperatures can vary greatly over the course of the year. Compared to springs in which water temperatures range between 14° to 17°C (57.2 to 62.6°F), river temperatures typically fluctuate between 5° to 23°C (41 to 78.8°F), and river flows within the species' range can range from less than 4,000 cfs to greater than 30,000 cfs throughout the course of a year. These river processes likely play a major role in structuring and/or limiting snail populations within the Snake River (Dodds 2002, pp. 418-425; EPA 2002a, pp. 9-10–9-12). While Bliss Rapids snails may reach moderate densities (10s to 100s per $m^2$) at some river locations, they are more frequently found at low densities (≤10 per sq m) (Richards and Arrington 2009, Figures 1-6, pp. 23-24; Richards et al. 2009b, pp. 35-39) if they are present. It is likely that annual river processes play a major role in the distribution and abundance of the Bliss Rapids snail throughout its range within the Snake River by killing or relocating snails, and by greatly altering the benthic habitat (Palmer and Poff 1997, p. 171; Dodds 2002, pp. 418-425; Liu and Hershler 2009, p. 1296). While declines in river volume due to a natural hydrograph are typically less abrupt than load-following, they are of much greater magnitude, and hence it is logical to assume these natural events play an important role in limiting snail populations within the river.

A genetic analysis of the Bliss Rapids snail based on specimens collected from throughout its range (Liu and Hershler 2009, p. 1294) indicated that spring populations were largely or entirely sedentary, with little to no movement between springs or between springs and river populations. Most spring populations were highly differentiated from one another as determined by DNA microsatellite groupings. By contrast, river populations exhibited no clear groupings, suggesting that they are genetically mixed (Liu and Hershler 2009, p. 1295) and without genetic barriers, or they have not been isolated long enough to establish unique genetic differentiation. This pattern supports the suggestion made by other biologists that the river-dwelling population(s) of the Bliss Rapids snail exist in either a continuous river population (Liu and Hershler 2009, pp. 1295-1297) or as a metapopulation(s) (Richards et al. 2009b, entire) in which small, semi-isolated populations (within the river) provide and/or receive recruits from one another to maintain a loosely connected population.

### Habitat

The Bliss Rapids snail is typically found on the lateral and undersides of clean cobbles in pools, eddies, runs, and riffles, though it may occasionally be found on submerged woody debris (Hershler et al. 1994, p. 239) where it is a periphyton (benthic diatom mats) grazer (Richards et al. 2006, p. 59). This species is restricted to spring-influenced bodies of water within and associated with the Snake River from King Hill RKM 879 (RM 546) to Elison Springs RKM 972 (RM 604). The snail's distribution within the Snake River is within reaches that are unimpounded and receive significant quantities (ca. 5,000 cfs) of recharge from the Snake River Plain Aquifer (Clark and Ott 1996, p. 555; Clark et al. 1998, p. 9). It has not been recovered from impounded reaches of the Snake River, but can be found in spring pools or pools with evident spring influence (Hopper 2006, *in litt.*). With few exceptions, the Bliss Rapids snail has not been found in sediment-laden habitats, typically being found on, and reaching its highest densities on clean, gravel to boulder substrates in habitats with low to moderately swift currents, but typically absent from whitewater habitats (Hershler et al. 1994, p. 237).

Previous observations have suggested that the Bliss Rapids snail is more abundant in shallower habitats, but most sampling has been in shallow habitat since deeper river habitat is more difficult to access. Clark (2009, pp. 24-25) used a quantile regression model that modeled a 50 percent decline in snail abundance for each 3 m (10 ft) of depth (e.g., snail density at 3 m was approximately 50 percent less than that at shoreline (p. 24)). Richards et al. (2009a, pp. 6-7) used an analysis of variance (ANOVA) to assess snail densities at 1-meter intervals and only found a statistical difference (increase) in densities in the first meter of depth, with no declining trends with increasing depth. Nonetheless, these authors suggest that greater than 50 percent of the river population could reside in the first 1.5 meter (5 ft) depth zone of the Snake River (Richards et al. 2009a, Appendix 1).

**Diet**

Richards (2004, pp. 112-120) looked at periphyton (benthic diatoms) consumption by the Bliss Rapids snail and the New Zealand mudsnail (*Potamopyrgus antipodarum*) in competition experiments. He described the Bliss Rapids snail as a “bulldozer” type grazer, moving slowly over substrates and consuming most, if not all, available diatoms. The dominant diatoms identified in his controlled field experiments consisted of the bacilliariophyt genera *Achananthus* sp., *Cocconeis* sp., *Navicula* sp., *Gomphonema* sp., and *Rhoicospheni*a sp., although the species composition of these and others varied greatly between seasons and location. At least one species of periphytic green algae was also present (*Oocystis* sp.). Richards (2004, p. 121) suggested that the Bliss Rapids snail appeared to be a better competitor (relative to the New Zealand mudsnail) in late successional diatom communities, such as the stable spring habitats where they are often found in greater abundance than the mudsnail.

## 2.3.2.4 Status and Distribution

In the Recovery Plan for the Snake River snails (USFWS 1995), the Service reported that the Bliss Rapids snails’ range extends along the Snake River from Indian Cove Bridge (RKM 845.4 (RM 525.4)) to Twin Falls (RKM 982.3 (RM 610.5)) and that it likely occurred upstream of American Falls in a disjunct population where it had been reported from springs (RKM 1207 (RM 750)) (USFWS 1995, p. 10). The current documented range of extant populations is more restricted; this species has been identified from the Snake River near King Hill (RKM 878.5 (RM 546)) to below Lower Salmon Falls Dam (RKM 922 (RM 573)), and from spring tributaries as far upstream as Ellison Springs (RKM 972 (RM 604)) (Bates et al. 2009, p. 100). The “American Falls” occurrence was later discounted after multiple surveys failed to relocate the species (USFWS 2008a, pp. 5-6). There is an isolated river population that occupies a limited bypass reach (Dolman Rapids) between the Upper and Lower Salmon Falls reservoirs (Stephenson 2006, p. 6).

Studies by the IPC found the species to be more common and abundant within the Snake River (RKM 879 to 920 (RM 546 to 572)) than previously thought, although typically in a patchy distribution with highly variable abundance (Bean 2006, pp. 2-3; Richards and Arrington 2009, Figures 1-6, pp. 23-24). Most, if not all, of the river range of the species is in reaches (Lower Salmon Falls and Bliss) where recent records show an estimated 5,000 cfs of water entering the Snake River from numerous cold springs from the Snake River Plain Aquifer (Clark and Ott 1996, p. 555; Clark et al. 1998, p. 9). This large spring influence, along with the steep, unimpounded character of the river in these reaches, improves water quality (temperature, dissolved oxygen, and other parameters) and helps maintain suitable habitat (low-sediment cobble) for the snail that likely contributes to the species' presence in these reaches (Hershler et al. 1994, p. 237). It is noteworthy that the species becomes absent below King Hill, where the river loses gradient, begins to meander, and becomes more sediment-laden and lake-like. Although Bliss Rapids snail numbers are typically lower within the Snake River than in adjacent spring habitats, the large amount of potential habitat within the river suggests that the population(s) within the river is/are low-density but large compared to the smaller, isolated, typically high-density spring populations (Richards and Arrington 2009, Figures 1-6, pp. 23-24). These river reaches comprise the majority of the species designated recovery area.

The species' range upstream of Upper Salmon Falls Reservoir RKM 941 to 972 (RM 585-604)) is restricted to aquifer-fed spring tributaries where water quality is relatively high and human disturbance is less direct. Within these springs, populations of snails may occupy substantial portions of a tributary (e.g., Box Canyon Springs Creek, where they are scattered throughout the 1.8 km (1.1 mi) of stream habitat) or may be restricted to habitats of only several square meters (e.g., Crystal Springs). Spring development for domestic and agricultural use has altered or degraded a large amount of these habitats in this portion of the species' range (Hershler et al. 1994, p. 241; Clark et al. 1998, p. 7), often restricting populations of the Bliss Rapids snail to spring source areas (Hershler et al. 1994, p. 241).

It is difficult to estimate the density and relative abundance of Bliss Rapids snail colonies. The species is documented to reach high densities in cold-water springs and tributaries in the Hagerman reach of the middle Snake River (Stephenson and Bean 2003, pp. 12, 18; Stephenson et al. 2004, p. 24), whereas colonies in the mainstem Snake River (Stephenson and Bean 2003, p. 27; Stephenson et al. 2004, p. 24) tend to have lower densities (Richards et al. 2006, p. 37). Bliss Rapids snail densities in Banbury Springs averaged approximately 32.53 snails per square foot (350 snails per square meter) on three habitat types (vegetation, edge, and run habitat as defined by Richards et al. 2001, p. 379). Densities greater than 5,800 snails per sq m (790 snails per sq ft) have been documented at the outlet of Banbury Springs (Morgan Lake outlet) (Richards et al. 2006, p. 99). In an effort to account for the high variability in snail densities and their patchy distribution, researchers have used predictive models to give more accurate estimates of population size in a given area (Richards 2004, p. 58). In the most robust study to date, predictive models estimated between 200,000 and 240,000 Bliss Rapids snails in a study area measuring 625 sq m (58.1 square ft) in Banbury Springs, the largest known colony (Richards 2004, p. 59). Due to data limitations, this model has not been used to extrapolate population estimates to other spring complexes, tributary streams, or mainstem Snake River colonies. However, with few exceptions (i.e., Thousand Springs and Box Canyon), Bliss Rapids snail colonies in these areas are much smaller in areal extent than the colony at Banbury Springs, occupying only a few square feet.

#### 2.3.2.5 Conservation Needs

Survival and recovery of the federally listed snails in and adjacent to the Snake River, Idaho, is considered contingent on "conserving and restoring essential main-stem Snake River and cold-water spring tributary habitats" (USFWS 1995, p. 27). Given the Bliss Rapids snail's habit of utilizing both river and spring habitats, the above stated recovery goal is critical. The generalized priority tasks for all of the listed Snake River snails, including the Bliss Rapids snail, consist of the following. For more information on threats to the Bliss Rapids snail see section 2.4.2.2, *Factors Affecting the Bliss Rapids Snail in the Action Area.*

*Priority 1*

- Securing, restoring, and maintaining free-flowing main-stem habitats between the C.J. Strike Reservoir and American Falls Dam, and securing, restoring, and maintaining existing cold-water spring habitats.

- Rehabilitating, restoring, and maintaining watershed conditions (specifically: cold, unpolluted, well-oxygenated flowing water with low turbidity. (*ibid.*, p. 1)).

- Monitoring populations and habitat to further define life history, population dynamics, and habitat requirements (USFWS 1995, pp. 27-28).

*Priority 2*

- Updating and revising recovery plan criteria and objectives as more information becomes available, recovery tasks are completed, or as environmental conditions change (USFWS 1995, p. 28).

Given the known limited distribution of the Bliss Rapids snail and its specific habitat requirements, maintaining or improving spring and river habitat conditions within its range is the primary need for this species' survival and recovery. The Bliss Rapids snail reaches its highest densities in cold-water springs dominated by cobble substrates and free, or relatively free, of fine sediments, and with good water quality. Protecting these habitats that contain Bliss Rapids snail populations is critical to their survival and recovery.

Ensuring that water quality within the Snake River is not degraded is important for sustaining the species' river-dwelling populations. Since water quality appears to be of crucial importance to the species, protection of the Snake River Plain Aquifer is a priority. The aquifer is the source of water for the springs occupied by the snail and serves a major role in maintaining river water quality within the species' range. More information regarding water quality is found in section 2.4.2.2, *Factors Affecting Bliss Rapids Snail in the Action Area*.

### 2.3.3 Banbury Springs Lanx

#### 2.3.3.1 Listing Status

The Banbury Springs lanx or limpet (*Lanx* species) was listed as endangered on December 12, 1992. Critical habitat has not been designated for this species. The recovery area for this species includes tributary cold-water spring complexes to the Snake River between RKM 941.5 to 948.8 (RM 584.8 to 589.3) (USFWS 1995, p. ii).

## 2.3.3.2 Species Description

This snail is a member of Lancidae, a small family of pulmonates (snails that lack gills) endemic to western North America. The species was first discovered in 1988 and has not been formally described. It is distinguished by a cap-shaped shell of uniform red-cinnamon color with a subcentral apex or point. Length from 2.4 to 7.1 mm (0.9 to 0.28 in), height ranges from 1.0 to 4.3 mm (0.03 to 0.17 in), and width ranges from 1.9 to 6.0 mm (0.07 to 0.24 in) (USFWS 1995, p. 12).

## 2.3.3.3 Life History

Very little is known of the life history of the Banbury Springs lanx. The species has been found only in spring-run habitats in swift-moving, well-oxygenated, clear, cold (15 $^{o}$ to 16 $^{o}$C (59 to 60.8 $^{o}$F)) waters on boulder or cobble-sized substrate. They are most often found on smooth basalt and avoid surfaces with large aquatic macrophytes or filamentous green algae. Beak Consultants (1989, p. 6) reported the species, originally identified as *Fisherola nuttalli*, at depths ranging from 46 to 61 centimeters (cm) (18 to 24 in) on boulder substrates. Frest and Johannes (1992, p. 29) found the species in water as shallow as 5 cm (2 in), but the snails were more typically found at depths of around 15 cm (6 in). Because lancids lack gills, gas exchange primarily occurs over the tissues of the mantle cavity. This makes these snails dependent on well-oxygenated water and particularly sensitive to fluctuations in dissolved oxygen (Frest and Johannes 1992, p. 27). Egg cases are attached to rocks between April and July and have been observed to contain up to six eggs each. Juveniles appear from May through July.

## 2.3.3.4 Status and Distribution

When it was listed, the Banbury Springs lanx was only found in three coldwater spring complexes along the Snake River, all within 7 kilometers (km) of each other; Thousand Springs, Box Canyon, and Banbury Springs. Since listing it has been discovered in one additional coldwater spring complex, Briggs Springs, less than 2 km (1.2 mi) upstream on the Snake River from the previously southernmost occupied spring complex, Banbury Springs. All lanx colonies are isolated from each other and restricted to their present locations, resulting in no possible conduit for natural dispersal or range expansion (USFWS 2006b, p. 7). The population size, abundance, and trends of the lanx are largely uncertain as little density and trend information exists (USFWS 2013a, p. 5).

**Thousand Springs**

At Thousand Springs, the lanx is found sporadically in an outflow of only one of the springs, which discharges into the North Channel, near the Minnie Milner Diversion (Frest and Johannes 1992, pp. 26- 27; Hopper 2006b, *in litt.*, pp. 1-2).

In the Thousand Springs Preserve near Minnie Milner Springs, Frest and Johannes (1992, p. 27) described the lanx colony as “sporadically distributed and cryptic.” Average population density in this area was between 16 to 48 individuals per square meter (sq m) (1.5 to 4.4 individuals per sq ft) and the total number of individuals in this area was estimated between 600 to 1,200 (Frest and Johannes 1992, p. 27). Service personnel found nine individuals while visually inspecting 40 cobbles in January of 2006 (Hopper 2006b, *in litt*., pp. 1 to 3).

Survey data from 2012 and 2013 indicate that this population of Banbury Springs lanx may be in danger of extirpation (Burak and Hopper 2013, p. 24). It is unknown what has caused this population to reach such low densities as habitat conditions and the limited water quality data collected to date do not indicate noticeable differences between this population and the remaining three. Even when this population was first discovered in 1991, it was not considered to be robust, with the total population estimated at 600-1,200 individuals at a density of 16-48 snails/sq m (Frest and Johannes 1992, p. 27). In 2013, the Service estimated the density to be 3.52 snails/sq m (0.3 snails/sq ft), with a population that is likely less than 200 individual's given that entire known distribution of this population was sampled.

At Thousand Springs, much of the spring water that originally cascaded down the basalt cliffs is now diverted into a concrete flume for delivery into the Thousand Springs hydroelectric project (Stephenson et al. 2004, p. 4). The Thousand Springs hydroelectric project is located on private land and was constructed in 1912. We do not have information regarding the historical or current discharge of water from the Thousand Springs complex but the diversion of much of the springflow into a power generating facility likely destroyed and/or modified suitable Banbury Springs lanx habitat. It is not known how the diversion has affected historical population density and/or spatial distribution of the species. However, at present the Banbury Springs lanx is only known to exist in one section of the North Channel near Minnie Milner Springs (Hopper 2006b, *in litt*., pp. 1-2).

**Box Canyon Springs**

Box Canyon Creek is fed by Box Canyon Spring. It is approximately 1.75 km (1.1 mi) in length and joins the Snake River just upstream of the Thousand Springs complex at RKM 946 (RM 588). In 2006, Box Canyon Creek discharge was the lowest in 50 years (USGS 2006, p. 1). Beginning in 2004, flows in Box Canyon Creek dropped below 300 cfs for the first time in its recorded history (USGS 2006, p. 1). The majority of the water originating from Box Canyon Creek is diverted upstream of the existing Banbury Springs lanx colony into a flume for delivery to a commercial aquaculture facility (Taylor 1985, p. 2; Langenstein and Bowler 1991, p. 185). The Banbury Springs lanx is currently known from Box Canyon Creek between Sculpin Pool and the diversion (Taylor 1985, p. 11; USGS 1994, *in litt*., pp. 1-2; Maret 2002, *in litt*., p. 3; Hopper 2006a, *in litt*., pp. 1-2). The diversion of approximately 86 percent of this creek's water (Langenstein and Bowler 1991, p. 185) constitutes a significant modification of potential and possibly historical Banbury Springs lanx habitat.

Within Box Canyon, Banbury Springs lanx have been found within stream habitat between Sculpin Pool on the downstream end to the hatchery water diversion/flume on the upstream end. This is approximately 150-175 m (492 to 574 ft) in length. In 2012 and 2013 the Service monitored the Banbury Springs lanx colony at Box Canyon. In 2012, 139 cobbles were sampled and 220 snails were found, equating to a density estimate of 1.57 snails/ cobble, which translates to 62.5 snail/sq m across the entire sampling area (Burak and Hopper 2012, p. 13). This is approximately 20.5 more snails/sq m than estimated in 2013. While there are only 2 years of data to compare, these results indicate population estimates for 2013 were less than 2012, and while not significant (U=10944.5, P=0.065), it may be the beginning of a downward trend. Additional monitoring is needed to confirm or discount this trend.

As was discovered during 2012 monitoring (Burak and Hopper 2012, p. 14), several snails were found at the top water level during 2013 monitoring, with several individuals above the water level, where the rock surface was still wet. This further supports the conclusion that this type of behavior is not an anomaly, and indicates Bliss Rapids snails are able to reside just outside the water column when microclimate conditions are ideal.

**Banbury Springs**

The Banbury Springs complex is the type locality, or the physical location from which the Banbury Springs lanx was originally collected and identified as a unique species (Reed et al. 1989, p. 2; Frest 2006, *in litt.*, p. 1; Figure 1). The actual springs of Banbury Springs originate from basalt cliffs and talus slopes about 50 m (164 ft) above the Snake River. The entire flow of these springs is captured in Morgan Lake, a man-made lake with a levee separating the lake from the Snake River. This lake creates lentic (still water) habitat and inundates the riffle/rapids habitat that likely existed previously at the confluence of Banbury Springs Creek and the Snake River. Currently, the Banbury Springs lanx is only found in the lower riffle complex in one of five spring outflows that enter into Morgan Lake (Hopper 2006a, *in litt.*, pp. 1-3).

Additional impacts to Banbury Springs lanx habitat occurred when the Boy Scouts of America previously used Morgan Lake for recreational activities such as canoeing and swimming (Wood 1998, *in litt.*, p. 1). This use of Banbury Springs was discontinued by 1998 but dilapidated bridges and remnant trails that crossed the riffle complex just upstream of where the lanx occur are still evident (Hopper 2006a, *in litt.*, pp. 1-3). Current recreational use of Banbury Springs is evidenced by relatively recent shotgun hulls, discarded by waterfowl hunters, observed in the streambed on top of a Banbury Springs lanx colony (Hopper 2006a, *in litt.*, pp. 1-3). Recreational users at Banbury Springs could potentially trample individual lanx at the lower section of the spring outflow.

Life history data (density) was collected by the IPC at the Banbury Springs site in 1995, 1996, 2000, 2001, 2002, and 2003 (Finni 2003a, p. 34; Finni 2003b, p. 24; Finni 2003c, p. 15; Stephenson and Bean 2003, p. 26; Stephenson et al. 2004, p. 23). However, the results are difficult to compare across years, because the methods have not been applied consistently. Generally, average density between years is comparable across the 6 years of surveys, with the exception of 2002 and 2003, where one or two outliers per year resulted in skewed averages and standard deviations.

In contrast, surveys conducted in 2013 indicate that the population at the Banbury Springs monitoring site is in a continued decline (Burak and Hopper 2013, p. 14). In 2008, 2011, and 2012, average density of snails/sq m was 65, 36.75, and 24.67 respectively (Burak and Hopper 2012, p. 11). This is approximately 49, 20.75, and 8.67 more snails/sq m than found in 2013, or approximately four times fewer snails/sq m just 5 years ago. Even though we do not know if these results can be extrapolated to the rest of the occupied habitat at Banbury Springs, the monitoring site was initially set up with the goal of incorporating the highest known Banbury Springs lanx density area at Banbury Springs and previous stream-wide surveys have indicated the species was much less commonly encountered than within the monitoring site. This continuing decline in the lanx population within the monitoring site is of great concern.

**Briggs Springs**

Surveys conducted by the USGS (1994, *in litt.*, pp. 3, 4) describe the Banbury Springs lanx as common with six or more individuals per cobble. A cursory survey performed by Service personnel found Banbury Springs lanx in the area described by USGS (1994, *in litt.* , pp. 3 -4) just upstream and downstream of the USGS gauging station at Briggs Springs Creek. We visually inspected 20 cobbles and found an average of 4.7 individuals per cobble (Hopper 2006b *in litt.*, pp. 1-3).

The Service monitored two sites at Briggs Springs in 2012 and one in 2013 (upper and lower) (Burak and Hopper 2013, p. 21). Briggs lower was the only site of all sites monitored in 2013 that the number of Banbury Springs lanx counted rose from 2012 to 2013, increasing from 88 snails counted in 2012 to 188 in 2013. Briggs upper decreased from 88 snails counted in 2012 to 59 counted in 2013.

## 2.3.3.5 Conservation Needs

The Service's 5-year status review for the lanx (USFWS 2006b, entire) includes the following recommendations for lanx conservation. For more information on threats to the Banbury Springs lanx see section 2.4.3.2, *Factors Affecting the Banbury Springs Lanx in the Action Area.*

**Update the Recovery Plan**

Update the Snake River Aquatic Species Recovery Plan to include new information that we have learned since the listing of this species in 1992. The existing recovery plan was finalized in 1995 and does not contain measurable recovery criteria specific to the Banbury Springs lanx; the existing criteria were written to encompass all species covered by the recovery plan. New recovery criteria should be formulated to include monitoring components as listed below that will enable a determination of whether each colony of Banbury Springs lanx is stable, declining, or increasing and whether the trend is increasing or stable across at least a 5-year period.

**Monitoring Program**

Implement a non-intrusive annual monitoring program at each of the four colonies (Thousand Springs, Box Canyon, Banbury Springs, and Briggs Springs) on a recurring basis. Comparisons can then be made across years to determine whether Banbury Springs lanx colonies are declining, stable, or increasing. Measurements should be performed in January when vegetation will be stunted allowing for more efficient detection; this time of year is when body sizes are largest and predates egg-laying. The Banbury Springs lanx occurs in low densities in some areas and monitoring should be halted if it is believed that the population is being reduced as a result of the monitoring effort.

**Life History Experiments**

Life history experiments with live Banbury Springs lanx should be performed in a laboratory setting to better understand the life history of this species in a controlled environment. Life history parameters of interest would include but not be limited to: growth rate, size at reproduction, number of egg capsules/individual, location of egg capsules, self-fertilization or fertilization from another individual, dispersal, feeding, temperature preference/maximums and minimums, and dissolved oxygen preference/maximum/minimum.

**Translocation**

As the Banbury Springs lanx are currently found at only four, isolated locations over 9.7 RKM (6 RM) of the Snake River, translocation of Banbury Springs lanx should be conducted to other suitable and protected coldwater spring habitats to ensure the continued existence of this species. Possible locations for translocation of the Banbury Springs lanx would be: (1) upstream of the waterfall at Box Canyon and the adjacent four spring locations at Banbury Springs; and, (2) Box Canyon, upstream of the falls, where the New Zealand mudsnail does not occur (note: caution should be exercised while transporting Banbury Springs lanx upstream of the falls to avoid contaminating this habitat with the mudsnail). As genetic studies are not yet available that show how colonies are related, we suggest that Banbury Springs lanx from Box Canyon be used to introduce the lanx upstream of the waterfall at Box Canyon, and that lanx from Banbury Springs be used to introduce snails to the adjacent springs at Banbury Springs. At Box Canyon, Banbury Springs lanx should be introduced near the spring origin to facilitate natural colonization of habitat downstream of the introduction site.

# 2.3.4 Bruneau Hot Springsnail

## 2.3.4.1 Listing Status

The Bruneau hot springsnail was listed as endangered on June 17, 1998 (63 FR 32981). Critical habitat for this species has not been designated. The Service completed a Five-year review on the status of the Bruneau hot springsnail and concluded that the snail should remain listed as endangered (USFWS 2002a, p. 28).

## 2.3.4.2 Species Description

Adult Bruneau hot springsnails have a small, globose (short, fat, rounded) to lowconic (short and cone-shaped, without many whorls) shell reaching a length of 5.5 mm (0.22 in) with 3.75 to 4.25 whorls (USFWS 2002a, pp. 1).

Fresh shells are thin, transparent, white-clear, appearing black due to pigmentation (Hershler 1990, p. 805). In addition to its small size (less than 2.8 mm [0.11 in] shell height), distinguishing features include a verge (penis) with a small lobe bearing a single distal glandular ridge and elongate, muscular filament (USFWS 2002a, p. 2).

## 2.3.4.3 Life History

The Bruneau hot springsnail is a member of the family Hydrobiidae. The family Hydrobiidae has a worldwide distribution that is represented in North America by approximately 285 species in 35 genera (Sada 2006, p. 1). In North America, most species occupy springs, and their abundance and diversity is notably high in the Great Basin, where approximately 80 species from the genus *Pyrgulopsis* occur (Hershler and Sada 2002, p. 255). Hydrobiids are dioecious (having separate sexes), and lay single oval eggs on hard substrate, vegetation, or another snail shell (Mladenka 1992, p. 3). *Pyrgulopsis* is the most common genus in the family with approximately 131 described species that are considered valid, 61 percent of which occur in the Great Basin (Hershler and Sada 2002, p. 255).

These tiny gill-breathing springsnails are aquatic throughout their life cycle (Hershler and Sada 2002, p. 255). Females from this genus are oviparous (producing egg capsules that are deposited on substrates) (Hershler and Sada 2002, p. 256). The Bruneau hot springsnail has a 1 to 1 male/female sex ratio (Mladenka 1992, p. 46), and reaches sexual maturity at approximately two months (maximum size at four months) with reproduction occurring year round at suitable temperatures (20-35 °C); 68-95 (°F)) (Mladenka 1992, p. 3). Male genitalia are evident by the time this species reaches a shell height of 1.4 mm (0.06 in ), and any snail lacking male genitalia at that size or greater is considered female (Mladenka and Minshall 2001, pp. 208-209). The egg capsules of the Bruneau hot springsnail are relatively small (approximately 0.3 mm (0.01 in) in diameter) (Mladenka and Minshall 2001, p. 208; Mladenka 1992, p. 40). After emergence, the Bruneau hot springsnail are transparent until they reach approximately 0. 7 mm (0.28 in) when black pigmentation appears in the body tissue (Mladenka and Minshall 2001, p. 208; Mladenka 1992, p. 40). Growth rates (field) ranged from 0.010 to 0.022 mm/day (0.0004 to 0.0009 in/day) (Mladenka and Minshall 2001, p. 208; Mladenka 1992, p. 40) while the number of juveniles per female ranged from 0 to 18.5 individuals/month (Mladenka 1992, p. 45).

The Bruneau hot springsnail is seldom found in standing or slow-moving water and was shown in the laboratory to tolerate higher current velocities than present in nature (Mladenka 1992, pp. 87 and 88). This species has a temperature tolerance between 11-35°C (52-95°F) (Mladenka 1992, p. 85).

This species appears to be an opportunistic grazer and seems to prefer colored algal mats, which contain higher numbers of diatoms relative to lighter algae (Mladenka 1992, p. 81). A movement study performed in the laboratory showed that the Bruneau hot springsnail is capable of crawling 1 centimeter per minute (cm/min) (0.3 in/min) (Myler and Minshall1998, pp. 53, 54). Additionally, this species prefers to move over wetted substrate (substrate covered with flowing water), and has a propensity to move upstream vs. downstream (Myler and Minshall 1998, pp. 53, 54). In a field substrate preference experiment, the Bruneau hot springsnail preferred cobbles (> 10 cm in diameter (4 in)) over gravel (2-10 mm) (0.08-0.4 in) and sand/silt (< 2 mm) (< 0.08 in) (Myler 2000, p. 26). In a field experiment where an artificial substrate (plexiglass 1 m by 1 m (39 in by 39 in)) was placed under thermal springflow near Mladenka's Site 2, the Bruneau hot springsnail was observed to colonize at a rate of 1 snail per hour with a carrying capacity of 300 snails per square meter (snails/sq m ) (Myler 2000, p. 42).

Water temperature appears to be the predominant factor that influenced abundance at long term monitoring sites (Mladenka 1992, p. 90). Bruneau hot springsnails have often been observed in the geothermal spring/river interface in surveys conducted since 1998 (Myler 2005, p. 8). Occurrence in this location likely facilitated individuals to optimize temperature preference. In a desiccation experiment performed in the laboratory, Bruneau hot springsnail mortality occurred between 2-4 hours (Mladenka 1992, p. 53), but it is unknown how this species disperses between suitable habitats under desiccated conditions. This species has been observed to drift into the Bruneau River when it is disturbed from its geothermal spring habitat (Myler 2005, p. 8). Drift as a mechanism of downstream dispersal is possible for this species. However, it is assumed that since this species has no locomotion abilities in the river current, many drifting individuals that do not settle in geothermal springs will likely perish due to their strict temperature requirements. Many questions regarding the dispersal and long-term exposure to cold river water for this species remain unanswered. Although the Bruneau hot springsnail have been observed in the

Bruneau River proper (Mladenka and Minshall 2003, pp. 7, 8), occurrences have been directly associated with geothermal upwelling on the river bottom (Myler 2005, pp. 3, 4). No evidence exists to suggest that the Bruneau hot springsnail is not a thermophilic species. In late summer (July to August) water temperatures in the Bruneau River are within the temperature tolerance of the Bruneau hot springsnail. However, we know of no surveys that have located Bruneau hot springsnails in cold water or outside of geothermal upwelling zones in the Bruneau River.

## 2.3.4.4 Status and Distribution

The Bruneau hot springsnail is endemic to thermal springs and seeps that occur along 8 km (5 mi) of the Bruneau River in southwest Idaho. The Bruneau hot springsnail currently occurs in geothermal springs on both the east and west sides of the Bruneau River with a distribution extending 4.4 km (2.73 mi) downstream of the confluence of Hot Creek and the Bruneau River, and 4.4 km (2.73 mi) upstream from the confluence of Hot Creek and the Bruneau River (Mladenka 1992, p. 68). As of November 2006, Hot Creek no longer flowed at the Indian Bathtub site and was completely dry. Hot Creek now begins flowing approximately 503 m (550 yards (yd)) downstream (Myler 2006, p. 7).

During the 15 year period from 1991 to 2006, the total number of geothermal springs along the Bruneau River upstream of Hot Creek occupied by Bruneau hot springsnails declined from 146 geothermal springs in 1991 to 66 in 2006 (Myler 2006, pp. 2 – 6, Figure 2). In 2011, the Service found that there were only 31 springs occupied by hot springsnails upstream of Hot Creek; snail density in 26 of these springs was categorized as low or very low (Hopper et al. 2012, p. 15). As documented in the 2006 monitoring report (Myler 2006, pp. 2-6, Figure 2): "In the past 10 years, the total number of geothermal springs surveyed along the Bruneau River downstream of Hot Creek have increased from 20 in 1996, to 88 in 2006 which we attribute to declining geothermal water levels and fragmentation of remaining geothermal springs sites." In other words, as the geothermal aquifer declines, geothermal springs often decrease in size and become fragmented into smaller geothermal springs and seeps. For example, what was counted as a single large spring in 1991-1993 is currently counted as multiple smaller springs and seeps with a smaller total area that represents a net decrease in habitat and species density. However, as of 2006, geothermal springs downstream of Hot Creek occupied by the snail had declined from 50 in 2003, to 26 in 2006 (Myler 2006, pp. 2-6, Figure 2). In 2012, the Service reported that the number of occupied springs downstream of Hot Creek had declined from 59 percent occupied (in 2010) to 19 percent (in 2011) on the west bank and a 22 percent decline in occupied springs on east bank (from 18 occupied to 14) (Hopper et al. 2012, pp. 10-12).

The relative density of Bruneau hot springsnails upstream of Hot Creek has also changed compared to surveys of 1991, 1993, 1996, 2003, and 2004 (Myler 2006, p. 6; Figure 4). In 2006, only 4 geothermal springs sites had medium densities of snails and no occupied sites had high densities of snails, compared to 33 medium and 11 high density sites (of 110 total occupied sites) located in 1996. The numbers of high and medium density snail sites show a decreasing trend since 1991, while the number of low density snail sites and sites without snails has increased (Myler 2006, p. 6; Figure 4). In the area downstream of Hot Creek, high and medium density sites have remained relatively constant, while the number of geothermal springs with low density or lacking snails have increased.

Thermal Infrared (TIR) images of the recovery area were collected by aircraft in November 2005 and showed 1,079 sq m of geothermal spring/seep habitat >14°C (57°F) upstream of Hot Creek. Downstream of Hot Creek (including Hot Creek), the measured geothermal habitat >14°C (57°F) measured 5,024 sq m and is attributed to a few very large springs. However, approximately 1,600 sq m of this downstream habitat had water temperatures that exceed the Bruneau hot springsnail's maximum temperature tolerance of 35 °C (95 °F). In addition, at least two large geothermal springs were detected that discharge underneath the Bruneau River as geothermal upwelling zones that are occupied by the Bruneau hot springsnail (Myler 2005, pp. 3-4). In 2004, the average water temperature in one thermal upwelling zone was 24.7°C (76.4°F) (Myler 2005, p. 4). In 2006, only two major geothermal upwelling zones were known as compared to 66 occupied geothermal springs and seeps (Myler 2006, pages 2-4).

As groundwater levels continue to decline, the Bruneau hot springsnail's remaining geothermal spring habitat flowing into the Bruneau River will continue to decline in number, and will become more fragmented. At some time in the future, the thermal upwelling zones in the Bruneau River may become more important in providing Bruneau hot springsnail habitat, but will also be affected by the declining geothermal aquifer and will likely follow the same decline as the geothermal springs. While the Bruneau hot springsnail has been found in recent surveys in these upwelling zones, we currently lack information on how these habitats are being used by this species. Further research in these geothermal upwelling areas and how the Bruneau hot springsnail uses them is currently planned for the future by the Service. We know that various non-native fishes (i.e. *Tilapia zilli* and *Gambusia affinis)* observed in laboratory studies (Myler and Minsahll 1998, p. 53) feed upon Bruneau hot springsnails, and also utilize parts of the Bruneau River that are influenced by geothermal water (Mladenka and Minshall 1993, p. 7; Myler 2005, p. 7). In addition, Bruneau hot springsnails in this habitat may be subject to increased scouring and removal from naturally occurring high runoff events in the Bruneau River.

In summary, the two largest Bruneau hot springsnail colonies (Hot Creek and Mladenka's Site 2) previously known from earlier reports (Taylor 1982b, p. 5; Mladenka 1992, p. 49) have been extirpated. Discharge from many of the geothermal springs along the Bruneau River is difficult to measure, therefore, the decline of the geothermal springflows is difficult to quantify. Photo points have been used for many of the surveys and definite reductions in geothermal spring discharges are easily observed from 1991 and 1993 surveys to present. Geothermal spring sites that have gone dry such as Indian Bathtub, Mladenka's Site 2, and Site U4E, demonstrate the drastic reduction in the geothermal aquifer at different locations.

## 2.3.4.5 Conservation Needs

Threats identified at the time of listing in 1998 still remain. The major threat to this species is the continued decline of the geothermal aquifer resulting in a decrease in suitable geothermal spring habitat for the Bruneau hot springsnail. In the 5-year status review the Service (USFWS 2007, p. 28) recommended that no change in the listing status be made to the Bruneau hot springsnail and that it should remain listed as endangered under the Act. For more information on threats to the Bruneau hot springsnail see section 2.4.2.2, *Factors Affecting the Bruneau Hot Springsnail in the Action Area.*

According to the 2007 5-year status review (USFWS 2007) recovery of the Bruneau hot springsnail is dependent upon meeting the five criteria listed below. The 5-year status review also provided the status for each criterion and these are included.

1. *Criterion:* Water levels in the geothermal aquifer are being maintained at 815 m (2,674 ft) above sea level (measured in October) at groundwater monitoring wells 03 BDC1, 03BDC2, and 04DCD1.

   *Status:* Geothermal water levels in wells 03 BDC1, 03BDC2, and 04DCD1 average 812 m above sea level and are showing a declining trend (Myler 2007, Appendix 4). This criterion has not been met.

2. *Criterion:* The geothermal springs number more than 200 in October, and are well distributed throughout the recovery area. (This value approximates the 204 geothermal springs from 1996 surveys (Mladenka and Minshall 1996)).

   *Status:* The total number of geothermal springs in 2006 was 154 (Myler 2006, pp. 2-4) and have declined since the 1996 surveys (Myler 2006, p. 5). This criterion has not been met.

3. *Criterion:* Greater than two-thirds of available geothermal springs (approximately 131 geothermal springs) are occupied by medium to high density populations of the Bruneau hot springsnail (1,650 to 10,000 $m^2$) (Rugenski and Minshall 2002).

   *Status:* In 2006, there were only 66 geothermal springs that were occupied by the Bruneau hot springsnail out of a total of 154 springs (Myler 2006, pp. 2-4). There were no geothermal springs in 2006 with high density (9,941/$m^2$± 4983), with medium density (1,618/$m^2$± 693), and 62 were low density (353/$m^2$± 293) (Myler 2006, p. 6). Given that only 4 out of 154 springs have medium to high density populations, the two-thirds criterion has not been met.

4. *Criterion:* Regulatory measures are adequate to permanently protect groundwater against further reductions.

   *Status:* Given that the geothermal aquifer and the number of geothermal springs are on a declining trend, regulatory mechanisms are inadequate or have not been implemented to protect the geothermal aquifer system from further reductions. This criterion has not been met.

# 2.3.5 Bull Trout

## 2.3.5.1 Listing Status

The coterminous United States population of the bull trout was listed as threatened on November 1, 1999 (64 FR 58910). The threatened bull trout occurs in the Klamath River Basin of south-central Oregon, the Jarbidge River in Nevada, north to various coastal rivers of Washington to the Puget Sound, east throughout major rivers within the Columbia River Basin to the St. Mary-Belly River, and east of the Continental Divide in northwestern Montana (Bond 1992, p. 4; Brewin and Brewin 1997, pp. 209-216; Leary and Allendorf 1997, pp. 715-720). The Service completed a 5-year status review in 2008 and concluded that the bull trout should remain listed as threatened (USFWS 2008b, p. 53).

The bull trout was initially listed as three separate Distinct Population Segments (DPSs) (63 FR 31647, 64 FR 17110). The preamble to the final listing rule for the U.S. coterminous population of the bull trout discusses the consolidation of these DPSs, plus two other population segments, into one listed taxon and the application of the jeopardy standard under Section 7 of the Act relative to this species (64 FR 58930):

> Although this rule consolidates the five bull trout DPSs into one listed taxon, based on conformance with the DPS policy for purposes of consultation under Section 7 of the Act, we intend to retain recognition of each DPS in light of available scientific information relating to their uniqueness and significance. Under this approach, these DPSs will be treated as interim recovery units with respect to application of the jeopardy standard until an approved recovery plan is developed[5]. Formal establishment of bull trout recovery units will occur during the recovery planning process.

Thus, as discussed above under the *Analytical Framework for the Jeopardy and Adverse Modification Determinations*, the Service's jeopardy analysis for the proposed action relative to the bull trout will involve consideration of how the EPA's proposed action is likely to affect the Columbia River interim recovery unit for the bull trout based on its uniqueness and significance as described in the DPS final listing rule cited above. However, in accordance with Service national policy, the jeopardy determination is made at the scale of the listed species. In this case, the coterminous U.S. population of the bull trout.

Though wide ranging in parts of Oregon, Washington, Idaho, and Montana, bull trout in the interior Columbia River basin presently occur in only about 45 percent of the historical range (Quigley and Arbelbide 1997, p. 1177; Rieman et al. 1997, p. 1119). Declining trends due to the combined effects of habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, angler harvest and poaching, entrainment into diversion channels and dams, and introduced nonnative species (e.g., brook trout, *Salvelinus fontinalis*) have resulted in declines in range-wide bull trout distribution and abundance (Bond 1992, p. 4; Schill 1992, p. 40; Thomas 1992, pp. 9-12; Ziller 1992, p. 28; Rieman and McIntyre 1993, pp. 1-18; Newton and Pribyl 1994, pp. 2, 4, 8-9; IDFG 1995, *in litt.*, pp. 1-3). Several local extirpations have been reported, beginning in the 1950s (Rode 1990, p. 1; Ratliff and Howell 1992, pp. 12-14; Donald and Alger 1993, p. 245; Goetz 1994, p. 1; Newton and Pribyl 1994, p. 2; Berg and Priest 1995, pp. 1-45; Light et al. 1996, pp. 20-38; Buchanan and Gregory 1997, p. 120).

Land and water management activities such as dams and other diversion structures, forest management practices, livestock grazing, agriculture, road construction and maintenance, mining, and urban and rural development continue to degrade bull trout habitat and depress bull trout populations (USFWS 2002b, p. 13).

[5] A final revised bull trout Recovery Plan is expected for release in September 2015.

## 2.3.5.2 Species Description

Bull trout (*Salvelinus confluentus*), member of the family Salmonidae, are char native to the Pacific Northwest and western Canada. The bull trout and the closely related Dolly Varden (*Salvelinus malma*) were not officially recognized as separate species until 1980 (Robins et al. 1980, p. 19). Bull trout historically occurred in major river drainages in the Pacific Northwest from the southern limits in the McCloud River in northern California (now extirpated), Klamath River basin of south central Oregon, and the Jarbidge River in Nevada to the headwaters of the Yukon River in the Northwest Territories, Canada (Cavender 1978, p. 165-169; Bond 1992, p. 2-3). To the west, the bull trout's current range includes Puget Sound, coastal rivers of British Columbia, Canada, and southeast Alaska (Bond 1992, p. 2-3). East of the Continental Divide bull trout are found in the headwaters of the Saskatchewan River in Alberta and the MacKenzie River system in Alberta and British Columbia (Cavender 1978, p. 165-169; Brewin and Brewin 1997, pp. 209-216). Bull trout are wide spread throughout the Columbia River basin, including its headwaters in Montana and Canada.

## 2.3.5.3 Life History

Bull trout exhibit resident and migratory life history strategies throughout much of the current range (Rieman and McIntyre 1993, p. 2). Resident bull trout complete their entire life cycle in the streams where they spawn and rear. Migratory bull trout spawn and rear in streams for 1 to 4 years before migrating to either a lake (adfluvial), river (fluvial), or, in certain coastal areas, to saltwater (anadromous) where they reach maturity (Fraley and Shepard 1989, p. 1; Goetz 1989, pp. 15-16). Resident and migratory forms often occur together and it is suspected that individual bull trout may give rise to offspring exhibiting both resident and migratory behavior (Rieman and McIntyre 1993, p. 2).

Bull trout have more specific habitat requirements than other salmonids (Rieman and McIntyre 1993, p. 4). Watson and Hillman (1997, p. 248) concluded that watersheds must have specific physical characteristics to provide habitat requirements for bull trout to successfully spawn and rear. It was also concluded that these characteristics are not necessarily ubiquitous throughout these watersheds, thus resulting in patchy distributions even in pristine habitats.

Bull trout are found primarily in colder streams, although individual fish are migratory in larger, warmer river systems throughout the range (Fraley and Shepard 1989, pp. 135-137; Rieman and McIntyre 1993, p. 2 and 1995, p. 288; Buchanan and Gregory 1997, pp. 121-122; Rieman et al. 1997, p. 1114). Water temperature above 15°C (59°F) is believed to limit bull trout distribution, which may partially explain the patchy distribution within a watershed (Fraley and Shepard 1989, p. 133; Rieman and McIntyre 1995, pp. 255-296). Spawning areas are often associated with cold water springs, groundwater infiltration, and the coldest streams in a given watershed (Pratt 1992, p. 6; Rieman and McIntyre 1993, p. 7; Rieman et al. 1997, p. 1117). Goetz (1989, pp. 22, 24) suggested optimum water temperatures for rearing of less than 10°C (50°F) and optimum water temperatures for egg incubation of 2 to 4°C (35 to 39°F).

All life history stages of bull trout are associated with complex forms of cover, including large woody debris, undercut banks, boulders, and pools (Goetz 1989, pp. 22-25; Pratt 1992, p. 6; Thomas 1992, pp. 4-5; Rich 1996, pp. 35-38; Sexauer and James 1997, pp. 367-369; Watson and Hillman 1997, pp. 247-249). Jakober (1995, p. 42) observed bull trout overwintering in deep beaver ponds or pools containing large woody debris in the Bitterroot River drainage, Montana, and suggested that suitable winter habitat may be more restrictive than summer habitat. Bull trout prefer relatively stable channel and water flow conditions (Rieman and McIntyre 1993, p. 6). Juvenile and adult bull trout frequently inhabit side channels, stream margins, and pools with suitable cover (Sexauer and James 1997, pp. 368-369).

The size and age of bull trout at maturity depend upon life history strategy. Growth of resident fish is generally slower than migratory fish; resident fish tend to be smaller at maturity and less fecund (Goetz 1989, p. 15). Bull trout normally reach sexual maturity in 4 to 7 years and live as long as 12 years. Bull trout are iteroparous (they spawn more than once in a lifetime), and both repeat- and alternate-year spawning has been reported, although repeat-spawning frequency and post-spawning mortality are not well documented (Leathe and Graham 1982, p. 95; Fraley and Shepard 1989, p. 135; Pratt 1992, p. 8; Rieman and McIntyre 1996, p. 133).

Bull trout typically spawn from August to November during periods of decreasing water temperatures. Migratory bull trout frequently begin spawning migrations as early as April, and have been known to move upstream as far as 250 kilometers (km) (155 miles (mi)) to spawning grounds (Fraley and Shepard 1989, p. 135). Depending on water temperature, incubation is normally 100 to 145 days (Pratt 1992, p. 1) and, after hatching, juveniles remain in the substrate. Time from egg deposition to emergence may exceed 200 days. Fry normally emerge from early April through May depending upon water temperatures and increasing stream flows (Pratt 1992, p. 1).

The iteroparous reproductive system of bull trout has important repercussions for the management of this species. Bull trout require two-way passage up and downstream, not only for repeat spawning, but also for foraging. Most fish ladders, however, were designed specifically for anadromous semelparous (fishes that spawn once and then die, and therefore require only one-way passage upstream) salmonids. Therefore, even dams or other barriers with fish passage facilities may be a factor in isolating bull trout populations if they do not provide a downstream passage route.

Bull trout are opportunistic feeders with food habits primarily a function of size and life history strategy. Resident and juvenile migratory bull trout prey on terrestrial and aquatic insects, macro zooplankton and small fish (Boag 1987, p. 58; Goetz 1989, pp. 33-34; Donald and Alger 1993, pp. 239-243). Adult migratory bull trout are primarily piscivores, known to feed on various fish species (Fraley and Shepard 1989, p. 135; Donald and Alger 1993, p. 242).

## Population Dynamics

The draft bull trout Recovery Plan (USFWS 2002b, pp. 47-48) defined core areas as groups of partially isolated local populations of bull trout with some degree of gene flow occurring between them. Based on this definition, core areas can be considered metapopulations. A metapopulation is an interacting network of local populations with varying frequencies of migration and gene flow among them (Meefe and Carroll 1994, p. 188). In theory, bull trout metapopulations (core areas) can be composed of two or more local populations, but Rieman and Allendorf (2001, p. 763) suggest that for a bull trout metapopulation to function effectively, a minimum of 10 local populations are required. Bull trout core areas with fewer than 5 local populations are at increased risk of local extirpation, core areas with between 5 and 10 local populations are at intermediate risk, and core areas with more than 10 interconnected local populations are at diminished risk (USFWS 2002b, pp. 50-51).

The presence of a sufficient number of adult spawners is necessary to ensure persistence of bull trout populations. In order to avoid inbreeding depression, it is estimated that a minimum of 100 spawners are required. Inbreeding can result in increased homozygosity of deleterious recessive alleles which can in turn reduce individual fitness and population viability (Whitesel et al. 2004, p. 36). For persistence in the longer term, adult spawning fish are required in sufficient numbers to reduce the deleterious effects of genetic drift and maintain genetic variation. For bull trout, Rieman and Allendorf (2001, p. 762) estimate that approximately 1,000 spawning adults within any bull trout population are necessary for maintaining genetic variation indefinitely. Many local bull trout populations individually do not support 1,000 spawners, but this threshold may be met by the presence of smaller interconnected local populations within a core area.

For bull trout populations to remain viable (and recover), natural productivity should be sufficient for the populations to replace themselves from generation to generation. A population that consistently fails to replace itself is at an increased risk of extinction. Since estimates of population size are rarely available, the productivity or population growth rate is usually estimated from temporal trends in indices of abundance at a particular life stage. For example, redd counts are often used as an indicator of a spawning adult population. The direction and magnitude of a trend in an index can be used as a surrogate for growth rate.

Survival of bull trout populations is also dependent upon connectivity among local populations. Although bull trout are widely distributed over a large geographic area, they exhibit a patchy distribution even in pristine habitats (Rieman and McIntyre 1993, p. 7). Increased habitat fragmentation reduces the amount of available habitat and increases isolation from other populations of the same species (Saunders et al. 1991, p. 22). Burkey (1989, p. 76) concluded that when species are isolated by fragmented habitats, low rates of population growth are typical in local populations and their probability of extinction is directly related to the degree of isolation and fragmentation. Without sufficient immigration, growth of local populations may be low and probability of extinction high. Migrations also facilitate gene flow among local populations because individuals from different local populations interbreed when some stray and return to nonnatal streams. Local populations that are extirpated by catastrophic events may also become reestablished in this manner.

Based on the works of Rieman and McIntyre (1993, pp. 9-15) and Rieman and Allendorf (2001, pp. 756-763), the 2002 draft bull trout Recovery Plan identified four elements to consider when assessing long-term viability (extinction risk) of bull trout populations: (1) number of local populations, (2) adult abundance (defined as the number of spawning fish present in a core area in a given year), (3) productivity, or the reproductive rate of the population, and (4) connectivity (as represented by the migratory life history form).

## 2.3.5.4 Status and Distribution

As noted above, in recognition of available scientific information relating to their uniqueness and significance, five interim recovery units of the coterminous United States population of the bull trout are considered essential to the survival and recovery of this species and are identified as: (1) Jarbidge River, (2) Klamath River, (3) Coastal-Puget Sound, (4) St. Mary-Belly River, and (5) Columbia River. Each of these segments is necessary to maintain the bull trout's distribution, as well as its genetic and phenotypic diversity, all of which are important to ensure the species' resilience to changing environmental conditions.

A summary of the current status and conservation needs of the bull trout within these units is provided below. A comprehensive discussion of these topics is found in the draft bull trout Recovery Plan (USFWS 2002b, entire; 2004a, b; entire; 2014b, entire).

Central to the survival and recovery of the bull trout is the maintenance of viable core areas (USFWS 2002b, p. 54). A core area is defined as a geographic area occupied by one or more local bull trout populations that overlap in their use of rearing, foraging, migratory, and overwintering habitat, and, in some cases, their use of spawning habitat. Each of the interim recovery units listed below consists of one or more core areas. One hundred and twenty one core areas are recognized across the United States range of the bull trout (USFWS 2005b, p. 9).

A core area assessment conducted by the Service for the 5 year bull trout status review determined that of the 121 core areas comprising the coterminous listing, 43 are at high risk of extirpation, 44 are at risk, 28 are at potential risk, 4 are at low risk and 2 are of unknown status (USFWS 2008b, p. 29).

## Jarbidge River

This interim recovery unit currently contains a single core area with six local populations. Less than 500 resident and migratory adult bull trout, representing about 50 to 125 spawners, are estimated to occur within the core area. The current condition of the bull trout in this segment is attributed to the effects of livestock grazing, roads, angler harvest, timber harvest, and the introduction of nonnative fishes (USFWS 2004b, p. iii). The draft bull trout Recovery Plan identifies the following conservation needs for this segment: (1) maintain the current distribution of the bull trout within the core area, (2) maintain stable or increasing trends in abundance of both resident and migratory bull trout in the core area, (3) restore and maintain suitable habitat conditions for all life history stages and forms, and (4) conserve genetic diversity and increase natural opportunities for genetic exchange between resident and migratory forms of the bull trout. An estimated 270 to 1,000 spawning fish per year are needed to provide for the persistence and viability of the core area and to support both resident and migratory adult bull trout (USFWS 2004b, p. 62-63). Currently this core area is at high risk of extirpation (USFWS 2005b, p. 9).

Since the 2004 draft recovery plan was written, updated information is available on the bull trout population in the Jarbidge River Distinct Population Segment (Allen et al. 2010, entire). The most recent study, conducted by the U.S. Geological Survey (USGS) in 2006 and 2007 to examine the distribution and movement of bull trout in the Jarbidge River system, captured 349 bull trout in 24.8 miles of habitat in the East and West Forks of the Jarbidge River, and in Fall, Slide, Dave, Jack, and Pine creeks. In 2007, they captured 1,353 bull trout in 15.5 miles of habitat in the West Fork Jarbidge River and its tributaries and 11.2 miles of habitat in the East Fork Jarbidge River and its tributaries (Allen et al. 2010, p. 6). The study results indicate that almost four times the number of bull trout estimated in the 2004 draft Recovery Plan inhabit the Jarbidge core area; and that these fish show substantial movements between tributaries, increased abundance with increasing altitude, and growth rates indicative of a high-quality habitat (Allen et al. 2010, p. 20).

## Klamath River

This interim recovery unit currently (as of 2002) contains three core areas and 12 local populations. The current abundance, distribution, and range of the bull trout in the Klamath River Basin are greatly reduced from historical levels due to habitat loss and degradation caused by reduced water quality, timber harvest, livestock grazing, water diversions, roads, and the introduction of nonnative fishes. Bull trout populations in this unit face a high risk of extirpation (USFWS 2002c, p. iv). The draft bull trout Recovery Plan (USFWS 2002c, p. v) identifies the following conservation needs for this unit: (1) maintain the current distribution of the bull trout and restore distribution in previously occupied areas, (2) maintain stable or increasing trends in bull trout abundance, (3) restore and maintain suitable habitat conditions for all life history stages and strategies, and (4) conserve genetic diversity and provide the opportunity for genetic exchange among appropriate core area populations. Eight to 15 new local populations and an increase in population size from about 3,250 adults currently to 8,250 adults are needed to provide for the persistence and viability of the three core areas (USFWS 2002c, p. vi).

## Coastal-Puget Sound

Bull trout in the Coastal-Puget Sound interim recovery unit exhibit anadromous, adfluvial, fluvial, and resident life history patterns. The anadromous life history form is unique to this unit. This interim recovery unit currently contains 14 core areas and 67 local populations (USFWS 2004c, p. iv; 2004d, pp. iii-iv). Bull trout are distributed throughout most of the large rivers and associated tributary systems within this unit. With limited exceptions, bull trout continue to be present in nearly all major watersheds where they likely occurred historically within this unit. Generally, bull trout distribution has contracted and abundance has declined, especially in the southeastern part of the unit. The current condition of the bull trout in this interim recovery unit is attributed to the adverse effects of dams, forest management practices (e.g., timber harvest and associated road building activities), agricultural practices (e.g., diking, water control structures, draining of wetlands, channelization, and the removal of riparian vegetation), livestock grazing, roads, mining, urbanization, angler harvest, and the introduction of nonnative species. The draft bull trout Recovery Plan (USFWS 2004c, pp. ix-x) identifies the following conservation needs for this unit: (1) maintain or expand the current distribution of bull trout within existing core areas, (2) increase bull trout abundance to about 16,500 adults across all core areas, and (3) maintain or increase connectivity between local populations within each core area.

## St. Mary-Belly River

This interim recovery unit currently contains six core areas and nine local populations (USFWS 2002d, p. v). Currently, bull trout are widely distributed in the St. Mary River drainage and occur in nearly all of the waters that were inhabited historically. Bull trout are found only in a 1.2-mile reach of the North Fork Belly River within the United States. Redd count surveys of the North Fork Belly River documented an increase from 27 redds in 1995 to 119 redds in 1999. This increase was attributed primarily to protection from angler harvest (USFWS 2002d, p. 37). The current condition of the bull trout in this interim recovery unit is primarily attributed to the effects of dams, water diversions, roads, mining, and the introduction of nonnative fishes (USFWS 2002d, p. vi). The draft bull trout Recovery Plan (USFWS 2002d, pp. v-ix) identifies the following conservation needs for this unit: (1) maintain the current distribution of the bull trout and restore distribution in previously occupied areas, (2) maintain stable or increasing trends in bull trout abundance, (3) maintain and restore suitable habitat conditions for all life history stages and forms, (4) conserve genetic diversity and provide the opportunity for genetic exchange, and (5) establish good working relations with Canadian interests because local bull trout populations in this unit are comprised mostly of migratory fish whose habitat is mainly in Canada.

### Columbia River

The Columbia River interim recovery unit includes bull trout residing in portions of Oregon, Washington, Idaho, and Montana. Bull trout are estimated to have occupied about 60 percent of the Columbia River Basin, and presently occur in 45 percent of the estimated historical range (Quigley and Arbelbide 1997, p. 1177). This interim recovery unit currently contains 97 core areas and 527 local populations. About 65 percent of these core areas and local populations occur in Idaho and northwestern Montana.

The condition of the bull trout populations within these core areas varies from poor to good, but generally all have been subject to the combined effects of habitat degradation, fragmentation and alterations associated with one or more of the following activities: dewatering, road construction and maintenance, mining and grazing, blockage of migratory corridors by dams or other diversion structures, poor water quality, incidental angler harvest, entrainment into diversion channels, and introduced nonnative species.

The Service has determined that of the total 97 core areas in this interim recovery unit, 38 are at high risk of extirpation, 35 are at risk, 20 are at potential risk, 2 are at low risk, and 2 are at unknown risk (USFWS 2005b, pp. 1-94).

The draft bull trout Recovery Plan (USFWS 2002b, p. v) identifies the following conservation needs for this interim recovery unit: (1) maintain or expand the current distribution of the bull trout within core areas, (2) maintain stable or increasing trends in bull trout abundance, (3) maintain and restore suitable habitat conditions for all bull trout life history stages and strategies, and (4) conserve genetic diversity and provide opportunities for genetic exchange.

## 2.3.5.5 Previous Consultations and Conservation Efforts

#### Consultations

Consulted-on effects are those effects that have been analyzed through section 7 consultation as reported in a biological opinion. These effects are an important component of objectively characterizing the current condition of the species. To assess consulted-on effects to bull trout, we analyzed all of the biological opinions received by the Region 1 and Region 6 Service Offices from the time of bull trout's listing until August 2003; this summed to 137 biological opinions. Of these, 124 biological opinions (91 percent) applied to activities affecting bull trout in the Columbia Basin interim recovery unit, 12 biological opinions (9 percent) applied to activities affecting bull trout in the Coastal-Puget Sound interim recovery unit, 7 biological opinions (5 percent) applied to activities affecting bull trout in the Klamath Basin interim recovery unit, and one biological opinion (< 1 percent) applied to activities affecting the Jarbidge and St. Mary-Belly interim recovery units (Note: these percentages do not add to 100, because several biological opinions applied to more than one interim recovery unit). The geographic scale of these consultations varied from individual actions (e.g., construction of a bridge or pipeline) within one basin to multiple-project actions occurring across several basins.

Our analysis showed that we consulted on a wide array of actions which had varying levels of effect. Many of the actions resulted in only short-term adverse effects, some with long-term beneficial effects. Some of the actions resulted in long-term adverse effects. No actions that have undergone consultation were found to appreciably reduce the likelihood of survival and recovery of the bull trout. Furthermore, no actions that have undergone consultation were anticipated to result in the loss of local populations of bull trout.

**Regulatory Mechanisms**

The implementation and effectiveness of regulatory mechanisms vary across the coterminous range. Forest practices rules for Montana, Idaho, Oregon, Washington, and Nevada include streamside management zones that benefit bull trout when implemented.

**State Conservation Measures**

State agencies are specifically addressing bull trout through the following initiatives:

- Washington Bull Trout and Dolly Varden Management Plan developed in 2000.
- Montana Bull Trout Restoration Plan (Bull Trout Restoration Team appointed in 1994, and plan completed in 2000).
- Oregon Native Fish Conservation Policy (developed in 2004).
- Nevada Species Management Plan for Bull Trout (developed in 2005).
- State of Idaho Bull Trout Conservation Plan (developed in 1996). The watershed advisory group drafted 21 problem assessments throughout Idaho which address all 59 key watersheds. To date, a conservation plan has been completed for one of the 21 key watersheds (Pend Oreille).

**Habitat Conservation Plans**

Habitat Conservation Plans (HCP) have resulted in land management practices that exceed State regulatory requirements. Habitat conservation plans addressing bull trout cover approximately 472 stream miles of aquatic habitat, or approximately 2.6 percent of the Key Recovery Habitat across Montana, Idaho, Oregon, Washington, and Nevada. These HCPs include: Plum Creek Native Fish HCP, Washington Department of Natural Resources HCP, City of Seattle Cedar River Watershed HCP, Tacoma Water HCP, and Green Diamond HCP.

**Federal Land Management Plans**

PACFISH is the “Interim Strategy for Managing Anadromous Fish-Producing Watersheds and includes Federal lands in Western Oregon and Washington, Idaho, and Portions of California.” INFISH is the “Interim Strategy for Managing Fish-Producing Watersheds in Eastern Oregon and Washington, Idaho, Western Montana, and Portions of Nevada.” Each strategy amended Forest Service Land and Resource Management Plans and Bureau of Land Management Resource Management Plans. Together PACFISH and INFISH cover thousands of miles of waterways within 16 million acres and provide a system for reducing effects from land management activities to aquatic resources through riparian management goals, landscape scale interim riparian management objectives, Riparian Habitat Conservation Areas (RHCAs), riparian standards, watershed analysis, and the designation of Key and Priority watersheds. These interim strategies have been in place since 1992 and are part of the management plans for Bureau of Land Management and Forest Service lands.

The Interior Columbia Basin Ecosystem Management Plan (ICBEMP) is the strategy that replaces the PACFISH and INFISH interim strategies when federal land management plans are revised. The Southwest Idaho Land and Resource Management Plan (LRMP) is the first LRMP under the strategy and provides measures that protect and restore soil, water, riparian and aquatic resources during project implementation while providing flexibility to address both short- and long-term social and economic goals on 6.6 million acres of National Forest lands. This plan includes a long-term Aquatic Conservation Strategy that focuses restoration funding in priority subwatersheds identified as important to achieving Endangered Species Act, Tribal, and Clean Water Act goals. The Southwest Idaho LRMP replaces the interim PACFISH/INFISH strategies and adds additional conservation elements, specifically, providing an ecosystem management foundation, a prioritization for restoration integrated across multiple scales, and adaptable active, passive and conservation management strategies that address both protection and restoration of habitat and 303(d) stream segments.

The Southeast Oregon Resource Management Plan (SEORMP) and Record of Decision is the second LRMP under the ICBEMP strategy which describes the long-term (20+ years) plan for managing the public lands within the Malheur and Jordan Resource Areas of the Vale District. The SEORMP is a general resource management plan for 4.6 million acres of Bureau of Land Management administered public lands primarily in Malheur County with some acreage in Grant and Harney Counties, Oregon. The SEORMP contains resource objectives, land use allocations, management actions and direction needed to achieve program goals. Under the plan, riparian areas, floodplains, and wetlands will be managed to restore, protect, or improve their natural functions relating to water storage, groundwater recharge, water quality, and fish and wildlife values.

The Northwest Forest Plan covers 24.5 million acres in Washington, Oregon, and northern California. The Aquatic Conservation Strategy (ACS) is a component of the Northwest Forest Plan. It was developed to restore and maintain the ecological health of watersheds and the aquatic ecosystems. The four main components of the ACS (Riparian Reserves, Watershed Analysis, Key Watersheds, and Watershed Restoration) are designed to operate together to maintain and restore the productivity and resiliency of riparian and aquatic ecosystems.

It is the objective of the Forest Service and the Bureau of Land Management to manage and maintain habitat and, where feasible, to restore habitats that are degraded. These plans provide for the protection of areas that could contribute to the recovery of fish and, overall, improve riparian habitat and water quality throughout the basin. These objectives are accomplished through such activities as closing and rehabilitating roads, replacing culverts, changing grazing and logging practices, and re-planting native vegetation along streams and rivers.

## 2.3.5.6 Conservation Needs

Refer to section 2.4.5.2, *Factors Affecting the Bull Trout in the Action Area*, for more specific information on threats to bull trout within the action area.

The 2014 revised draft bull trout Recovery Plan (USFWS 2014b, p. vi) states “that the ultimate goal of this recovery strategy is to manage threats and ensure sufficient distribution and abundance to improve the status of bull trout throughout their extant range in the coterminous United States so that protection under the Endangered Species Act is no longer necessary. When this is achieved, we expect that:

- Bull trout will be geographically widespread across representative habitats and demographically stable in each recovery unit;
- The genetic diversity and diverse life history forms of bull trout will be conserved to the maximum extent possible; and
- Cold water habitats essential to bull trout will be conserved and connected." [6]

The 2014 revised draft bull trout Recovery Plan (USFWS 2014b, p. ix) identifies the following tasks needed for achieving recovery: (1) protect, restore, and maintain suitable habitat conditions for bull trout that promote diverse life history strategies and conserve genetic diversity, (2) prevent and reduce negative effects of non-native fishes and other non-native taxa on bull trout. (3) work with partners to conduct research and monitoring to implement and evaluate bull trout recovery activities, consistent with an adaptive management approach using feedback from implemented, site-specific recovery tasks.

Another threat now facing bull trout is warming temperature regimes associated with global climate change. Because air temperature affects water temperature, species at the southern margin of their range that are associated with cold water patches, such as bull trout, may become restricted to smaller, more disjunct patches or become extirpated as the climate warms (Rieman et al. 2007, p. 1560). Rieman et al. (2007, pp. 1558, 1562) concluded that climate is a primary determining factor in bull trout distribution. Some populations already at high risk, such as the Jarbidge, may require "aggressive measures in habitat conservation or restoration" to persist (Rieman et al. 2007, p. 1560). Conservation and restoration measures that would benefit bull trout include protecting high quality habitat, reconnecting watersheds, restoring flood plains, and increasing site-specific habitat features important for bull trout, such as deep pools or large woody debris (Kinsella 2005, entire).

# 2.3.6. Bull Trout Critical Habitat

## 2.3.6.1 Legal Status

The Service published a proposed critical habitat rule on January 14, 2010 (75 FR 2260) and a final rule on October 18, 2010 (75 FR 63898). The rule became effective on November 17, 2010. A justification document was also developed to support the rule and is available on our website (http://www.fws.gov/pacific/bulltrout).

The Service designated reservoirs/lakes and stream/shoreline miles in 32 critical habitat units (CHU) within the coterminous geographical area occupied by the species at the time of listing as

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[6] The 2002 draft Recovery plan (USFWS 2002a, p. 49) identified the following conservation needs (goals) for bull trout recovery: (1) maintain the current distribution of bull trout within core areas as described in recovery unit chapters, (2) maintain stable or increasing trends in abundance of bull trout as defined for individual recovery units, (3) restore and maintain suitable habitat conditions for all bull trout life history stages and strategies, and (4) conserve genetic diversity and provide opportunity for genetic exchange.

bull trout critical habitat (see Table 2). Designated bull trout critical habitat is of two primary use types: (1) spawning and rearing; and (2) foraging, migrating, and overwintering (FMO).

**Table 2. Stream/shoreline distance and reservoir/lake area designated as bull trout critical habitat by state.**

| State | Stream/Shoreline Miles | Stream/Shoreline Kilometers | Reservoir/ Lake Acres | Reservoir/ Lake Hectares |
|---|---|---|---|---|
| Idaho | 8,771.6 | 14,116.5 | 170,217.5 | 68,884.9 |
| Montana | 3,056.5 | 4,918.9 | 221,470.7 | 89,626.4 |
| Nevada | 71.8 | 115.6 | - | - |
| Oregon | 2,835.9 | 4,563.9 | 30,255.5 | 12,244.0 |
| Oregon/Idaho | 107.7 | 173.3 | - | - |
| Washington | 3,793.3 | 6,104.8 | 66,308.1 | 26,834.0 |
| Washington (marine) | 753.8 | 1,213.2 | - | - |
| Washington/Idaho | 37.2 | 59.9 | - | - |
| Washington/Oregon | 301.3 | 484.8 | - | - |
| **Total** | **19,729.0** | **31,750.8** | **488,251.7** | **197,589.2** |

This rule also identifies and designates as critical habitat approximately 1,323.7 km (822.5 miles) of streams/shorelines and 6,758.8 ha (16,701.3 acres) of lakes/reservoirs of unoccupied habitat to address bull trout conservation needs in specific geographic areas in several areas not occupied at the time of listing. These unoccupied areas were determined by the Service to be essential for restoring functioning migratory bull trout populations based on currently available scientific information. These unoccupied areas often include lower mainstem river environments that can provide seasonally important migration habitat for bull trout. This type of habitat is essential in areas where bull trout habitat and population loss over time necessitates reestablishing bull trout in currently unoccupied habitat areas to achieve recovery.

The final rule continues to exclude some critical habitat segments based on a careful balancing of the benefits of inclusion versus the benefits of exclusion. Critical habitat does not include: (1) waters adjacent to non-Federal lands covered by legally operative incidental take permits for habitat conservation plans (HCPs) issued under section 10(a)(1)(B) of the Endangered Species Act of 1973, as amended, in which bull trout is a covered species on or before the publication of this final rule; (2) waters within or adjacent to Tribal lands subject to certain commitments to conserve bull trout or a conservation program that provides aquatic resource protection and restoration through collaborative efforts, and where the Tribes indicated that inclusion would impair their relationship with the Service; or (3) waters where impacts to national security have been identified (75 FR 63898). Excluded areas are approximately 10 percent of the stream/shoreline miles and 4 percent of the lakes and reservoir acreage of designated critical habitat. Each excluded area is identified in the relevant CHU text, as identified in paragraphs (e)(8) through (e)(41) of the final rule. It is important to note that the exclusion of waterbodies from designated critical habitat does not negate or diminish their importance for bull trout conservation. Because exclusions reflect the often complex pattern of land ownership, designated critical habitat is often fragmented and interspersed with excluded stream segments.

### 2.3.6.2 Conservation Role and Description of Critical Habitat

The conservation role of bull trout critical habitat is to support viable core area populations (75 FR 63943). The core areas reflect the metapopulation structure of bull trout and are the closest approximation of a biologically functioning unit for the purposes of recovery planning and risk analyses. CHUs generally encompass one or more core areas and may include FMO areas, outside of core areas, that are important to the survival and recovery of bull trout.

As previously noted, 32 CHUs within the geographical area occupied by the species at the time of listing are designated under the final rule. Twenty-nine of the CHUs contain all of the physical or biological features identified in this final rule and support multiple life-history requirements. Three of the mainstem river units in the Columbia and Snake River basins contain most of the physical or biological features necessary to support the bull trout's particular use of that habitat, other than those physical and biological features associated with Primary Constituent Elements (PCEs) 5 and 6, which relate to breeding habitat (see list below).

The primary function of individual CHUs is to maintain and support core areas, which (1) contain bull trout populations with the demographic characteristics needed to ensure their persistence and contain the habitat needed to sustain those characteristics (based on Rieman and McIntyre 1993, p. 19); (2) provide for persistence of strong local populations, in part, by providing habitat conditions that encourage movement of migratory fish (based on MBTSG 1998, pp. 48-49; Rieman and McIntyre 1993, pp. 22-23); (3) are large enough to incorporate genetic and phenotypic diversity, but small enough to ensure connectivity between populations (based on MBTSG 1998, pp. 48-49; Rieman and McIntyre 1993, pp. 22-23); and (4) are distributed throughout the historic range of the species to preserve both genetic and phenotypic adaptations (based on MBTSG 1998, pp. 13-16; Rieman and Allendorf 2001, p. 763; Rieman and McIntyre 1993, p. 23).

The Olympic Peninsula and Puget Sound CHUs are essential to the conservation of amphidromous bull trout, which are unique to the Coastal-Puget Sound interim recovery unit. These CHUs contain marine nearshore and freshwater habitats, outside of core areas, that are used by bull trout from one or more core areas. These habitats, outside of core areas, contain PCEs that are critical to adult and subadult foraging, migrating, and overwintering.

In determining which areas to propose as critical habitat, the Service considered the physical and biological features that are essential to the conservation of bull trout and that may require special management considerations or protection. These features are the PCEs laid out in the appropriate quantity and spatial arrangement for conservation of the species. The PCEs of designated critical habitat are:

1. Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia.
2. Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats, including, but not limited to, permanent, partial, intermittent, or seasonal barriers.
3. An abundant food base, including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish.

4. Complex river, stream, lake, reservoir, and marine shoreline aquatic environments and processes that establish and maintain these aquatic environments, with features such as large wood, side channels, pools, undercut banks and unembedded substrates, to provide a variety of depths, gradients, velocities, and structure.
5. Water temperatures ranging from 2 to 15 °C (36 to 59 °F), with adequate thermal refugia available for temperatures that exceed the upper end of this range. Specific temperatures within this range will depend on bull trout life-history stage and form; geography; elevation; diurnal and seasonal variation; shading, such as that provided by riparian habitat; streamflow; and local groundwater influence.
6. In spawning and rearing areas, substrate of sufficient amount, size, and composition to ensure success of egg and embryo overwinter survival, fry emergence, and young-of-the-year and juvenile survival. A minimal amount of fine sediment, generally ranging in size from silt to coarse sand, embedded in larger substrates, is characteristic of these conditions. The size and amounts of fine sediment suitable to bull trout will likely vary from system to system.
7. A natural hydrograph, including peak, high, low, and base flows within historic and seasonal ranges or, if flows are controlled, minimal flow departures from a natural hydrograph.
8. Sufficient water quality and quantity such that normal reproduction, growth, and survival are not inhibited.
9. Sufficiently low levels of occurrence of nonnative predatory (e.g., lake trout, walleye, northern pike, smallmouth bass); interbreeding (e.g., brook trout); or competing (e.g., brown trout) species that, if present, are adequately temporally and spatially isolated from bull trout.

### 2.3.6.3 Current Rangewide Condition of Bull Trout Critical Habitat

The condition of bull trout critical habitat varies across its range from poor to good. Although still relatively widely distributed across its historic range, the bull trout occurs in low numbers in many areas, and populations are considered depressed or declining across much of its range (67 FR 71240). This condition reflects the condition of bull trout habitat. Refer to section 2.4.6.2, *Factors Affecting Bull Trout Critical Habitat in the Action Area*, for more specific information on the condition of bull trout critical habitat in the action area.

The primary land and water management activities impacting the physical and biological features essential to the conservation of bull trout include timber harvest and road building, agriculture and agricultural diversions, livestock grazing, dams, mining, urbanization and residential development, and nonnative species presence or introduction (75 FR 2282).

There is widespread agreement in the scientific literature that many factors related to human activities have impacted bull trout and their habitat, and continue to do so. Among the many factors that contribute to degraded PCEs, those which appear to be particularly significant and have resulted in a legacy of degraded habitat conditions are as follows:

1. Fragmentation and isolation of local populations due to the proliferation of dams and water diversions that have eliminated habitat, altered water flow and temperature regimes, and

impeded migratory movements (Dunham and Rieman 1999, p. 652; Rieman and McIntyre 1993, p. 7), affecting the condition of PCEs 2, 4, and 5.

2. Degradation of spawning and rearing habitat and upper watershed areas, particularly alterations in sedimentation rates and water temperature, resulting from forest and rangeland practices and intensive development of roads (Fraley and Shepard 1989, p. 141; MBTSG 1998, pp. ii - v, 20-45), affecting the condition of PCEs 5 and 6. 3. The introduction and spread of nonnative fish species, particularly brook trout and lake trout, as a result of fish stocking and degraded habitat conditions, which compete with bull trout for limited resources and, in the case of brook trout, hybridize with bull trout (Leary et al. 1993, p. 857; Rieman et al. 2006, pp. 73-76), affecting the condition of PCE 9.
4. In the Coastal-Puget Sound region where amphidromous bull trout occur, degradation of mainstem river FMO habitat, and the degradation and loss of marine nearshore foraging and migration habitat due to urban and residential development, affecting the condition of PCE 2, 3, and 4.
5. Degradation of FMO habitat resulting from reduced prey base, roads, agriculture, development, and dams, affecting PCEs 2, 3, and 4.

The bull trout critical habitat final rule also aimed to identify and protect those habitats that provide resiliency for bull trout use in the face of climate change. Over a period of decades, climate change may directly threaten the integrity of the essential physical or biological features described in PCEs 1, 2, 3, 5, 7, 8, and 9. Protecting bull trout strongholds and cold water refugia from disturbance and ensuring connectivity among populations were important considerations in addressing this potential impact. Additionally, climate change may exacerbate habitat degradation impacts both physically (e.g., decreased base flows, increased water temperatures) and biologically (e.g., increased competition with nonnative fishes).

## 2.3.7 Kootenai River White Sturgeon

### 2.3.7.1 Listing Status

On June 11, 1992, the Service received a petition from the Idaho Conservation League, North Idaho Audubon, and the Boundary Backpackers to list the Kootenai sturgeon as threatened or endangered under the Act. The petition cited lack of natural flows affecting juvenile recruitment as the primary threat to the continued existence of the wild Kootenai sturgeon population. Pursuant to section 4(b)(A) of the Act, the Service determined that the petition presented substantial information indicating that the requested action may be warranted, and published this finding in the Federal Register on April 14, 1993 (58 FR 19401).

A proposed rule to list the Kootenai sturgeon as endangered was published on July 7, 1993 (58 FR 36379), with a final rule following on September 6, 1994 (59 FR 45989).

### 2.3.7.2 Species Description

White sturgeon are included in the family Acipenseridae, which consists of 4 genera and 24 species of sturgeon. Eight species of sturgeon occur in North America with white sturgeon being one of the five species in the genus *Acipenser*. Kootenai sturgeon are a member of the species *Acipenser transmontanus*.

White sturgeon were first described by Richardson in 1863 from a single specimen collected in the Columbia River near Fort Vancouver, Washington (Scott and Crossman 1973, p. 100). These sturgeon have a characteristic elongated body, with a large, broad head, small eyes and flattened snout. This fish has a ventral mouth with four barbels in a transverse row on the ventral surface of the snout. White sturgeon are distinguished from other *Acipenser* by the specific arrangement and number of scutes (bony plates) along the body (USFWS 1999, p. 3). The white sturgeon is light grey in color, and can grow quite large; the largest white sturgeon on record, weighing approximately 1,500 pounds was taken from the Snake River near Weiser, Idaho in 1898 (USFWS 1999, p. 3). Scott and Crossman (1973, p. 98) describe a white sturgeon reported to weigh over 1,800 pounds from the Fraser River near Vancouver, British Columbia, date unknown. Individuals in landlocked populations tend to be smaller. The largest white sturgeon reported among Kootenai sturgeon was a 159 kilogram (350 pound) individual, estimated at 85 to 90 years of age, captured in Kootenay Lake in September 1995 (USFWS 1999, p. 3). White sturgeon are generally long lived, with females living from 34 to 70 years (USFWS 1999, p. 3).

## 2.3.7.3 Life History

As noted in the Kootenai Sturgeon Recovery Plan (USFWS 1999, p. 4), Kootenai sturgeon are considered opportunistic feeders. They are primarily bottom feeders but larger individuals will also take prey in the water column (Scott and Crossman 1973, p. 99). Smaller sturgeons feed predominantly on chironomids; for larger sturgeons, fish and crayfish become the predominant foods, with chironomids remaining a significant portion of their diet (Scott and Crossman 1973, p. 99). Partridge (1983, pp. 23-28) found Kootenai sturgeon more than 70 centimeters (28 inches) in length feeding on a variety of prey items including clams, snails, aquatic insects, and fish.

A natural barrier at Bonnington Falls in British Columbia has isolated the Kootenai River white sturgeon from other white sturgeon populations in the Columbia River basin for approximately 10,000 years (Apperson 1992, p. 2), resulting in a genetically distinct population with unique behaviors (e.g. this population is active at lower temperatures than Snake River and Columbia River populations, and displays a “short two-step migration” to spawning areas) (Paragamian et al. 2001, p. 22).

Pre-Libby Dam reports and documents unanimously state that the spawning location of Kootenai sturgeon was in a stretch of the river just downstream of Kootenai Falls, Montana (USFWS 2011, p. 12). A Corps of Engineers environmental statement (USCOE 1971, p. 11) states, “Little is known about the spawning habitat requirements of the white sturgeon, which spawns downstream from Kootenai Falls in Montana.” A 1974 report by Montana Fish Wildlife and Parks (MFWP 1974, p. 30) states, “Sturgeon from the Kootenai River in Idaho or Kootenay Lake, British Columbia spawn in the Kootenai River in Montana in the vicinity of Kootenai Falls.” The report also predicted, “A changed flow regime reducing high spring flows may eliminate spawning runs of this fish into Montana and may reduce population numbers in the downstream areas.” All other currently available historical reports and documents give similar descriptions of the pre-Libby Dam spawning location of Kootenai sturgeon and that construction and operations of the dam would negatively affect Kootenai sturgeon’s spawning behavior and success (MFWP 1983).

Currently, most Kootenai River white sturgeon spawning is occurring over sandy/silty substrates within an 18 RKM (11.2 RM) reach of the Kootenai River, from Bonners Ferry downstream to below Shorty's Island, known as the meander reach (Paragamian et al. 2001, p. 28; Paragamian 2012, p. 160 ). Spawning over sand and silt substrates results in suffocation of fertilized eggs and in the 1994 listing rule this suffocation was identified as the primary cause of recruitment failure for the sturgeon. This threat remains (USFWS 2011, p. 12). However, at that time sturgeon managers believed the sand and silt was covering rocky substrates that had only become inundated since the construction and operation of Libby Dam (USFWS 2011, p. 12). The view that increased flows would flush away the sand and silt and expose the underlying rocky substrates is reflected in the Service's 1995 and 2000 Federal Columbia River Power System (FCRPS) biological opinions, the 1999 recovery plan, and the 2001 critical habitat designation. Subsequent coring and other data from the meander reach revealed that lacustrine clays lie underneath the sand and silt in the meander (current spawning) reach, indicating that the reach has always been comprised of substrates atypical for successful white sturgeon spawning and incubation (Barton 2004). A few isolated pockets of gravel were identified at the mouths of Deep Creek and Myrtle Creek. It is unlikely that these areas of gravel were sufficient to sustain the entire original population of Kootenai sturgeon.

The overall conclusion from the substrate data and the historical information is that it's likely at least a portion of the Kootenai sturgeon population spawned in the canyon reach of the Kootenai River, most likely in the vicinity of Kootenai Falls. However, this new information does not address what actions would be necessary, or if it is even possible to restore this migration and spawning behavior in Kootenai sturgeon. The new information indicates that the earlier view that "flushing flows" were the primary action needed to restore recruitment in Kootenai sturgeon were incorrect.

Reproductively active Kootenai sturgeon respond to increased river depth and flows by ascending the Kootenai River. Although about a third of Kootenai sturgeon in spawning condition migrate upstream to the Bonners Ferry area annually, few remain there to spawn. Kootenai sturgeon have spawned in water ranging in temperature from 2.9 to 13°C (37.3 to 55.4°F). However, most Kootenai sturgeon spawn when the water temperature is near 50°F (10°C) (Paragamian et al. 1997, p. 30). The size or age at first maturity for Kootenai sturgeon in the wild is quite variable (PSMFC 1992, p. 11). In the Kootenai River system, females have been estimated (based upon age length relationships) to mature at age 30 and males at age 28 (Paragamian et al. 2005, p. 525). Only a portion of Kootenai sturgeon are reproductive or spawn each year, with the spawning frequency for females estimated at 4 to 6 years (Paragamian et al. 2005, p. 525). Spawning occurs when the physical environment permits egg development and cues ovulation. Fecundity of Kootenai white sturgeon is up to 200,000 eggs in a single spawning event (Paragamian and Beamesderfer 2004, p. 382). Kootenai sturgeon spawn during the period of historical peak flows, from May through July (Apperson and Anders 1991, p. 50; Marcuson 1994, p. 18). Spawning at near peak flows with high water velocities disperses and prevents clumping of the adhesive, demersal (sinking) eggs.

Following fertilization, eggs adhere to the rocky riverbed substrate (which, as discussed above, is not present in the current Kootenai River spawning reach) and hatch after a relatively brief incubation period of 8 to 15 days, depending on water temperature (Brannon et al. 1985, pp. 58-

64). Here they are afforded cover from predation by high near-substrate water velocities and ambient water turbidity, which preclude efficient foraging by potential predators.

Upon hatching the embryos become "free-embryos" (the larvae stage after hatching with continued dependence upon yolk materials for energy but active foraging begins). Free-embryos initially undergo limited downstream redistribution(s) by swimming up into the water column and are then passively redistributed downstream by the current. This redistribution phase may last from one to six days depending on water velocity (Brannon et al. 1985, pp. 58-64; Kynard and Parker 2006, p. 2). The inter-gravel spaces in the substrate provide shelter and cover during the free-embryo "hiding phase".

As the yolk sac is depleted, free-embryos begin to increase feeding, and ultimately become free-swimming larvae, entirely dependent upon forage for food and energy. At this point the larval sturgeon are no longer highly dependent upon rocky substrate or high water velocity for survival (Brannon et al. 1985, pp. 58-64; Kynard and Parker 2006, p. 3). The timing of these developmental events is dependent upon water temperature. With water temperatures typical of the Kootenai River, free-embryo Kootenai sturgeon may require more than seven days post-hatching to develop a mouth and be able to ingest forage. At 11 or more days, Kootenai sturgeon free-embryos would be expected to have consumed much of the energy from yolk materials, and they become increasingly dependent upon active foraging.

The duration of the passive redistribution of post-hatching free-embryos, and consequently the linear extent of redistribution, is dependent upon near substrate water velocity, with greater linear dispersion anticipated under higher water velocity conditions. However, larvae enter the “hiding phase” sooner when they are in faster currents, thereby limiting their downstream distribution (Brannon et al. 1985, pp. 58-64). Working with Kootenai sturgeon, Kynard and Parker (2006, p. 3) found that under some circumstances this dispersal phase may last for up to 6 days. This prolonged dispersal phase would increase the risk of predation on the embryo and diminish energy reserves. Juvenile and adult rearing occurs in the Kootenai River and in Kootenay Lake.

## 2.3.7.4 Status and Distribution

Distinct population segment of Kootenai River white sturgeon is restricted to approximately 270 RKM (168 RM) of the Kootenai River in Idaho, Montana, and British Columbia, Canada. One of 18 land-locked populations of white sturgeon known to occur in western North America, the range of the Kootenai sturgeon extends from Kootenai Falls, Montana, located 50 RKM (31 RM) below Libby Dam, Montana, downstream through Kootenay Lake to Corra Linn Dam which was built on Bonnington Falls at the outflow from Kootenay Lake in British Columbia. The downstream waters of Kootenay Lake drain into the Columbia River system. Approximately 45 percent of the species' range is located within British Columbia.

Bonnington Falls in British Columbia, a natural barrier downstream from Kootenay Lake, has isolated the Kootenai sturgeon since the last glacial advance roughly 10,000 years ago (Apperson 1992, p. 2). Apperson and Anders (1990, pp. 35-37; 1991, pp. 48-49) found that at least 36 percent (7 of 19) of the Kootenai sturgeon tracked during 1989 over-wintered in Kootenay Lake. Adult Kootenai sturgeon forage in and migrate freely throughout the Kootenai River downstream of Kootenai Falls at RKM 312 (RM 193.9). Juvenile Kootenai sturgeon also forage in and migrate freely throughout the lower Kootenai River downstream of Kootenai Falls and within

Kootenay Lake. Apperson and Anders (1990, pp. 35-37; 1991, pp. 48-49) observed that Kootenai sturgeon no longer commonly occur upstream of Bonners Ferry, Idaho. However, there are no structural barriers preventing Kootenai sturgeon from ascending the Kootenai River up to Kootenai Falls, and this portion of the range remains occupied as documented by Stephens et al. (2010, pp. 14-16), and Stephens and Sylvester (2011, pp. 21-34).

Paragamian et al. (2005, p. 518) indicated that "the wild population now consists of an aging cohort of large, old fish" and cited Jolly-Seber population estimates that indicated Kootenai sturgeon had declined from approximately 7,000 adults in the late 1970s to 760 in 2000. Their results also showed that at the estimated "mortality rate of 9 percent per year, fewer than 500 adults remained in 2005 and there may be fewer than 50 remaining by 2030."

However, in recent years field crews have not noticed an increased difficulty in capturing unmarked sturgeon, as would be expected with a declining population with what should be a high proportion of marked/tagged fish. A 2009 draft report on a review conducted by Cramer Fish Sciences (CFS) for the Kootenai Tribe of Idaho indicated that due to differences in capture probabilities between sturgeon in Kootenay Lake and sturgeon in the Kootenai River, earlier population estimates were biased and as a result, underestimated the adult population and overestimated the mortality rate (Beamesderfer et al., 2009, entire). The draft report estimated the existing adult Kootenai sturgeon population to be approximately 1,000 fish, with a 95 percent confidence interval of 800 to 1,400. The draft report also estimated the annual rate of decline to be four percent (Beamesderfer et al. 2009, p. 2).[7]

Based on data from the period 1992 through 2001, it is estimated that currently an average of only about 10 juvenile sturgeon currently may be naturally reproduced in the Kootenai River annually (Paragamian et al. 2005, p. 524). This suggests that high levels of mortality are now occurring in habitats used for egg incubation and free-embryo development, which are unlikely to sustain a wild population of the Kootenai sturgeon. Natural reproduction at this level cannot be expected to provide any population level benefits, nor would reproduction at this level (20 juveniles per thousand sturgeon per year) have been adequate to sustain the population of 6,000 to 8,000 sturgeon that existed in 1980. The last year of significant natural recruitment was 1974.

In summary, natural spawning in the Kootenai River has not resulted in sufficient levels of recruitment into the aging population of the Kootenai sturgeon to reverse the strong negative population trend that has been observed over the last 40 years. This recruitment failure appears to be related to changes in riverbed substrate and reduced river flows, reduced water velocities, lowered water depths, and downstream movement of the velocity transition points with reduced flows since Libby Dam became operational. While water depth appears to be a significant factor, it is unclear how other altered parameters may be involved in causing the sturgeon to spawn primarily at sites below Bonners Ferry in the meander reach. These sites have unsuitable

[7] In general, the Service agrees with the draft report that recapture biases have skewed previous population estimates and that there are likely more adult Kootenai sturgeon than previously estimated. However, due to choices of models, issues regarding tag loss, and other questions, Service staff are currently working with CFS staff on the report to ensure the revised estimate is robust enough to be cited as "best available science."

sandy riverbed substrates, insufficient rocky substrate (Barton 2004, pp. 18–21; Anders et al. 2002, pp. 73, 76), and water velocities insufficient to provide protection from predation for eggs and free embryos and to assure normal dispersal behavior among free embryos (Parsley et al. 1993, pp. 220– 222, 224–225; Miller and Beckman 1996, pp. 338–339). The upstream braided reach provides suitable rocky substrates, but a large portion of the braided reach has become wider and shallower due to loss of energy from reduced flows, reduced backwater effects, and bed load accumulation (the accumulation of large stream particles, such as gravel and cobble carried along the bottom of the stream) (Barton 2004, p. 17; Barton et al. 2005). The increase in bed load is a result of the broadening of the braids and water velocity reductions.

Hatchery origin Kootenai sturgeon have been released into the Kootenai River since 1990. Releases from 1990 to 1993 were largely experimental and were made up of small year classes. Since 1995, the Kootenai Tribe of Idaho's Kootenai sturgeon aquaculture program has released over 170,000 hatchery origin juvenile sturgeon into the Kootenai basin. Typically between 10,000 and 35,000 juveniles representing as many as 18 family groups are released each year. The larger releases have primarily occurred since 2004. Recapture data indicates that hatchery juvenile Kootenai sturgeon survive at high rates after release, with 60 percent survival the first year after release and 90 percent the following years (Ireland et al. 2002).

However, an analysis by Justice et al. (2009) showed that hatchery origin Kootenai sturgeon released at <25 cm (9.84 in) (roughly corresponding to age-2 juveniles) survived at significantly lower rates than those released at larger sizes. Further, since 2005 sturgeon managers have released either fertilized eggs or free-embryos into reaches of the Kootenai River that have more suitable rocky substrates. Annually, over one million fertilized eggs or free-embryos are released, yet to date these experimental releases have not produced a detected increase in captured unmarked juvenile Kootenai sturgeon (Rust 2010, *in litt.*).

These data have led sturgeon managers to hypothesize that Kootenai sturgeon are experiencing a second survival bottleneck at the larval-to-age-2 stage (the first bottleneck being suffocation of eggs and free-embryos from sand and silt in the braided reach). It is generally thought that the cause of this bottleneck is nutrient/food related, in that there is an insufficient food supply in the Kootenai River for larval and age-1 sturgeon.

## 2.3.7.5 Conservation Needs

Based on the best scientific information currently available, the habitat needs for successful spawning and recruitment fundamental to conserving Kootenai sturgeon are described below. Refer to section 2.4.7.2 for information on factors affecting the sturgeon in the action area.

*Water Velocity*

High "localized" water velocity is one of the common factors of known sites where white sturgeon spawn and successfully recruit in the Columbia River Basin (ODFW 2011). Mean water velocities exceeding 1 m/s (3.3 ft/s) (f/s) are important to spawning site selection. These water velocities provide: trigger cue for adult spawning behavior; cover from predation (Miller and Beckman 1996, Anders et al. 2002); normal free-embryo behavior and redistribution (Kynard 2005); and shelter (living space) for eggs and free-embryos through the duration of the incubation period.

*Water Depth*

The best information currently available indicates that water depth is a factor affecting both migratory behavior and spawning site selection among Kootenai sturgeon. Water depth appears to be a factor in sturgeon migration and spawning site selection. Parsley and Beckman (1994) summarized mean water column depths of sites where sturgeon eggs were found in the lower Columbia River, and observed a range of depths from 4 to 24.1 m (13.2 to 79.2 ft), with most between 5 and 18.1 m (16.5 and 59.4 ft). Paragamian and Duehr (2005) reported depths at which Kootenai sturgeon were found during the spawning period ranging from 2 to 10 m (6.5 to 32.8 ft), with an average depth of 7 m or 23 feet. Of 209 radio contacts with tagged Kootenai sturgeon in spawning condition, 75 percent were within the lower one-third of the water column, and they tended to be found even closer to the bottom during the actual spawning period (Paragamian and Duehr 2005).

These studies suggest that Kootenai sturgeon require thalweg water depths of no less than 5 m (16.5 ft) and ideally up to 7 m (23 ft) at any point between staging areas near Shorty's Island and potential spawning sites throughout the spawning period, in order to facilitate migration of sturgeon in spawning condition for breeding.

These sturgeon also appear to require water depths throughout the breeding period (approximately May 6 through July 3) of 5 m (16.5 ft) and ideally up to 7 m (23 ft) at spawning sites which are located upstream of continuous rock substrates that are approximately 8 river km (5 river mi) in length.

*Rocky Substrate*

Rocky substrate and associated inter-gravel spaces provide both structural shelter and cover for egg attachment, embryo incubation, and normal free-embryo incubation and behavior involving downstream redistribution by the river current.

*Water Temperature/Quality*

Suitable water and substrate quality are necessary for the viability of early life stages of Kootenai sturgeon, including both incubating eggs and free-embryos, and for normal breeding behavior. Lower than normal water temperatures in the spawning reach may affect spawning behavior, location, and timing. Preferred spawning temperature for the Kootenai sturgeon is near 10°C (50°F), and sudden drops of 1.9 to 3.0°C (3.5 to 5.5°F) cause males to become reproductively inactive, at least temporarily. Water temperatures also affect the duration of incubation of both embryos (eggs) and free-embryos.

# 2.3.8 Kootenai River White Sturgeon Critical Habitat

## 2.3.8.1 Legal Status

On September 6, 2001 the Service designated critical habitat for the Kootenai sturgeon. That final rule designated18 RKM (11.2 RM ) of the Kootenai River (Bonner County, Idaho) in the meander reach as critical habitat, from RKM 228 (RM 141.4) to RKM 246 (RM 152.6); that is, from Bonner's Ferry to below Shorty's Island and bounded by the ordinary high water lines (66 FR 46548).

On February 21, 2003, the Center for Biological Diversity filed a complaint against the Corps and the Service (CV 03-29-M-DWM) in Federal Court in the District of Montana, stating, among other issues, that designated critical habitat for the Kootenai sturgeon was inadequate, as it failed to include areas of rocky substrate.

On May 25, 2005, the District Court of Montana ruled in favor of the plaintiffs, and remanded the critical habitat designation to the Service for reconsideration with a due date of December 1, 2005. We filed a motion to alter or amend the judgment, and the Court extended the deadline for releasing a revised critical habitat designation to February 1, 2006. In the interim, the Court ruled that the 2001 designation of critical habitat remained in effect. In response to the District Court ruling and to meet the Court's deadline, we published an interim rule designating an additional reach of the Kootenai River, the braided reach, as critical habitat for the Kootenai River sturgeon on February 8, 2006 (71 FR 6383), resulting in a total of 29.5 RKM (18.3 RM) designated. Although the interim rule designating critical habitat for the Kootenai sturgeon constituted a final rule with regulatory effect, it also opened a comment period on the substance of the rule.

On July 9, 2008, the Service issued a final rule (73 FR 39506) designating 29 RKM (18.3 RM) of the Kootenai River as revised critical habitat within Boundary County, Idaho. This designation maintains as critical habitat the 11 RKM (7.1 RM) ''braided reach,'' and the 18 RKM (11.2 RM ) ''meander reach,'' from the February 8, 2006, interim rule (71 FR 6383). Included within this designation is the 1.5 km (0.9 mi) transition zone that joins the meander and braided reaches at Bonners Ferry, as described in the interim rule. The critical habitat areas described below constitute our best assessment at this time of areas determined to be occupied at the time of listing that contain the physical and biological features essential for the conservation of the species and that the Service has determined require special management.

**Summary of Changes from the Interim Rule**

In developing this revised final critical habitat rule for the Kootenai sturgeon, we reviewed peer review and public comments received on the interim rule and draft economic analysis published in the Federal Register on February 8, 2006 (71 FR 6383), as well as a second round of peer review comments received specifically on the PCEs. The following rule modification description was extracted directly from the final rule.

Based on comments received, including peer review comments, this final rule modifies the interim rule in the following ways:

1. We have made the PCEs more explicit to more clearly communicate the best available scientific information regarding the conservation needs of the species[8].
2. We have modified the depth PCE (PCE 1) from a minimum of 5 m (16 ft ) to a minimum of 7 m (23 ft ) to more accurately reflect the best available science, indicating that mean

[8] Although the Service identified sediment and water quality components as a PCE (#4) in the 2001 Critical Habitat Rule, of importance for the Effects Section of this Opinion is the fact that the Service removed sediment and water quality as a PCE in the 2008 Revised Final Rule after determining these were not limiting factors(Flory 2014, pers. comm).

water depth of at least 7 m (23 ft ) is necessary for spawning site selection by white sturgeon in the Kootenai River (for example, Paragamian et al. 2001, Table 2, p. 27, p. 29, and Figure 4, p. 29; Paragamian and Duehr 2005, p. 263, 265).

3. In the interim rule, we stated that we added 11.1 RKM (6.9 RM) to the critical habitat designation, but later stated that this additional reach extends from RKM 257 (RM 159.7) to RKM 245.9 (RM 152.6)), which is actually 11.4 RKM (7.1 RM). The area designated as critical habitat in the interim rule remains unchanged in this revised final rule. This final rule simply corrects the RKM totals to indicate that we added 11.4 RKM (7.1 RM) to our 2001 designation of 18 RKM (11.2 RM), for a total of 29.5 RKM (18.3 RM).

4. We have combined the two former units, the braided reach and the meander reach, into a single designation because the two units are contiguous, and clarified the location of the river reaches within the designation: (i) The braided reach begins at RKM 257.0 (RM 159.7), below the confluence of the Moyie River, and extends downstream within the Kootenai River to RKM 246.0 (RM 152.6 ) below Bonners Ferry; (ii) The meander reach begins at RKM 246.0 (RM 152.6 ) below Bonners Ferry, and extends downstream to RKM 228.0 (RM 141.4 ) below Shorty's Island; and, (iii) This designation includes the 1.5 km (0.9 mi ) ''transition zone,'' described in the February 2006 interim rule (71 FR 6383) that joins the meander and braided reaches at Bonners Ferry.

## 2.3.8.2 Conservation Role and Description of Critical Habitat

For inclusion in a critical habitat designation, the habitat within the geographical area occupied by the species at the time of listing must contain the physical and biological features essential to the conservation of the species, and be included only if those features may require special management considerations or protection. Critical habitat designations identify, to the extent known using the best scientific data available, habitat areas that provide essential life cycle needs of the species. Under the Act, we can designate critical habitat in areas outside the geographical area occupied by the species at the time it is listed only when we determine that those areas are essential for the conservation of the species.

The final designation focuses solely on spawning and rearing habitats, the factors that we understand to be currently limiting to sturgeon conservation (Paragamian et al. 2001, pp. 22–33; Paragamian et al. 2002, pp. 608, 615). All of the following PCEs must be present during the spawning and incubation period for successful spawning, incubation, and embryo survival to occur. However, although the PCEs to support successful spawning must occur simultaneously in time and space, it is not necessary for them to be present through the entire spawning period, nor must they be present throughout the entire designated area. The PCEs are:

1. A flow regime, during the spawning season of May through June, that approximates natural variable conditions and is capable of producing depths of 23 ft (7 m) or greater when natural conditions (for example, weather patterns, water year) allow. The depths must occur at multiple sites throughout, but not uniformly within, the Kootenai River designated critical habitat.

2. A flow regime, during the spawning season of May through June, that approximates natural variable conditions and is capable of producing mean water column velocities of 3.3 ft/ s (1.0 m/s) or greater when natural conditions (for example, weather patterns,

water year) allow. The velocities must occur at multiple sites throughout, but not uniformly within, the Kootenai River designated critical habitat.

3. During the spawning season of May through June, water temperatures between 47.3 and 53.6 °F (8.5 and 12 °C), with no more than a 3.6 °F (2.1 °C) fluctuation in temperature within a 24- hour period, as measured at Bonners Ferry.
4. Submerged rocky substrates in approximately 5 continuous river miles (8 river kilometers) to provide for natural free embryo redistribution behavior and downstream movement.
5. A flow regime that limits sediment deposition and maintains appropriate rocky substrate and inter-gravel spaces for sturgeon egg adhesion, incubation, escape cover, and free embryo development. Note: the flow regime described above under PCEs 1 and 2 should be sufficient to achieve these conditions.

As stated previously, this critical habitat designation is focused on Kootenai sturgeon spawning habitats and egg attachment and egg incubation habitats, as these areas are currently the limiting habitat components essential to Kootenai sturgeon conservation (Paragamian et al. 2001, pp. 22–33; Paragamian et al. 2002, pp. 608, 615). Maintaining the PCEs in this designated area is consistent with our recovery objective to re-establish successful natural recruitment of Kootenai sturgeon (USFWS 1999, p. iv). However, the presence of PCE components related to flow, temperature, and depth are dependent in large part on the amount and timing of precipitation in any given year. These parameters vary during and between years, and at times some or all of the parameters are not present in the area designated as critical habitat. Within the critical habitat reaches, the specific conditions are variable due to a number of factors such as snowmelt, runoff, and precipitation.

This designation recognizes the natural variability of these factors, and does not require that the PCEs be available year-round, or even every year during the spawning period. At present, the PCEs are achieved only infrequently, such as in 2006 during the ‘‘stacked flow’’ operations when the Kootenai River reached river stage 1,763.61 MSL (feet above mean sea level; 537.5 m) at Bonners Ferry (USCOE 2007, p. 6), resulting in the first documented movement of tagged female Kootenai sturgeon into the braided reach above Bonners Ferry. The designation means that sufficient PCE components to support successful spawning must be present and protected during the spawning season of May through June at multiple sites throughout, but not uniformly within, the Kootenai River designated critical habitat in all years when natural conditions (for example, weather patterns, water year) make it possible.

We recognize that, due to existing morphologic constraints and limitations at Libby Dam, the depth PCE described in this rule (23 ft; 7 m) is currently not achievable on an annual basis in the braided reach. Since the construction of Libby Dam and the subsequent altered hydrograph, the braided reach has become shallower and wider (Barton et al. 2005, unpublished data), thus limiting the ability to achieve the depth PCE in the braided reach in most years. To address this issue, the Kootenai Tribe of Idaho, in cooperation with regional partners and Federal managers, is pursuing the Kootenai River Ecosystem Restoration Project. This restoration project has as one of its goals to “restore and maintain Kootenai River habitat conditions that support all life stages” of Kootenai sturgeon including addressing sturgeon depth requirements (Kootenai Tribe of Idaho (KTOI) 2009). Until this project is implemented, we recognize that the ability to meet the depth PCE in the braided reach is limited. However, we also acknowledge that the depth PCE has been achieved intermittently under current operating conditions (stacked flows in 2006).

### 2.3.8.3 Current Rangewide Condition of Kootenai River White Sturgeon Critical Habitat

Both of the designated critical habitat reaches provide the physical and biological features that are essential to the Kootenai sturgeon for spawning, egg attachment, incubation, and juvenile rearing, and both require special management to ensure that the appropriate water depths, velocities, and temperature are achieved during the spawning period in all years when natural conditions allow.

**Braided Reach**

The braided reach begins at RKM 257 (RM 159.7), below the confluence with the Moyie River, and extends downstream within the Kootenai River to RKM 246 (RM 152.6 ) below Bonners Ferry. Within this reach the valley broadens, and the river forms the braided reach as it courses through multiple shallow channels over gravel and cobbles (Barton 2004, pp. 18–19). This reach was occupied by Kootenai sturgeon at the time of listing, and is currently occupied by foraging and migrating sturgeon. Tagged female sturgeon moved into the braided reach above Bonners Ferry during the spawning period in 2006, although it is not known whether spawning occurred in the area (Kootenai Sturgeon Recovery Team 2006, pp. 1–2). Gravel and cobble are exposed along the bottom of the Kootenai River in the braided reach (Barton 2004, pp. 18–19; Berenbrock 2005, p. 7), and water velocities in excess of 1 m/s (3.3 ft/s) are likely achieved on a seasonal basis due to the high surface gradient in this reach (Berenbrock 2005, Figure 11, p. 23). At present, the braided reach provides the temperatures, depths, and velocities required to trigger spawning only occasionally, and these features require special management for spawning sturgeon.

**Meander Reach**

The meander reach begins at RKM 246 (RM 152.6) below Bonners Ferry, and extends downstream to RKM 228 (RM 141.4) below Shorty’s Island. This reach was occupied by Kootenai sturgeon at the time of listing, is used by foraging and migrating sturgeon, and is currently the primary spawning reach for Kootenai sturgeon (Paragamian et al. 2002, p. 608, and references therein). Although most of the reach is composed primarily of sand substrates unsuitable for successful spawning, some limited areas of gravel and cobble are present or at least exposed intermittently (Paragamian et al. 2002, p. 609; Barton 2004, pp. 18–19). Although appropriate spawning depths are available on occasion in this reach (Paragamian et al. 2001, Table 2, p. 26; Barton 2004, Table 1, p. 9), the temperatures and velocities required for successful spawning require special management to be achieved on more than an infrequent basis.

In summary, natural spawning in the Kootenai River has not resulted in sufficient levels of recruitment into the aging population of the Kootenai sturgeon to reverse the strong negative population trend that has been observed over the last 30 years. This recruitment failure appears to be related to changes in riverbed substrate and reduced river flows, reduced water velocities, lowered water depths, and downstream movement of the velocity transition points with reduced flows since Libby Dam became operational. While water depth appears to be a significant factor, it is unclear how other altered parameters may be involved in causing the sturgeon to spawn primarily at sites below Bonners Ferry in the meander reach. These sites have unsuitable sandy riverbed substrates, insufficient rocky substrate (Barton 2004, pp. 18–21; Anders et al. 2002, pp. 73, 76), and water velocities insufficient to provide protection from predation for eggs and free embryos and to assure normal dispersal behavior among free embryos (Parsley et al. 1993, pp. 220– 222, 224–225; Miller and Beckman 1996, pp. 338–339). The braided reach provides suitable rocky substrates, but a large portion of the braided reach has become wider and shallower due to loss of energy from reduced flows, reduced backwater effects, and bed load accumulation (the accumulation of large stream particles, such as gravel and cobble carried along the bottom of the stream) (Barton 2004, p. 17; Barton et al 2005 and unpublished data). The increase in bed load is a result of the broadening of the braids and water velocity reductions.

# 2.4 Environmental Baseline of the Action Area

This section assesses the effects of past and ongoing human and natural factors that have led to the current status of the species, its habitat and ecosystem in the action area. Also included in the environmental baseline are the anticipated impacts of all proposed Federal projects in the action area that have already undergone section 7 consultations, and the impacts of state and private actions which are contemporaneous with this consultation.

Actions that form the environmental baseline for this consultation include but are not limited to: dam operation and the resulting impacts to the environment [creation of reservoirs, disruption of river flows, redistribution and retention of sediments, solar heating, reduced DO, creation of physical (dams) and habitat (reservoirs) barriers to dispersal]; diversion and nutrient loading of spring and river waters; complete dewatering of some riverbed areas (water diverted for urban and agriculture use); and degradation of water quality due to point and non-point sources of pollutants or nutrient enrichment (e.g., run-off and aquifer recharge from range or farm land). These activities represent a combination of State, private, and Federal actions, conducted on State, private, and/or Federal lands.

Aside from anadromous and resident salmonids and the white sturgeon, little is known regarding the distribution and abundance of the endemic biota of the Snake River prior to dam construction. Early accounts reference the abundance of salmon that used this river and its tributaries as spawning grounds (Evermann 1896, pp. 262-276). Fish movement, and that of other aquatic species, was unimpeded by dams and human use of the river at that time had not resulted in the suite of water degrading uses that now affect the river. Given the early distribution of salmon, it is very likely that most of the Snake River snails were far more widespread throughout the river system and historical collections indicate this to be the case. As with the salmon that once thrived in the Mid-Snake and its tributaries, the native snail fauna has undoubtedly been negatively impacted by the multitude of human alterations to this river.

# 2.4.1 Snake River Physa Snail

## 2.4.1.1 Status of Snake River Physa Snail in the Action Area

Because the range of the Snake River physa is contained entirely within the action area, refer to section 2.3.1 of this Opinion for the baseline status for the Snake River physa snail.

## 2.4.1.2 Factors Affecting the Snake River Physa Snail in the Action Area

The Service's final rule classifying Snake River physa as endangered ((57 FR 59244) identified the following threats to the species: construction of new hydropower dams, operation of existing hydropower dams, water quality degradation, water diversions and groundwater withdrawals for agriculture and aquaculture, small hydroelectric development, lack of State regulations, pollution regulations, Federal consultation regulations, and competition with the non-native New Zealand mudsnail. The information contained in the following sections updates what the Service stated at the time of listing. Additionally, factors that may affect the Snake River physa seldom act independently, but rather interact synergistically and/or cumulatively, and should be regarded holistically instead of as separate threats. These threats and conservation actions are discussed in more detail in this section.

Refer to section 2.3.1.5 for more information on the conservation needs of the Snake River physa.

**Construction of New Hydropower Dams**

Proposed hydroelectric projects within the range of Snake River physa as discussed in the 1993 final listing rule were never approved for construction. The A.J. Wiley project and Dike Hydro Partners preliminary permits have lapsed; the Kanaka Rapids, Empire Rapids, and Boulder Rapids permits were denied by the Federal Energy Regulatory Commission (FERC) in 1995. There was a notice of surrender of the preliminary permit for the River Side Project in 2002 and two other proposed projects, the Eagle Rock and Star Falls Hydroelectric Projects, were denied preliminary permits by the FERC. In 2003, a notice was provided of surrender of the preliminary permit for the Auger Falls Project. Information provided by the state of Idaho indicates that all proposals and preliminary permits for the construction of new dams along the mid-Snake River have either lapsed or been denied by the FERC (Caswell 2007, *in litt.*). Today, the Service is unaware of any hydroelectric development proposals within the species known range that would threaten the Snake River physa.

While there are no immediate or specific plans for dam and reservoir development within the range of the Snake River physa, the Idaho Water Resource Board (IWRB) has proposed the need to consider such development in the future. Development of specific new dams or reservoirs within the Snake River is not mentioned in the 2012 Idaho State Water Plan, though that plan does state that future surface water development will continue to play an important role in the State's future (IWRB 2012, pp. 18-20), and the "existing capacity is insufficient to provide the water supply and management flexibility needed…", and that "New Snake River surface storage projects should be investigated and constructed if determined to be feasible" (IWRB 2012, p. 55). Any water development/management activities that would directly alter lotic habitats (e.g., construction of new reservoirs), or reduce flows within the Snake River will pose a threat to the free-flowing river habitats important to the species.

### Operation of Existing Hydropower Dams

The impacts from the presence of dams and reservoirs, and subsequent alterations of flows are well documented and generally known to have negative impacts on macroinvertebrate species (Fisher and LaVoy 1972, p. 1473; Kroger 1973, pp. 479-480; Brusven et al. 1974, pp. 75-76; Gislason 1980, pp. 83-85; Gersich and Brusven 1981, p. 235; Armitage 1984, pp. 141-142; Brusven 1984, pp. 167-168; Poff et al. 1997, pp. 776-777). In the following section, we will discuss the threat of the operation of existing dams on the Snake River physa through two avenues; daily fluctuations of water levels due to hydropower operations (*Peak-Loading*), and; seasonal fluctuations of water levels due to irrigation water delivery (*Dam Operations for Irrigation Purposes*).

#### *Peak-Loading*

"Peak-loading (the operation of dams that are directly in response to electricity demands) is a frequent and sporadic practice that results in dewatering mollusk habitats in shallow, littoral shoreline areas" (57 FR 59252). Peak-loading operations within the range of the Snake River physa occur at the Bliss Dam (RKM 901 (RM 560)), Lower Salmon Falls Dam (RKM 922 (RM 573)), C.J. Strike Dam (RKM 789 (RM 490), and Swan Falls Dam (RKM 736.6 (RM 457.7)) (USFWS 2004a, pp. 19, 20; USFWS 2012b, p. 5).

Irving and Cuplin (1956, entire) provided information on the effects that hydropower peak-loading had on the aquatic organisms of the Mid-Snake River (approximately RKM 943 to RKM 711 (RM 586 to RM 442)). Their work showed a pronounced decrease in number (reduced by 84 percent) and biomass (reduced by 92 percent), of benthic invertebrates in the shallow tailwaters of both the Lower Salmon Falls and Bliss Dams, as compared to reaches of the river where flows were maintained at more natural levels.

Subsequent studies have also reported negative impacts to benthic invertebrates such as stranding and desiccation, and all of these studies inferred or noted reduced abundance of benthic invertebrates in de-watered areas (Fisher and LaVoy 1972, p. 1472; Kroger 1973, p. 478; Brusven *et al.* 1974, p. 78; Brusven and MacPhee 1976, p. iv). Members of the family Physidae are a relatively mobile group of aquatic snails, and being members of the "lung-breathing" Class Pulmonata, are typically capable of some limited respiration out of aquatic habitats. Under certain conditions, members of the aquatic pulmonates, and notably the Physidae, may actively leave the water to avoid predators (Dillon 2000, pp. 307-309). Covich *et al.* (1994, p. 287) observed protean physa remain out of the water for hours and days to avoid predation. Although a number of these snails died from desiccation, about 87 percent survived. Similarly, it is plausible that physids may be able to re-enter, or follow water should their habitats suddenly be dewatered. Since the Snake River physa primarily occurs in deeper habitats, it is less likely to be within the regularly dewatered zone caused by peak-loading from hydroelectric dams. However, peak-loading likely limits available habitats for Snake River physa in regularly de-watered areas of the river channel, restricting them to deeper portions that are located well within continuously watered habitats.

At Bliss and Lower Salmon Falls Dams, peak-loading operations can result in river stage changes downstream of the dams of up to 1.5 to 1.8 m (5 and 6 ft) per day for the two dams respectively (USFWS 2012b, p. 9). As stated above, the Snake River physa does not appear to be common downstream of Bliss Dam and Lower Salmon Falls Dam. Downstream of C.J. Strike Dam, fluctuations up to 1.2 m (4 ft) in the tailwaters may result during each peak-loading episode associated with loading operations (USFWS 2004a, p. 20). Given the sparse occurrence data of Snake River physa downstream of C.J. Strike Dam, and the rarity of the species in this reach, it is difficult to assess the threat of peak loading from C.J. Strike Dam on Snake River physa.

While peak-loading operations occur to a certain extent below Swan Falls Dam (RKM 736.6 (RM 457.7); its primary operation is to re-regulate flows from C.J. Strike Dam, which is located approximately 52 RKM (32 RM) upstream), its operation has been determined not to rise to the level of impacting the Snake River physa in a manner that would result in population level effects, though low summer flows, nutrient loading, and sediment deposition are considered the most significant threat to the species downstream of this dam (USFWS 2012b, p. 43). If habitat conditions worsen downstream of Swan Falls Dam, additional impacts to the species habitat may occur, though at this time, without further information it is difficult to project if this will occur and how it would affect the species persistence in this area (USFWS 2012b, p. 43).

*Dam Operations for Irrigation Purposes*

Unlike Snake River dams whose operations require peak-loading in response to electricity demand, the primary purpose of other Snake River dams is to provide storage water for irrigation (e.g. Minidoka Dam, Milner Dam). One of the primary differences between these two operational regimes on Snake River physa habitat is that dams operated for irrigation purposes can dewater large areas of river habitat for a much greater duration of time than for peak-loading operations. Therefore the potential effects of irrigation dewatering on the Snake River physa possess similarities to those experienced during peak-loading operations (see above under *Peak-Loading*). However, whereas peak-loading entails more frequent, short-term dewatering episodes, irrigation management imposes infrequent (e.g., seasonal) but extended periods of dewatering, often dewatering larger benthic areas.

The most robust known population of Snake River physa occurs in 18.5 km (11.5 mi) of the Snake River downstream of Minidoka Dam (RKM 1086 (RM 675)), which is operated by the USBOR. This dam is operated to provide irrigation water during summer months, so summer discharges are kept at a higher rate than during the winter months, and therefore the river below the dam mimics more of a natural hydrograph with flows increasing in spring, peaking during summer, and tapering off through the fall. Downstream of Minidoka Dam, Snake River physa have been found predominately in permanently watered habitat greater than 1.2 m (3.9 ft) in depth (Gates and Kerans 2010, p. 4). In addition, Gates and Kerans (2010, p. 5) found that even after 5 months of water immersion of the littoral zone during elevated irrigation flows, most mollusk species were more commonly recorded in deeper areas of the channel, those habitats watered year-round. It is possible that the area where this population of Snake River physa occurs has experienced consistent seasonal dewatering (4-6 months/ year) of approximately 30% of the riverbed since 1910, the year Minidoka Dam began diverting flows for irrigation (Gates and Kerans 2010, p. 9).

USBOR has committed to a minimum flow of 11.2 cubic meters per second (cms) (400 cubic feet per second (cfs)) outflow from Minidoka Dam, so the deepest portions of the riverbed remains submerged year round (USFWS 2005a, p. 27). This is important as the Snake River physa is mostly found within the deepest portions of the Snake River within this reach. If this minimum flow requirement was removed, and flows during winter fell below this minimum, additional portions of the riverbed would be exposed to freezing temperatures. This would further impact the only known robust[9] population of Snake River physa.

Substrate composition was also found to significantly differ between watered and dewatered sampled habitat downstream of Minidoka Dam, with more silt occurring in the seasonally dewatered areas of the river bed (Gates and Kerans 2010, p. 36), which is not a suitable substrate for the Snake River physa. Although Snake River physa have continued to persist in this reach, continued dam operations at Minidoka Dam likely limit suitable habitat potentially available for the species.

There are other dams within the range of the species that divert water out of the Snake River for irrigation purposes. During low-water years Milner Dam (RKM 1028.5 (RM 639.1)) diverts all measurable flows from the river during the irrigation season to provide water to fulfill nonfederal water rights holdings for agriculture (USFWS 2005a, p. 29; IWRB 2012, pp. 42-48; see Figure 2). This results in approximately 2.6 km (1.6 mi) of the Snake River immediately downstream of Milner Dam being cut off from river flows, some of which are put back into the stream channel further downstream, via a bypass (irrigation) canal through a hydroelectric plant. Milner Dam has been in operation since 1905 (Yost 2013, *in litt.*), meaning impacts related to reduced or no river flow have occurred there for over a century. Water quality downstream of Milner Dam is also substantially compromised since a significant proportion of the source water downstream of the dam is from irrigation return flows (Clark et al. 1998, pp. 8, 18). This reach of the Snake River is documented to be water quality limited until significant volumes of groundwater enter into the river from the Eastern Snake River Plain Aquifer (ESPA) in the Thousand Springs to King Hill area ("north-side springs"; approximately RKM 940-982 (RM 584-610)) (Clark et al. 1998, pp. 18-19). While it is unknown what the status of Snake River physa is between Milner Dam and Lower Salmon Falls Dam (the next Snake River dam downstream of Milner Dam) due to the lack of surveys, the reduced water quality and poor river habitat condition in this reach would not be expected to support the species.

[9] The description of the Snake River physa population directly below the Minidoka Dam as "robust" means that this population of snails is sufficiently large numerically to have been repeatedly found and monitored and is considered stable, thus seeming to be maintaining their population over time. The status of other Snake River physa populations has been more difficult to verify.

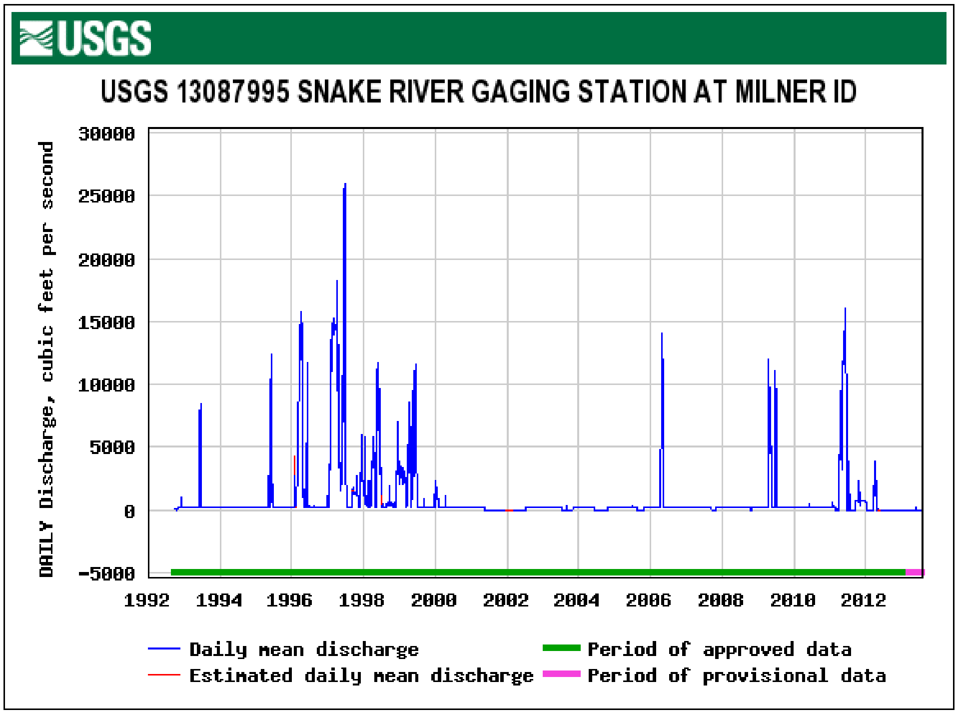


**Figure 1. Snake River Flows at Milner Dam from 1993 (time of listing) through early 2013.**

While water is diverted for agricultural purposes at C.J. Strike Dam, the primary reason for its operation is to provide hydroelectricity. It is unknown how much water is diverted for agriculture purposes at C.J. Strike Dam, however, under the current license requirements, discharge from this dam cannot drop below 110 cms (3,900 cfs), helping to ensure some minimal flows in the Snake River (USFWS 2004a, p. 20). Given that information on the distribution and abundance of the Snake River physa downstream of C.J. Strike Dam is limited, it is difficult to assess the effects of these diversions at this dam on the species in this reach.

In summary, Snake River physa have been documented downstream of five dams on the Snake River, indicating that the species can exist to a certain extent with existing dams and their operations. Downstream of Minidoka Dam, the largest known Snake River physa population (along with most mollusk species) is found predominantly in habitat that is not seasonally dewatered. The relationship between the Snake River physa and other Snake River dams within its current known range is much less clear due to limited surveys and occurrence information, though existing information indicates that Snake River physa populations below the other dams are not as large or robust as the population downstream of Minidoka Dam. While hydroelectric operations may not be directly affecting the Snake River physa, their operations, in concert with other threats such as degraded water quality, likely limits the suitable habitat available to the species, especially where water levels can fluctuate substantially over short time periods (e.g. daily) from normal flows, or from the lack of flushing type flows during the summer months. Therefore we have determined operation of existing dams is a factor affecting the Snake River physa.

## Degraded Water Quality

Factors that are known to degrade water quality in the Snake River include reduced water velocity, warming due to impoundments, and increases in the amounts of nutrients, sediment, and pollutants reaching the river (USFWS 2005a, p. 114). Reduced flow/ discharge increases water residence time in reservoirs, and allow for temperature increases in both reservoirs and in unimpounded reaches. These factors often lead to increases in primary productivity, phytoplankton levels, nutrient concentrations (FERC 2010, p. 35), and proliferation of algal and rooted macrophytes.

Several water quality assessments have been completed for the Snake River by the U.S. Environmental Protection Agency (EPA), USBOR, U.S. Geological Survey (USGS), and IPC. All generally demonstrate that the water quality in the Snake River of southern Idaho is good for some months of the year (e.g. meeting Idaho's water quality criteria for the protection of aquatic life), but may be poor during summer high temperatures and low flows when water quality criteria such as dissolved oxygen may not be attained (Clark et al. 1998, p. 23; Clark and Ott 1996, p. 553; Clark 1997, pp. 8, 9, 19; Meitl 2002, pp. 32, 33; Clark et al. 2004, p. 38; Kosterman et al. 2008, p. 45). The Idaho River Ecological Assessment Framework (Grafe 2002, entire) and the Idaho Assessment of Ecological Condition [Rivers] (Kosterman et al. 2008, p. 45), document changes in the ecological condition[10] of the Snake River, with a decline in water quality and ecological condition from southeastern Idaho upstream of Heise (RKM 1370 (RM 851)) to southwestern Idaho near Weiser (RKM 565 (RM 351)).

In the Snake River downstream of Twin Falls, approximately 144 cms (5,100 cfs) of groundwater originating from the ESPA enters the Snake River, greatly increasing base flows (EPA 2002a, pp. 4-9) so that discharge at King Hill (RKM 882 (RM 548)) does not drop below 156 cms (5,500 cfs). These aquifer springs provide relatively clean and cool water that is also ideal for commercial trout production. This reach of the Snake River has numerous licensed aquaculture facilities responsible for approximately 76 percent of the commercial trout production in the U.S. with several of these operations including fish-processing facilities (EPA 2002a, pp. 4-10). Both aquaculture operations and fish-processing facilities contribute wastes which make their way into the Snake River, including ammonia, bacteria, dead fish, fish feces, suspended sediments, and residual quantities of drugs and chemicals used to control disease outbreaks (EPA 2002a; pp. 4-20). Falter and Hinson (2003, pp. 26, 27) reported "significantly higher concentrations" (i.e. elevated, not increasing) of nitrogen and phosphorous, as well as higher levels of trace elements including zinc, copper, cadmium, lead, and chromium in sediments downstream of aquaculture facilities when compared to areas upstream of those facilities. The impact of these effluents and trace elements to the growth, survival, and reproduction of Snake River physa is unknown, but recent studies have shown another native Snake River species, the Jackson Lake springsnail (*Pyrgulopsis robusta*) is highly sensitive to copper (a common component in algaecides), and pentachlorophenol, a restricted-use pesticide/wood preservative (Ingersoll 2006, p. 3). Both aquaculture facilities and irrigation

[10] Ecological condition can be defined as "the state of the physical, chemical, and biological characteristics of the environment, and the processes and interactions that connect them" (EPA 2008a, p. 6-3).

conveyances typically require the periodic use of algaecides to keep facilities and canals free of filamentous algal growth. Some of these compounds contain copper and are known to be highly toxic to snails, and may also affect diatoms (unicellular algae), the likely primary food source for Snake River physa. Lastly, benthic macroinvertebrate densities and biomass in Snake River studies have been shown to generally increase downstream of aquaculture discharges with a concomitant decrease in species richness, indicating an overall decline in habitat quality immediately downstream of aquaculture facilities (Falter and Hinson 2003, p. 13).

Over 23,310 square kilometers ($km^2$) (9,000 square miles ($mi^2$)) of irrigated land are located within the Snake River drainage or that of its tributaries (Johnson et al. 2013, *in litt.*). Most of the crops grown in this area are subject to modern agricultural practices which include the use of herbicides, insecticides, fungicides, and fertilizers (which may include copper); a proportion of which make their way into the Snake River via irrigation return flows and through ground water recharge (Clark et al. 1998, p. 2).

Cattle production and confinement has increased substantially in south central Idaho within the range of the Snake River physa (Cassia, Gooding, Jerome, Minidoka, and Twin Falls Counties). From 1992 through 2012, total cattle numbers in these counties increased by over 100 percent, from an estimated 467,500 to 946,500 head (both dairy and beef combined; USDA 2013, *in litt.*). Wastewater from confined animal feeding operations has been identified as a major contributor to water quality degradation in surface waters, groundwater, and springs in southern Idaho (Clark et al. 1998, p. 19; Bahr and Carlson 2000a, p. 2; Schorzman et al. 2009, p. 19). Nitrate values from monitored wells in southern Idaho between 1990 and 2003 indicate an increasing trend in concentrations overall, although there were decreases at some wells (Neely 2005, pp. 5-11). Clark et al. (1998, p. 3) report that 10 percent of the wells sampled between Burley and Hagerman contained nitrate concentrations in excess of 10 mg/L, quantities regarded as harmful to human health.

Several other environmental pollutants have been documented in the Snake River within the range of Snake River physa. Water samples collected at locations in the middle and upper Snake River including Box Canyon (RKM 946 (RM 588)), between 1989 and 2000, had concentrations of cadmium and lead exceeding the state of Idaho's acute or chronic criteria (Hardy et al. 2005, pp. 17, 64, 65). Research at Montana State University revealed concentrations of lead, cadmium, and arsenic in the tissues of native Snake River snails (Richards. 2002, *in litt.*), but observations of effects from these concentrations were not reported. In additional studies, Rattray et al. (2005) detected trace elements including barium, chromium, lithium, manganese, and zinc in water samples that supply the major springs on the north side of the Snake River (Rattray et al. 2005, pp. 7, 8). While many of these pollutants are present in relatively low concentrations throughout the species' range, and in some locations exceed EPA aquatic life standards, the effect of most of these pollutants on Snake River physa is unknown.

The human population has also grown within southern Idaho. For example, from 2000 through 2011, the human population in Cassia, Gooding, Jerome, Minidoka, and Twin Falls Counties in southern Idaho grew 15 percent (U.S. Census Bureau 2013, *in litt.*), with the city of Twin Falls growing by 20 percent from 2000 to 2010 (City of Twin Falls Data 2013, *in litt.*). Sewage treatment facilities from these municipalities have permitted National Pollutant Discharge Elimination System (NPDES) discharges of nutrients, ammonia, suspended solids, organic matter, and industrial wastes into the Snake River (Clark et al. 1998, p. 7; EPA 2002a, pp. 4-19).

Other nonpoint discharges from urban areas, such as parking lot run-off and urban-use pesticides (Clark et al. 1998, p. 7), do not undergo treatment but can be reasonably expected to make their way into the Snake River and/or its tributaries. Although urban run-off likely contributes to declines in water quality in the Snake River, it is not considered to be a major source of pollutants (Clark et al. 1998, p. 19).

One avenue to assess recent trends of water quality throughout the range of the Snake River physa is through evaluation of existing nutrient and contaminant loads through the Total Maximum Daily Load (TMDL) monitoring program (see Section 2.3.2.4 - *Inadequacy of Existing Regulatory Mechanisms* for detailed information regarding TMDLs). The Snake River downstream of Minidoka Dam (the uppermost range of the Snake River physa and site of the most robust known population) to Milner Dam was listed as not meeting the State's criteria for sediment, dissolved oxygen, total phosphorus (TP; a nutrient source for macrophyte growth), and oil and grease (IDEQ 2000, p. 46). Two of these, total suspended solids (TSS) and TP, were found at higher concentrations with increasing proximity to Milner Reservoir relative to concentrations further upstream at Minidoka Dam, likely due to the result of numerous drains and tributaries that empty into the Snake River as one moves downstream (IDEQ 2000, pp. 64-65). The recent 5-year review for the TMDL indicates that this stretch of the Snake River continues to be listed as not supporting water quality standards for TP, and may not be supporting TSS, though additional data is needed. TP values are actually higher than those recorded before the TMDL was established (IDEQ 2012, pp. 26 and 72), indicating that water quality may further be deteriorating since the TMDL was established.

In 2010, IDEQ completed the 5-year review for the TMDL for the Middle Snake River Watershed Management Plan (1997), Upper Snake Rock Watershed Management Plan (2000), and the Upper Snake Rock Modification (IDEQ 2010, entire). This review covers the section of the Snake River and certain tributary segments from near Milner Dam (RKM 1027.6 (RM 638.5)) at Murtaugh, Idaho to King Hill, Idaho (RKM 877.1 (RM 545.0); IDEQ 2010, p. xii), where the primary pollutants of concern are TSS and TP (IDEQ 2010, p. xi). Although this section is the species type locality, more recent surveys have been unsuccessful in locating the species in this section of the Snake River. Generally, water quality has improved in this section of the Snake River (Buhidar 2006, *in litt.*; IDEQ 2010, p. xiii) although TP is still elevated (IDEQ 2010, pp. 7, 36).

The Mid Snake River/Succor Creek Subbasin TMDL implementation plan was completed in July of 2005, with the latest 5-year review completed in September, 2011 (IDEQ 2011). This TMDL encompasses a large portion of southwest Idaho, and includes the Snake River between Swan Falls Dam (RKM 736.6 (RM 457.7)) and the Oregon State line (RKM 654.2 (RM 406.5)). Previously (1995-2003), this section of the Snake River yielded collections of Snake River physa (IPC 2012, *in litt.*). The 5-year review for this TMDL indicates water quality is declining, with sediment, temperature, bacteria, and phosphorus the main sources of pollution (IDEQ 2011, p. v). Total Phosphorous (the only pollutant in the Snake River with an allocation in this TMDL) levels within this Snake River subbasin appear to have increased and are above criteria, although the trend is not clear (IDEQ 2011, p. 31).

Downstream of Minidoka Dam, the river reach containing the most robust known population of Snake River physa in the Snake River and the population appears to have been stable over the past 6 years, this area of the Snake River is still experiencing higher pollutant levels such as TP and potentially TSS due to numerous drains and tributaries entering the Snake River. What likely counteracts the degraded water quality conditions downstream of the Minidoka Dam is that flushing flows are higher during the summer and early autumn months, likely keeping the pebble and gravel beds free of fine sediments and macrophytes during the period of highest insolation and summer temperatures. As stated in Section 2.3.1.4, Snake River physa have been collected with less sampling effort within the Minidoka reach versus the Lower Salmon Falls Dam to Ontario, Oregon reach, indicating the species is less abundant outside the Minidoka reach. This is likely due to various reasons, including suitable habitat availability, water quality deterioration, and altered flow regimes (for example, flows are maintained at higher rates, and for longer periods, during summer downstream of Minidoka Dam, while the inverse is true downstream of Swan Falls Dam).

In summary, surface water quality in the Snake River has been impacted by the cumulative effects of decades of agricultural, municipal, and industrial activities within the watershed, and by the regulation of flows. As discussed above in Section 2.3.1 Biology and Habitat, the current ranges of water temperatures in the Snake River do not seem to limit Snake River physa; the species appears to tolerate the range of temperatures observed. However, additional factors such as sediments or suspended solids introduced into the Snake River from livestock use, agricultural run-off, fish production wastes, and other land uses (Bowler et al. 1992, p. 45; Hardy et al. 2005, p. 7), are likely filling the interstitial spaces between bed substrates and providing an environment favorable for macrophyte growth in the river. However, while degraded water quality (primarily due to increased sediment and nutrients) does not currently appear to be negatively affecting Snake River physa habitat uniformly across its range, it likely reduces available suitable habitat (i.e. relatively clean gravel to pebble, and possibly gravel to cobble with limited fines and macrophytes) in several Snake River reaches outside of the Minidoka reach, within the range of the species. Therefore, we have determined degraded water quality is a threat factor which is modifying or curtailing the Snake River physa's habitat or range.

**Ground Water Withdrawals**

Over a 95-year period of recordkeeping, spring flows from the ESPA contributed between 30-85 percent of flow in the Snake River at King Hill (Richards et al. 2006, pp. 84, 85). Prior to the 1950's, irrigation water was moved from rivers and streams with the use of surface conveyance canals. Seepage from these canals into the fractured basalt resulted in recharge of the ESPA and corresponding increases in spring discharge (Kjelstrom 1992, entire). Based on analyses reported by Richards and others (2006, p. 84), and Ondrechen (2004, *in litt.*), spring discharges in the early 2000's may have been 15 percent greater than they were in the early 1900's, however, spring discharges began a sharp decline with the increased use of groundwater for irrigation, and a corresponding decrease in flood irrigation due to the use of central pivot sprinklers, which contribute little to groundwater recharge (Ondrechen 2004, *in litt.*; University of Idaho 2007, *in litt.*). Current estimates of groundwater use for Idaho are > 34 billion liters (9 billion gallons) per day, with agricultural uses accounting for about 60 percent of this total (IDEQ 2013a, *in litt.*). These large withdrawals have been documented to be contributing to the depletion of the overall ground water storage in the ESPA (University of Idaho 2007, *in litt.*). Springs flows from the ESPA provide an important contribution in maintaining/ improving water quantity and quality in the Snake River within the range of the Snake River physa; however, due to known Snake River physa populations occur both above and below the primary ESPA spring discharge, the point at which reduced spring discharge will have adverse effects on the species cannot be predicted at this time.

**Surface and Ground Water Management**

The Idaho Department of Water Resources (IDWR) manages water in the state of Idaho. Among the IDWR's responsibilities is the development of the State Water Plan (Water Plan) (IWRB 2012, entire). The Water Plan outlines objectives for the conservation, development, management, and optimum use of all unappropriated waters in the State. One of these objectives is to ''maintain, and where possible enhance water quality and water-related habitats'' (IWRB 2012, p. 6). It is the intent of the Water Plan that any water savings realized by conservation or improved efficiencies is appropriated to other beneficial uses (e.g., agriculture, hydropower, or fish and wildlife).

The Water Plan also states that the capacity of water storage, flood control, and flow regulation on the Snake River is insufficient for future beneficial uses (IWRB 2012, p. 55) and further states that construction of new reservoirs, enlargement of existing reservoirs, and development of off-stream storage sites may be necessary to meet future demands (IWRB 2012, p. 19). Given the non-protected status of the river reach that constitutes the range of the Snake River physa (see Factor A - Section 2.3.2.1), there exists no assurances that future development of water resource projects will not negatively impact habitat or water quality upon which the species depends.

The ESPA discharges approximately 144 cms (5,100 cfs) of groundwater to the Snake River in the Thousand Springs area (approximately RKM 940-982 (RM 584-610)), greatly increasing the Snake River's base flows (EPA 2002a, pp. 4-9). The storage in the ESPA has been declining since the 1950's due to several reasons, including more efficient water delivery through canals (thus decreasing seepage into the ground), increased groundwater pumping, drought, and climate change (IWRB 2013, p. 2). This has resulted in declines in the average spring outflows in the Thousand Springs area over the past 50 years (Clark and Ott 1996, pp. 553-555). While the Snake River physa is found within the Snake River itself, it has not been found in areas where springs enter the Snake River.

The IDWR and other State agencies have created additional regulatory mechanisms that limit future surface and ground water development in the ESPA, including the continuation of various moratoria on new consumptive water rights, and the designation of Water Management Districts (Caswell 2007, *in litt.*). The State is attempting to stabilize aquifer levels and enhance cold water spring outflows from the ESPA by implementing water conservation measures identified in the Comprehensive Aquifer Management Plan (CAMP) for this area (IDWR 2009, entire). The long-term objective of the CAMP is to incrementally achieve a net ESPA water budget of 600,000 acre feet annually by the year 2030 through a mix of management strategies, including aquifer recharge, ground-to-surface water conversions, demand reduction strategies, and weather modification (IWRB 2013, p. 3).

While aquifer recharge may reduce the rate of groundwater depletion in the ESPA, it also may affect ESPA groundwater quality if measures are not taken to ensure water utilized for recharge purposes is relatively clean. As stated above, the Snake River physa is found within the Snake River itself and has not been found in areas where springs enter the Snake River. Therefore, it is difficult to assess possible impacts to the Snake River physa if groundwater quality is affected by aquifer recharge activities. Overall though, since adoption of the CAMP, progress is being made towards strategy implementation (IWRB 2013, p. 3), although it is too early to determine if these strategies are effective at reducing the rate of groundwater depletion in the ESPA.

In summary, there are no assurances that current State regulations and policies will protect the Snake River physa and its habitat from water projects that occur in the Snake River and the ESPA. While there are no known water development projects within the range of the Snake River physa, future development projects would be a concern if they impacted the remaining free-flowing reaches of the Snake River within the species' range. Conservation measures in the ESPA CAMP have been developed and implemented, but it is too early to determine if they can stabilize ESPA water levels and its discharges into the Snake River. While we anticipate ground water levels in the ESPA will continue to decline even if water conservation measures are implemented, the Snake River also receives substantial amounts of water from areas outside of the ESPA. Given this complexity, we remained concerned with a declining water resource and the potential effects to Snake River physa and its habitat.

Various State-managed water quality programs are being implemented within the range of the Snake River physa. These programs are tiered off the CWA, which requires States to establish water-quality standards that provide for (1) the protection and propagation of fish, shellfish, and wildlife, and (2) recreation in and on the water. As required by the CWA, Idaho has established water-quality standards (e.g., for water temperature and dissolved oxygen) for the protection of cold-water biota (e.g., salmonids) in many reaches of the Snake River. The CWA also specifies that States must include an antidegradation policy in their water quality regulations that protects water-body uses and high quality waters. Idaho's antidegradation policy, updated in the State's 1993 triennial review, is detailed in their Water Quality Standards (IDEQ NA, pp. 15-16).

While point source pollution regulations are enforceable through the CWA, nonpoint source water pollution is primarily addressed through non-regulatory means under the CWA (EPA 2013a, *in litt.*). The IDEQ works closely with the EPA to manage point and non-point sources of pollution to water bodies of the State through the National Pollutant Discharge Elimination System (NPDES) program under the CWA. IDEQ has not requested the authority from the EPA to issue NPDES permits, and therefore all NPDES permits within the state of Idaho are issued by the EPA Region 10 (EPA 2013b, *in litt.*). These NPDES permits are written to meet all applicable water-quality standards established for a water body to protect human health and aquatic life.

One statewide NPDES permit developed by EPA for activities capable of discharging waste on a relatively large basis within the range of the Snake River physa is for the numerous aquaculture facilities located on tributaries and springs that flow into the Snake River (EPA 2007a, entire; Helder 2013, *in litt.*). In Idaho, there are approximately 115 permitted aquaculture facilities, 70 percent of which operate in the Magic Valley, discharging into the Snake River or its tributaries within the range of the Snake River physa (IDEQ 2013b, *in litt.*). Aquaculture facilities that produce less than 9,072 kilograms (20,000 pounds) of fish annually are not required to obtain an NPDES permit (EPA 2007a, p. 9). These smaller facilities lie outside of this regulatory nexus, and as such their discharges are not regulated. The Service is unaware how many unpermitted aquaculture facilities discharge to the Snake River or its tributaries within the range of the Snake River physa.

Under Section 303(d) of the 1972 CWA, States are required to develop lists of impaired waters not meeting State water quality standards (EPA 2013c, *in litt.*). Waters that do not meet water-quality standards due to point and non-point sources of pollution are listed on EPA's 303(d) list of impaired water bodies. IDEQ, under authority of the State Nutrient Management Act, is coordinating efforts to identify and quantify contributing sources of pollutants (including nutrient and sediment loading) to the Snake River basin via the TMDL approach. In water bodies that are currently not meeting water quality standards, the TMDL approach applies pollution-control strategies through several of the following programs: State Agricultural Water Quality Program, CWA section 401 Certification, USBLM Resource Management plans, the State Water Plan, and local ordinances. Several TMDLs have been approved by the EPA in Snake River stream segments within the range of the Snake River physa (Buhidar 2006, *in litt.*), and most apply to TSS, TP, or temperature.

Within the range of the Snake River physa in the Snake River, there are 4 TMDLs approved by the EPA since the Snake River physa was listed: 1) Snake River-King Hill-C.J. Strike Reservoir Subbasin, 2) Snake River (Middle)-Succor Creek Subbasin, 3) Snake River (Middle)-Upper Snake Rock Subbasin, and 4) Snake River (Middle) Subbasin. Status reviews of these TMDLs indicate mixed success, with certain areas of the Snake River showing improving water quality, while other areas are decreasing in quality. Overall, the majority of the stream segments within the range of Snake River physa habitat with existing TMDLs are not meeting the water quality standards established by the TMDL for one or more pollutants, particularly TSS and TP.

In summary, within the state of Idaho, point-source discharges are regulated through the NPDES permitting process, while non-point source discharges are addressed through TMDLs using waste load calculations for that waterbody; however, there is no implementation authority for the non-point discharges. Some stream segments within the range of the Snake River physa and under existing TMDLs are not meeting water quality standards for one or more pollutants. Although regulatory pollution control methods authorized under the CWA have been implemented within the range of the Snake River physa, water quality remains degraded, with no indication that it will improve in the near future. Therefore, the inadequacy of existing regulatory mechanisms regarding Federal and State pollution control regulations continues to be a factor affecting the Snake River physa.

### State Invertebrate Species Regulations

There has been no change in State regulations regarding the protection of invertebrates since the time of the 1993 listing. The IDFG, under Idaho Code section 36-103, is mandated to preserve, protect, perpetuate, and manage all wildlife. However, these regulations do not extend protection to invertebrate species. The only regulations provided for Snake River physa are provided by the Endangered Species Act. In 2005, Idaho finalized the State's Comprehensive Wildlife Conservation Strategy (CWCS; IDFG 2005, entire), which is a conservation strategy for the State's species of greatest conservation need (SGCN). As part of the CWCS, the Snake River physa is included in the State's list of SGCN (IDFG 2005, pp. 423-425), though there is no regulatory authority associated with this designation. In summary, there are no State regulations in place that are specific to the Snake River physa; therefore State invertebrate species regulations for the Snake River physa continue to be inadequate.

**Invasive Species Regulations**

Numerous authorities and regulations are utilized to manage existing populations of invasive species, and seek to prevent introduction and establishment of new species and populations. Regulation of invasive species management in Idaho falls under multiple State laws, including; 22-1900, Invasive Species Act; Idaho Rule 02.06.09, Rules Governing Invasive Species; 22-2012, 22-2016 Plant Pest Act; 22-2409, Noxious Weed Law; 36-104, 36-106, 36-1102; 13.01.10. Fish and Game Authorities; IDAPA 13.01.03, Public Use of Land Owned or Controlled by Idaho Department of Fish and Game; 25-214, Disease Inspection and Suppression; 25-3900, Deleterious Animals; 38-602, Forest Pests (Idaho State Department of Agriculture (ISDA) 2012, p. 32). Various Federal authorities exist that address invasive species issues, including, but not limited to; the Lacey Act; the Nonindigenous Aquatic Nuisance Prevention and Control Act; and the National Invasive Species Act (Idaho Invasive Species Council (IISC) 2012, p. 33).

For aquatic nuisance species, Idaho developed the Idaho Aquatic Nuisance Species Plan, a supplement to Idaho's Strategic Action Plan for Invasive Species (IISC 2007, entire; IISC 2012, entire). In 2009, the Idaho Legislature enacted the Invasive Species Prevention Sticker Rules (IDAPA 26.01.34), which require owners of motorized and non-motorized boats to purchase and have an Invasive Species Sticker on their boats to launch and operate on Idaho's waters (IISC 2012, p. 8). Concurrent with passage of the Invasive Species Prevention Stickers, the ISDA, along with other local governments have initiated mandatory inspection and decontamination stations at various major highway entrances throughout the State to reduce the spread of aquatic invasive species into Idaho (ISDA 2012, pp. 5-7). Since 2009, these stations have operated every year during the boating season and have resulted in the inspection of over 154,000 watercraft, with 93 boats being identified as potentially harboring the invasive zebra (Zebra (*Dreissena polymorpha*) and/ or Quagga mussels (*Dreissena rostriformis*) (ISDA 2012, p. 1). These two species have not been found in Idaho but are known to severely impact aquatic habitats when they become established. While it is unknown how many boats with these species and other invasive species may have come into the State undetected, this program has been effective at stopping a number of contaminated boats from potentially entering the Snake River within the range of the Snake River physa.

The state of Idaho and the Federal Government have implemented various measures for stopping and controlling the spread of invasive species that may affect the Snake River physa or its habitat. One measure, mandatory State boat inspection stations, has had some level of success at containing the introduction of invasive species into Idaho's waters, though it is unknown how many fouled boats are not being stopped by these inspection stations. Until additional action is taken to reduce the incidences of fouled-boats leaving contaminated waters in other States, there will be a continued threat of new invasive species becoming established within Idaho, even given the continued operation of the mandatory boat inspection stations within the State. Therefore, the inadequate Federal and State invasive species regulatory mechanisms will continue to be a risk factor for Snake River physa.

**New Zealand Mudsnail Competition and Aquatic Invasive Species**

The 1993 listing rule stated that the non-native invasive New Zealand mudsnails did compete for habitat with the Snake River physa in the mainstem Snake River (57 FR, p. 59254). The New Zealand mudsnail appears to flourish in Snake River reaches under a variety of environmental conditions, including low dissolved oxygen and on substrates of mud or silt, but it is also found at high densities in some cold-water spring tributaries to the Snake River (e.g. up to 500,000 snails/$m^2$ (46,500/ $ft^2$) at Banbury Springs; Richards et al. 2001, p. 375). New Zealand mudsnails have been documented in dark mats at densities of nearly 0.62/ $mm^2$ (400 individuals/ $in^2$) in free-flowing habitats within the range of the Snake River physa (57 FR 59254). Although the New Zealand mudsnail can tolerate various water velocities, they appear to reach their highest densities in slower moving waters (Richards et al. 2001, pp. 378, 389).

Some researchers have suggested that the New Zealand mudsnail competes with native species for food and/or space (Kerans et al. 2005, pp. 135, 136; Hinson 2006, p. 41) and can dominate ecosystem nutrient and energy flow (Hall et al. 2003, p. 411). Research has shown that New Zealand mudsnails influence the growth of sympatric freshwater snails (Richards 2004, entire) and can displace native species (Hall et al. 2006, entire). Competition from the New Zealand mudsnail was shown to negatively impact growth rates of the Bliss Rapids snail (*Taylorconcha serpenticola*), also a listed species endemic to the Snake River drainage, under experimental conditions (Richards 2004, pp. 117-118). In enclosure experiments, increasing New Zealand mudsnail densities also resulted in lower Bliss Rapids snail densities (Richards 2004, pp. 117-118).

The New Zealand mudsnail was collected by Gates and Kerans (2010, p. 25) in the Minidoka reach in approximately the same numbers as the Snake River physa (total abundance of 294 and 271 respectively), but whether the Snake River physa and New Zealand mudsnail compete for the same resources has not been assessed. This reach of the Snake River is free flowing and doesn't contain the optimum habitat for New Zealand mudsnails which are found in slower moving water. Considering that the two species were found in about the same numbers where Snake River physa was most abundant may suggest that under what are assumed to be optimum habitat conditions for Snake River physa (in the Minidoka reach), competition from New Zealand mudsnail appears to be minimal. In areas supporting high numbers of New Zealand mudsnail that overlap with Snake River physa habitat, it is possible that the New Zealand mudsnail could have a competitive edge over Snake River physa. However, at this time we don't have the information that New Zealand mudsnails are impacting, or are an overall threat to Snake River physa. It is likely additional aquatic invasive species will colonize or occur within the range of the Snake River physa, (see Section 2.3.2.4 - Inadequacy of existing regulatory mechanisms – Invasive Species Regulations), and the effects they will have on Snake River physa.

**Small Population Size, Habitat Fragmentation, and Loss of Connectivity**

The two general areas of the Snake River where Snake River physa have been found since the time of listing are downstream of Minidoka Dam (RKM 1086-1067.8 (RM 675-663.5)) and downstream of Lower Salmon Falls Dam (RKM 922 (RM 573)) to Ontario, Oregon (RKM 592 (RM 368)). The largest known population is found within the 18.5 RKM (11.5 RM) reach of river directly downstream of Minidoka Dam to the beginning of the reservoir pool at Milner Dam. At certain times of the year, the entire flow of the Snake River is diverted at Milner Dam to provide water for irrigation. This leaves the river essentially dry for approximately 2.6 km (1.6 mi) downstream of Milner Dam. This is important to note because the next known occurrence of Snake River physa is downstream of Lower Salmon Falls Dam (RKM 922 (RM 573)). While the Minidoka reach population is relatively robust, the entire flow of the Snake River is essentially severed as a source for downstream populations when Milner Dam is diverting the entire flow of the Snake River. While there have been reports of Snake River physa occurring upstream of Minidoka Dam (PEI 1991), both historic collection (Keebahugh 2014) and more recent surveys (Newman 2012, *in litt.*) have not confirmed presence. Therefore, the Minidoka reach population is regarded as isolated, with limited possibility for dispersal into, or out of the population.

Further downstream, from C.J. Strike Reservoir (RKM 789 (RM 490)) downstream to Ontario, Oregon (RKM 592 (RM 368)), the Snake River physa is patchily distributed. Unlike the Minidoka reach where the population is relatively robust, this area has had very limited collections of Snake River physa (Keebaugh 2009). Currently, C.J. Strike and Swan Falls dams limit connectivity within this area (compared to the Minidoka reach population).

Overall, while the two general population areas for the Snake River physa are isolated at times with limited connectivity opportunities, we support continued investigation to determine if the small population size, habitat fragmentation, and loss of connectivity are factors having a direct impact on the species at this time.

### Climate Change

Air temperatures have been warming more rapidly over the Rocky Mountain West compared to other areas of the coterminous U.S. (Rieman and Isaak 2010, p. 3). Data from stream flow gauges in the Snake River watershed in western Wyoming, and southeast and southwest Idaho indicate that spring runoff is occurring between 1 to 3 weeks earlier compared to the early twentieth century (Rieman and Isaak 2010, p. 7). These changes in flow have been attributed to interactions between increasing temperatures (earlier spring snowmelt) and decreasing precipitation (declining snowpack). Global Climate Models project air temperatures in the western U.S. to further increase by 1 to 3 °C (1.8 to 5.4 °F) by mid-twenty-first century (Rieman and Isaak 2010, p. 5), and predict significant decreases in precipitation for the interior west. Areas in central and southern Idaho within the Snake River watershed are projected to experience moderate to extreme drought in the future (Rieman and Isaak 2010, p. 5).

As discussed earlier, Snake River physa appear to tolerate a range of water temperatures in the Snake River. If Snake River water temperatures rise as a result of climate change, indirect impacts to the species may occur, including effects on metabolic processes, foraging behavior, and dynamics with predators and/ or invasive species (Poff et al. 2002, entire; Williamson et al. 2008, p. 248; and Rahel and Olden 2008, entire). In addition, indirect impacts of climate change include the possible synergy of higher temperatures with contaminants (Sokolova and Lannig 2008, p. 183), the increased incidence of cyanobacteria (i.e. blue green algae) blooms due to higher temperatures, higher atmospheric carbon dioxide, and increased nutrient enrichment (Paerl and Huisman 2008, entire; Paerl et al. 2011, p. 1743). Further, habitats supporting Snake River physa could be reduced due to low summer flows and warmer temperatures leading to an extended growing season for macrophytes.

The vulnerability to climate change are projected to be highest in river basins with the largest hydrologic response to warming and lowest management flexibility – that is, fully allocated, mid-elevation, temperature-sensitive, mixed rain-snow watersheds with existing water conflicts among users of summer water, such as the Snake River basin (National Climate Assessment and Development Advisory Committee (NCADAC) 2013, p. 726). The Snake River is a highly regulated river system that serves multiple uses, including, but not limited to, irrigation, hydropower, and aquaculture. Even though the Snake River is a highly managed riverine system, if precipitation decreases within the Snake River basin, as the models and literature forecast, and groundwater flows decline due to continued depletion of the aquifer, there may be less water within the river itself, especially as competition for this limited resource increases (Meyer et al. 1999, p. 1373). With these changes, we anticipate suitable habitat for the Snake River physa will become limited and this species will further contract its range. Therefore we have determined future projected climate change effects are a factor affecting the habitats and range of the Snake River physa.

## 2.4.2 Bliss Rapids Snail

### 2.4.2.1 Status of Bliss Rapids Snail in the Action Area

Because the range of the Bliss Rapids snail is contained entirely within the action area, refer to section 2.3.3 of this Opinion for the baseline status of this snail.

### 2.4.2.2 Factors Affecting Bliss Rapids Snail in the Action Area

Our understanding of the threats to the Bliss Rapids snail has changed since we listed the species in 1992. Some threats are now known to be removed (i.e., new hydropower dam construction) while other threats have emerged (i.e., depletion of groundwater that supports the spring colonies). As discussed in the following sections, we believe, based on the best available data, that it is reasonable to expect the primary threats (i.e., reduced ground water levels, water quality and pollution concerns, competition from nonnative species, and climate change) to Bliss Rapids snails will continue to occur throughout the range of the species and to affect all colonies into the future.

Refer to section 2.3.2.5 for more information on the conservation needs of the Bliss Rapids snail.

**Construction of New Hydropower Dams**

In our 1992 final rule listing the Bliss Rapids snail as a threatened species, we stated: “Six proposed hydroelectric projects, including two high dam facilities, would alter free flowing river reaches within the existing range of [the Bliss Rapids snail]. Dam construction threatens the [Bliss Rapids snail] through direct habitat modification and moderates the Snake River's ability to assimilate point and non-point pollution. Further hydroelectric development along the Snake River would inundate existing mollusk habitats through impoundment, reduce critical shallow, littoral shoreline habitats in tailwater areas due to operating water fluctuations, elevate water temperatures, reduce dissolved oxygen levels in impounded sediments, and further fragment remaining mainstem populations or colonies of [the Bliss Rapids snail]” (57 FR 59251).

Proposed hydroelectric projects discussed in the 1992 final listing rule are no longer moving forward. The A.J. Wiley project and Dike Hydro Partners preliminary permits have lapsed; the Kanaka Rapids, Empire Rapids, and Boulder Rapids permits were denied by the Federal Energy Regulatory Commission (FERC) in 1995; there was a notice of surrender of the preliminary permit for the River Side Project in 2002; and two other proposed projects, the Eagle Rock and Star Falls Hydroelectric Projects, were denied preliminary permits by the FERC. In 2003, a notice was provided of surrender of the preliminary permit for the Auger Falls Project. Information provided by the state of Idaho indicates that all proposals and preliminary permits for the construction of new dams along the mid-Snake River have either lapsed or been denied by the FERC (Caswell 2006, *in litt.*).

**Operation of Existing Hydropower Dams**

The Bliss Rapids snail occurs in riverine and spring or spring-influenced habitats but is not known to occur in reservoir habitats. In the December 14, 1992, final listing rule we stated: "Peak- loading, the practice of artificially raising and lowering river levels to meet short-term electrical needs by local run-of-the-river hydroelectric projects also threatens [the Bliss Rapids snail]. Peak- loading is a frequent and sporadic practice that results in dewatering mollusk habitats in shallow, littoral shoreline areas … these diurnal water fluctuations [prevent the Bliss Rapids snail] from occupying the most favorable habitats" (57 FR 59252). Peak loading operations within the range of river colonies of the Bliss Rapids snail occur below the Bliss Dam (RKM 901 (RM 560)) and the Lower Salmon Falls Dam (RM 573) (USFWS 2004a, pp. 19, 20). For example, at the Bliss Dam (Stephenson and Bean 2003, p. 30) the Snake River can experience daily fluctuation of water levels from hydropower generating activities (peak loading) up to 2.1 m (7 ft). It appears that Bliss Rapids snails are found primarily in areas less than 0.9 m (3 ft) deep, although this may be an artifact of more intensive sampling at shallow depths (Richards et al. 2006, pp. 43, 52-56). Nevertheless, our current understanding based on the best available information, is that a majority of Bliss Rapids snails in the Snake River occupy shallow water. Furthermore, Bliss Rapids snails in these shallow-water areas are susceptible to the effects from peak loading operations, including desiccation and freezing when water levels drop and expose snails to atmospheric conditions.

Laboratory studies have shown that peak-loading during winter months, a time when the species is reproducing, is likely to result in mortality of individual Bliss Rapids snails. Air temperatures within the range of Bliss Rapids snails in Idaho regularly fall below 0°C (32°F) between November and March (Richards 2006, p. 28). In a laboratory study conducted by Richards (2006, p. 12), half of the Bliss Rapids snails subjected to a temperature of minus 7°C (19°F) died in less than an hour. In a field study, Richards (unpublished data, cited in Richards et al. 2006, pp. 125-126) found that Bliss Rapids snails could survive for many hours to several days in moist conditions (i.e., undersides of cobbles) when air temperatures were above freezing (0° C (32° F )) (Richards et al. 2006, p. 125). Although the mortality rate outside of these conditions has not been documented in field studies or after an actual peak loading event, work by Richards et al. 2014, p. 961) utilizing laboratory-controlled aquaria, found Bliss Rapids snail mortality to be up to 100 percent under conditions characteristic (summer high and winter low temperatures) of some hydropower operations in the middle Snake River. Based on the above information, peak loading likely affects individual Bliss Rapids snails through desiccation and freezing and may have population level effects as well.

**Degraded Water Quality**

In the 1992 final listing rule the Service stated: "The quality of water in [snail] habitats has a direct effect on the species survival. The [Bliss Rapids snail] require[s] cold, well-oxygenated unpolluted water for survival. Any factor that leads to deterioration in water quality would likely extirpate [the Bliss Rapids snail]" (57 FR 59252). New information has become available indicating some improvements to Snake River water quality. Significant nutrient and sediment reduction has occurred in the Snake River following implementation of the Idaho Nutrient Management Act and regulated Total Maximum Daily Load (TMDL) reductions from the mid-

1990s to the present (Richards et al. 2006, pp. 5-6, 86). The Mid-Snake River reach also receives a large infusion of clean, cold-water spring flows and supports the highest densities and occurrence of Bliss Rapids snails.

Hypereutrophy (planktonic algal blooms and nuisance rooted aquatic plant growths), prior to listing in 1992, was very severe during drought cycles when deposition of sediments and organic matter blanketed river substrate often resulting in unsuitable habitat conditions for Bliss Rapids snails. Although some nutrient and sediment reduction has been documented in the Snake River since listing (Richards et al. 2006, p. 5), there are still large inflows of agriculture and aquaculture runoff entering the river at Twin Falls to Lower Salmon Falls dam (RKM 922 (RM 573)). As a result, nutrient and sediment concentrations can be relatively high in this portion of the river, especially during lower summer flows (Richards et al. 2006, p. 91). Phosphorus concentrations, the key nutrient leading to hypereutrophic conditions in the middle Snake River, exceeded EPA guidelines for the control of nuisance algae at numerous locations along the Snake River from 1989 to 2002, including areas immediately upstream of Bliss Rapids snail colonies (Hardy et al. 2005, p. 13). Several water quality assessments have been completed by the EPA, USBR, and IPC, and all generally agree that water quality in the Snake River of southern Idaho meets Idaho water quality standards for aquatic life for some months of the year, but may not meet these standards when temperatures are high and flows are low (Meitl 2002, p. 33). Idaho Department of Environmental Quality's (IDEQ) 2005 performance and progress report to the EPA states that projects are meeting the Idaho non-point source pollution program goals (IDEQ 2006, entire.). Others report that water quality has not improved appreciably between 1989 and 2002 (Hardy et al. 2005, pp. 19-21, 49, 51).

Several reaches of the Snake River are classified as water-quality- impaired due to the presence of one or more pollutants (e.g., Total Phosphorus (TP), sediments (TSS), total coliforms) in excess of State or Federal guidelines. Nutrient-enriched waters primarily enter the Snake River via springs, tributaries, fish farm effluents, municipal waste treatment facilities, and irrigation returns (EPA 2002a, pp. 4-18 to 4-24). Irrigation water returned to rivers is generally warmer, contains pesticides or pesticide byproducts, has been enriched with nutrients from fish farms and land-based agriculture (e.g., nitrogen and phosphorous), and frequently contains elevated sediment loads. Pollutants in fish farm effluent include nutrients derived from metabolic wastes of the fish and unconsumed fish food, disinfectants, bacteria, and residual quantities of drugs used to control disease outbreaks. Furthermore, elevated levels of fine sediments, nitrogen, and trace elements (including cadmium, chromium, copper, lead, and zinc), have been measured immediately downstream of several aquaculture discharges (Hinson 2003, pp. 44-45). Additionally, concentrations of lead, cadmium, and arsenic have been previously detected in snails collected during a research study in the Snake River (Richards 2002, *in litt.*). The effects of these elevated levels of nutrients and trace elements on Bliss Rapids snails, both individually and synergistically, are not fully understood. However, studies have shown another native Snake River snail, the Jackson Lake springsnail (*Pyrgulopsis robusta*), to be relatively sensitive to copper (a common component in algaecides) and pentachlorophenol, a restricted use pesticide/wood preservative (Ingersoll 2006, *in litt.*).

### Water Diversions and Ground Water Withdrawals

Threats to cold water spring-influenced habitats from ground water withdrawal and diversions for irrigation and aquaculture are not as they were perceived when the Bliss Rapids snail was listed in 1992. At that time the threat from ground water withdrawal was identified only at Box Canyon, and the scope of this threat was underestimated. Based on the best available data, we now know that this threat is likely to affect the Bliss Rapids snail throughout its range. In concert with the historical losses of habitat to surface diversions of spring water for irrigation and aquaculture, the continuing decline of the groundwater aquifer is one of the primary threats to the long-term viability of the Bliss Rapids snail.

Average annual spring flows increased from about 4,400 cubic feet per second (cfs) in 1910, to approximately 6,500 cfs in the early 1960s, because widespread flood irrigation caused artificial recharge of the aquifer (Richards et al. 2006, pp. 84, 87). As a result of more efficient irrigation practices from 1960 to the present (i.e., switching from flood irrigation or direct surface diversion to more efficient center-pivot irrigation systems utilizing ground water), more water was pumped from the aquifer while water percolation into the aquifer declined, resulting in declines (from the high values of the 1960s) of average annual spring flows to about 5,000 cfs (Richards et al. 2006, pp. 84, 87). Although the current spring flow levels total about 15 percent higher than average spring flows measured in 1910, they are declining (USFWS 2008a, pp. 23-24). We anticipate spring flows will likely continue to decline in the near future, even as water-conservation measures are implemented and are being developed as water demands in the vicinity continue to increase. The state of Idaho has taken steps to improve ground water recharge and limit new ground water development within the eastern Snake River plain; however, the Snake River Plain aquifer level continues to decline (USFWS 2008a, p. 26).

Effects from the over-allocation of ground water and the subsequent declining ground water levels appear to be more of a threat than previously thought. Evidence indicates that springs from the Eastern Snake River Aquifer where the Bliss Rapids snail resides depend on ground water levels and that the ground water levels are declining (USFWS 2008a, p. 26) even with ongoing measures attempting to address the decline (Caswell 2007, *in litt.*). Spring sites are important since Bliss Rapids snail colonies that occur in springs have been shown to be a source of genetic diversity to riverine colonies and to contain four times as many private (i.e., unique) alleles (n=16) compared to riverine populations (Liu and Hershler 2009, p. 1296). Colonies in springs or at their outflows are also the most dense, may account for most of the reproductive output of the species, and likely act as refugia from competition with invasive New Zealand mudsnails (see below). Finally, if spring colonies are lost, particularly those at the upstream end of the species' distribution, the probability of recolonization is likely to be extremely small (USFWS 2008b, p. 36).

### Inadequacy of Existing Regulatory Mechanisms

In the 1992 final listing rule, we found inadequate regulatory mechanisms to be a threat because: (1) regulations were inadequate to curb further water withdrawal from ground water spring outflows or tributary spring streams; (2) it was unlikely that pollution control regulations would reverse the trend in nutrient loading in the near future; (3) there was a lack of State-mandated protections for invertebrate species in Idaho; and (4) regulations did not require FERC or the U.S. Army Corps of Engineers to address Service concerns regarding licensing hydroelectric projects or permitting projects under the Clean Water Act (CWA) for unlisted snails. Below, we address each of these concerns in turn.

*Ground Water Withdrawal Regulations*

The Idaho Department of Water Resources (IDWR) manages water in the state of Idaho. Among the IDWR's responsibilities is the development of the State Water Plan (IDWR 2006a, *in litt.*). The State Water Plan was updated in 1996 and included a table of federally threatened and endangered species in Idaho, such as the Bliss Rapids snail. The State Water Plan outlines objectives for the conservation, development, management, and optimum use of all unappropriated waters in the State. One of these objectives is to "maintain, and where possible enhance water quality and water-related habitats" (IDWR 2006a, *in litt.*). It is the intent of the State Water Plan that any water savings realized by conservation or improved efficiencies is appropriated to other beneficial uses (e.g., agriculture, hydropower, or fish and wildlife).

Another IDWR regulatory mechanism is the ability of the Idaho Water Resource Board to designate "in-stream flows" (IDWR 2006b, *in litt.*). The IDWR currently has 89 licensed water rights for minimum in-stream flows in Idaho (IDWR 2006b, *in litt.*). Of these, 11 potentially have conservation benefits for Bliss Rapids snails (i.e., provide for minimum in-stream flows near tributary spring outflows that provide habitat for Bliss Rapids snails). However, individuals that hold water rights with earlier priority dates have the right to fill their needs before the minimum stream flow is considered. If there is not enough water available to satisfy all of the water rights, then the senior water rights are satisfied first, and so on in order, until there is no water left. It is the junior water right holders that do not get water when there is not enough to satisfy all the water rights. Senior diversions can legally dewater the stream in a drought year or when low flows occur, leaving no water for the minimum stream flow (IDWR 2013, *in litt.*), therefore impacting species such as the Bliss Rapids snail.

The IDWR and other State agencies have also created additional regulatory mechanisms that limit future surface and ground water development; they include the continuation of various moratoria on new consumptive water rights and the designation of Water Management Districts (Caswell 2007, *in litt.*). The State is attempting to stabilize aquifer levels and enhance cold water spring outflows from the Eastern Snake River Plain by implementing water conservation measures contained in the Comprehensive Aquifer Management Plan (CAMP) for this area (IDWR 2009). The goal of the CAMP is to "sustain the economic viability and social and environmental health of the Eastern Snake Plain by adaptively managing a balance between water use and supplies" (IDWR 2009, p. 4). The CAMP will include several alternatives in an attempt to increase water supply, reduce withdrawals from the aquifer, and decrease overall demand for groundwater (IDWR 2009, p. 7).

In addition, the state of Idaho established moratoria in 1993 (the year after listing of the Bliss Rapids snail) that restricted further surface-water and groundwater withdrawals for consumptive uses from the Snake River Plain aquifer between American Falls Reservoir and C.J. Strike Reservoir. The 1993 moratoria were extended by Executive Order in 2004 (Caswell 2006, *in litt.*, attachment 1). However, these actions have not yet resulted in stabilization of aquifer levels. Depletion of spring flows and declining groundwater levels are a collective effect of drought conditions, changes in irrigation practices (the use of central-pivot sprinklers contribute little to groundwater recharge), and groundwater pumping (University of Idaho 2007, *in litt.*). The effects of groundwater pumping downstream in the aquifer can affect the upper reaches of the aquifer, and the effects of groundwater pumping can continue for decades after pumping ceases (University of Idaho 2007, *in litt.*). Thus, we anticipate groundwater levels will likely continue to decline in the near future, even as water-conservation measures are implemented, and are being developed. Furthermore, species associated with these springs that are dependent upon the presence of water, such as the Bliss Rapids snail, will likely experience local extinctions without the opportunity for recolonization (USFWS 2008a, pp. 36-37). Loss of a colony from any individual habitat patch, without subsequent recolonization, increases the extinction risk for the species as a whole, a phenomenon dubbed the “extinction ratchet” (Burkey and Reed 2006, p. 11).

*Pollution Control Regulations*

Since the 1992 final listing rule, reductions in TSS and TP loading have improved water quality in localized reaches of the Snake River (Buhidar 2006, *in litt.*). Various State-managed water quality programs are being implemented within the range of the Bliss Rapids snail. These programs are tiered off the Clean Water Act (CWA), which requires States to establish water-quality standards that provide for (1) the protection and propagation of fish, shellfish, and wildlife, and (2) recreation in and on the water. As required by the CWA, Idaho has established water-quality standards (e.g., for water temperature and dissolved oxygen) for the protection of cold-water biota (e.g., invertebrate species) in many reaches of the Snake River. The CWA also specifies that States must include an antidegradation policy in their water quality regulations that protects water-body uses and high-quality waters. Idaho's antidegradation policy, updated in the State's 1993 triennial review, is detailed in their Water Quality Standards (IDEQ NA, pp. 15-16).

The IDEQ works closely with the EPA to manage point and non-point sources of pollution to water bodies of the State through the National Pollutant Discharge Elimination System (NPDES) program under the CWA. IDEQ has not been granted authority by the EPA to issue NPDES permits directly; all NPDES permits are issued by the EPA Region 10[11]. These NPDES permits are written to meet all applicable water-quality standards established for a water body to protect human health and aquatic life. Waters that do not meet water-quality standards due to point and non-point sources of pollution are listed on EPA's 303(d) list of impaired water bodies. States must submit to EPA a 303(d) list (water-quality-limited waters) and a 305(b) report (status of the State's waters) every 2 years. IDEQ, under authority of the State Nutrient Management Act, is coordinating efforts to identify and quantify contributing sources of pollutants (including nutrient and sediment loading) to the Snake River basin via the Total Maximum Daily Load (TMDL) approach. In water bodies that are currently not meeting water-quality standards, the TMDL approach applies pollution-control strategies through several of the following programs: State Agricultural Water Quality Program, Clean Water Act section 401 Certification, BLM Resource Management plans, the State Water Plan, and local ordinances. Several TMDLs have been approved by the EPA in stream segments within the range of the Bliss Rapids snail in the Snake River or its tributaries (Buhidar 2006, *in litt.*), although most apply only to TSS, TP, or temperature. Therefore, these stream segments do not yet have water quality attributes that are protective of the Bliss Rapids snail until the TMDL approach has sufficient time to bring the stream segment water quality in line with approved standards.

[11] See: https://www.deq.idaho.gov/permitting/water-quality-permitting/npdes.aspx

*Federal Consultation Regulations*

In Idaho, the EPA retains authority for the issuance of permits through the NPDES, which is designed to manage point source discharges. There are more than 80 licensed aquaculture facilities on the Snake River permitted by the EPA (EPA 2002a, pp. 4-19, 4-20). Updated draft permits for aquaculture and fish processing facilities throughout Idaho have been made available for public review (71 FR 35269). Draft permits have been issued for aquaculture facilities on Billingsley Creek, Riley Creek, Niagara Springs, and Thousands Springs, all within the known range of the Bliss Rapids snail. Facilities that produce less than 9,072 kilograms (20,000 pounds) of fish annually are not required to obtain an NPDES permit (EPA 2006, p. 3-1). These smaller facilities lie outside of this regulatory nexus, and as such their discharges are not regulated or reported.

Since the species was listed in 1992, Federal agencies, including the Army Corps of Engineers and the FERC, have been required to comply with section 7 of the Act on any projects or managed activities that may affect the Bliss Rapids snail. Some conservation benefits to the species are being realized through section 7 consultation with other Federal agencies, but without the Act’s protection there are no regulatory assurances that these conservation benefits would continue.

IPC and the Service cooperated in a Settlement Agreement (Agreement) approved by the FERC. This Agreement was designed to assess potential effects of the IPC's operations in the Wiley and Dike Reaches, and was approved as part of the biological opinion and license issuance for the Lower Salmon Falls and Bliss Projects. These studies and their analyses were scheduled to be completed in 2009.

The BLM manages more than 245 million acres of land in the 11 western States, including land adjacent to the Snake River in Idaho. The BLM manages activities on Federal lands such as outdoor recreation, livestock grazing, mining development, and energy production to conserve natural, historical, cultural, and other resources on the public lands[12]. In Idaho, the BLM has been consulting with the Service pursuant to section 7 of the Act on ongoing BLM actions that may affect the Bliss Rapids snail. Through these consultation efforts, coordinated and cooperative conservation measures have been added to proposed actions (e.g., new or renewed grazing permits on public lands) to minimize impacts to the species. Programmatic guidance and direction, documented through a conservation agreement between the BLM and Service, has increased the likelihood that conservation benefits may be realized for new, re-authorized, and ongoing actions; however, without the continued protections of the Endangered Species Act, there are no regulatory assurances that these conservation measures would continue.

**Other Natural or Manmade Factors Affecting the Continued Existence of the Bliss Rapids Snail**

The final listing rule stated that New Zealand mudsnails (*Potamopyrgus antipodarum*) were not abundant in cold water springflows with colonies of Bliss Rapids snails, but that they did compete with the Bliss Rapids snail in the mainstem Snake River (57 FR 59254; December 14,

[12] http://www.blm.gov/wo/st/en/info/About_BLM.print.html (Accessed February 12, 2014).

1992). We have no direct evidence that New Zealand mudsnails have displaced colonies of Bliss Rapids snails, but New Zealand mudsnails have been documented in dark mats at densities of nearly 400 individuals per square inch in free-flowing habitats within the range of the Bliss Rapids snail (57 FR 59254). Richards et al. (2006, pp. 61, 64, 68) found that Bliss Rapids snails may be competitively excluded by New Zealand mudsnails in most habitats, and that Bliss Rapids snail densities would likely be higher in the absence of New Zealand mudsnails. Both species are mostly scraper-grazers on algae and have similar resource requirements (Richards et al. 2006, pp. 59, 66). Furthermore, New Zealand mudsnails have become established in every cold water spring-fed creek or tributary to the Hagerman Reach of the Snake River that has been surveyed (74 FR 47543). However, New Zealand mudsnails do not appear able to colonize headwater spring habitats, which may provide Bliss Rapids snails refugia from competition with New Zealand mudsnails (Frest and Johannes 1992, p. 50; Richards et al. 2006, pp. 67-68).

The physiological tolerances of the New Zealand mudsnail, including temperature and water velocity (Winterbourn 1969, pp. 457, 458; Lysne and Koetsier 2006b, p. 81); life history attributes such as high fecundity and growth rates (Richards 2004, pp. 25-34); and wide variety of habitat use such as springs, rivers, reservoirs, and ditches (Cada 2004, pp. 27, 28; USBR 2002, pp. 3, 11; Hall et al. 2003, pp. 407, 408; Clark et al. 2005, pp. 10, 32-35; Richards 2004, pp. 47-67), may provide the New Zealand mudsnail a competitive advantage over Bliss Rapids snails outside of cold headwater springs.

### Climate Change

Air temperatures have been warming more rapidly over the Rocky Mountain West compared to other areas of the coterminous U.S. (Rieman and Isaak 2010, p. 3). Data from stream flow gauges in the Snake River watershed in western Wyoming, and southeast and southwest Idaho indicate that spring runoff is occurring between 1 to 3 weeks earlier compared to the early twentieth century (Rieman and Isaak 2010, p. 7). These changes in flow have been attributed to interactions between increasing temperatures (earlier spring snowmelt) and decreasing precipitation (declining snowpack). Global Climate Models project air temperatures in the western U.S. to further increase by 1 to 3 °C (1.8 to 5.4 °F) by mid-twenty-first century (Rieman and Isaak 2010, p. 5), and predict significant decreases in precipitation for the interior west. Areas in central and southern Idaho within the Snake River watershed are projected to experience moderate to extreme drought in the future (Rieman and Isaak 2010, p. 5).

While the effects of global warming on the Bliss Rapids snail are not fully understood, it has the potential to affect their habitat. For the bull trout which tends to have lower thermal requirements than other salmonids, Rieman et al. (2007) predicted that global warming could reduce suitable habitat in the interior Columbia River basin by up to 92 percent (range 18 to 92 percent) (Rieman et al. 2007, p. 1559). While it is reasonable to suspect that populations of snails within the Snake River may be affected by elevated water temperatures, aquifer springs are less likely to immediately exhibit increased temperatures. If warmer winters deplete surface water reserves, either through earlier snow melt or greater proportions of precipitation as rain, then it is plausible that there will be an increased demand for groundwater, which could further reduce spring flows. Climate change will affect water use in the action area, but the magnitude of this effect will partially depend on how local government and water users respond to these changes. How this will affect Bliss Rapids snails and their habitat is uncertain, but has the potential to be adverse.

# 2.4.3 Banbury Springs Lanx

## 2.4.3.1 Status of the Banbury Springs Lanx in the Action Area

Because the range of the Banbury Springs lanx is contained entirely within the action area, refer to section 2.3.3 of this Opinion for the baseline status of the lanx.

## 2.4.3.2 Factors Affecting the Banbury Springs Lanx in the Action Area

Banbury Springs lanx habitat in Thousand Springs, Box Canyon, Banbury Springs, and Briggs Springs has been impacted by habitat modification. The free-flowing, coldwater environments where the Banbury Springs lanx evolved have been affected by, and continue to be vulnerable to, adverse habitat modification and deteriorating water quality from one or more of the following human activities: hydroelectric development, water withdrawal and diversion, water pollution (point and non-point sources), and aquaculture.

Refer to section 2.3.3.5 for more information on the conservation needs of the Banbury Springs lanx.

**Habitat Modification**

See section 2.3.3.4 (Status and Distribution) for a description of conditions/modification of lanx habitat at Thousand Springs, Box Canyon, Banbury Springs, and Briggs Springs. Modification of potential habitat in the Snake River is described below.

*Potential Snake River Habitat*

The Banbury Springs lanx currently is known to occur only in coldwater spring complexes and tributaries, in riffles and along the margins of rapids where water quality is considered relatively good. Prior to anthropogenic (human caused) alterations between Briggs Springs and Thousand Springs, the Snake River at least seasonally may have provided a conduit where the Banbury Springs lanx could move between coldwater springs. The Service hypothesizes that 11 dams constructed in the middle Snake River contributed to the restricted range of the Banbury Springs lanx and precluded immigration, emigration, and genetic exchange between the four extant colonies by inundation of habitat, reduction of flow, and sediment accumulation. As a result, the Banbury Springs lanx is now restricted to four isolated colonies with no possible conduit for dispersal or range expansion.

Three dams on the middle Snake River (Milner, Upper Salmon Falls, and Lower Salmon Falls) affect Banbury Springs lanx dispersal and potential habitat in the Snake River. Milner [RM 640 (RKM 1030)] is an irrigation dam, which in many years can deplete the Snake River of flow at that location on a seasonal basis (EPA 2002a, p. 4-8). Even though this dam is 50 RM (80 RKM) upstream from the closest Banbury Springs lanx location, a reduction of flow of that magnitude (i.e., total loss of flow) typically has negative ramifications on downstream habitat. The Upper Salmon Falls [RM 582.5 (RKM 937)] and Lower Salmon Falls [RM 572.9 (RKM 933)] hydroelectric projects have replaced free-flowing river habitat with slow moving water storage reservoirs. The reservoir created by the Upper Salmon Falls Dam extends in the Snake River upstream of Thousand Springs [RM 583.9 (RKM 937.7)]. The drop in water velocity in a reservoir often results in elevated surface water temperatures and subsequent reductions in dissolved oxygen concentrations (Hynes 1970, pp. 444-448). In addition, water-transported sediments, that would under free-flowing river flows be flushed downstream and deposited in pools, eddies, and other still-water environments, are now settled out in slower moving reservoir waters (Hynes 1970, pp. 448-449; Simons 1979, p. 95).

Since the four colonies of Banbury Springs lanx biologically represent a single species, the Service hypothesizes that they were likely at one time part of a larger, continuous interbreeding population. The knowledge of events that isolated these colonies from one another are speculative since we do not have a detailed understanding of the species' historic distribution. It is possible, like other Snake River gastropods, that they are a relict population of a lake-dwelling species formerly of Pleistocene Lake Idaho. However, the species' morphology and current habitat preference (groundwater dependence) do not suggest it was a strict lacustrine (lake) species. Given this species' morphology and observed habitat preferences, it is more likely that the Banbury Springs lanx is a riverine species and that its historic distribution probably included appropriate habitats within the Snake River prior to anthropogenic activities, which altered flows and reduced water quality. Since anthropogenic impacts have occurred recently in terms of genetic evolutionary timescale, it is doubtful detailed genetic studies would identify genetic differentiation between the four isolated colonies.

**Groundwater Quality**

In addition to the destruction and/or modification of the Banbury Springs lanx habitat discussed above (i.e., modification of Thousand Springs, Box Canyon, Banbury Springs, and Briggs Springs), poor groundwater quality is an anthropogenic factor which likely impacts this species and limits its geographic distribution. Springflow diversions and irrigation return flows are believed to degrade water quality and are detrimental to the Banbury Springs lanx due to the resulting flow reduction, increased water temperature, decreased dissolved oxygen, elevated nutrient concentrations, and the accumulation of pollutants and sediment, as described below.

*Springflow reduction*

USGS records show that the average spring outflows in the Hagerman reach of the Snake River have declined over the past 50 years (Clark and Ott 1996, pp. 553-555). These declines have been observed at locations occupied by Banbury Springs lanx (e.g., Box Canyon and Briggs Springs). In part, these declines are due to groundwater pumping of the Snake River Plain aquifer for agricultural and urban use, as well as a gradual replacement of flood irrigation practices with the use of center-pivot sprinkler systems, which contribute to little or no aquifer recharge. Furthermore, as groundwater pumping continues in the Snake River Plain, aquifer levels have shown a declining trend over the last 50 years in Gooding County. Groundwater pumping continues today and the potential exists for severe aquifer depletion in the future with continuing and new demands from water users such as municipalities and irrigators. Senior water right holders (e.g., fish hatcheries and irrigators) are expected to maintain the same quantities of water withdrawal from the springs, thereby continuing to reduce flow in the natural spring channel. The cumulative effects of these actions translate into a continued decline in the coldwater springflows upon which the Banbury Springs lanx depends.

*Water Temperature and Dissolved Oxygen*

Water temperature is considered one of the most influential environmental factors controlling the occurrence and distribution of macroinvertebrates (Ward and Stanford 1979, p. 35). Although water temperature may not be a major issue of concern for the Banbury Springs lanx in the four coldwater spring complexes where it resides, anthropogenic activities in the springs such as impoundments and/or diversions can alter natural thermal characteristics of water bodies (Ward and Stanford 1979, p. 42). This is problematic because the capacity of water to hold dissolved oxygen decreases with increasing water temperatures (Mason 1996, p. 34). As specialized respiratory organs are lacking, the Banbury Springs lanx are particularly sensitive to dissolved oxygen fluctuations (Baker 1925, p. 148) and have stringent dissolved oxygen requirements. It has been suggested that any factor that reduces dissolved oxygen concentrations in the water column (e.g., siltation, flow reduction, removal of riparian vegetation, and increased water temperature) for even a few days is likely to prove fatal to all or the majority of the population (Reed et al. 1989, pp. A1-4-A1-5).

*Accumulation of Nutrients, Sediments, and other Pollutants*

The two primary nutrients associated with plant growth, and of interest in freshwater systems, are nitrogen and phosphorus (Smith 1996, p. 300). Excessive additions of nitrogen and phosphorus constitute pollutants in water (Clark et al. 1998, p. 12) and can limit the ability of streams to support the beneficial uses (coldwater biota) (Hardy et al. 2005, p. 12). The main sources of excessive nutrient and sediment loads are agriculture in the form of crop production, cattle grazing, confined animal feeding operations, aquaculture facilities, and municipal wastewater treatment facilities (Bowler et al. 1992, pp. 45 - 47; EPA 2002a, pp. 4.22 - 4.24). Nitrogen and phosphorus are also introduced to the environment from numerous natural and anthropogenic sources (Smith 1996, pp. 206, 212; Clark et al. 1998, p. 12; Hardy et al. 2005, p. 12), including atmospheric deposition and the weathering of bedrock material, but also from sewage disposal and urban runoff. Excess nutrients enter groundwater by way of infiltration, percolation, and lateral flow through alluvial deposits and bedrock material. There it can be sequestered and accumulate in groundwater aquifers which eventually flow into spring habitats. Nutrient levels in springs may be linked to seasonal fertilizer application and irrigation practices. Data collected by the USGS from 1985 to 1990 on nutrient concentrations in springs within the Hagerman reach and their contribution to nutrient loading into the Snake River show that concentrations of nitrite+nitrate fluctuate seasonally and coincide with higher spring discharges during and following irrigation season (Clark 1994, pp. 19-24). Of this amount only 20 percent was derived from leguminous plants (e.g., alfalfa) while 29 percent was from cattle manure and 45 percent was from synthetic fertilizers (Clark 1994, p. 8). Similarly, the Idaho Department of Environmental Quality (IDEQ) reported that the majority of nitrogen concentrations in their study originated from agricultural fertilizers and livestock sources (Baldwin et al. 2000, p. 21). The report also stated that nitrate+nitrite concentrations increased significantly during the 1990s at spring sites along the north bank of the Snake River (Baldwin et al. 2000, p. 25), including the springs identified to be within the Banbury Springs lanx recovery area.

The total contribution of nitrogen and phosphorous entering the middle Snake River from agricultural lands via groundwater springs has been estimated to be 27,000 kilograms (kg) (59,529 pounds (lbs)) of nitrogen daily (EPA 2002a, pp. 4.22-4.24). This accounts for 64 percent of the detected total nitrogen in the system (MacMillan 1992 and Clark 1994, in EPA 2002a, p. 4.22). Recent reports developed by the Idaho Department of Agriculture (IDA) stated that groundwater aquifers within the middle Snake River region continue to be impacted by nitrates and pesticides (Bahr and Carlson 2000b, p. 10; Carlson and Bahr 2000, p. 3; Baldwin et al. 2000, pp. 22-23; Fox and Carlson, 2003, p. 7). One report stated that 53 percent of wells tested had levels of nitrates greater than 5 milligrams per liter (mg/L) and one well had concentrations greater than EPA's drinking water standard of 10 mg/L (Carlson and Bahr 2000, p. 3). Another report showed that 19 percent of wells tested approached the EPA's established drinking water limit of 10 mg/L, and 6 percent of the 761 tested wells surpassed the EPA standard (Bahr and Carlson 2000b, p. 10). The reports concluded that agricultural practices are likely a contributor of nitrates and pesticides to groundwater sources. Similarly, a review of springflow effects on chemical loads in the Snake River demonstrated that 36 percent of nitrogen in the system at King Hill, Idaho, was derived proximately from springflows and ultimately from irrigated agriculture (Clark and Ott 1996, pp. 556-560). More recently, Rattray et al. (2005) reported elevated levels of nutrients from groundwater samples collected from the Eastern Snake River Plain aquifer (Rattray et al. 2005, p. 8). They reported that at all sites, concentrations of nitrite+nitrate were greater than the laboratory reporting level (LRL), and at one site near Jerome, Idaho, the concentration of nitrite+nitrate exceeded the EPA's maximum contaminant level (MCL) for drinking water (Rattray 2005, p. 8).

Approximately 80 aquaculture facilities are located in the Hagerman Valley (Bowler et al. 1992, p. 46; EPA 2002a, p. 4.19), of which at least 3 utilize or divert coldwater spring and tributary flows where the Banbury Springs lanx resides. These facilities have directly affected spring habitats that are or may have been occupied by the Banbury Springs lanx and other coldwater spring adapted fauna. The two hatcheries that occur on the tributary springs where the lanx is found belong to a private facility which grows rainbow trout for human consumption, and does not have any mitigation responsibilities to the government or Tribes. Hatchery operations contribute significant quantities of nutrients and sediment to lower sections of coldwater springs as well as the Snake River (Bowler et al. 1992, pp. 45-47). Most of these nutrients are derived from metabolic wastes of the fish and unconsumed fish food. A number of aquaculture facilities also include fish-processing facilities and some of the processing wastes make their way into the Snake River (EPA 2002a, p. 4.20). Other wastes and residues from fish farms include disinfectants, bacteria, and residual quantities of drugs used to control disease outbreaks. Of the standard contaminants, aquaculture facilities contribute a sizable proportion of the total measured nutrients (e.g., greater than 5,000 kilograms per day (kg/day) nitrogen, and greater than 700 kg/day phosphorus) as well as an estimated 13,500 kg/day of suspended sediment in the mid-Snake River area (EPA 2002a, p. 4.22). Recent research found elevated levels of nitrogen and phosphorous, as well as elevated levels of trace elements, including zinc, copper, cadmium, lead, and chromium in sediments from the Snake River (Falter and Hinson 2003, p. 26 to 27). Benthic (occurring on the bottom of a stream) macroinvertebrate densities and biomass in Snake River studies have been shown to generally increase downstream of aquaculture discharges with a concomitant decrease in species richness, indicating an overall decline in habitat quality immediately downstream of aquaculture facilities (Falter and Hinson 2003, p. 13). The cumulative effects of these alterations (e.g., increased sediment, nutrients, and contaminants) are undesirable consequences with regard to benthic species habitats (Bowler et al. 1992, p. 45). In addition, the recent discovery of antibiotics originating from fish farms in streams of the United States is of concern (USGS 2003, *in litt.,* p. 1-4). Researchers from the USGS collected 189 water samples from 14 fish farms across the Nation and found antibiotics in 27 of those samples from 5 fish farms (USGS 2003, *in litt*., pp. 1-4). Although no information exists that directly links these pollutants as impacting Banbury Springs lanx, there are studies (Bowler et al. 1992, p.

45; USGS 2003, *in litt.*, pp. 1-4; Falter and Hinson 2003, p. 13) that show a decrease in species richness below aquaculture facilities and the lanx has not been found in these locations.

Another pollutant of concern to the Banbury Springs lanx is sediment. Past construction of the diversion structures in Box Canyon and Briggs Springs for aquaculture facilities likely impounded lanx habitat that is now inundated with fine sediment. Similar habitat modifications occurred at Banbury Springs with the impoundment of what is now Morgan Lake, which restricts the current distribution of that colony of lanx. Dr. Terrence Frest in an affidavit (Reed et al. 1989, p. A2-3) indicated that “immediate and irreparable harm” to lanx could occur with even a few hours of siltation because members of the subfamily Lancinae breathe through a heavily vascularized mantle and excessive siltation could compromise the animal’s oxygen exchange capacity.

The return of diverted irrigation water to the coldwater springs and tributaries plays a major role in degrading water quality (Frest and Johannes 1992, pp. 16-17; Bowler et al. 1992, pp. 45-47; Clark et al. 1998, p. 2; EPA 2002a, p. 4.21), which may impact benthic organisms in the Snake River. Irrigation return flow returns to coldwater springs within the range of the Banbury Springs lanx (Frest and Johannes 1992, pp. 16-17; Clark and Ott 1996, pp. 553-555). Irrigation water generally has increased temperatures (with a subsequent decrease in dissolved oxygen), contains pesticide residues, has been enriched with nutrients from agriculture (nitrogen and phosphorous), and frequently contains elevated sediment loads (Frest and Johannes 1992, pp. 16, 17; Bowler et al. 1992, p. 45; Clark et al. 1998, pp. 2-3;EPA 2002a, p. 4.22). In Sand Springs and the Thousand Springs complex Frest and Johannes (1992, pp. 16, 17) observed certain areas at the base of talus slopes discharging relatively warm, silty water that contained agricultural contaminants. Clark et al. (1998, pp. 2-3) found pesticides in animal tissues, streams, irrigation canals, and irrigation returns in the Snake River Basin in concentrations exceeding the aquatic-life criteria established by EPA. Similarly, Falter and Hinson (2003, pp. 68-69) found that sediments, nitrogen, phosphorous, and trace elements were generally higher downstream from irrigation returns while species richness was generally higher upstream.

Industrial wastes in groundwater are also a potential threat to the Banbury Springs lanx. Beginning in the 1950s, the Idaho National Laboratory (INL), a Department of Energy facility, pumped mixed waste and wastewater from nuclear industrial processing into the ground for disposal (Rattray et al. 2005, p. 1). This practice continued until 1984; currently waste and wastewater from the facility are handled differently and not pumped into the aquifer. These contaminants include tritium, strontium-90, cesium-137, gross alpha-particle radioactivity, and gross beta-particle radioactivity (Rattray et al. 2005, pp. 6-7). The presence of contaminants from nuclear industrial processes in the Snake River Plain aquifer is of concern because they someday will likely reach the coldwater springs upon which the Banbury Springs lanx depends. It is not currently known how these contaminants from the nuclear industrial process will impact the Banbury Springs lanx. However, Clark and Ott (1996, pp. 556-559) reported tritium in several of the coldwater springs in the mid Snake River area in extremely minute quantities. The source of this nuclear contaminant remains unknown.

Presently, there are other environmental pollutants affecting coldwater springs complexes where the Banbury Springs lanx occurs. Box Canyon Springs has concentrations of cadmium and lead that exceed the state of Idaho’s acute criteria for aquatic life (Hardy et al. 2005, p. 65). Recent research at Montana State University revealed elevated concentrations of lead, cadmium, and arsenic in *Fluminicola* tissues collected from Banbury Springs (Richards et al. 2002, *in litt.*, p. 4). Rattray et al. (2005) detected trace elements including barium, chromium, lithium, manganese, and zinc in water sources that supply the major springs on the north side of the Snake River (Rattray et al. 2005, pp. 7-8), including the Thousand Springs complex and Box Canyon Springs. The effects of metal bioaccumulation in stream organisms are widely documented in the primary literature (Eisler 1998, pp. 16-20; Brumbaugh et al. 2001, p. 19; Maret et al. 2003, pp. 1-2). Pollutants such as mercury, other trace elements, and pesticides can enter tributaries and springs (and eventually the Snake River) from atmospheric deposition, agriculture, and industrial inputs (Maret and Ott 1997, p. 2; Rattray et al. 2005, p. 4). Although the direct and long-term effects of these pollutants upon Banbury Springs lanx colonies are not known, the pollutants are present in the spring system in which the lanx resides.

**Inadequacy of Existing Regulatory Mechanisms**

The Idaho Department of Water Resources (IDWR) regulates water development in the Snake River Basin. At present, there are maintenance flow requirements for fish and wildlife on several tributary streams to the Snake River; however, coldwater springs used for aquaculture can be completely appropriated for hatchery operations if it falls within their water right although liability for take of a listed species remains under Section 9 of the Endangered Species Act. At Box Canyon, the Banbury Springs lanx occurs downstream of the aquaculture diversion and further reduction or diversion of this coldwater springflow would reduce suitable, available habitat and potentially harm this species. Present management regulations may be inadequate, and water withdrawals from groundwater aquifers, spring outflows, or tributary streams may be at a level that affects the sustainability of Banbury Springs lanx.

In Idaho, the EPA retains authority for the issuance of permits through the National Pollutant Discharge Elimination System (NPDES), which is designed to manage point-source discharges. There are approximately 80 private or public-owned aquaculture facilities on the middle Snake River now permitted under the NPDES and over 20 additional facilities have applied for permits (EPA 2002a, pp. 4.19-4.20; Meitl 2002, pp. 23-25). Briggs Springs and Box Canyon have active NPDES permits for point-source discharges downstream of existing Banbury Springs lanx colonies. Given the increase in permit applications and the record of Clean Water Act violations in Idaho and the Pacific Northwest (Meitl 2002, pp. 23-25), threats to aquatic species, including the Banbury Springs lanx, from unexpected point-source discharges are likely to continue and increase the immediate future (i.e., within the next five years).

The IDEQ is responsible for managing point and non-point sources of pollution to waterbodies of the State. These sources contribute to a stream's inclusion in the EPA's list of impaired water bodies pursuant to section 303(d) of the Clean Water Act. Additionally, IDEQ under authority of the State Nutrient Management Act coordinates efforts to identify and quantify contributing sources of pollutants (including nutrient and sediment loading) to the middle Snake River and other Idaho watershed areas using a Total Maximum Daily Load (TMDL) approach (Baldwin et al. 2000, pp. 14-21). The TMDL approach is used to develop pollution control strategies in waterbodies that are currently not meeting water quality standards through several of the following programs: State Agricultural Water Quality Program, CWA section 401 Certification, Bureau of Land Management land management plans, the State Water Plan, and local ordinances. Factors addressed under TMDLs are mostly limited to phosphorus, total suspended solids, dissolved oxygen, flow, temperature, pesticides, metals, and petroleum compounds.

TMDLs do not address groundwater, although protection of surface water would logically improve/conserve groundwater quality into the future.

### Other Natural or Manmade Factors affecting the Continued Existence of the Lanx

Invasive species may affect the continued existence of the Banbury Springs lanx in Idaho. The most notable example in the range of Banbury Springs lanx is the New Zealand mudsnail (mudsnail) (*Potamopyrgus antipodarum*) which was discovered in North America in the Snake River in 1987 and has since spread rapidly throughout Idaho and to other western states (Bowler 1991, pp. 175-176; Richards et al. 2004, p. 114). Frest and Johannes (1992, pp. 45-46) found the mudsnail in 43 sites on the Thousand Springs Preserve. Currently, the mudsnail occurs in all four coldwater spring tributaries where the Banbury Springs lanx is found but in very low densities at occupied Banbury Spring lanx sites (Hopper 2006a, *in litt.*, pp. 1-2; Hopper 2006b, *in litt.*, pp. 1-2). However, near habitat margins where Banbury Springs lanx disappear, observed mudsnail densities increase.

The mudsnail appears to flourish in watercourses with relatively low dissolved oxygen and with substrates of mud or silt, but has also been recorded at high densities within some of the cold-water spring complexes of the middle Snake River (e.g., up to 500,000/m2 at Banbury Springs; Richards et al. 2001, p. 375). Although the mudsnail may be able to withstand high water velocities (Lysne and Koetsier 2006a, pp. 81-83), they appear to reach the greatest densities in slower moving waters (Richards et al. 2001, p. 378). The New Zealand mudsnail's physiological tolerances (e.g., temperature and water velocity; Winterbourne 1969, p. 454; Lysne and Koetsier 2006a, pp. 81-83), life history attributes (e.g., high fecundity, growth, and dispersal rates; Winterbourne 1970, p. 147; Richards 2004, pp. 25-34), and habitat uses (e.g., springs, rivers, reservoirs, and ditches; Cada 2004, p. 27; Hall et al. 2003, p. 407; Clark et al. 2005, p. 10; Richards 2004, pp. 47-67) may confer to the mudsnail a competitive advantage over the Banbury Springs lanx. Given the potential for an ecosystem-wide impact given the species' specific habitat requirements (Hall et al. 2003, p. 407), the New Zealand mudsnail seems likely to continue to present a threat to native populations of aquatic species by occupying marginal habitats where native species may have been found.

### Climate Change

Air temperatures have been warming more rapidly over the Rocky Mountain West compared to other areas of the coterminous U.S. (Rieman and Isaak 2010, p. 3). Data from stream flow gauges in the Snake River watershed in western Wyoming, and southeast and southwest Idaho indicate that spring runoff is occurring between 1 to 3 weeks earlier compared to the early twentieth century (Rieman and Isaak 2010, p. 7). These changes in flow have been attributed to interactions between increasing temperatures (earlier spring snowmelt) and decreasing precipitation (declining snowpack). Global Climate Models (GMCs) project air temperatures in the western U.S. to further increase by 1 to 3 °C (1.8 to 5.4 °F) by mid-twenty-first century (Rieman and Isaak 2010, p. 5), and predict significant decreases in precipitation for the interior west. Areas in central and southern Idaho within the Snake River watershed are projected to experience moderate to extreme drought in the future (years 2035 to 2060) (Rieman and Isaak 2010, p. 5).

While the effects of global warming on the Banbury Springs lanx are not fully understood, it has the potential to affect their habitat. For another cold water dependent species, the bull trout, which tends to have lower thermal requirements than other salmonids, Rieman et al. (2007) predicted that global warming could reduce suitable habitat in the interior Columbia River basin by up to 92 percent (range 18 to 92 percent) (Rieman et al. 2007, p. 1559). While it is reasonable to suspect that populations of snails within the Snake River may be affected by elevated water temps, aquifer springs are less likely to immediately exhibit increased temperatures. If warmer winters deplete surface water reserves, either through earlier snow melt or greater proportions of precipitation as rain, then it is plausible that there will be an increased demand for groundwater, which could further reduce spring flows. Climate change will affect water use in the action area, but the magnitude of this effect will partially depend on how local government and water users respond to these changes. How this will affect the Banbury Springs lanx and their habitat is uncertain, but it is reasonable to anticipate potential adverse effects.

# 2.4.4 Bruneau Hot Springsnail

## 2.4.4.1 Status of Bruneau Hot Springsnail in the Action Area

See section 2.3.4 of this Opinion for the status of the Bruneau Hot Springsnail in the action area.

## 2.4.4.2 Factors Affecting Bruneau Hot Springsnail in the Action Area

The Service's 5-year status review (USFWS 2007, entire) found threats identified at the time of listing in 1998 still remain. As described in the following sections, the protected geothermal habitat along the Bruneau River upstream of Hot Creek is declining and existing colonies of the hot springsnail in this area are becoming more and more fragmented and isolated, primarily associated with irrigation groundwater withdrawal and the inadequacy of regulatory mechanisms to address the trend. As the geothermal aquifer continues to decline, the habitats downstream of Hot Creek become more important to the long-term survival of this species. Less significant threats to the geothermal habitat downstream of Hot Creek include: livestock grazing, surface water diversion, and recreation. Additionally, predation by two introduced species of warm water exotic fish threatens the long-term existence of this snail.

Refer to section 2.3.4.5 for more information on the conservation needs of the Bruneau hot spring snail.

**Habitat Curtailment**

*Groundwater Withdrawal and Springflow Reduction*

Groundwater withdrawal for irrigation has resulted in a decline of the geothermal aquifer underlying the Bruneau, Sugar, and Little valleys in north-central Owyhee County, Idaho which threatens the Bruneau hot springsnail through the reduction or loss of geothermal habitat. Increased agricultural use of groundwater since the mid-1960s has resulted in a steady decrease in local water table levels. Mineral deposits high on the basalt cliffs provide some evidence of once higher water levels (Myler 2000, p. 2). It appears that thermal springs were so plentiful that the Bruneau hot springsnail, within its historic range along Hot Creek and the Bruneau River, was able to migrate and colonize new locations or re-colonize former areas. Within the historical limits set by the elevation of surfacing hot water, the original population probably was not confined to isolated springs (Myler 2000, p. 2). The total number of geothermal springs along the Bruneau River upstream of Hot Creek (with and without Bruneau hot springsnails) declined from 1991 to 2006 (Myler 2006, pp. 2-6) and there are currently fewer high and low snail density sites with the Bruneau hot springsnail compared to 1991 (Myler 2006, p. 6; Figure 4).

Data from wells that monitor the geothermal aquifer near Indian Bathtub demonstrate that groundwater withdrawal for agriculture has had the most noticeable impact on the geothermal aquifer in that area (Myler 2007, Appendix 4, p. 1). By contrast, some monitoring wells located further from Indian Bathtub do not show such declines (Myler 2007, Appendix 4, p. 2). It is possible that because the geothermal aquifer is a confined pressure related system, certain wells in the immediate vicinity might cause a cone of depression or change the pressure equilibrium of the aquifer system. As with any aquifer, many questions remain regarding the dynamics of aquifer withdrawal and recharge, but geothermal spring/seep habitat on which the Bruneau hot springsnail depends is declining as well as the geothermal aquifer levels near Indian Bathtub (Myler 2007, Appendix 4). Because the water table has dropped dramatically, much of the geothermal spring habitat previously inhabited by the Bruneau hot springsnail is dry, resulting in a reduction in number of habitats, habitat area, and isolation of colonies.

The two largest Bruneau hot springsnail colonies (Hot Creek and Mladenka's Site 2), previously known from earlier reports (Taylor 1982b, p. 5; Mladenka 1992, p. 49), have been extirpated. Discharge from many of the geothermal springs along the Bruneau River is difficult to measure, therefore, the decline of the geothermal springflows is difficult to quantify. Photo points have been used for many of the surveys and definite reductions in geothermal spring discharges are easily observed from 1991 and 1993 surveys to present. Geothermal spring sites that have gone dry such as Indian Bathtub, Mladenka's Site 2, and Site U4E, demonstrate the drastic reduction in the geothermal aquifer at different locations. These sites are briefly discussed below.

As previously stated, in Hot Creek, approximately 1,000,000 Bruneau hot springsnails were estimated to occur in the "Low Indian Bathtub Hot Spring" in 1982, with as many as 60 snails/in$^2$ observed on the wetted rockfaces surrounding Indian Bathtub (Taylor 1982b, p. 5). Indian Bathtub, which is located at the base of Hot Creek Falls, was reduced to less than one-half its size by a major sediment deposition event in 1991 (Varricchione et al. 1997, p. 58). Field experiments performed by Myler (2000a, p. 26) in experimental exclosures placed in Hot Creek have shown that the Bruneau hot springsnail prefers large cobbles (> 10 cm diameter ( 4 in)) over gravel (2-10 mm (0.08-0.4 in)), and sand/silt(< 2 mm (< 0.08 in)). Trench analysis performed in Hot Creek in 1997, showed that larger substrate has been buried by finer gravel, sand, and silt (< 10 mm) (4 in) (Varricchione et al. 1997, p. 46). Another flood event occurred in Hot Creek in July 1992 which drastically reduced hot springsnails from Hot Creek by filling much of the Indian Bathtub area with sediment (Royer and Minshall 1993, p. 1), and by 1997, the population had been totally extirpated (Varricchione et al. 1997, p. 58). Currently, Hot Creek discharges 503 m (550 yards) downstream of Indian Bathtub (Myler 2006b, p. 7).

At Mladenka's Site 2 abundant thermal springwater once flowed down rock cliffs and created habitat for greater than 100,000 Bruneau hot springsnails (Mladenka 1992, p. 49). This site is currently dry except for seasonal flow that discharges from the base of the cliff (Myler 2006, p. 4). Site U4E also supported high densities of the Bruneau hot springsnail in 1991 and discharged one cubic foot per second (cfs) of geothermal water (Mladenka 1992, p. 71). In 1993, site U4E still supported a high density of Bruneau hot springsnails, but geothermal discharge had declined to a trickle. In 1996, Site U4E only discharged geothermal water below the surface of the

Bruneau River; and by 2000, the geothermal water at this location was gone and Bruneau hot springsnails were absent (Myler 2000, p. 12).

*Livestock Grazing*

Prior to 1998, livestock grazing was considered a threat factor that impacted some geothermal spring habitats where the Bruneau hot springsnail occurred near Hot Creek. In the 1990s, the BLM constructed fences to exclude livestock grazing in this area, and presently, cattle are excluded from Hot Creek and all geothermal spring habitats along the Bruneau River upstream of Hot Creek. Riparian vegetation has rebounded and is providing stream cover as well as defense against instream erosion. Indian Bathtub has not noticeably changed since it was filled with sediment in 1992. Presently, livestock grazing is considered a low ranking threat factor to the Bruneau hot springsnail colonies and the 24 geothermal habitats it occupies in Hot Creek or along the Bruneau River upstream of Hot Creek. Surveys conducted in 2004-2006 of geothermal springs and seep habitats along the Bruneau River downstream of Hot Creek document trampling by livestock and streambeds that are embedded in fine sediment (Myler 2005, pp. 7, 8; Myler 2006, p. 8). If the current declining trend of the geothermal aquifer continues and more geothermal spring habitats go dry upstream of Hot Creek, the importance of the habitat along the Bruneau River downstream of Hot Creek will become important to the long-term survival of the Bruneau hot springsnail.

*Surface Water Diversion*

Surface water withdrawals and diversions only occur along the Bruneau River downstream of Hot Creek. Within the recovery area, which extends approximately 2 km (1.2 mi) downstream of Hot Creek, there are two major diversions dams, Harris Dam and Buckaroo Dam. These dams take nearly all of the flowing water from the Bruneau River and send it to two canals to be used for irrigation in the lower Bruneau Valley. It is not known how the Bruneau hot springsnail disperses between geothermal springs; however, they have been observed to drift into the Bruneau River when disturbed (Myler 2006, p. 8). Therefore, removing the majority of the flow downstream of Hot Creek may impede the ability of this species to migrate or disperse to other geothermal springs located downstream. However, surface water diversion is a low ranking threat that only applies to habitat along the Bruneau River downstream of Hot Creek.

*Recreation*

The original 1993 listing stated that recreational access also impacts habitats of the Bruneau hot springsnail along the Bruneau River (58 FR 5938-5946). This activity continues to occur at one geothermal spring where small dams have been constructed to form thermal pools for bathing. The 1998 Notice of Determination determined that recreational use of thermal springs was not a significant threat to the Bruneau hot springsnail or its geothermal spring habitat (63 FR 32981-32996). Presently, only one known geothermal spring in the recovery area is used by recreational bathers, but is above the thermal maximum of 35°C (95°F), that the Bruneau hot springsnail can tolerate. Therefore, recreational use of the geothermal springs and seeps is considered a low ranking threat to the Bruneau hot springsnail. However, with the declining geothermal aquifer there remains concern that other bathing pools may be constructed in occupied Bruneau hot springsnail habitat.

### Disease or Predation

There is currently no information regarding the threat of disease to the continued existence of Bruneau hot springsnails. We believe that disease is not likely to affect the species unless an unknown pathogen is transmitted to the snails.

Introduced exotic redbelly Tilapia *(Tilapia zilli),* and mosquito fish *(Gambusia affinis)* populations thrive in Hot Creek and in the geothermal springs that discharge into the Bruneau River throughout the entire range of the Bruneau hot springsnail (Mladenka and Minshall 1993, p. 7; Myler 2005, p. 7). *T. zilli* is an omnivorous feeder (i.e. detritus, algae, invertebrates, and fish) and G. *affinis* also is known for a broad feeding preference (i.e. diatoms and other algae, crustaceans, and invertebrates) (Myler 2000, p. 11). A fish gut content analysis conducted on *T. zilli* and G. *affinis* collected from Hot Creek in 1995 did not find the Bruneau hot springsnail in stomachs (Varricchione and Minshall 1995, p. 1 ). However, an extensive survey conducted for the Bruneau hot springsnail from the origin of Hot Creek to the confluence with the Bruneau River in 1998, did not find hot springsnails (Myler and Minsahll 1998, p. 47), which suggests that the snails may not have been present to be eaten when the fish gut analysis was conducted in 1995.

Recent laboratory studies suggest that *Tilapia zilli* will use the snails as a food source. A laboratory fish feeding experiment was conducted in 1998 (Myler and Minsahll 1998) where *T. zilli* were captured from Hot Creek and placed in two aquaria. In the first aquarium, *T. zilli* were fed aquarium fish food, and in the second, fish were starved for 48 hours (Myler and Minsahll 1998, p. 14). Twenty Bruneau hot springsnails were then added into each aquarium and within two hours, all 40 snails had been consumed in both aquaria (Myler and Minsahll 1998, p. 53). A stomach analysis performed following this study revealed no hot springsnails in the stomachs of *T. zilli* (Myler and Minsahll 1998, p. 53).

In 1999, a controlled fish feeding experiment was performed in enclosures in Hot Creek with *T. zilli* and *P. bruneauensis* (Myler 2000, pp.11-17). All the Bruneau hot springsnails were absent within five days (Myler 2000, p. 26). At the end of five days, a stomach analysis was performed that revealed no Bruneau hot springsnails in the stomachs of *T. zilli* (Myler 2000, p. 26), indicating that shells are broken down by mastication, stomach acids, or rapid digestive processes.

Since *T. zilli* occur in the geothermal springs along the Bruneau River and in Hot Creek (Mladenka and Minshall 1993, p. 7; Myler 2005, p. 7) they likely threaten the continued existence of the Bruneau hot springsnail through predation. In addition, Mladenka observed *G. affinis* to eat Bruneau hot springsnails in the laboratory (Mladenka peer review comments to the 5-year status review). As madicolous habitat (thin sheets of water flowing over rock faces) goes dry (e.g., Indian Bathtub, Mladenka's Site 2, and Site U4E) Bruneau hot springsnails are in direct contact with these exotic fish and therefore are more susceptible to predation as the geothermal water levels continue to decline.

### Inadequacy of Existing Regulatory Mechanisms

The IDWR regulates water development in the Bruneau-Grand View area. The Bruneau-Grand View area was declared a Ground-Water Management Area in 1982 by IDWR due to increases and projected increases in groundwater withdrawal, and declines in spring flows from the geothermal aquifer system (Harrington and Bendixen 1999, p. 29). Present management and regulations that govern water use affecting the geothermal aquifer have not been adequate in reversing the continuing declining trend of the geothermal aquifer upon which the Bruneau hot springsnail depends (USFWS 2007, p. 27).

The IDEQ is responsible for managing point and non-point sources of pollution into waterbodies of the State. These sources contribute to a stream's inclusion in the EPA's list of impaired water bodies pursuant to section 303( d) of the CWA. Additionally, IDEQ under authority of the State Nutrient Management Act, coordinates efforts to identify and quantify contributing sources of pollutants (including nutrient and sediment loading) into Idaho watersheds areas using a Total Maximum Daily Load (TMDL) approach (Lay 2000, pp. 4-32). The TMDL approach is used to develop pollution control strategies in waterbodies that are currently not meeting water quality standards through several of the following programs: State Agricultural Water Quality Program, CWA section 401 Certification, BLM land management plans, the State Water Plan, and local ordinances. Currently the Bruneau River is under a TMDL which includes nutrients, total suspended solids, and temperature (Lay 2000, pp. 4-32). Although the Bruneau TMDL does not address groundwater, by addressing surface water pollutants, it may indirectly improve/conserve groundwater quality.

**Climate Change**

Air temperatures have been warming more rapidly over the Rocky Mountain West compared to other areas of the coterminous U.S. (Rieman and Isaak 2010, p. 3). Data from stream flow gauges in the Snake River watershed in western Wyoming, and southeast and southwest Idaho indicate that spring runoff is occurring between 1 to 3 weeks earlier compared to the early twentieth century (Rieman and Isaak 2010, p. 7). These changes in flow have been attributed to interactions between increasing temperatures (earlier spring snowmelt) and decreasing precipitation (declining snowpack). Global Climate Models (GCMs) project air temperatures in the western U.S. to further increase by 1 to 3 °C (1.8 to 5.4 °F) by mid-twenty-first century (Rieman and Isaak 2010, p. 5), and predict significant decreases in precipitation for the interior west. Areas in central and southern Idaho within the Snake River watershed are projected to experience moderate to extreme drought in the future (years 2035-2060) (Rieman and Isaak 2010, p. 5).

While the effects of global warming on the Bruneau hot springsnail are not fully understood, it has the potential to affect their habitat. For example, extreme drought and earlier spring run-off (due to decreased snowpack and earlier spring melt) will diminish recharge of the subsurface aquifers (Rieman and Isaak 2010, p. 7) including aquifers that support the Bruneau hot springsnail. If warmer winters deplete surface water reserves, either through earlier snow melt or greater proportions of precipitation as rain, then it is plausible that there will be an increased demand for groundwater, which could further reduce spring flows. Climate change will affect water use in the action area, but the magnitude of this effect will partially depend on how local government and water users respond to these changes. How this will affect the Bruneau hot springsnail and their habitat is uncertain, but it is reasonable to anticipate potential adverse effects.

## 2.4.5 Bull Trout

### 2.4.5.1 Status of the Bull Trout in the Action Area

Bull trout are found throughout the action area in spawning and early rearing habitat (local populations) as well as in habitat used for foraging, migrating, and overwintering (FMO). Spawning and early rearing habitat is typically found in headwater areas while mainstem rivers provide FMO habitat. Bull trout use these habitat types in 35 core areas within the action area (or approximately 30 percent of the core areas within the coterminous distribution of bull trout).

The analysis presented in this Opinion will assess bull trout baseline status at the larger draft recovery units and core area levels as opposed to the smaller, local population scale. The draft recovery plan (USFWS 2002a, p. 98) identified a bull trout core area as the closest approximation of a biologically functioning unit for bull trout. Core areas contain both spawning and early rearing and FMO habitat. Core areas constitute the basic unit on which to gauge recovery (USFWS 2002b, p. 98).

**Table 3. Status of bull trout core areas (by draft recovery units) within the action area from the Services 5-year Status Review (USFWS 2008b).**

| Draft Recovery/ Management Unit | Core Area | Population Abundance Category (individuals) | Distribution Range Rank (Stream length miles) | Short-term Trend Rank | Threat Rank | Final Rank |
|---|---|---|---|---|---|---|
| Coeur d'Alene | Coeur d'Alene Lake | 50-250 | 125-620 | Stable | Substantial, imminent | High risk |
| Northeast Washington – not located within Idaho but included because of potential downstream effects. | Pend Oreille River | 1-50 | 25-125 | Unknown | Substantial, imminent | High risk |
| Clark Fork | – Lake Pend Oreille | 2500-10000 | 620-3000 | Stable | Moderate, non-imminent | Potential risk |
| | Priest Lakes | 50-250 | 25-125 | Rapidly declining | Substantial, imminent | High risk |
| Kootenai | Kootenai River | 250-1000 | 125-620 | Stable | Moderate, imminent | At risk |
| Clearwater | NF Clearwater | 250-1000 | 125-620 | Declining | Moderate, imminent | At risk |

| Draft Recovery/ Management Unit | Core Area | Population Abundance Category (individuals) | Distribution Range Rank (Stream length miles) | Short-term Trend Rank | Threat Rank | Final Rank |
|---|---|---|---|---|---|---|
| | Fish Lake (NF) | 1-50 | 125-620 | Declining | Moderate, imminent | High risk |
| | Lochsa R. | 50-250 | 125-620 | Stable | Moderate, imminent | At risk |
| | Fish Lake (Lochsa) | 1-50 | 125-620 | Unknown | Widespread, low-severity | At risk |
| | Selway R. | unknown | 125-620 | Unknown | Widespread, low-severity | Potential risk |
| | SF Clearwater | 1000-2500 | 125-620 | Unknown | Substantial, imminent | At risk |
| | Middle-Lower | unknown | 125-620 | Unknown | Substantial, imminent | At risk |
| Salmon | Upper Salmon | unknown | 620-3000 | Unknown | Moderate, imminent | Potential risk |
| | Pahsimeroi R. | unknown | 125-620 | Unknown | Widespread, low-severity | At risk |
| | Lake Cr. | 50-250 | 25-125 | Unknown | Widespread, low-severity | At risk |
| | Lemhi R. | 250-1000 | 125-620 | Unknown | Substantial, imminent | At risk |
| | Middle Salmon R. – Panther | unknown | 125-620 | Unknown | Moderate, imminent | At risk |
| | Opal Lake | unknown | 125-620 | Unknown | Moderate, imminent | Potential risk |
| | Middle Fork Salmon | unknown | 620-3000 | Unknown | Slightly | Low risk |
| | Middle Salmon-Chamberlain | unknown | 125-620 | Unknown | Widespread, low-severity | Potential risk |
| | SF Salmon | unknown | 125-620 | Unknown | Moderate, imminent | At risk |

| Draft Recovery/ Management Unit | Core Area | Population Abundance Category (individuals) | Distribution Range Rank (Stream length miles) | Short-term Trend Rank | Threat Rank | Final Rank |
|---|---|---|---|---|---|---|
| | Little-Lower Salmon | 50-2250 | 125-620 | Unknown | Substantial, imminent | High risk |
| Hells Canyon Complex | Pine-Indian-Wildhorse | 250-1000 | 125-620 | Very rapid decline | Substantial, imminent | High risk |
| SW Idaho | Arrowrock | unknown | 125-620 | Declining | Moderate, imminent | At risk |
| | Anderson Ranch | 250-1000 | 125-620 | Unknown | Substantial, imminent | At risk |
| | Lucky Peak | 1-50 | 25-125 | Unknown | Substantial, imminent | High risk |
| | Upper SF Payette R. | unknown | 125-620 | Unknown | Moderate, imminent | At risk |
| | MF Payette R. | unknown | 25-125 | Unknown | Substantial, imminent | At risk |
| | Deadwood R. | 125-1000 | 25-125 | Unknown | Substantial, imminent | High risk |
| | NF Payette R. | 1-50 | 2.5-25 | Very rapid decline | Substantial, imminent | High risk |
| | Squaw Creek | 250-1000 | 25-125 | Unknown | Substantial, imminent | High risk |
| | Weiser R. | unknown | <2.5 | Rapidly declining | Substantial, imminent | High risk |
| Little Lost | Little Lost | unknown | 25-125 | Unknown | Substantial, imminent | At risk |
| Imnaha/Snake | Sheep | unknown | 2.5-25 | Unknown | Unthreatened | Unknown risk |
| | Granite | unknown | 2.5-25 | Unknown | Unthreatened | Unknown risk |

| Draft Recovery/ Management Unit | Core Area | Population Abundance Category (individuals) | Distribution Range Rank (Stream length miles) | Short-term Trend Rank | Threat Rank | Final Rank |
|---|---|---|---|---|---|---|
| Jarbidge River (interim recovery unit) | Jarbidge River | 50-250 (recent surveys show abundance is four times higher than this level (Allen et al. 2010, p. 20) | 125-620 | Unknown | Substantial/ imminent | High risk |

The summary of Table 3 below, shows the number and name of core areas at each level of extirpation risk, by draft recovery unit:

1 core area at *low risk*: Salmon River - Middle Fork of the Salmon River

5 at *potential risk*: Clark Fork – Lake Pend Oreille, Clearwater – Selway River, Salmon – Upper Salmon River, Salmon – Opal Lake, Salmon – Middle Salmon-Chamberlain.

16 *at risk*: Kootenai – Kootenai River, Clearwater – NF Clearwater, Clearwater – Lochsa River, Clearwater – Fish Lake (Lochsa), Clearwater – SF Clearwater River, Clearwater – Middle-Lower, Salmon – Pahsimeroi R., Salmon – Lake Cr., Salmon – Lemhi R., Salmon – Middle Salmon R. – Panther, Salmon – SF Salmon, SW Idaho – Arrowrock, SW Idaho – Anderson Ranch, SW Idaho – Upper SF Payette R., SW Idaho – MF Payette R., SW Idaho – Little Lost

11 at *high risk*: Northeast Washington – Pend Oreille River, Coeur d’Alene – Coeur d’Alene Lake, Clark Fork – Priest Lakes, Clearwater – Fish Lake (NF), Salmon – Little-Lower Salmon, SW Idaho – Lucky Peak, SW Idaho – Deadwood R., SW Idaho – NF Payette R., SW Idaho – Squaw Creek, SW Idaho – Weiser R, Jarbridge River interim recovery unit Jarbridge River

2 at *unknown risk*: Imnaha-Snake – Sheep Creek, Granite Creek

These figures show that 77 percent of the core areas in the action area are “at risk” or “at high risk” of extirpation.

## 2.4.5.2 Factors Affecting the Bull Trout in the Action Area

As previously described in the Status of the Species section of this Opinion, bull trout distributions, abundance, and habitat quality have declined rangewide primarily from the combined effects of habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, angler harvest, entrainment, and introduced non-native fish species such as brook trout. There are numerous natural and anthropogenic influences on bull trout throughout the state of Idaho. Although restoration actions and ongoing research efforts have positively affected bull trout, the majority of anthropogenic influences have contributed to the species decline by reducing bull trout numbers, reproduction, and distribution.

### Current Threats to the Bull Trout and Bull Trout Critical Habitat

For more information regarding factors affecting specific core areas within the action area, please refer to the individual chapters in the Service's 2002 Bull Trout Draft Recovery Plan for the Columbia River (USFWS 2002a, entire) and the 2004 Jarbidge River Draft Recovery Plan (USFWS 2004a, entire; Allen et al. 2010, entire). The individual chapters in the Service's draft plans identified the categories of activities that have had the most significant adverse impacts on bull trout in each recovery unit, and are summarized below.

Because bull trout is a wide-ranging species, threats to local populations vary with area. In general, as stated above, population declines have resulted from the combined effects of habitat degradation and fragmentation, the blockage of migratory corridors, poor water quality, angler harvest and poaching, entrainment into diversion channels and dams, and introduced nonnative species. Specific land and water management activities that depress bull trout populations and degrade habitat include dams and other diversion structures, forest management practices, livestock grazing, agriculture, agricultural diversions, road construction and maintenance, mining, and urban and rural development (see USFWS 2002a, pp. vi-v). To provide some specificity, we have grouped threats to bull trout by draft recovery unit (within the action area) and described them below. Note: Critical habitat units were patterned after draft recovery units and have similar boundaries.

### Draft Coeur d'Alene Lake Basin Recovery Unit.

Bull trout are found primarily in the upper portions of the St. Joe River subbasin (PBTTAT 1998; USFWS 1998), which contains spawning and rearing habitats (USFWS 2002e, p. 8). Migratory bull trout also use the St. Joe River and Coeur d'Alene Lake for foraging, migrating, and overwintering habitat. The distribution and abundance of bull trout in the Coeur d'Alene Lake basin have been effectively limited by landscape-level changes that degraded physical and chemical habitat quality and resulted in fragmentation of habitat patches and isolation of populations. Dramatic changes in riparian, wetland, stream, and forest ecosystems have resulted from several suppressing factors that include livestock grazing, dam construction, logging, mining, introduction of and management for exotic species, channelization, urbanization, construction of transportation networks, and irrigation withdrawals. In many instances, habitat degradation and consequent reduction in bull trout populations have resulted from the cumulative effects of changes to terrestrial and aquatic ecosystems. Over time, these cumulative effects may be the most harmful to bull trout populations because of their potential to alter ecosystem processes that have defined bull trout existence.

Mining activities have contributed to aquatic and riparian habitat degradation and impaired water quality in Coeur d'Alene Lake and portions of the Coeur d'Alene River and St. Joe River subbasins. Aquatic conditions have been, and continue to be, unsuitable for resident fishes and other aquatic life in the South Fork Coeur d'Alene River and mainstem Coeur d'Alene River downstream to Coeur d'Alene Lake, primarily due to mine pollution (Ellis 1932, p. 117, Dixon 1999, p. 16; Rahel 1999 pp. 18-19; Reiser 1999, pp. 6-1 – 6-5). In addition, Coeur d'Alene Lake currently exceeds state water quality criteria for lead, zinc, and cadmium at various times during a typical year and is not fully protective of aquatic life. Rahel (1999, p. 18) concluded that fish populations downstream of Canyon Creek in the South Fork Coeur d'Alene River showed a clear spatial pattern of being reduced when compared with the population level further upstream, as well as population levels in a reference stream. This observation includes reduced abundance of trout and the absence of native sculpin species and mountain whitefish. The alteration of the fish community was most closely associated with metals rather than changes in other habitat features. Reiser (1999, p. 6-1) found that wild trout populations in Nine Mile Creek, Canyon Creek, and the South Fork Coeur d'Alene River are controlled by elevated metal concentrations. Dixon (1999, p. 16) concluded that there is clear evidence that metals are causing injury to fish in the Coeur d'Alene River subbasin. He also concluded that there is substantial evidence of direct lethal and sublethal toxicity to fish in the Coeur d'Alene subbasin.

One of the largest superfund sites in the nation (Bunker Hill) is located in the South Fork Coeur d'Alene River drainage near Kellogg, Idaho. Heavy metal contamination continues to exclude fish in some reaches of the lower portion of the river. Woodward (1999, p. 5) concluded that the water column concentrations of cadmium and zinc in the Coeur d'Alene River will reduce survival, growth, and abundance of fish. He also concluded that fish feeding on invertebrates in the river below locations of mine waste release have a diet source with elevated metals and are therefore at risk of reduced fitness. The Department of Interior, Department of Agriculture, the Coeur d'Alene Tribe, and the state of Idaho have partnered to implement restoration actions in the Basin in response to the environmental degradation from mining activities (see http://restorationpartnership.org/index.html, accessed October 22, 2014). [See USFWS 2002e, pp. 13-24 for more details on threats to bull trout.]

*Core Area Status*

The only core area within this draft recovery unit, Coeur d'Alene Lake Basin (encompassing the entire Coeur d'Alene Lake, the St. Joe and Coeur d'Alene River subbasins), is at "high risk" of extirpation (Refer to Table 3 for the status of all the following core areas).

**Draft Northeast Washington Recovery Unit**.

The construction and operation of Albeni Falls, Box Canyon, and Boundary Dams on the Pend Oreille River have fragmented habitat and negatively impacted migratory bull trout. Other dams and diversions without fish passage facilities in tributaries to the Pend Oreille River further fragmented habitat and reduced connectivity. Impacts from past timber harvest have altered habitat conditions in portions of the draft recovery unit; the legacy of these activities still persists where high densities of roads, impassable culverts, channel changes, and compaction of hill slopes remain. Livestock grazing has degraded habitat in both upland and riparian areas of most tributaries in the watershed on public and private land. Nonnative species have been introduced in the draft recovery unit and continue to impact bull trout populations through competition and hybridization. [See USFWS 2002f (pp. 14-22) for more details on threats to bull trout.]

*Core Area Status*

The Pend Oreille River core area (the only core area within this recovery unit, located in Washington State) is at "high risk" of extirpation.

**Draft Clark Fork Recovery Unit.**

Dams have been one of the most important factors in reducing the bull trout population of the draft Clark Fork Recovery Unit. Large hydroelectric dams have permanently interrupted established bull trout migration routes, eliminating access from portions of the tributary system to Lake Pend Oreille and Flathead Lake. Additionally, these dams have impacted the habitat that was left behind, altering reservoir and lake levels, water temperature, and water quality. Smaller irrigation storage dams further fragmented some of the watershed and impair migration. The risk of local population extirpation from isolation and fragmentation of habitat in the draft recovery unit is increasing, particularly where populations of bull trout are in decline. Major dams were the catalyst for much of this disruption, and fragmentation has continued at a finer scale, caused by habitat decline and introductions of nonnative species. At a few locations, however, benefits have resulted from some dams forming isolation barriers that have prevented the movement of nonnative fish. While bull trout are present in most historical core areas, substantial evidence indicates that local populations have been extirpated in major portions of this draft recovery unit, and many populations are at low enough levels to seriously reduce the chances of recolonization.

For over 100 years, forestry practices have caused major impacts to bull trout habitat throughout the draft Clark Fork Recovery Unit. Because forestry is the primary landscape activity in the basin, the impacts have been widespread. The negative primary effects of past timber harvest, such as road construction, log skidding, riparian tree harvest, clear-cutting, and splash dams, have been reduced by the more progressive practices that have since been developed. The legacy of the past century has resulted in lasting impacts to bull trout habitat, however, including increased sediment in streams, increased peak flows, hydrograph and thermal modifications, loss of instream woody debris, channel instability, and increased access by anglers and poachers. These impacts continue, and are irreversible in some drainages.

Agricultural impacts are also a significant and widespread threat to bull trout in this draft recovery unit. Diversions for irrigation can destabilize stream channels, severely interrupt migratory corridors (blockages and dewatering) and, in some cases, entrain fish that become lost to the ditches. Another, potentially more serious issue, is the increased water temperature regime common to streams that are heavily diverted and/or subject to receiving irrigation return flows. Some of the worst agricultural impacts occur in the upper drainages, and these problems are then transmitted to the receiving waters downstream.

Transportation systems are also a threat to bull trout in this draft recovery unit. Construction methods during the late 19$^{th}$ and early 20$^{th}$ century, primarily channelization and meander cutoffs, caused major impacts to many of these streams, impacts that are still being manifested. Such impacts seldom occur with new roads. However, significant problems remain that are associated with passage barriers, sediment production, unstable slopes, improper maintenance, increased water temperatures from reduced shading, and high road densities, all of which impact bull trout.

Extreme water quality degradation from mining in the upper portions of the Clark Fork River drainage dates back to the 19$^{th}$ century and will continue to impact bull trout for many years. Over a century of mining and smelting activity in the upper Clark Fork watershed resulted in designation of one of the nation's largest Superfund site with the EPA. Descriptions of the river from early researchers clearly indicate that certain reaches were void of fish prior to 1900 as a result of mining-related pollution (Evermann 1901, p. 16). The entire 40 km (25 mi) length of Silver Bow Creek remains fishless, and fish populations in the upper 193 km (120 mi) of the Clark Fork River remain depressed in some reaches due to copper contamination from mine tailings (Phillips and Lipton 1995, p. 1991). Most other drainages in the upper Clark Fork River basin have also been impacted by gold mining activity (placer and hydraulic). Permits are being sought to operate an underground copper/silver mine and mill that could produce 10,000 tons of ore per day in the Rock Creek drainage of the Lower Clark Fork Recovery Subunit near Noxon. The Rock Creek drainage has been identified as one of two spawning and rearing streams for migratory bull trout. There are areas in the Lake Pend Oreille basin that have been impacted by underground and open-pit mining operations and the resulting effluent from these closed or abandoned mines.

Impacts from unmanaged growth and residential sprawl may be one of the largest threats to the recovery of bull trout in this draft recovery unit. Human population growth in western Montana and northern Idaho has accelerated. Increasing human populations have a direct impact on all of the other risk categories that affect bull trout. Both legal and illegal angling have direct impacts on bull trout populations, despite the implementation of restrictive fishing regulations and strong educational efforts. The problem of illegal take of bull trout is intensified in stream corridors where roads provide access to highly visible (and therefore vulnerable) spawning stocks. [See USFWS 2002g (pp. 29-115) for more details on threats to bull trout.]

*Core Area Status*

The two core areas within this recovery unit, Priest Lakes and Lake Pend Oreille, are at high risk and potential risk of extirpation, respectively.

**Draft Kootenai River Recovery Unit.**

Of the factors listed above, habitat degradation and fragmentation, and land and water management activities are likely the most limiting factors for bull trout in this draft recovery unit. Libby Dam has been one of the most important factors affecting bull trout in this draft recovery unit. Completion of the dam in 1972 severed the migratory corridor between the upper Kootenai River watershed (Montana and British Columbia) and the lower Kootenai River basin in northern Idaho. The dam blocks all upstream migration and essentially bisects the United States portion of the Kootenai River drainage into two reaches. The habitat in the riverine reach has been altered as a result of Libby Dam and is characterized by unnatural flow patterns, water temperatures, and water quality parameters.

Forestry practices also rank as a high risk to bull trout in the draft Kootenai River Recovery Unit, largely because forestry is the dominant land use in the basin. Although the current forestry practices have improved, the risk of adverse effects to bull trout is still high because of the existing road system, mixed land ownership, lingering results of past activities, and inconsistent application of best management practices.

Mining has caused site-specific impacts on local populations of bull trout, but widespread negative impacts to water quality due to mining (such as those occurring in the draft Clark Fork Recovery Unit) have not occurred in this draft recovery unit. There are several active and proposed mining operations in the watershed, some of large dimension. Fisheries management risks include poaching, introduction of nonnative species, and growing angler use of both the lake and river. Illegal harvest of bull trout has been well documented in the draft Kootenai River Recovery Unit and is considered a high risk because of the traditional focus on well-known and limited spawning areas. Introduced species are widespread throughout the drainage, and the proliferation of brook trout is currently thought to present the greatest nonnative species risk to bull trout due to the threat of hybridization. [See USFWS 2002h (pp. 19-33) for more details on threats to bull trout in this area.]

*Core Area Status*

The Kootenai River core area is "at risk" of extirpation.

**Draft Clearwater River Recovery Unit**

Land and water management activities that depress bull trout populations and degrade habitat in the draft Clearwater River Recovery Unit include operation and maintenance of dams and other diversion structures, forest management practices, livestock grazing, agriculture, agricultural diversions, road construction and maintenance, mining, and introduction of nonnative species. Impassable dams and diversion structures isolate and fragment bull trout local populations. Forestry activities impact bull trout through decreased recruitable large woody debris, increased water temperatures from reduced shading, and lack of pools and habitat complexity. Livestock grazing degrades aquatic habitat by removing riparian vegetation, destabilizing streambanks, widening stream channels, promoting incised channels and lowering water tables, reducing pool frequency, increasing soil erosion, and altering water quality. Agriculture practices impact bull trout through added inputs of nutrients, pesticides, herbicides, and sediment, and reduced riparian vegetation. Introduced brook trout threaten bull trout through hybridization, competition, and possible predation.

Agriculture practices within the lower Clearwater basin are extensive and have both an ongoing and legacy effect on fisheries and water quality in the Lower and Middle Fork Clearwater River Core Area. Farming practices include the use of fertilizers, insecticides, and herbicides, and drain ditches, channel straightening, and field tiling to improve drainage. Soil erosion rates are among the highest in the country. Changes in land cover from grass/herbaceous/tree to tilled cropland, combined with stream channel alterations and increased runoff, have cumulatively changed the form and hydrologic function of all the tributaries in the lower Clearwater basin (CBBTTAT 1998, p. 27).

Mining degrades aquatic habitat used by bull trout by altering water chemistry (e.g., pH); altering stream morphology and flow; and causing sediment, fuel, heavy metals and other toxics to enter streams (Martin and Platts 1981, p. 1, Spence et al. 1996, p. 7). The South Fork Clearwater River Core Area in particular has a complex mining history that included periods of intense mining by varied methods including dredging, hydraulic, draglines, drag shovels, and hand operations. Mines are distributed throughout the draft recovery unit, with the lowest number of occurrences in the Selway River Core Area. The majority of mines pose a low relative degree of environmental risk, however, there are mines with high ecological hazard ratings located in the South Fork Clearwater River Core Area (Crooked, Red, and American Rivers and Newsome Creek watersheds) and in the Orofino drainage of the Middle-Lower Clearwater River Core Area (CSS 2001, pp. 57, 58-59). In the Moose Creek watershed within the North Fork Clearwater Core Area, tailing piles and channelization have been identified as threats to bull trout. [See USFWS 2002i (pp. 42-82) for more details on threats to bull trout.]

*Core Area Status*

Of the seven core areas within the Clearwater River recovery unit, the Selway River is at "potential risk"; the North Fork Clearwater River, Lochsa River, Fish Lake (Lochsa), South

Fork Clearwater River, and the Middle-Lower Clearwater River are “at risk”; and the Fish Lake (North Fork Clearwater) is at “high risk” of extirpation.

**Draft Salmon River Recovery Unit**

Dramatic changes have occurred in riparian, wetland, stream, and forest ecosystems mostly outside wilderness areas in the draft Salmon River Recovery Unit. These changes have resulted from several suppressing factors that include livestock grazing, logging, roads, mining, introduction and management for nonnative species, and irrigation withdrawals. In many instances, habitat degradation and consequent reduction in bull trout populations outside of wilderness areas have resulted in cumulative effects of change to terrestrial and aquatic ecosystems. Legacy effects of forest management practices are prevalent throughout the draft recovery unit (e.g., excessive bank instability, erosion, and sedimentation). Livestock grazing impacts riparian vegetation and bull trout habitat in most core areas in the draft recovery unit, with the most prevalent impacts occurring in the Upper Salmon River, Middle Salmon River-Panther, Upper Salmon River, and Pahsimeroi Core areas.

Water diversions, primarily for agriculture, are one of the most prevalent threats to bull trout in the Lemhi River, Pahsimeroi River, Upper Salmon River, and Middle Salmon River-Panther Core areas. There are an estimated 773 known diversions in the Salmon River basin (Servheen 2001, p. 101).

Agricultural practices, such as cultivation, irrigation, and applications of pesticides can also release sediment, nutrients, and pesticides into streams, and reduce riparian vegetation. In 1988, the IDEQ conducted an assessment of nonpoint source pollution of the Salmon River basin. Of 4,080 km of streams assessed, 1,374 km were determined to be negatively affected by agricultural practices (USFWS 1998, p. 41).

Effects of roads on bull trout include adverse impacts of excessive amounts of fine sediment, reduced large woody debris recruitment, habitat degradation in and near streams, increased water temperatures from reduced shading, and increased human access which may induce angling mortality and introductions of nonnative fishes. Approximately 11 percent of the draft Salmon River Recovery Unit has high road density (greater than 1.05 km per square km), 25 percent of the area has moderate road density (0.4 to 1.05 km per square km), 37 percent of the area has low road density, and 27 percent of the area is roadless (Servheen 2001, p. 28). In the Upper Salmon River Core Area heavy recreational and residential development associated with Redfish Lake has released chemical and nutrient pollutants and degraded bull trout habitat (USFS 1999, p. V-68). Other residential development in the Sawtooth Valley continues to impact bull trout habitat by filling flood channels and by diverting water from bull trout streams (USRITAT 1998, p. 39). Brook trout hybridization and brook trout competition for habitat are also known threats to bull trout in the draft recovery unit. Brook trout were stocked in the draft Salmon River Recovery Unit from 1913 to 1998 (Servheen 2001, p. 59).

Although active mining operations are less abundant than they were in the past, mining in the Salmon River basin is widespread and impacts to tributary streams are significant. Acid or other mine drainage occurs in the Thompson Creek drainage (Pat Hughes, Buckskin, and Thompson Creeks), and Jordan/Pinyon, Big Deer, Blackbird, Panther, Patterson, Warren, Crooked, Sugar, Meadow Creeks, East Fork of the South Fork of the Salmon River. Mine tailings and debris exist in the lower Yankee Fork River, the Slate Creek watershed. Blackbird Creek Mine is a

Superfund Site (Site), located on Blackbird Creek and continues to release contaminants into the Panther Creek watershed. Final remedial activities commenced in 2003. Downstream of the discharge into Panther Creek aquatic life including bull trout has been heavily impacted or absent for many miles, but by 2007 benthic macroinvertebrates and fish had begun to reoccupy the affected stream reaches (EPA 2008b, p. 36). In 2008, the Forest Service approved the Idaho Cobalt Project, a cobalt and copper mine on Forest Service and private lands within and adjacent to the Blackbird Mine Site; the date when construction and active mining will start is unknown (EPA 2013a, p. 3-2 – 3-3). Bull trout occupy Blackbird Creek upstream of the mining impacts and are just starting to reoccupy Big Deer Creek downstream of the South Fork of Big Deer Creek as cleanup efforts continue. Stibnite Mine (Meadow Creek drainage) has been considered as a potential Superfund Site for more than a decade. Drainage from the mine site has resulted in arsenic and antimony concentrations in the upper East Fork South Fork Salmon river to exceed State water quality criteria from 1978 to 1996. Concentrations of these metals present in 1997 were considered stressful to salmonid populations (Wagoner and Burns 2001, p. 28). [See USFWS 2002j (pp. 31-54) for more details on threats to bull trout.]

*Core Area Status*

Of the nine core areas in the Salmon River recovery unit, the Middle Fork Salmon River is at “low risk”; the Upper Salmon River, Opal Lake, and Middle Salmon River-Chamberlain is at “potential risk”; the Pahsimeroi River, Lake Creek, Middle Salmon River-Panther, and South Fork Salmon River “at risk”; and the Little-Lower Salmon River is at “high risk” of extirpation.

**Draft Hells Canyon Complex Recovery Unit**

Currently, habitat fragmentation and degradation are likely the most limiting factors for bull trout throughout the Hells Canyon Complex. In the Snake River, large dams of the Hells Canyon Complex lack fish passage and have isolated bull trout among three basins, the Pine Creek and Indian Creek watersheds, Wildhorse River, and Powder River. Dams, irrigation diversions, and road crossings have formed impassable barriers to fish movement within the basins, further fragmenting habitats and isolating bull trout. Land management activities that degrade aquatic and riparian habitats by altering stream flows and riparian vegetation, such as water diversions, past and current mining operations, timber harvest and road construction, and improper grazing practices, have negatively affected bull trout in several areas of the draft recovery unit. Bull trout are also subject to negative interactions with nonnative brook trout in streams where the species occur together.

Extensive mining activities were historically conducted and continue in the draft Hells Canyon Complex Recovery Unit. Degradation of aquatic and riparian habitats important for bull trout caused by mining include removal of riparian vegetation, stream channelization, sedimentation, and input of potentially toxic substances. Most mining activities in the draft recovery unit have occurred in the Pine Creek and Powder River basins. Mine tailings were placed on the banks of Pine Creek and East Fork Pine Creek and are considered hazardous waste by the Oregon Department of Environmental Quality. It is unknown whether toxic materials are leaching from the tailing piles and affecting fishes currently residing in the area (Powder Basin Watershed Council (PBWC) 2000, p. 66). [See USFWS 2002k (pp. 15-27) for more details on threats to bull trout.]

*Core Area Status*

The Pine-Indian-Wildhorse core area is at "high risk" of extirpation.

**Draft Southwest Idaho Recovery Unit**.

Habitat fragmentation and degradation are likely the most limiting factors for bull trout throughout the draft Southwest Idaho Recovery Unit. Although reservoirs formed by dams in some basins have allowed bull trout to express adfluvial life histories, dams, irrigation diversions, and road crossings have formed impassable barriers to fish movement within the basins, further fragmenting habitats and isolating bull trout. Land management activities that degrade aquatic and riparian habitats by altering stream flows and riparian vegetation, such as water diversions, past and current mining operations, timber harvest and road construction, and improper grazing practices, have negatively affected bull trout in several areas of the draft recovery unit. Bull trout are also subject to negative interactions with nonnative brook trout in some streams. [See USFWS 2002l for more details on threats to bull trout.]

*Core Area Status*

Of the eight core areas in this unit, Arrowrock, Anderson Ranch, and Middle Fork Payette River are "at risk" while Lucky Peak, Deadwood River, North Fork Payette River, Squaw Creek, and Weiser River are at "high risk" of extirpation.

**Draft Little Lost River Recovery Unit**.

Within the draft Little Lost River Recovery Unit, elevated stream temperatures are probably the most limiting factor for bull trout. Land management activities, such as water diversions and improper grazing practices, that degrade aquatic and riparian habitats by altering stream flows and riparian vegetation may elicit or exacerbate unsuitable water temperature regimes for bull trout. Other factors that negatively affect bull trout in the draft recovery unit include habitat fragmentation and isolation due to fish passage barriers, interactions with nonnative brook trout, and possibly harvest of fish due to poaching or to misidentification by anglers. [See USFWS 2002m (pp. 11-21) for more details on threats to bull trout.]

*Core Area Status*

The Little Lost River core area is the only core area in this unit and is at "high risk" of extirpation.

**Draft Imnaha-Snake Rivers Recovery Unit**

There has been a combination of human-induced factors that have adversely affected bull trout including forest management practices, irrigation withdrawals, livestock grazing, past bull trout harvest, and introduction of nonnative species. Lasting effects of some of these activities still act to limit bull trout production in the Imnaha, Sheep Creek, and Granite Creek core areas. Dams in the Snake River have impaired the connectivity between bull trout local populations from the draft Imnaha-Snake Rivers Recovery Unit and those from below Lower Granite Dam or above Hells Canyon Dam.

Past forest practices such as logging (Little Sheep Creek watershed), thinning of riparian vegetation, destruction of riparian vegetation, and increased sedimentation from forest roads

(Imnaha River watershed) have impacted bull trout by decreasing the function of the existing riparian vegetation in many areas.

Livestock grazing has contributed to the decline of bull trout through impacts to both upland and riparian areas of many tributaries in the draft recovery unit (Big Sheep Creek watershed).

The construction and operation of dams and diversions for agriculture have contributed to the decline of bull trout populations. Barriers have been constructed in Big Sheep Creek, Little Sheep Creek, and McCully Creek; all of these diversions lack fish passage facilities. The diversion at McCully Creek has effectively isolated bull trout local populations since the 1880's. Unscreened or inadequately screened irrigation diversions may strand bull trout in canals, sometimes resulting in mortality. In addition, water withdrawals from streams for irrigation, particularly in late summer, exacerbate natural low-flow conditions and in some streams. When irrigation water is returned to streams and rivers, it carries sediment and nonpoint pollution from agricultural chemicals which degrade water quality. [See USFWS 2002n (pp. 21-28) for more details on threats to bull trout.]

*Core Area Status*

The Sheep Creek and Granite Creek core areas (the two core areas in Idaho) are at an "unknown risk" of extirpation.

**Jarbidge River (Interim Recovery Unit)**

The limiting factors for bull trout discussed here are specific to the Jarbidge River Distinct Population Segment and include a combination of historical and current human-induced and natural factors. These limiting factors include dams and diversions, increasing water temperatures, forest management practices, livestock grazing, transportation networks (road construction and maintenance), mining, residential development, fisheries management, isolation and habitat fragmentation, recreation, and random naturally-occurring events (*e.g.*, landslides and floods). [See USFWS 2004b (pp. 21-28) for more details on threats to bull trout.]

*Core Area Status*

The only core area in this unit, Jarbidge River, is at "high risk" of extirpation. Note: recent surveys show bull trout abundance is four times higher than determined in the 2004 draft Recovery Plan (Allen et al. 2010, p. 20).

**Climate Change**

Changes in hydrology and temperature caused by changing climate have the potential to negatively impact aquatic ecosystems in Idaho, with salmonid fishes being especially sensitive. Average annual temperature increases due to increased carbon dioxide are affecting snowpack, peak runoff, and base flows of streams and rivers (Mote et al. 2003, p. 45). Increases in water temperature may cause a shift in the thermal suitability of aquatic habitats (Poff et al. 2002, p. iii). For species that require colder water temperatures to survive and reproduce, warmer temperatures could lead to significant decreases in available suitable habitat. Increased frequency and severity of flood flows during winter can affect incubating eggs and alevins in the streambed and over-wintering juvenile fish. Eggs of fall spawning fish, such as bull trout, may suffer high levels of mortality when exposed to increased flood flows (ISAB 2007, p. iv).

# 2.4.6 Bull Trout Critical Habitat

## 2.4.6.1 Status of Bull Trout Critical Habitat in the Action Area

The Service published a final rule designating critical habitat for bull trout rangewide on October 18, 2010 (effective November 17, 2010). Figure 3, below, shows bull trout critical habitat within the action area. In Idaho, there are 8,771.6 stream miles of critical habitat and 170,217.4 lake or reservoir acres designated. Most of the critical habitat occurs on federal lands managed by the Forest Service or BLM. Across the action area, streams may provide spawning and rearing critical habitat or foraging, migrating, and overwintering (FMO) critical habitat, depending on site specific stream characteristics and local bull trout population life history expressions

**Coeur d'Alene River Basin Unit Critical Habitat Unit (CHU)**

Located in Kootenai, Shoshone, Benewah, Bonner, and Latah Counties in Idaho, the Coeur d'Alene River Basin CHU includes the entire Coeur d'Alene Lake basin in northern Idaho. A total of 821.5 km (510.5 mi) of streams and 12,606.9 ha (31,152.1 ac) of lake surface area are designated as critical habitat. There are no subunits within the Coeur d'Alene River Basin CHU. This unit provides spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit, for justification of why this CHU is designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 801-811).

**Clark Fork River Basin CHU**[13]

The Clark Fork River Basin CHU includes the northeastern corner of Washington (Pend Oreille County), the panhandle portion of northern Idaho (Boundary, Bonner, and Kootenai Counties), and most of western Montana (Lincoln, Flathead, Sanders, Lake, Mineral, Missoula, Powell, Lewis and Clark, Ravalli, Granite, and Deer Lodge Counties). This unit includes 12 CHSUs, organized primarily on the basis of major watersheds: Lake Pend Oreille, Pend Oreille River, and lower Priest River (Lake Pend Oreille); Priest Lakes and Upper Priest River (Priest Lakes); Lower Clark Fork River; Middle Clark Fork River; Upper Clark Fork River; Flathead Lake, Flathead River, and Headwater Lakes (Flathead); Swan River and Lakes (Swan); Hungry Horse Reservoir, South Fork Flathead River, and Headwater Lakes (South Fork Flathead); Bitterroot River; Blackfoot River; Clearwater River and Lakes; and Rock Creek. The Clark Fork River Basin CHU includes 5,356.0 km (3,328.1 mi) of streams and 119,620.1 ha (295,586.6 ac) of lakes and reservoirs designated as critical habitat. The subunits within this unit provide spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and subunits, and for justification of why this CHU, any CHSUs, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 827-913).

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[13] Includes Pend Oreille River core area in the Northeast Washington 2002 draft Recovery Unit, referenced in the bull trout baseline section 2.4.5.

**Kootenai River Basin CHU**

The Kootenai River Basin CHU is located in the northwestern corner of Montana and the northeastern tip of the Idaho panhandle and includes the Kootenai River watershed upstream and downstream of Libby Dam. The Kootenai River flows in a horseshoe configuration, entering the United States from British Columbia, Canada, and then traversing across northwest Montana and the northern Idaho panhandle before returning to British Columbia from Idaho where it eventually joins the upper Columbia River drainage. The Kootenai River Basin CHU includes two CHSUs: the downstream Kootenai River CHSU in Boundary County, Idaho, and Lincoln County, Montana, and the upstream Lake Koocanusa CHSU in Lincoln County, Montana. The entire Kootenai River Basin CHU includes 522.5 km (324.7 mi) of streams and 12,089.2 ha (29,873.0 ac) of lake and reservoir surface area designated as critical habitat. The subunits within this unit provide spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and subunits, and for justification of why this CHU, any CHSUs, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 815-820).

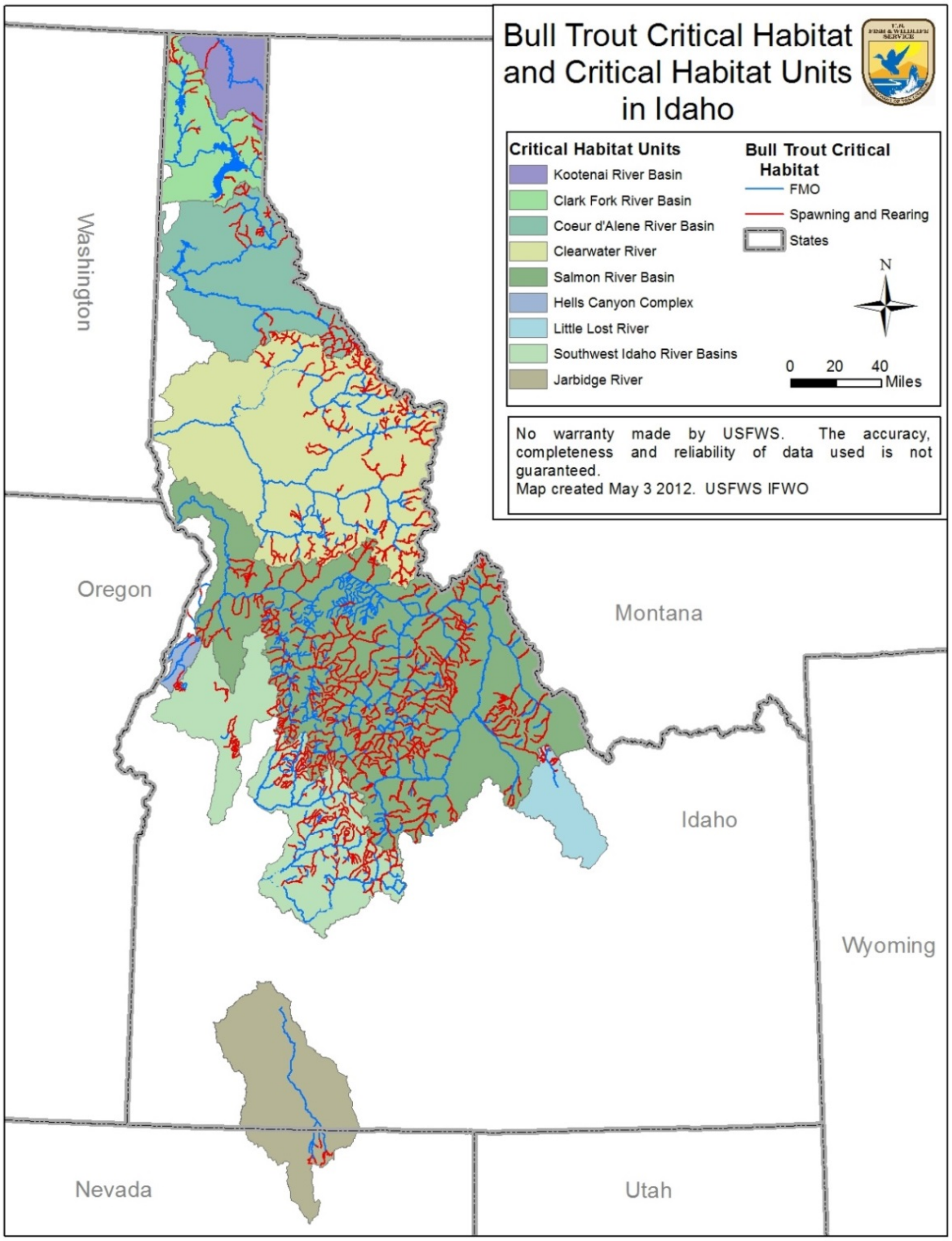


**Figure 2. Bull trout critical habitat in Idaho, by Critical Habitat Unit and type of designation (i.e., spawning and early rearing or foraging, migrating, and overwintering.**

### Clearwater River CHU

The Clearwater River CHU is located east of Lewiston, Idaho, and extends from the Snake River confluence at Lewiston on the west to headwaters in the Bitterroot Mountains along the Idaho–Montana border on the east in Nez Perce, Latah, Lewis, Clearwater, Idaho, and Shoshone Counties. In the Clearwater River CHU, 2,702.1 km (1,679.0 mi) of streams and 6,721.9 ha (16,610.1 ac) of lake and reservoir surface area are designated as critical habitat. The subunits within this unit provide spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and subunits, and for justification of why this CHU, any CHSUs, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 527-573).

### Salmon River Basin CHU

The Salmon River basin extends across central Idaho from the Snake River to the Montana–Idaho border. The Salmon River Basin CHU extends across portions of Adams, Blaine, Custer, Idaho, Lemhi, Nez Perce, and Valley Counties in Idaho. There are 10 CHSUs: Little-Lower Salmon River, Opal Lake, Lake Creek, South Fork Salmon River, Middle Salmon–Panther River, Middle Fork Salmon River, Middle Salmon Chamberlain River, Upper Salmon River, Lemhi River, and Pahsimeroi River. The Salmon River Basin CHU includes 7,376.5 km (4,583.5 mi) of streams and 1,683.8 ha (4,160.6 ac) of lakes and reservoirs designated as critical habitat. The subunits within this unit provide spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and subunits, and for justification of why this CHU, any CHSUs, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 671-791).

### Hells Canyon Complex Unit CHU

The Hells Canyon Complex is located in Adams County, Idaho, and Baker County, Oregon. This CHU contains 377.5 km (234.6 mi) of streams designated as critical habitat. The subunits within this unit provide spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and subunits, and for justification of why this CHU, CHSUs, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 505-510).

### Southwest Idaho River Basins CHU

The Southwest Idaho River Basins CHU is located in southwest Idaho in the following counties: Adams, Boise, Camas, Canyon, Elmore, Gem, Valley, and Washington. This unit includes eight CHSUs: Anderson Ranch, Arrowrock Reservoir, South Fork Payette River, Deadwood River, Middle Fork Payette River, North Fork Payette River, Squaw Creek, and Weiser River. The Southwest Idaho River Basins CHU includes approximately 2,150.0 km (1,335.9 mi) of streams and 4,310.5 ha (10,651.5 ac) of lake and reservoir surface area designated as critical habitat. The subunits within this unit provide spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and subunits and for justification of why this CHU, any CHSUs, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 613-669).

**Little Lost River CHU**

Located within Butte, Custer, and Lemhi Counties in east-central Idaho, near the town of Arco, Idaho, designated critical habitat in the Little Lost River CHU includes 89.2 km (55.4 mi) of streams. This unit provides spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and for justification of why this CHU, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 795-798).

**Sheep and Granite Creeks CHU**

This CHU is located within Adams and Idaho Counties in Idaho, approximately 21.0 km (13.0 mi) east of Riggins, Idaho. In the Sheep and Granite Creeks CHU, 47.9 km (29.7 mi) of streams are designated as critical habitat. This unit provides spawning, rearing, foraging, migratory, and overwintering habitat. For a detailed description of this unit and for justification of why this CHU, or in some cases individual waterbodies, are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 499-501).

**Jarbidge River CHU**

The Jarbidge River CHU encompasses the Jarbidge and Bruneau River basins, which drain into the Snake River within C.J. Strike Reservoir upstream of Grand View, Idaho. The Jarbidge River CHU is located approximately 70 miles north of Elko within Owyhee County in southwestern Idaho and Elko County in northeastern Nevada. The Jarbidge River CHU includes 245.2 km (152.4 mi) of streams designated as critical habitat. The Jarbidge River CHU contains six local populations of resident and migratory bull trout and provides spawning, rearing, foraging, migratory, connecting, and overwintering habitat. For a detailed description of this unit and for justification of why this CHU, any CHSUs, or in some cases individual waterbodies are designated as critical habitat, and for documentation of occupancy by bull trout, see USFWS 2010a (pp. 603-610).

## 2.4.6.2 Factors Affecting Bull Trout Critical Habitat in the Action Area

The factors affecting bull trout critical habitat are addressed in section 2.4.5.2 above.

**Climate Change**

An additional factor affecting bull trout critical habitat is global climate change which threatens bull trout throughout its range in the coterminous United States. Downscaled regional climate models for the Columbia River basin predict a general air temperature warming of 1.0 to 2.5 °C (1.8 to 4.5 °F) or more by 2050 (Rieman et al. 2007, p. 1552). This predicted temperature trend may have important effects on the regional distribution and local extent of habitats available to salmonids (Rieman et al. 2007, p. 1552), although the relationship between changes in air temperature and water temperature are not well understood. The optimal temperatures for bull trout appear to be substantially lower than those for other salmonids (Rieman et al. 2007, p. 1553). Coldwater fish do not physically adapt well to thermal increases (McCullough et al. 2009, pp. 96–101). Instead, they are more likely to change their behavior, alter the timing of certain behaviors, experience increased physical and biochemical stress, and exhibit reduced growth and survival (McCullough et al. 2009, pp. 98–100). Bull trout spawning and initial rearing areas are currently largely constrained by low fall and winter water temperatures, and define the spatial structuring of local populations or habitat patches across larger river basins; habitat patches represent networks of thermally suitable habitat that may lie in adjacent watersheds and are disconnected (or fragmented) by seasonally unsuitable habitat or by actual physical barriers (Rieman et al. 2007, p. 1553). With a warming climate, thermally suitable bull trout spawning and rearing areas are predicted to shrink during warm seasons, in some cases very dramatically, becoming even more isolated from one another under moderate climate change scenarios (Rieman et al. 2007, pp. 1558–1562; Porter and Nelitz 2009, pp. 5–7). Climate change will likely interact with other stressors, such as habitat loss and fragmentation (Rieman et al. 2007, pp. 1558–1560; Porter and Nelitz 2009, p. 3); invasions of nonnative fish (Rahel et al. 2008, pp. 552–553); diseases and parasites (McCullough et al. 2009, p. 104); predators and competitors (McMahon et al. 2007, pp. 1313–1323; Rahel et al. 2008, pp. 552–553); and flow alteration (McCullough et al. 2009, pp. 106–108), rendering some current spawning, rearing, and migratory habitats marginal or wholly unsuitable. For example, introduced congeneric populations of brook trout are widely distributed throughout the range of bull trout. McMahon et al. (2007, p. 1320) demonstrated the presence of brook trout has a marked negative effect on bull trout, an effect that is magnified at higher water temperatures (16–20 °C (60–68 °F)). Changes and complex interactions are difficult to predict at a spatial scale relevant to bull trout conservation efforts, and key gaps exist in our understanding of whether bull trout (and other coldwater fishes) can behaviorally adapt to climate change.

However, we predict that over a period of decades, climate change may directly threaten the integrity of the essential physical or biological features described in PCEs 1, 2, 3, 5, 7, 8 and 9.

## 2.4.7 Kootenai River White Sturgeon

### 2.4.7.1 Status of the Kootenai River White Sturgeon in the Action Area

See Section 2.3.7 above for a discussion of the status of the Kootenai River white sturgeon in the action area.

### 2.4.7.2 Factors Affecting the Kootenai River White Sturgeon in the Action Area

At the time of listing, the significant modifications to the natural hydrograph in the Kootenai River caused by flow regulation at Libby Dam was considered the primary reason for the Kootenai River white sturgeon's continuing lack of recruitment and declining numbers (59 FR 45996). The 2011 5–year status review (USFWS 2011) indicates that additional information has been collected since the time of listing pointing to a second survival bottleneck related to lack of nutrients and food for larval and age 1 sturgeon. Information has also been collected on the presumed presence of rocky substrates in the current spawning reach (i.e., the meander reach) (USFWS 2011, p. 16). These constraining factors as well as Libby Dam construction and operation are discussed below.

See section 2.3.7.5 above for a discussion of the conservation needs of the Kootenai River white sturgeon.

## Libby Dam

### *Construction*

Libby Dam was authorized for hydropower, flood control, and other benefits by Public Law 516, Flood Control Act of May 17, 1950, substantially in accordance with the report of the Chief of Engineers dated June 28, 1949 (Chief's Report) as contained in the House Document No. 531, 81st Congress, 2nd session. The Corps began construction of Libby Dam in 1966 and completed construction in 1973. Commercial power generation began in 1975. Libby Dam is 422 ft tall and has three types of outlets: (1) three sluiceways; five penstock intakes, three of which are currently inoperable; and (3) a gated spillway. The crest of Libby Dam is 3,055 ft long, and the widths at the crest and base are 54 ft and 310 ft, respectively. A selective withdrawal system was installed on Libby Dam in 1972 to control water temperatures in the dam discharge by selecting various water strata in the reservoir forebay.

Koocanusa Reservoir (known also as Lake Koocanusa or Libby Reservoir) is a 90-mile-long storage reservoir (42 miles extend into Canada) with a surface area of 46,500 acres at full pool. The reservoir has a usable storage of approximately 4,930,000 acre-feet and gross storage of 5,890,000 acre-feet.

The authorized purpose of Libby Dam is to provide power, flood control, and navigation and other benefits. With the five units currently installed, the electrical generation capacity is 525,000 kilowatts. The maximum discharge with all five units in operations is about 26,000 cfs. The surface elevation of Koocanusa Reservoir ranges from 2,287 feet to 2,459 feet at full pool. The spillway crest elevation is 2,405 feet.

### *Operations*

Presently, Libby Dam operations are dictated by a combination of power production, flood control, recreation, and special operations for the recovery of ESA-listed species, including the Kootenai sturgeon, bull trout, and salmon in the mid-and lower Columbia River.

The Corps currently manages Libby Dam operations not to volitionally exceed 1,764 mean sea level at Bonners Ferry, the flood stage designated by the National Weather Service. In accordance with the NMFS biological opinion, the Corps manages Libby Dam to refill Lake Koocanusa to elevation 2459 feet (full pool) by July 1, when possible (NMFS 2000, p. 3-2).

The Service's 1995 Federal Columbia River Power System (FCRPS) biological opinion recommended a flow regime that approached average annual pre-dam conditions, and would result in a pattern more closely resembling the pre-dam hydrograph (Figure 3) (USFWS 1995, pp. 6-10). The Service's 2000 FCRPS opinion and 2006 opinion on Libby Dam continued this recommendation. However, the actual volume of these augmented freshets has been relatively insignificant when compared to the magnitude of the natural pre-dam freshet.

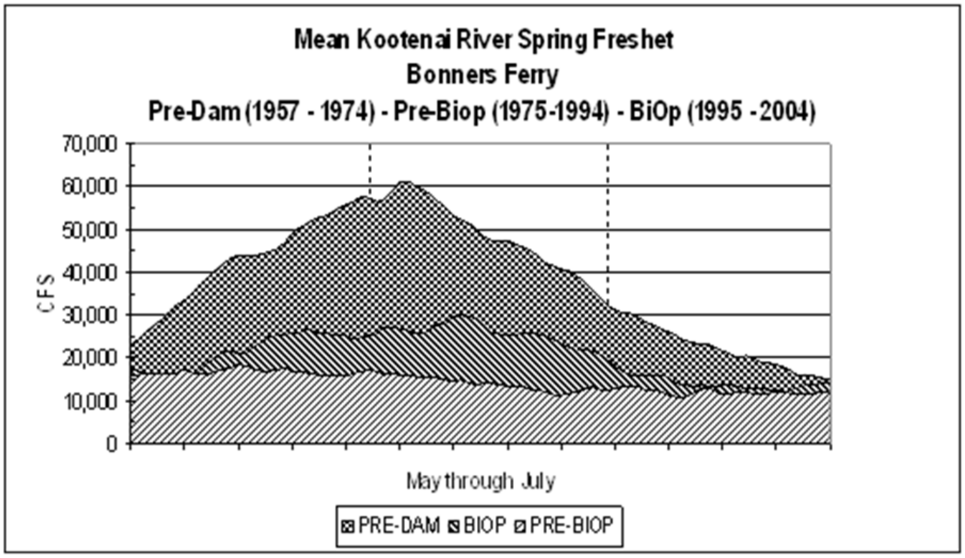


**Figure 3. Mean seasonal (May through July) hydrograph (calculated; Bonners Ferry) for pre-dam (1957 – 1974), pre-biological opinion (BiOp) (1975-1994), and BiOp (1995-2004).**

The Service's 2000 FCRPS opinion and 2006 opinion on Libby Dam included RPA's that recommended the implementation of Variable-Flow Flood Control (VARQ) operations at Libby Dam. In 2002, VARQ operations at Libby Dam began and continued on an "interim" basis until the completion of an Environmental Impact Statement (EIS) in April, 2006, and the signing of a Record of Decision (ROD) to implement VARQ operations in June, 2008.

The Service's 2006 opinion on Libby Dam also recommended that Libby Dam operations provide for minimum tiered volumes of water, based on the seasonal water supply, for augmentation of Kootenai River flows during periods of sturgeon spawning and early life stage development. Less volume is dedicated for sturgeon flow augmentation in years of lower water supply. Measurement of sturgeon volumes excludes the 4,000 cfs minimum flow releases from the dam.

*Northwest Power and Conservation Council Proposed Libby Operational Changes*

In its 2000 Columbia River Basin Fish and Wildlife Program, the first revision of the program since 1995, the Northwest Power and Conservation Council (Council) committed to revise the 1995 program's recommendations regarding mainstem Columbia and Snake River dam operations in a separate rulemaking. That rulemaking commenced in 2001. On April 8, 2003, the Council adopted the new mainstem amendments which included operations of these projects. These amendments are advisory and call for the following at Libby Dam:

- Continue to implement the VARQ flood control operations and implement Integrated Rule Curve operations as recommended by Montana Fish, Wildlife & Parks.
- With regard to operations to benefit Kootenai sturgeon, the Council recommended a refinement to operations in the 2000 FCRPS biological opinion that specify a "tiered" strategy for flow augmentation from Libby Dam to simulate a natural spring freshet.

- Refill should be a high priority for spring operations so that the reservoirs have the maximum amount of water available during the summer.
- Implement an experiment to evaluate the following interim summer operation:
  - Summer drafting limits at Libby Dam should be 10 feet from full pool by the end of September in all years except during droughts when the draft could be increased to 20 feet.
- Draft Koocanusa Reservoir as stable or "flat" weekly average outflows from July through September, resulting in reduced drafting compared to the NMFS FCRPS biological opinion.

**Kootenay Lake and Backwater Effect**

Corra Linn Dam located downstream on the Kootenay River, at the outlet of Kootenay Lake, in British Columbia, controls lake level for much of the year with the notable exception occurring during periods of high flows, such as during the peak spring runoff season. During the spring freshet, Grohman Narrows (RM 23), a natural constriction upstream from the dam near Nelson, British Columbia regulates flows out of the lake. Kootenay Lake levels are managed in accordance with the International Joint Commission (IJC) Order of 1938 that regulates allowable maximum lake elevations throughout the year. During certain high flow periods when Grohman Narrows determines the lake elevation, Corra Linn Dam passes inflow in order to maximize the flows through Grohman Narrows. Regulation of lake inflows by Libby Dam and Duncan Dam (on the Duncan River flowing into the north arm of the lake) maintains Kootenay Lake levels generally lower during the spring compared to pre-dam conditions.

Historically, during spring freshets, water from Kootenay Lake backed up as far as Bonners Ferry and at times further upstream (Barton 2004, p. 4). However, since hydropower and flood control operations began at Corra Linn and Libby Dams, the extent of this "backwater effect" has been reduced an average of over 7 feet during the spring freshet (i.e. water from Kootenay Lake currently extends further downstream than historically) (Barton 2004, p. 5).

**Survival Bottlenecks**

At the time of the 1994 listing determination, the primary cause of recruitment failure was identified as the suffocation of fertilized eggs as a result of spawning taking place over sand and silt substrates in the meander reach of the Kootenai River. This threat remains. However, at that time sturgeon managers believed the sand and silt was covering rocky substrates that had only become inundated since the construction and operation of Libby Dam. The view that increased flows would flush away the sand and silt and expose the underlying rocky substrates is reflected in the Service's 1995 and 2000 FCRPS biological opinions, the 1999 recovery plan, and the 2001 critical habitat designation. Subsequent coring and other data from the meander reach revealed that lacustrine clays lie underneath the sand and silt in the meander reach, indicating that the reach has always been comprised of substrates atypical for successful white sturgeon spawning and incubation (Barton 2004). A few isolated pockets of gravel were identified at the mouths of Deep Creek and Myrtle Creek. It is unlikely that these areas of gravel were sufficient to sustain the entire original population of Kootenai sturgeon (USFWS 2011, p. 13).

The overall conclusion from the substrate data and the historical information is that it's likely at least a portion of the Kootenai sturgeon population spawned in the canyon reach of the Kootenai River, most likely in the vicinity of Kootenai Falls. However, this new information does not address what actions would be necessary, or if it is even possible to restore this migration and spawning behavior in the Kootenai River white sturgeon. The new information indicates that the earlier view that "flushing flows" were the primary action needed to restore recruitment in the Kootenai River white sturgeon were population incorrect (USFWS 2011, p. 13).

More recently, sturgeon managers are hypothesizing that Kootenai River white sturgeon are experiencing a second survival bottleneck at the larval-to-age 2 state because sturgeon recapture data indicates that hatchery origin Kootenai River white sturgeon released at <9.86 inches survive at far lower rates than those released at larger sizes (Justice et al. 2009). Further, since 2005, sturgeon managers have released either fertilized eggs or free-embryos into reaches of the Kootenai River that have more suitable rocky substrates. Annually, over one million fertilized eggs or free-embryos are released, yet to date these experimental releases have not produced a detected increase in captured unmarked juvenile Kootenai River white sturgeon (Rust 2010). It is generally thought that the cause of this bottleneck is nutrient/food related, in that there is an insufficient food supply in the Kootenai River for larval and age-1 sturgeon.

Beginning in 2008, U.S. Geological Survey crews have been conducting surveys and inventories of the Kootenai Basin and have found that in the Kootenai River, there is very little zooplankton and macroinvertebrate production, relative to abundances in Kootenay Lake (Parker, pers. comm. 2010). Although modest efforts at nutrient restoration in the Kootenai River are ongoing, they appear to be insufficient.

The Kootenai Tribe of Idaho (KTOI) is in the planning phase of the Kootenai River Ecosystem Restoration Project, which involves actions specifically targeted at remedying the lack of nutrients and food available for Kootenai sturgeon (KTOI 2009). Reconnecting floodplains, restoring side channels, restoring kokanee populations, and restoring riparian functions in the Kootenai basin are all included in the planned project and, if successfully implemented, are anticipated to increase the primary productivity in the Kootenai River. Whether this will be sufficient to support a self-sustaining population of the Kootenai River sturgeon remains to be seen.

Additionally, the Corps in partnership with the KTOI, are conducting a feasibility study under Section 1135 of the Water Resources Development Act to evaluate habitat restoration opportunities in the Kootenai River. Restoration measures specific to restoring suitable spawning and early life stage habitats to address the primary bottleneck for the reproduction and survival of the species and avert the potential near-term extinction of the species are being considered.

### Other Factors Affecting the Sturgeon's Environment within the Action Area

Beginning in the early 1900's to 1961, in order to provide a measure of protection from spring floods, a series of dikes were constructed along the Kootenai River (below Libby Dam) and its tributaries. Other factors affecting the Kootenai River white sturgeon within the action area include floodplain development, contaminant runoff from mining activities, over-harvest, municipal water use, livestock grazing, and timber harvest as described in NPCC 2005, p. 110.

## Climate Change

Global Climate Models (GCMs) project air temperatures in the western U.S (including the Kootenai River area) to further increase by 1 to 3 °C (1.8 to 5.4 °F) by mid-twenty-first century (Rieman and Isaak 2010, p. 4). Dalton et al. (2013) report that increasing air temperatures and changes in precipitation from global warming will alter streamflow magnitude and timing, water temperatures, and water quality with hydrologic impacts varying by the type of watershed. "Snow-dominant watersheds are projected to shift toward mixed rain-snow conditions, resulting in earlier and reduced spring peak flow, increased winter flow, and reduced late-summer flow; mixed rain-snow watersheds are projected to shift toward rain-dominant conditions; and rain-dominant watersheds could experience higher winter streamflows if winter precipitation increases, but little change in streamflow timing" (Dalton et al. 2013, p. xxiii).

The changes and impacts described by Dalton et al. (2013) are evident in the Kootenai River basin. Data from stream flow gauges indicate that spring runoff is occurring between 15 and greater than 20 days earlier compared to the mid twentieth century (Rieman and Isaak 2010, p. 7). These changes in flow have been attributed to interactions between increasing temperatures (earlier spring snowmelt) and declining snowpack. The Alder et al. (2014) predict increasing precipitation (as rain) and decreasing snowpack for the Kootenai River basin. Water temperatures in the Kootenai River are also expected to increase. An analysis of the NorWeST stream temperature data (https://www.sciencebase.gov/flexviewer/NorWeST/) showed that in the braided and meander (spawning) reaches of the Kootenai River mean August stream temperatures will increase from 16.7°C (62.1°F) currently to 18.2°C (64.8°F) in 2040 and 19.3°C (66.76°F) in 2080.

For the Columbia River white sturgeon population, Jones et al. (2011, pp. 82-83) concluded that while "the thermal tolerance range of adult white sturgeon may be quite broad, disease and parasites may be more prevalent in warmer waters, and several studies have documented some temperature requirements for spawning and egg incubation and survival." Parsley et al. (1993) reported that spawning of the Columbia River white sturgeon typically occurs from April through July with water temperatures between 10 – 18°C (50 – 64°F); most spawning occurring at 14°C (57°F). Egg mortality increases when incubation reaches 18°C (64°F) and total egg mortality occurs at 68°F (Wang et al. 1985, p. 48). The Kootenai River white sturgeon also spawn in May or June; however, water temperatures are much cooler, about 8.5 - 12.5°C (47.3-54°F) (Paragamian et al. 2001; Paragamian and Wakkinen 2002). Eggs incubated at cooler than optimal temperatures develop normally but take longer to hatch (Wang et al. 1985, p. 48). In addition to water temperature, climate change may also cause reduced discharge and water velocities in the Kootenai River. Paragamian (2012) reports that for optimum white sturgeon spawning, discharge in the Kootenai River should be above 630 cubic meters per second (cms) (22, 248 cfs). Given the importance of both water temperature and discharge for successful sturgeon spawning and recruitment, any increase in temperature or decrease in discharge due to climate change would adversely affect the sturgeon and its habitat.

### 2.4.8 Kootenai River White Sturgeon Critical Habitat

#### 2.4.8.1 Status of Kootenai River White Sturgeon Critical Habitat in the Action Area

See the *Status of Kootenai River White Sturgeon Critical Habitat* section (2.3.8) above.

#### 2.4.8.2 Factors Affecting Kootenai River White Sturgeon Critical Habitat in the Action Area

As the same factors are affecting both Kootenai River white sturgeon and sturgeon critical habitat in the action area, see the *Factors Affecting the Kootenai River White Sturgeon* section above for details on these factors, including the factor related to climate change.

A warming climate as described above for bull trout and the sturgeon may also significantly impact sturgeon critical habitat, specifically PCE 3 which requires that during the spawning season of May through June, water temperatures between 8.5 and 12 °C (47.3 and 53.6 °F), with no more than a 2.1 °C (3.6 °F) fluctuation in temperature within a 24- hour period, as measured at Bonners Ferry.

## 2.5 Effects of the Proposed Action

Effects of the action considers the direct and indirect effects of an action on the listed species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action. These effects are considered along with the environmental baseline and the predicted cumulative effects to determine the overall effects to the species. Direct effects are defined as those that result from the proposed action and directly or immediately impact the species or its habitat. Indirect effects are those that are caused by, or will result from, the proposed action and are later in time, but still reasonably certain to occur. An interrelated activity is an activity that is part of the proposed action and depends on the proposed action for its justification. An interdependent activity is an activity that has no independent utility apart from the action under consultation.

### 2.5.1. Foundation of Analyses

ESA section 7(a)(2) states that each Federal agency shall, in consultation with the Secretary, insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of a listed species or result in the destruction or adverse modification of designated critical habitat. A biological assessment (Assessment) is prepared to analyze the likely effects of the action on the species or habitat based on the best available information including that related to biological studies, review of literature, and the views of species experts.

For the EPA proposed action of approving Idaho’s water quality standards, there are no direct effects of the proposed approval to listed species or critical habitat, that is, approving the standards in and of themselves will not change the environmental baseline or directly affect listed species or critical habitat. However, there are indirect effects of approving the standards because the approval sets the context for implementation of the standards via CWA section

303(d) evaluations and listings, and development of TMDLs, NPDES permits, and water quality management plans designed to meet the standards over time. As a consequence, the analysis of effects to listed species and critical habitat in this document is addressed in a summary context, rather than categorized as direct or indirect effects of the proposed action.

The following analysis also relies on Service national policy regarding best available scientific and commercial data; see page 1-6 of the *Endangered Species Consultation Handbook* (USFWS and NMFS 1998). Under that policy, in the absence or uncertainty of relevant data needed to complete the analysis of effects of the action, where significant data gaps exist there are two options: (1) extend or postpone the consultation until sufficient information is developed for a more complete analysis; or (2) develop the biological opinion with available information giving the benefit of the doubt to the species. In this case option 2 was applied.

## 2.5.1.1 Comparison of 2004 and 2015 Opinions

The Service completed a draft opinion on the proposed action in 2004 which, as described in the *Consultation History* section of this Opinion, was never finalized. In that 2004 draft opinion, the Service disagreed with most of EPA's NLAA determinations and found that in most cases an LAA finding was warranted. There were many more LAA findings in the 2004 opinion than in the current Opinion. One of the primary reasons for this difference is that in the 2004 draft opinion we relied heavily on the *Common Factors that Affect Toxicity of Criteria to Listed Species* (the *Common Factors* described below in section 2.5.1.5) in evaluating the effects of the proposed action on listed species and critical habitat due in large part to the absence of applicable, primary research results. In contrast, in the current Opinion with more than a decade of additional research to draw on, we are able to rely more on related species-specific (unfortunately, not listed species-specific) analyses using the best available toxicological data to evaluate potential effects and make our findings. Although we refer to and use the *Common Factors* assessment in some sections of this Opinion, they were typically considered as a component of, not the primary basis for, our findings.

The number of jeopardy and adverse modification determinations also differs between our 2004 draft Opinion and this Opinion because we have acquired updated species information since 2004 that warrants those changes. For example, at the time of drafting the 2004 Opinion, available information indicated that the Snake River physa had a very limited distribution and very low population numbers (<50 individuals). In other words, information at the time indicated that the species was at a very high risk of extirpation. We now know that the Snake River physa is more widely distributed and has higher population numbers located in strongholds such as the Minidoka reach of the Snake River. The current distribution of this species has also expanded to include the Snake River near Ontario, Oregon.

## 2.5.1.2 Development of Water Quality Criteria by EPA

Detailed information on the development of water quality criteria are presented in Stephan et al.'s (1985a) "guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses." Protection of aquatic organisms and their uses in turn was defined as "prevention of unacceptable long-term and short-term effects on (1) commercially, recreationally, and other important species and (2) (a) fish and benthic invertebrate assemblages in rivers and streams, and (b) fish, benthic invertebrate, and zooplankton assemblages in lakes, reservoirs, estuaries, and oceans."

The 1985 guidelines rely on many fundamental assumptions, judgements, and procedures that in turn are inherent to their degree of protectiveness for listed species. Among these assumptions were that:

(1) chemicals will have similar effects to organisms in laboratory and field settings;

(2) It is acceptable to extrapolate from compilations of severely toxic effects from short-term, "acute" tests to less severe effects in long-term, "chronic" exposures.

(3) If 95 percent of the species in acceptable datasets were protected, that would be sufficient to protect aquatic ecosystems in general;

(3) It is not necessary to protect all of the species all of the time, in order to sufficiently protect aquatic communities and socially valued species. Aquatic organisms may have ecologically redundant functions in communities. The loss of some species might not be important if other species would fill the same ecological function. Further, aquatic ecosystems have resiliency and can recover from occasional criteria exceedances (Stephan et al., 1985a; Stephan 1985b, entire)

These and more assumptions, judgments and procedures from the criteria development guidelines were evaluated in some detail in NMFS (2014a, pp. 61-117). NMFS's evaluation is largely salient to the species under review in this opinion as well. While some analyses in NMFS (2014a, pp. 61-117) cover similar ground as the following "Common Factors" section of the present opinion, for brevity, most are not repeated in the present opinion since they are available online in the NMFS review. While the NMFS review was generally not unfavorable, scenarios were identified which could result in insufficient protection to listed species or habitats. Thus a conservative view is appropriate when interpreting the specific literature on species and substances later in this opinion.

## 2.5.1.3 Assumptions in Effects Analyses

Because this action and subsequent analyses are focused on assessing the protectiveness of aquatic life criteria for toxic substances, the Assessment and this opinion analyze the protectiveness of the aquatic life criteria. As most of the criteria are expressed in two parts, with an acute criterion that is intended to protect against short-term pulses of contaminants, and a chronic criterion that is intended to protect against long-term or indefinite exposures, the evaluations of specific criteria follow that short-term, long-term structure. Acute criteria were evaluated through comparisons of criteria concentrations with reports of effects to species of interest resulting from short-term exposures (96-hours or less). Similarly, chronic criteria were evaluated through comparisons of criteria concentrations with reports of effects to species of interest resulting from longer-term exposures (>96-hours).

Because the effects analyses analyze the protectiveness of regulatory criteria, the analyses effectively address the question, "what if" concentrations were at criteria concentrations for the allowed durations. This has led to commenters suggesting that the protectiveness of criteria should not be evaluated by comparing effects concentrations to criteria concentrations. Rather, commenters argued that the comparisons be made to existing conditions in the action area, rather than concentrations that could be authorized by criteria, but in most cases are not actualized. Under this reasoning, if the existing concentrations of the proposed substances are suitable for the listed species and habitats, then the regulatory criteria would by definition be suitable.

Acknowledging that such a tactic would result in identifying fewer likely adverse effects than would evaluating the criteria directly, the Service believes that such an approach to defining the action would be inconsistent with the salient parts of the definition of an action which describes programs or permits authorized by the action agency that directly or indirectly cause modifications to the land, water, or air. [50 CFR §402.02]. Therefore, in most cases we evaluate the potential effects of the action as authorized. The exceptions are certain cases discussed later where the authorization in the present action to indirectly allow discharges of certain manufactured pesticides and organic chemicals is countermanded by other regulatory actions such as banning or restricting pesticides under the Federal Insecticide Fungicide and Rodenticide Act (FIFRA) or the Toxic Substances Control Act (TSCA).

The analyses in the Assessment for the protectiveness of numeric criteria similarly assumed that listed species are exposed to concentrations of pollutants at the water quality criteria as proposed to be authorized, which may be higher or lower than conditions which currently exist in Idaho's waters (EPA 1999b, p. 120). EPA made this assumption because the aquatic life criteria will be applied statewide without deference to species' ranges, and because the purpose of the consultation is to evaluate the protectiveness of the criteria. Therefore, our analysis of effects was also based on this assumption.

## 2.5.1.4 Structure, Organization, and Methods of the Effect Analyses

The effect analyses for the proposed action are complex. For the purposes of the *Effects of the Action* section of this Opinion, the analyses were separated into two parts. In the first part, we present the "common factors" (see section 2.5.1.5 below) that may affect the toxicity of each of the 11 inorganic substances considered herein.

The second part of the analyses consists of a narrative that discusses the effects of each standard for each inorganic metal to each listed species/critical habitat considered in this Opinion. These analysis could result in three potential effect outcomes for each standard for each inorganic metal: (1) no effect; (2) not likely to adversely affect; and (3) likely to adversely affect. For each inorganic toxic metal subject to a standard, a potential outcome is possible at both an acute (brief/temporary in nature) and chronic (recurring/permanent in nature) exposure level.

EPA's proposed approval of Idaho water quality standards also addresses 11 organic compounds. Of these, nine are pesticides (endosulfan, aldrin, dieldrin, chlordane, DDT, endrin, heptachlor, lindane, and toxaphene) and two are industrial chemicals (PCBs, PCPs) that are no longer being released, are banned, or are very restricted in use. For these reasons, the Service finds they are unlikely to be found in the environment in concentrations sufficient to cause adverse effects to listed species or critical habitat. On that basis, the Service concurs with EPA's finding that this aspect of the proposed action is not likely to adversely affect listed species or critical habitat. NMFS (2014a) provided similar rationale and findings in their Opinion for the same organic compounds.

## 2.5.1.5 Common Factors that Affect Toxicity of Criteria to Listed Species

Certain factors, such as the effects of water quality parameters on toxicity, are common to the analyses for all of the proposed water quality criteria. Most of these common factors relate to information not considered - or not available to be considered - by EPA when it completed its assessment to determine the criteria. Rather than repeat the same analysis for each chemical, for each species, the Service grouped the common factors into the following 8 categories:

1. Surrogate sufficiency
2. The effects of chemical mixtures (i.e., additive, less than additive, etc.)
3. Sediments and multiple routes of exposure
4. Dietary effects or bioaccumulation effects on fish and wildlife species
5. Use of a low-end cap in the equation for hardness-dependent metals
6. Adjustments to the calculated criteria for toxic metals
7. Use of conversion factors and translators to derive criteria for toxic metals
8. Choice/use of endpoints

**1. Surrogate Sufficiency**

Comparative toxicity testing of chemicals usually uses a relatively small group of standard laboratory organisms that are readily cultured and tested in controlled laboratory settings. Direct toxicity testing of listed species is infrequent because of technical, ethical, and administrative challenges. Technical challenges include culture and handling difficulties; listed species may not thrive in laboratory settings, and substantial effort to develop culture and testing methods may be needed. The capturing and killing of listed organisms in order to determine their risks of being harmed by contaminants may be ethically unjustifiable, and administrative permissions to do so may not be forthcoming.

Some direct testing of listed species has been conducted, usually by obtaining culture organisms of the same taxonomic species from a non-listed DPS, conservation hatchery programs, or field collections from locally abundant populations (e.g., Ingersoll and Mebane 2014; Kiser et al., 2010; Besser et al., 2005a, 2009); Dwyer et al. 2005; and Hansen et. al 2002c). However, in most cases, some sort of extrapolation of effects from similar, non-listed species was needed as a surrogate for effects to the listed species. Because different studies usually obtained different results, we developed a rough priority scheme for evaluating the relevance of different “surrogate” study results to the listed species of interest.

- Taxonomic similarity: We generally assume that all things otherwise being equal, closely related species would have similar sensitivities to the same contaminants. This assumption has been a long-standing concept in risk assessment, such as the practice in EPA’s criteria development guidelines to average species sensitivities within a genus for criteria development (Stephan et al. 1985a). This approach and concept has been expanded with some success to make extrapolations for the effects of acute criteria concentrations across chemicals and less-closely related taxa. For instance, for chemicals with similar modes of action, Raimondo et al. (2010) developed extrapolation models that were usually accurate within a factor of five for species within a family.

- Similarity of species traits: Species that may not be closely related taxonomically, may share similar traits that affect their risks to chemicals. These include traits related to similar life histories, intrinsic sensitivity, and factors related to population sensitivity (Rubach et al. 2011).

For example, the endangered Snake River physa snail is far too rare to practically or ethically use in destructive toxicity testing, yet the closely related snail *Physa gyrina* is common in ponds and has been used in toxicity testing. Toxicity test data with *Physa gyrina* would be assumed directly relevant to the endangered Snake River physa snail. However, taxonomic closeness may not always be the only factor considered in selecting surrogate species. For example, different sturgeon species have different sensitivities to chemicals, and in some cases a rainbow trout would make a better surrogate for a sturgeon in the genus Acipenser than would a much closer taxonomic relative within the family Acipenseridae (Dwyer et al. 2005).

In some cases, no reasonably comparable data for a surrogate species may exist for a chemical. In these cases, crude assumptions may need to be made that relative species-sensitivities are similar across chemicals. For example, a species that is sensitive to the insecticide diazinon might also be sensitive to ammonia or nickel. If this were the case, and if sensitivities to chemicals are correlated, then these interspecies-correlations could be used to estimate toxicity of untested chemicals and species. Interspecies correlation estimates (ICE) have been formalized through a modeling framework to contrast the possible relative acute sensitivity of listed species to “standard” surrogate species such as the rainbow trout to untested chemicals (Raimondo et al. 2013). This ICE modeling approach has obvious limitations and uncertainties, such as the assumption that relative sensitivities are maintained across chemicals with different modes of toxic actions, and that correlations determined from short-term, acute toxicity tests can be extrapolated to long-term indefinite exposures. While such assumptions may not be correct, the approach does generate numbers, which in the absence of data, might be all that is available for completing effect analyses for some species and chemical combinations. For example, the ICE model outputs for acutely toxic effect concentrations of 35 and 62 µg/L of a generic chemical to the rainbow trout resulted in a corresponding toxic effect concentration estimate for the genus *Acipenser* (to which the Kootenai River white sturgeon belongs) of 21 and 40 µg/L of that chemical.

Because the ICE modeling approach of Raimondo et al. (2013) may represent the best available information in some instances, a limited evaluation of the ICE predictions was made. The evaluation used six data pairs where comparable effect data were on hand for both a surrogate species and a threatened or endangered species, and where an appropriate ICE model was available for the test pair (Table 4). The results showed considerable variability in predictions with ICE estimates ranging from over-predicting toxicity by up to 74 percent greater than actual toxicity to under-predicting toxicity by up to 240 percent. In this context, under-predicting toxicity means that the actual effects concentration was lower than the predicted effects concentration, and thus the substance was more toxic than predicted, and vice versa for over-predicted toxicity. The ICE predictions were considered “correct” in regard to the protectiveness of criteria in half the cases compared (Table 4).

**Table 4. Comparison of actual and Interspecies Correlation Estimate (ICE) predicted toxicities relative to the bull trout and the Kootenai River white sturgeon.**

| ESA Listed Species for which ICE predictions are made ("unknown") | Surrogate Species ("known") | Chemical | Endpoint | Actual Effect Concentration for Surrogate (µg/L) | Actual effect concentration for Listed Species (µg/L) | ICE Predicted effect Concentration for Listed Species (µg/L) | % Prediction Error | Relevant criterion value (µg/L) | Would ICE interpretation have led to a "correct" interpretation of criterion protectiveness for the endpoint? | Source for Actual Effects |
|---|---|---|---|---|---|---|---|---|---|---|
| Bull trout | Rainbow trout | Cd | 96-h LC50 (as SMAV) | 2.0 | 2.1 | 3.4 | -62% | 1.5 | No | (Mebane 2006) |
| Bull trout | Rainbow trout | Cu | 96-h LC50 (as SMAV) | 22.0 | 68.0 | 29 | 57% | 4.7 | Yes | (EPA 2007) |
| White sturgeon | Rainbow trout | Cd | EC10, 28-d exposure | 1.5 | 2.4 | 0.63 | 74% | 0.55 | Yes | (Wang et al. 2014a) |
| White sturgeon | Rainbow trout | Cu | EC10, 28-d exposure | 13.0 | 2.0 | 7 | -250% | 9 | No | (Wang et al. 2014a) |
| White sturgeon | Rainbow trout | Pb | EC10, 28-d exposure | 55.0 | 13.0 | 35 | -169% | 2.5 | Yes | (Wang et al. 2014a) |
| White sturgeon | Rainbow trout | Zn | NOEC, 28-d exposure | 135.0 | 181.0 | 96 | 47% | 118 | No | (Wang et al. 2014a) |

ICE predicted effects used the ICE “Endangered Species Module - Aquatic Species” available at http://www.epa.gov/ceampubl/fchain/webice/iceTNESpecies.html, accessed 29 December2014)

## 2. The Effects of Chemical Mixtures

In point or non-point pollution, chemicals occur together in mixtures, but criteria for those chemicals are developed in isolation, without regard to additive toxicity or other chemical or biological interactions. Whether the toxicity of chemicals in mixtures is likely greater or less than that expected of the same concentrations of the same chemicals singly is a complex and difficult problem. While long recognized, the “mixture toxicity” problem is far from being resolved. Even the terminology for describing mixture toxicity is dense and has been inconsistently used (e.g., Sprague 1970; Marking 1985; Borgert 2004; Vijver et al. 2010). One scheme for describing the toxicity of chemicals in mixtures is whether the substances show additive, less than additive, or more than additive toxicity. The latter terms are roughly similar to the terms “antagonism” and “synergism” that are commonly, but inconsistently used in the technical literature.

For both metals and organic contaminants that have similar mechanisms of toxicity (e.g., different metals, different chlorinated phenols), assuming chemical mixtures to have additive toxicity has been considered reasonable and is usually protective (Norwood et al. 2003; Meador 2006). This conclusion is in conflict with the way effluent limits are calculated for discharge of toxic chemicals into receiving water. Each projected effluent chemical concentration occurring during design flow is divided by its respective criterion, along with adjustments for variability and mixing zone allowances (EPA 1991). Thus, each substance would be allowed to reach one “concentration unit” and any given discharge or cleanup scenario would likely have several concentration units allowed, which is sometime referred to as cumulative criterion units.

Experimental approaches in the literature usually report “toxic units” (TUs) based on observed toxicity in single substance tests, rather than criterion units. In this “concentration addition” scheme, toxicity of different chemicals is additive if the concentrations and responses can be summed on the basis of TUs. For instance, assume for simplicity that cadmium is more toxic than copper to a species, with an EC50 of 4 µg/L for cadmium, and an EC50 of 8 µg/L for copper. Under this analysis, we will also refer to each single metal EC50 as a TU. The toxicity of mixtures could be estimated as follows:

4 µg/L Cd + 0 µg/L Cu = $\frac{4\ \mu\text{g/L}}{4\mu\text{g/L/TU}} + \frac{0\ \mu\text{g/L}}{8\mu\text{g/L/TU}}$ =1 TU, (obviously, for a single substance), *or*

2 µg/L Cd + 4 µg/L Cu = $\frac{2\mu\text{g/L}}{4\mu\text{g/L/TU}} + \frac{4\mu\text{g/L}}{8\mu\text{g/L/TU}}$ =0.5+0.5=1 TU (for two substances)

Using this approach, some studies have shown significant additive toxicity. For instance, Spehar and Fiandt (1986) exposed the rainbow trout and *Ceriodaphnia dubia* simultaneously to a mixture of five metals and arsenic, each at their acute CMC, which are intended to be protective. There were no survivors. In chronic tests, adverse effects were observed at mixture concentrations of one-half to one-third the approximate chronic toxicity threshold of fathead minnows and daphnids, respectively, suggesting that components of mixtures at or below no effect concentrations may contribute significantly to the toxicity of a mixture on a chronic basis (Spehar and Fiandt 1986).

A common outcome in metals mixture testing has been that metal combinations have been less toxic than the sum of their single-metal toxicities, i.e., show less than additive toxicity or are antagonistic (Finlayson and Verrue 1982; Hansen et al. 2002d; Norwood et al. 2003; Vijver et al. 2011; Mebane et al. 2012; Balistrieri and Mebane 2014). The other possibility, more than additive toxicity (also called synergistic effects) are rare with metals although it has been shown with pesticides (Norwood et al. 2003; Laetz et al. 2009).

The EPA's approach to the mixture toxicity problem in effluents, including effects of substances without numeric criteria or unmeasured substances, has been to recommend an integrated approach to toxics control (EPA 1991, 1994). The EPA has long recognized that numerical water quality criteria are an incomplete approach to protecting or restoring the integrity of water. A major part of EPA's strategy for measuring and controlling such potential issues has been through the concept of an integrated approach to toxics control, where meeting numerical criteria is but one of three elements. The other two elements are the concept of regulating whole effluents through whole- effluent toxicity (WET) testing or through biological monitoring of ambient waters that receive point or nonpoint discharges (EPA 1991, 1994). Because of assumptions that chemicals will inevitably occur in ambient waters in mixtures rather than occurring chemical by chemical in the fashion that criteria are developed, it is not possible to know all the potential contaminants of concern in effluents and receiving waters, let alone measure them, and it is not feasible to predict effects by chemical concentrations alone. Thus, the EPA developed procedures for testing the whole-toxicity of effluents and receiving waters, including procedures for identifying and reducing toxicity (e.g., Mount and Norberg-King 1983; Norberg-King 1989; Mount and Hockett 2000). In practice, some consideration of the potential for aggregate toxicity through WET testing is made by EPA for major permits that they administer in Idaho.

### 3. Sediments and Multiple Routes of Exposure

The water quality criteria under consultation were derived to protect against contaminant exposures in a single medium, the water column. However, chemical contamination of the environment typically occurs in multiple media, such as the water column, water-sediment interface, interstitial pore waters of sediments, periphyton (biofilms), and through the food web. Chemicals move between media, and environmental controls established for one medium, such as the water column, have an impact on other media(Reiley et al., 2003, pp. 41-42).

Aquatic and aquatic-dependent organisms that routinely ingest sediment while feeding or that live in or on sediments (e.g., aquatic snails and the white sturgeon) are subjected to an additional route of exposure to toxic chemicals not currently considered by the EPA in developing and promulgating water quality criteria for the protection of aquatic life. The Assessment (EPA 2000, pp. 1-2) states that the consideration of exposure to chemicals is limited to passage of dissolved constituents through the gills and does not include ingestion of pollutants. Exclusive use of water column criteria may underestimate the toxicity of an aquatic system by excluding ingestion of particulates and ingestion of prey that consume particulates as a pathway for toxic chemical exposure (EPA 2000, p. 18). Most organic and inorganic contaminants adsorb to organic particulates and settle out in sediments, so at sites with past or continuing discharges of contaminants into the water column, a repository and continuing source of exposure likely exists (Hoffman et al. 1995, p. 4). The Service has assumed that this additional route of exposure is likely to increase the adverse effects of each contaminant addressed in this Opinion on listed species and critical habitat.

The distribution of solutes in the pore water of sediments will adjust quickly to fluctuations in bottom water currents and oxygen concentrations, and consequently there can be rapid changes in the fluxes across the sediment-water interface (Sundby 1994, pp. 147-149). Although these pollutants may not be readily transferred to the water column, they are available for food-chain transfer through ingestion of sediment from benthic prey, sequestration by plants or epiphytes, or ingestion of sediment while feeding (Baudo and Muntau 1990, p. 6; Power and Chapman 1992, pp. 6-9;). Organic compounds are of particular concern in regard to accumulation in sediment. They generally have a long half-life and persist in the soils for an extended period of time. A good example is aldrin/dieldrin; residue from these compounds remain in the soil for a long duration. The half-life for aldrin is estimated to be between 2 to 5 years, depending on the composition of the soil, and more than 56 percent of the original weight of aldrin in the soil converts to dieldrin. The half-life for dieldrin varies depending on the rate at which it was used. At a rate of 0.6 kg/ha, the half-life is approximately 2.6 years, while at 9.0 kg/ha, the half-life is 12.5 years (Jorgenson 2001, p. 123).

A number of studies have also documented arsenic-contaminated diets having adverse effects on salmonids (Woodward et al. 1994, p. 51; Farag et al. 1994, p. 2021; Woodward et al. 1995, p. 1994, Hansen et al. 2004, p. 1902-1911). EPA's Assessment (2000, p. 19) states that the application of water column criteria is intended to protect water column organisms from exposure to metals from the water column. Little connection exists between the establishment of water column concentrations to protect against toxicity to aquatic organisms and the degree to which metals might accumulate in sediment and/or accumulate in benthic organisms that serve as prey for fish and other organisms (EPA 2000, p. 19).

### 4. Dietary Effects or Bioaccumulation Effects on Listed Fish and Wildlife Species

Bioaccumulation and biomagnification are two commonly used terms that are frequently confused in the environmental literature. Bioaccumulation refers to the simple presence of a chemical in a living organism, and biomagnification refers to the stepwise increase in contaminant residues in tissues from one trophic level to the next. Neither bioaccumulation or biomagnification alone indicate adverse effects to aquatic life; rather, only the biological responses to the chemicals or their metabolites are indicative of such effects. Still, chemicals that strongly biomagnify will result in greater exposure in higher trophic level animals, and biomagnifying chemicals are generally of heightened concern (Spacie et al. 1995). Among the inorganic contaminants, mercury appears to be unique in its capacity to consistently biomagnify across trophic levels. Biomagnification is a well-known property of certain organic contaminants. Persistent organic pollutants (POPs), including DDT, PCBs, heptachlor, pentachlorophenol, aldrin, dieldrin, and chlordane, are widely known to biomagnify. For example, mortalities and reproductive failures in fish and fish-eating birds were linked to unusually high concentrations of DDT or its metabolites in the fat of their prey. Although use and sale of many of the POPs has been restricted or canceled, POPs exhibit markedly long half-lives in the environment (Mattina et al. 1999, p. 2425). The degree of accumulation in an aquatic organism depends on its position in the food chain, on the availability and persistence of the contaminant in water, and especially on the physical-chemical properties of the contaminant (Spacie and Hamelink 1985, p. 495; EPA 2000, p. 19).

Some metals are essential micronutrients for aerobic life with many proteins requiring a metal co-factor for proper function, most notably iron, zinc, copper, selenium, and cobalt. All animals have the capacity to regulate their internal concentrations of essential trace metals, known as homeostasis. Freshwater animals upregulate to avoid deficiency by decreasing excretion in dilute waters when scarce, and downregulating to avoid toxicity by increasing excretion when abundant. Toxicity to essential trace elements results when the homeostatic mechanisms are overwhelmed by high concentrations (Wood 2011a, pp. 23-24). Exposure through the diet can result in adverse effects even with substances that do not biomagnify across trophic levels. Among inorganic contaminants, in addition to mercury which does biomagnify, arsenic and selenium have been implicated in causing dietary toxicity (Hansen et al. 2004; Janz et al. 2010; Erickson et al. 2011b). In high enough doses, other inorganic contaminants such as copper, nickel, lead, and zinc can cause toxicity in aquatic organisms. For instance, following a substantive review of the issue, Schlekat et al. (2005, p. 141) noted that while laboratory and field studies have documented adverse effects of metals in the diets of fish, amphibians, and invertebrates, they observed that "we found no studies that demonstrate adverse effects resulting from diet-borne metals in systems in which water quality criteria were apparently being met. However, this could be a reflection of poorly designed approaches or a lack of appropriate data rather than an indication that such effects are not possible" (Schlekat et al. 2005, p. 141).

### 5. Use of a Low-end Cap in the Equation for Hardness-dependent Metals

In the National Toxics Rule, EPA described and required minimum and maximum hardness values (25 mg/L and 400 mg/L of $CaCO_3$, respectively) to be used when calculating hardness-dependent freshwater metals criteria (EPA 2000, p. 21). Most of the data EPA used to develop the criteria formulas were in that hardness range and therefore, were most accurate when used in that context. Although most stream water quality in Idaho falls within that range of hardness values, there are some, such as the North Fork Payette and Upper Middle Fork Salmon that average below 25 mg/L of $CaCO_3$ (19 mg/L and 16 mg/L of $CaCO_3$, respectively). Toxicities of several contaminants addressed in this Opinion are hardness-dependent, with toxicity increasing with decreasing hardness. Using a hardness cap of 25 mg/L for all streams when some have lower hardness values will result in artificially elevated aquatic life criteria. From Appendix F of the Assessment (EPA 1999a), 5 of 82 (6 percent) mean hardness values reported for certain Idaho streams/reaches are < 20 mg $CaCO_3$/L and 54 of 82 (66 percent) minimums are < 20 mg $CaCO_3$/L. This means that for the streams/reaches reported in Appendix F of the Assessment, hardness values in 66 percent of listed reaches will fall below the cap at some point during the year and are likely to exhibit contaminant concentrations (such as for Cd) above levels observed to cause adverse effects to aquatic organisms. Five percent of the reported streams/reaches had mean hardness values below the cap and thus are likely to frequently exhibit contaminant concentrations above levels observed to cause adverse effects to aquatic organisms. For calculating effluent limits for National Pollution Discharge Elimination System (NPDES) permits and load allocations for Total Maximum Daily Loads (TMDL), EPA uses the fifth percentile of the ambient and or effluent hardness values that are taken from instantaneous data (EPA 2000, p. 21). However, the hardness values used in these calculations never fall below 25 mg/L. EPA states that this provides a conservative approach on a site-specific basis for determining an acceptable discharge of metals. However, it is not clear from the Assessment how criteria are adjusted to fit these conditions, and if other circumstances could apply that would not provide protection, such as an area that receives a significant amount of metals-related discharge from non-point sources (such as the Snake River).

Although the state of Idaho was withdrawn from the NTR on April 12, 2000 (65 FR 19659), the State has not opted to use ambient hardness values outside the 25 to 400 mg/L range when calculating criteria for hardness-dependent metals. Therefore, current formulas for calculating metals criteria within this range (particularly at the low end) are not protective in all waters of Idaho, especially those with bull trout. For contaminants with hardness-dependent toxicity, the Service has used the formulas provided by EPA (1999b, pp. 40-41) to calculate the proposed criteria at concentrations below the 25 mg/L cap. In situations where the calculated criteria are below adverse effect thresholds for other aquatic species, the Service assumes adverse effects are likely to occur to listed species as well.

### 6. Adjustments to the Calculated Criteria for Toxic Metals

Part of the proposed action is to approve aquatic life criteria that are formula-based for the following metals: arsenic, chromium, copper, lead, mercury, nickel, silver, and zinc. To determine criteria for these metals that are applicable to a given water body, site-specific data must be obtained, input to a formula, and numeric criteria computed. There are three types of site-specific data that may be necessary to determine and/or modify the criteria for a metal at a site: (1) water hardness; (2) conversion factors and translators; and (3) water effects ratios (WERs). The following is a discussion of the Service's concerns regarding the application of these data and the potential implications for the proposed metals criteria.

<u>Hardness</u>

The following discussion is adapted from NMFS (2014a):

Some of the metals criteria under review in this consultation are hardness-dependent, meaning that rather than establishing a criterion as a concentration value, the criteria are defined as a mathematical equation using the hardness of the water as an independent variable. Thus, in order to evaluate the protectiveness of the hardness-dependent criteria, it was first necessary to evaluate the hardness-toxicity relations. The criteria that vary based on site-specific hardness are copper, chromium (III), lead, nickel, silver, and zinc. Hardness measurements for calculating these criteria are expressed in terms of the concentration of $CaCO_3$, expressed in mg/L, required to contribute that amount of calcium plus magnesium. In the criteria equations, hardness and toxicity values are expressed as natural logarithms to simplify the math. In a general sense, these are referred to by the shorthand "ln (hardness) vs. ln (toxicity)" relations.

In the 1980s, hardness was considered a reasonable surrogate for the factors that affected toxicities of several metals. It was generally recognized that pH, alkalinity and hardness were involved in moderating the acute toxicity of metals. While it wasn't clear which of these factors was more important, because pH, alkalinity, and hardness were usually correlated in ambient waters, it seemed reasonable to use hardness as a surrogate for other factors that might influence toxicity (Stephan et al. 1985a). In the case of copper, dissolved organic matter or carbon (DOM or DOC) were also recognized as being important. It was assumed that DOC would be low in laboratory waters and might be high or low in ambient waters, and that hardness-based copper criteria would be sufficiently protective in waters with low DOC and conservative in waters with high DOC (EPA 1985a). Most of these relations were established in acute testing, and they were assumed to hold for long-term exposures (chronic criteria). Whether that assumption is reliable was and continues to be unclear. For instance, in at least two major sets of chronic studies with metals conducted in waters with low and uniform DOC concentrations, water hardness did not appear to have a significant effect on the observed toxicity in most cases (Sauter et al. 1976; Chapman et al. 1980).

In the two decades since the NTR metals criteria were established, a much better understanding has been developed of the mechanisms of acute toxicity in fish and factors affecting bioavailability and toxicity of metals in water. Generally, acute toxicity of metals is thought to be moderated by complexation of metals, competition for binding sites on the surface of the fish's gill, and binding capacity of the gill before a lethal accumulation ($LA_{50}$) results (Wood et al. 1997; Playle 1998). The interplay of these factors has been modeled through biogeochemical gill surface models or biotic ligand models (BLMs) (Di Toro et al. 2001; Niyogi and Wood 2004). For brevity, BLMs as used here refers to both.

While BLMs are conceptually applicable for developing water quality guidelines for many metals, the BLM approach is most advanced for copper. The EPA's (2007b) recommended national criteria for copper are based on a BLM. Santore et al. (2001) validated acute toxicity predictions of the copper BLM by demonstrating that it could predict the acute toxicity of copper to the fathead minnow and *Daphnia* within a factor of two under a wide variety of water quality conditions. The predictive capability of the BLM with taxonomically distinct organisms is evaluated in detail in NMFS (2014a), Appendix C. Predictions, based on toxicity tests involving the fathead minnow, rainbow trout, Chinook salmon, planktonic invertebrates (various daphnids), and benthic invertebrates (freshwater mussels and the amphipod *Hyalella* sp.) in a variety of natural and synthetic waters, were always strongly correlated with measured acute toxicity. In several field studies, adverse effects to macroinvertebrate communities appear likely to have occurred at concentrations lower than those allowed by EPA's (2007b) chronic copper criterion. Still, the 2007 BLM-based copper criterion was at least as or more protective for macroinvertebrate communities than were EPA's 1985c and 1995 hardness-based criteria for copper (EPA 1985c, 1996)

For copper, the research leading to development of a BLM generally refutes the relevance of the hardness-toxicity relation in ambient waters (e.g., Meador 1991; Welsh et al. 1993; Erickson et al. 1996; Markich et al. 2005). This is because the important factors that influence copper bioavailability are, in rough order of importance, DOC $>\approx$ pH $>$Ca $>$ Na $\approx$ alkalinity $\approx$ Mg. Hardness is likely correlated with pH, calcium, Na, and alkalinity in natural waters, but DOC and hardness are not expected to rise and fall together.

For lead, the situation is probably similar with hardness being less important than DOC in many waters where DOC is abundant, although the BLM for lead is less advanced. With lead, calcium hardness was an important modifier of toxicity in laboratory waters with low DOC concentrations. However, at DOC concentrations reflective of many ambient waters ($>\approx$ 2.5 mg/L DOC), DOC was more important (Grosell et al. 2006a; Meyer et al. 2007; Mager et al. 2011).

In contrast, for nickel and zinc, the BLM and experimental data generally support the hardness-toxicity assumption in that acute toxicity to fish is influenced by water chemistry variables that are usually correlated with hardness (e.g., calcium, pH, Na, alkalinity, magnesium, in rough order of importance). The DOC is less important (Niyogi and Wood 2004).

For zinc, or copper under conditions of low organic carbon, the ratio of calcium to magnesium impacts the protective influence of hardness. Under the NTR and Idaho criteria, hardness is determined for a site, expressed as mg/L of $CaCO_3$, and input to the criteria equations for each metal. In natural waters, considerable variation can occur in the calcium: magnesium ratio contributing to site-specific water hardness. Studies show significant differences in toxicity for some metals depending on this ratio. In general, calcium provides greater reductions in toxicity than magnesium. For example, in the case of zinc, the presence of calcium is protective against toxicity whereas magnesium, sodium, sulfate ions and the carbonate system appear to give little to no protection (Carroll et al. 1979; Davies et al. 1993; Alsop et al. 1999). Welsh et al. (2000) and Naddy et al. (2002) determined that calcium also afforded significantly greater protection to fish against copper toxicity than magnesium.

The calcium to magnesium ratio in natural waters of Idaho varies by about two orders of magnitude (NMFS 2014a, Appendix A). Median molar ratios of calcium to magnesium across a USGS/IDEQ network of 56 sites across Idaho monitored from 1989 to 2002 range from 0.56 to 9.73, and median ratios at all sites except one exceeded 1.3 (Hardy et al. 2005).

The Service recognizes and acknowledges that water hardness and the hardness acclimation status of a fish will modify toxicity and toxic response. However, the use of hardness alone as a universal surrogate for all water quality parameters that may modify toxicity, while perhaps convenient, will clearly leave gaps in protection when hardness does not correlate with other water quality parameters such as DOC, pH, chloride, or alkalinity and will not provide the combination of comprehensive protection and site specificity that a multivariate water quality model could provide. In our review of the best available scientific literature, we have found no conclusive evidence that water hardness, by itself, in either laboratory or natural water, is a consistent, accurate predictor of the aquatic toxicity of all metals in all conditions.

<u>Water Effect Ratios</u>

The Service recognizes and acknowledges that water hardness and the hardness acclimation status of a fish will modify toxicity and toxic response. However, the use of hardness alone as a universal surrogate for all water quality parameters that may modify toxicity, while perhaps convenient, will clearly leave gaps in protection when hardness does not correlate with other water quality parameters such as DOC, pH, chloride, or alkalinity and will not provide the combination of comprehensive protection and site specificity that a multivariate water quality model could provide. In our review of the best available scientific literature, we have found no conclusive evidence that water hardness, by itself, in either laboratory or natural water, is a consistent, accurate predictor of the aquatic toxicity of all metals in all conditions.

Along with hardness, WER's are used in the formulas to derive Idaho's acute and chronic criteria for copper, chromium (III), lead, nickel, silver, and zinc. A WER is a means to account for a difference between the toxicity of the metal in laboratory dilution water and its toxicity in the water at the site. The WER is assigned a value of 1 until a different water-effect ratio is derived from suitable tests representative of conditions in the affected waterbody. Except in waters that are extremely effluent-dominated, WERs can be $\geq$ 1 and result in higher numeric criteria. A WER may be more important than hardness of site water or metal-specific conversion factors and translators in determining a criterion and hence the level of metal-loading allowed.

For the reasons stated below, the Service believes that the EPA procedures for determining WERs for metals may underestimate toxicity and thereby underestimate adverse effects to listed species and critical habitat.

1. Differences in the calcium to magnesium ratio in hardness between laboratory water and site water can significantly alter the WER. EPA guidelines for WER determinations (EPA 1994, entire) instruct users to reconstitute laboratory waters according to protocols that result in a calcium to magnesium ratio of ~0.7 across the range of hardness values (EPA 1991). This proportion (~0.7) of calcium to magnesium is far less than the ratio found in most natural waters (Welsh et al. 2000). The Service agrees with Welsh et al. (2000) that imbalances in calcium to magnesium ratios between site waters and dilution waters may result in WERs which are overestimated because calcium ions are more protective of metals toxicity than are magnesium ions.

2. Toxicity testing for WER development is not required across the same range of test organisms used in criteria development. EPA metal criteria are based on over 900 records of laboratory toxicity tests (EPA 1992) using hundreds of thousands of individual test organisms, including dozens of species across many genera, trophic levels, and sensitivities to provide protection to an estimated 95 percent of the genera most of the time (EPA 1985a, p. 9). The use of a ratio-based WER, based on findings for two or three test species, limits the reliability of the resultant site-specific criteria and may not be protective for families or genera not represented in the WER testing.

3. The inherent variability associated with living organisms used in toxicity testing can be magnified when used in a ratio. The inherent variability of toxicity testing can also have a significant effect on the final WER determination, especially because it is used in a ratio. As discussed above, the EPA has developed its criteria based on a relatively large database. However, even with such a large database, variability in test results can still cause difficulty in determining a criterion value. If 95 percent confidence intervals for the tests overlap, they are likely not significantly different and should not be used to determine a WER. Thus, toxicity tests should be conducted and carefully evaluated to minimize experimental variance when collecting data to calculate WERs.

Because of the above uncertainties regarding the accuracy of WERs, the Service believes the adverse effects to listed species and critical habitat caused by criterion concentrations for toxic metals that rely on WERs may be more severe than anticipated by EPA; in the Assessment (EPA 1999a), EPA determined that the majority of effects to listed species and critical habitat that may be caused by compliance with the proposed aquatic life criteria were insignificant or discountable.

### 7. Conversion Factors and Translators

Adoption of the NTR by Idaho in 1994, originally included criteria as total recoverable metals. In May 1995, EPA issued a stay on the effectiveness of the metals criteria as total recoverable and promulgated revised criteria expressed as dissolved metals (60 FR 22228). At that time, EPA also promulgated conversion factors (CFs) for converting between dissolved to total recoverable metals criteria. As of 1997, Idaho's criteria are expressed as dissolved metals (IDAPA 16.01.02.250.07.a.iv). The formula-based metals are included in this discussion as a group because the key issues of how dissolved metal criteria are derived and the implications of this derivation are similar for each of them.

The policy of converting total recoverable criteria to dissolved metal criteria through the use of formulas is based on the premise that the dissolved fraction of a metal in water is the most bioavailable and therefore the most toxic (EPA 1993a, p. 2; 1997, p. ES-7). EPA formulas for computing criteria are adjusted via a CF so that criteria based on total metal concentrations can be "converted" to a dissolved basis. Metals for which a CF has been applied include arsenic, chromium (III), chromium (VI), copper, lead, mercury, nickel, silver, and zinc. The term "dissolved" metal refers to metal concentrations determined in samples that have been filtered (0.45-micron pore size) prior to acidification and analysis. Particulate metals can be adsorbed to or incorporated into silt, clay, algae, detritus, plankton, etc., which can be removed from the test water by filtration through a 0.45 micron filter. A CF value is always less than 1 (except for arsenic which is currently 1.0) and is multiplied by a total recoverable criterion to yield a (lower) dissolved metal criterion.

The EPA Office of Water Policy and Technical Guidance has noted that particulate metals contribute some toxicity and that there is considerable debate in the scientific community on this point (EPA 1993a, p. 2). While the Service agrees that dissolved metal forms are generally more toxic than particulate metal forms, this is not equivalent to saying that particulate metals are non-toxic, do not contribute to organism exposure, or do not require criteria guidance by the EPA. Few studies have carefully manipulated particulate metal concentrations along with other water constituents to determine their role(s) in modulating metal toxicity. Erickson et al. (1996, p. 190) performed such a study while measuring growth and survival endpoints in fish and suggested that copper adsorbed to metal particulates cannot be considered to be strictly non-toxic. Playle (1998, p. 159) cautions that it is premature to dismiss particulate-associated metals as biologically unavailable and recommends the expansion of fish gill-metal interaction models to include these forms. The Service is concerned that investigations have not been performed with test waters that contain both high particulate metal concentrations and dissolved metal concentrations near criteria concentrations.

Particulates may act as a sink for metals, but they may also act as a source. Through chemical, physical, and biological activity these metals can become bioavailable (Moore and Ramamoorthy 1984, pp. 205-234). Particulate and dissolved metals may end up in sediments but are not rendered entirely non-toxic or completely immobile, thus they still may contribute to the toxicity of the metal in natural waters. Particulate metals have been removed from the regulatory "equation" through at least two methods: the use of a CF to determine the dissolved metal criteria, and the use of a translator to convert back to a total metal concentration for use in waste load limit calculations. When waste discharge limits are developed and TMDLs are determined for a receiving watershed, the dissolved criterion must be "translated" back to a total concentration because effluent limits will continue to be based on a total recoverable metal criterion.

The Service believes that the current use of CFs and site-specific translators in formula-based metal criteria may result in establishing water quality criteria for toxic metals that may cause adverse effects to listed aquatic species and critical habitat because organisms may be exposed to particulate metals through sediment or food-web exposure (common factors #3 and #4), and particulate bound metals cannot be considered inert.

### 8. Choice/Use of Endpoints

To assess the toxicity of a compound to an organism, an endpoint must be chosen. An endpoint is the adverse biological response that is measured in toxicity tests (Rand 1995, p. 941). There are issues that must be considered in choosing an endpoint and using it to derive water quality criteria that are protective of aquatic and aquatic-dependent species. The endpoint must be appropriate to address the question at hand, and prior to conducting toxicity tests, study design decisions must be made. The resolution of each of these issues will influence/determine the applicability of the resultant criteria.

Historically, lethality/organism mortality was the endpoint of choice, and remains in fairly common use today in acute toxicity testing. Lethality provides an endpoint that is easy to measure and unambiguous; a typical lethal endpoint is the LC50, or the concentration at which 50 percent of the test organisms die. The main value of an LC50 lies in its provision of a relative starting point for hazard assessment (Mayer and Ellersieck 1986, p. 2). Tests using 48-hour or 96-hour LC50s are commonly used by EPA to derive acute water quality criteria.

While this endpoint is widely used in short-term tests, it does not capture sub-lethal adverse impacts to organism health that may be important to survival, especially of a listed species. Adverse effects include sublethal toxicity, including, but not limited to changes in growth, reproductive, and physiological performance (Kramer et al. 2011). To prevent excessive acute lethality rather than to permit it, the LC50 values should be extrapolated to LC10, LC01, or other appropriate values, or a correction factor should be applied to prevent low-level mortality (Suter 1993, p. 225). For listed species, use of sublethal effects as endpoints is more appropriate to prevent unauthorized take. The ESA requires Federal agencies to avoid jeopardizing the continued existence of listed species (and adversely modifying critical habitat), which is likely to require use of sublethal endpoints such as incipient toxicity levels (the levels at which effects first become apparent) for some species. Use of lethality as the endpoint for deriving water quality criteria does not necessarily account for lower level effects to their sensitive olfactory system, which is critical in fishes for key life history functions such as avoiding predation, aiding their return to spawning grounds, successful reproduction, and species perpetuation (Tierney et al., 2010). The behavior of fish is extremely sensitive to many metals, often at levels that are close to or even below ambient water quality criteria (AWQC). The mechanism may involve attraction or avoidance at very low levels, followed by interference with chemosensory, mechanosensory, and/or cognitive functions at slightly higher levels (Scott and Sloman, 2004; Wood 2011a). However, as Wood (2011a) notes, *"unfortunately, this information has been ignored or discounted by most regulatory authorities, such that behavioral disturbance cannot be used as an endpoint in deriving AWQCs, and such information is usually overlooked in ecological risk assessments"* (Wood 2011a, p. 39).

Summary for Common Factors Affecting Toxicity

The common factors described above point out the numerous instances where EPA may have underestimated the potential adverse effects of the proposed criteria on listed species and critical habitat. Significant factors, such as other water quality parameters, alternate exposure pathways, bioaccumulation of toxins, and additive mixture toxicity effects should be considered by EPA when determining the effects of the proposed criteria on listed species and critical habitat. While there are reasons why the effects of chemicals to the listed species and habitats addressed in this opinion could be either more or less severe in the wild than in typical water-only laboratory tests relied upon for most criteria (NMFS 2014a, pp. 65-70), each of the common factors discussed here may act to increase toxicity of a constituent above that which is demonstrated in standard laboratory tests. In the wild, organisms are likely to be exposed to most, if not all, of these factors, and effects may manifest at lower concentrations than indicated by laboratory tests. Unfortunately, empirical testing that adjusts for all of these factors has not been completed, and may not even be feasible to complete. Thus, the available information was interpreted conservatively, with an eye towards erring on the side of species protection when the available information (primarily laboratory studies) was incomplete or ambiguous for assessing potential adverse effects in the wild.

#### 2.5.1.6 Application of Human Health Criteria

In addition to Idaho's aquatic life criteria, EPA has also approved Idaho criteria designed to protect human health from recreational, fish consumption, and drinking water uses which are also applicable to the waters in the action area. In practice, when multiple criteria are applicable to the same water body, the most stringent criteria will drive discharge limits and other pollution management efforts (IDEQ NA; subsection 70.1, *Applicability of standards, multiple criteria*.

In some cases, EPA (1999a) determined that while the aquatic life criteria may have the potential to adversely affected listed aquatic species, an added level of protection was provided by the human health criteria for the substances, which also applied to all occupied or critical habitats for listed species, and were sometimes more stringent than the aquatic life criteria. This rationale applied to arsenic, acute aldrin/dieldrin, chlordane, PCP and DDT aquatic life criteria.

For our analysis, if review of the aquatic life criteria indicated that adverse effects to listed species or their habitats and critical habitat were likely, then we reviewed the human health-based ambient water quality criteria concentrations for the same substance to see if the human-health concentrations would be protective of listed species and critical habitat.

#### 2.5.1.7 Note on EPA's Interspecies Correlation Estimations (ICE)

As described above, to address data gaps in species sensitivity, the EPA and collaborators developed the Interspecies Correlation Estimations or ICE application model "to extrapolate acute toxicity to taxa with little or no acute toxicity data for a chemical of interest, including threatened and endangered species (Asfaw et al. 2003; Raimondo et al. 2013)." ICE models are least square regressions of the relationship between surrogate and predicted taxon based on a database of acute toxicity values: median effect or lethal water concentrations for aquatic species (EC/LC50; µg/L) and median lethal oral doses for wildlife species (LD50; mg/kg bodyweight). Web-based ICE (Web-ICE, version 3.2) provides interspecies extrapolation models for acute toxicity in a user-friendly internet platform (Raimondo et al.2013).

The Service chose not to use the ICE models in our analyses of acute toxicity to listed species and critical habitat in this Opinion, but relied on the primary literature to assess acute toxic effects. This approach is foundationally similar while providing a more transparent comparison of species-specific assessment of effects.

In addition, NMFS (2014a) states that "Caution is needed when using species mean acute values (SMAVs) or genus mean acute values (GMAVs) as summary statistics for ranking species sensitivity or setting criteria. Reviews of the protectiveness of chemical concentrations or criteria that rely in large part upon published mean acute values for species of special concern such as threatened species, or their surrogates, may be subject to considerable error if the underlying data points are not examined and the associated environmental conditions influencing the primary research are not reported or are unclear. This may include analyses such as SSD, interspecies correlation estimates (ICE, Asfaw et al. 2003), or any other relative sensitivity comparisons that uses mean acute values at the family, genus, or species level" (NMFS 2014a, p. 72).

## 2.5.2 Arsenic Aquatic Life Criteria

The proposed acute criterion for arsenic is not to exceed 340 µg/L; the proposed chronic criterion is not to exceed 150 µg/L. The EPA-approved (on July 7, 2010) human health/ recreational use criterion for arsenic is 10 µg/L[14]. While arsenic is not a metal, aquatic life criteria are expressed as "dissolved" metals, i.e., determined from filtered samples. The Idaho Water Quality Standards (IWQS) are unclear as to whether the above human health criterion for arsenic is expressed as dissolved or total arsenic. The IWQS state that the criterion for arsenic addresses "inorganic arsenic only" (IDEQ NA, pp. 137, 141). The latter provision is not further explained and is curious because organic arsenic compounds are likely to have different levels of bioavailability (i.e., the degree and rate at which a substance is absorbed into a living organism or system) and toxicity than the inorganic forms of arsenic. This finding is supported by Plant et al. (2007, p. 33). However, as discussed below, organic arsenic may be less toxic than inorganic arsenic in the diet of fish. Presumably the application of the human-health recreational use standard for arsenic in Idaho was intended as total (unfiltered) arsenic since the "fishable and swimmable" components of the IWQS address exposures from incidental consumption of water while swimming or eating fish. Neither swimmers nor fish can be expected to filter their water prior to ingestion.

The term "total arsenic" (or any trace element) may be ambiguous because it can refer to two different things. In common usage in applied water quality practice, "total arsenic" refers to the total mass of arsenic determined from an unfiltered samples, which is the sum of particulate-bound and dissolved or quasi-dissolved fractions that can pass a 0.45 µm filter. In chemistry, "total arsenic" refers to all different species or forms of arsenic determined in a (usually) filtered sample, such as the sum of trivalent, pentavalent, or the many organic arsenic compounds. Here we try to make the context clear whether "total" refers to dissolved vs. particulate fractions, or total inorganic and organic forms of arsenic in a filtered sample.

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[14] The proposed action initially included a 50 µg/L criterion for arsenic (EPA 1999a) that was intended to be protective of recreational uses (i.e., consumption of fish and water by humans). In 2010, the State of Idaho lowered the recreational use criterion for arsenic to10 µg/L, which was approved by the EPA on July 7, 2010. Because IDEQ has inclusive rules for designated aquatic life and recreational uses, the human-health related criteria also apply in all waters in Idaho, including those designated as critical habitat for the bull trout and the Kootenai River white sturgeon, and waters inhabited by listed aquatic snails, bull trout, salmon and steelhead in Idaho (IDEQ NA, p. 135).

The high dietary toxicity of arsenic to humans and livestock has been recognized for hundreds of years. Relative to mammals, arsenic is carcinogenic, mutagenic, and teratogenic, and at high enough dietary exposures can be directly lethal. Compared to mammalian toxicology, relatively little work has been done with fish at environmentally relevant exposures (Sorensen 1991, pp. 66-94).

Adverse effects in fish caused by arsenic are most likely from dietary rather than waterborne exposures and involves an interaction between arsenic and selenium. Arsenic and selenium interact with each other in various metabolic functions and each element can substitute for the other to some extent, which could partly explain the reported protective effect of selenium against some arsenic-linked diseases (Plant et al. 2007, pp. 18-20).

The human health/ recreation use criterion for arsenic applies to all waters in Idaho with one exception. The IWQS provide an exception for Bucktail Creek, a small stream contaminated by historic mining wastes. Bucktail Creek is a tributary to Big Deer Creek, which is a tributary to Panther Creek, which in turn is a tributary to the Salmon River, in the Middle Salmon-Panther hydrologic unit. Panther Creek is designated as critical habitat for the bull trout (USFWS 2010a, p. 745).

**General Environmental Effects of the Proposed Arsenic Criteria**

The total recoverable criteria for arsenic are identical to the proposed acute and chronic criteria because the Conversion Factor (CF) for arsenic is 1.0. Arsenic toxicity does not vary significantly with hardness (Borgmann et al. 2005, Table 3).

When the toxicity of arsenic is limited to consideration of direct effects in water-only exposures, arsenic is essentially non-toxic at environmentally relevant concentrations. However, as discussed in the following sections, arsenic can be very toxic when organisms are exposed to it through the foodweb.

Arsenic occurs naturally in the environment. It is bioaccumulated (i.e., accumulation of a chemical in tissues as a result of ingestion of water-borne chemicals or as food) by organisms but is not biomagnified, which is the process where tissue concentrations of a chemical increase through the food chain (Eisler 1988, p. 12). The chemical form of arsenic in surface waters is dependent on factors such as the redox potential, pH, and biological process-related “speciation” of arsenic in water. In well oxygenated waters typical of flowing waters, arsenic is commonly found as arsenate (Mok and Wai 1989; McIntyre and Linton 2011, p. 332). In fish, tolerance of arsenic appears to increase with temperature (McGeachy and Dixon 1990, p. 2228), whereas in invertebrates the opposite is true (Bryant et al. 1985, p. 135).

## 2.5.2.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

The following factors were considered in the following analysis of the proposed criteria for arsenic on listed aquatic snails: (1) the lack of species-specific arsenic toxicity data (or data on closely-related species that are similar in life history); (2) the limited and/or isolated distribution of each of the four listed aquatic snail species within their habitats; and (3) the degraded conditions of existing habitats. The information presented by EPA in the Assessment is primarily based on laboratory tests that are typically conducted in the absence of confounding factors normally experienced by snails in their native habitats. The toxicity of arsenic can be altered by a number of factors including temperature, speciation, suspended solid concentration, the presence of mixtures, and the duration of exposure. In addition, we are not aware of information on the effects of mixtures on arsenic toxicity to aquatic snails, or on the combined effects of arsenic absorption from both the water column and through dietary uptake from grazing or sediment ingestion. For this analysis, we assumed that bottom-feeding aquatic snails are likely ingesting sediment while grazing and this is likely an additional route of exposure to arsenic and other potential contaminants.

The limited data on arsenic toxicity available for snails indicate there is little risk of snail mortality from direct, water-only, long-term exposures to arsenic. Spehar et al. (1980, p. 53; p. 55, Table 1) exposed the pulmonate snails *Helisoma campanulata* (Planorbidae) and *Stagnicola emarginata (*Lymnaeidae) to four arsenic compounds at up to 1000 µg/L for 28 days and observed no reductions in survival. The arsenic compounds were taken up by the snails and reached tissue residues up 80 mg/kg dw (Spehar et al. 1980, p. 55, Table 1). These tissue concentrations are far higher than tissue concentrations associated with damage to fish (see section 2.5.1.3 addressing the bull trout below). Similarly, with the snail *Apelxa hypnorum* (Physidae), an LC50 for arsenic of 24,500 µg/L was obtained from 4-day water-only exposures (Holcombe et al. 1983, Table 3). Ambient arsenic concentrations in surface water are unlikely to approach concentrations that would cause acute toxicity in aquatic snails, or even concentrations that meet the proposed acute criterion of 340 µg/L (Table 5).

Based on our review of best available information presented in the Assessment and elsewhere, no evidence was found of direct adverse effects to snails from long-term exposure to arsenic at concentrations less than the proposed chronic criterion concentration of 150 µg/L. However, indirect effects may occur due to the effects of elevated arsenic concentrations on the snails' presumed primary food sources: algae, detritus, and periphyton; this matter is further discussed below.

**Table 5. Selected concentrations of arsenic in stream water, sediment, and in the tissues of aquatic invertebrates from field studies. Selected undiluted mine effluent concentrations from within the action area are included for comparison. Unless otherwise noted, concentrations are averages, values in parentheses are ranges.**

| Location and notes | Arsenic Concentration (µg/L) in Filtered Water | Arsenic Concentration (µg/L) in Unfiltered Water | Arsenic Concentration (mg/kg dw) in Sediment | Arsenic Concentration (mg/kg dw) in Invertebrate Tissues |
|---|---|---|---|---|
| Effects thresholds (j) | | | 7-33 | ~ 20 |
| "Typical" USA river waters, not in enriched areas | | 0.1 – 2 (l) | | |
| Idaho rivers–statewide assessment (h) | | 2.3 (0.06 – 17) | | |
| Stream sediments, USGS national median | | | 6.3 (l) | |
| Gold Cr (Chloride Gulch, mining-affected), ID (m) | 12 | | 537 | 97 |
| Upper Gold Cr (mining-affected) | 5.5 | | 50 | 41 |
| Gold Cr (Delta, mining-affected | 1.1 | | 28 | 28 |
| Gold Cr (West Gold, reference), ID (m) | 0.9 | | 2.6 | 5.4 |
| Panther Cr, ID, mining influenced reaches (prior to cleanup (a, f, I, n) | 1 - 6 | 102 (max) | 27-888 | 76 (f) |
| Blackbird Creek, ID (1993)(a) | 1.1 | 158 (max) | 939 | |
| South Fork Coeur d'Alene (b, c) | 0.4 – 4 | 13 (max) | 180 | 42 (c) |
| Clark Fork River at Galen, MT (b,d) | 15 (3-53) | 20 (4-80) | 170 (3) | 21(e) |
| Snake River leaving Yellowstone NP, WY (b,e) | 34 (8-55) | | 38 | 11 (f) |
| Snake River at King Hill, ID (b,e) | 3 (0.5 – 7) | 4 (2-9) | 5 (4-7) | 1 (0.5 – 2) (f) |
| Hecla Grouse Creek gold mine, near Custer, Idaho (k) | 2.4 (<1-5) | 7 (<5 – 55) | | |
| Thompson Creek molybdenum mine, nr Clayton, Idaho (l) | 2 – 4 | | | |

blank cells = no data. Literature sources: (a) Beltman et al. (1994); Maest et al. (1994); (b) USGS Water-Quality Data for the Nation, *http://nwis.waterdata.usgs.gov/nwis/qw*; (c) Farag et al. (1998); (d) Hansen et al. (2004); (e) Ott (1997); (f) Community sample; (g) caddisfly *Hydropsyche* sp.; (h) Essig (2010); (i) Mebane (2002); (j) Effects thresholds for invertebrate residues are from this review; values for sediment are threshold and probable effect concentrations presented in MacDonald et al. (2000); (k) R. Tridle, Hecla Mining Company, unpublished data, Jan 2008; (l) Thompson Creek mine "NPDES" wastewater permit factsheets, accessed January 2008 from *http://yosemite.epa.gov/r10/water.nsf*; and Plant et al. (2007), (m) Kiser et al. (2010), (n) Mok and Wai (1989)

Two of the major uses of arsenic are in the production of herbicides and wood preservatives. Inorganic arsenic compounds have been used widely for centuries as insecticides, herbicides, algicides, and desiccants (Eisler 1988, p. 5). The literature on the effects of arsenic compounds to individual algae species or communities is more abundant than, for example, the effects of those compounds on invertebrates. EPA's (1985a, Table 4) ambient water quality criteria for arsenic listed effect data for 16 algae species and two aquatic plant species. Their compilation indicated a huge range of sensitivities with growth inhibition concentrations of arsenic ranging from 48 to 202,000 µg/L. Two of the 16 effect concentrations listed in EPA (1985a, table 4) were lower than the chronic criterion values with growth inhibition at 48 µg/L of arsenic for *Scenedesmus obliquus*, a green alga, and growth inhibition of two phytoplankton species at an arsenic concentration of 75 µg/L. The original source publications for these two studies are Vocke et al. (1980) and Planas and Healey (1978), respectively. However, based on our review of these publications, the above results were discounted because of data quality concerns. Neither study included any analytical verification of their actual exposure concentrations (Planas and Healey 1978).

Two more recent and (more analytically robust) studies of the effects of arsenic on algae species are reported by Knauer et al. (1999) and Rahman et al. (2014). Rahman et al. (2014) tested the exposure of different inorganic and organic arsenic compounds with the green algae Chlorella and found that As(V), arsenate, was most toxic, but the 50 percent growth inhibition concentration of 1150 µg/L of arsenate was well above the proposed chronic criterion value for arsenic. In contrast to the classic beaker tests used by Rahman et al. (2014) and most others, Knauer et al. (1999) tested natural phytoplankton communities in large limnocorrals suspended in lakes along an arsenic contamination gradient. In their control lake with low concentrations of arsenic (≈ 1.2 µg/L total arsenic), photosynthesis was inhibited by 50 percent at about 22 µg/L, with threshold reductions as low as 4 µg/L of arsenic, as arsenate (Knauer et al. 1999). Arsenate was more toxic to phytoplankton than was arsenite or organic forms of arsenic. Some lakes were contaminated with arsenic concentrations up to 14 µg/L. In the lakes with elevated arsenic concentrations, phytoplankton communities were much more tolerant of additional arsenic exposure than the algal species, suggesting either selection for tolerant taxa, or the phytoplankton had developed an adaptive resistance to arsenic (Knauer et al. 1999).

Based on the above information, although direct mortality of the three Snake River aquatic snails and the Bruneau hot springsnail is not likely to occur from the proposed acute and chronic arsenic criteria, significant effects to their food base are likely to occur. Snails graze upon algae, and as discussed above, arsenic has been shown to adversely affect natural algal communities with profound (50 percent) impairment of photosynthesis at arsenic concentrations as low as 22 µg/L (Knauer et al. 1999). On that basis, we conclude there is likely to be a significant alteration in available algae food sources for Snake River aquatic snails and the Bruneau hot springsnail throughout their ranges caused by arsenic concentrations below the proposed chronic criterion levels.

## 2.5.2.2 Bull Trout

Based on our review of best available information, no studies were found that reported acute toxicity to juvenile or adult salmonids at arsenic concentrations close to the proposed acute criterion. All of the studies we reviewed indicate that arsenic toxicity following short-term, water-only exposures occurs only at very elevated concentrations that are much higher than the proposed acute criterion. For example, acute LC50s (lethal concentrations killing 50 percent of tested fish) for the brook trout (*Salvelinus fontinalis*), a close relative of the bull trout, ranged from 14,900 to 10,440 µg/L in 4- to 10-day exposures (EPA 1985b, Tables 1 and 6, pp. 20, 36). EPA's EcoTox database lists a total of nine acute tests with brook trout with LC50s ranging from 18,000 to 54,100 µg/L (EPA 2013b). Although none of the values in the EcoTox database matched those from EPA (1986), even though both were attributed to the same original source, lethal concentrations of arsenic identified in both studies are much higher than the proposed acute arsenic criterion of 340 µg/L.

Based on a recent comprehensive review of arsenic toxicology in fishes by McIntyre and Linton (2011), waterborne exposure to arsenic is not likely to cause toxic effects to exposed fish, although the toxicity tests considered in that paper are not that environmentally meaningful. The results of Birge et al. (1980) suggest that chronic arsenic toxicity from waterborne exposure occurs to developing embryos of listed salmonids at concentrations below the proposed chronic criterion. Rainbow trout embryos were exposed to arsenic for 28 days (4 days post-hatching) at 12°C to13°C and a hardness of 93 mg/L to 105 mg/L $CaCO_3$ in static tests. Arsenic concentrations of 42 to 134 µg/L were estimated to be associated with the onset of embryo mortality, at LC1 and LC10 levels, respectively (Birge et al. 1980, Table 2). However, no further details of the results of this test were reported beyond these statistical effect estimates, making these results impossible to critically review. Studies reviewed in Eisler (1988, see Table 4) and EPA (1985b, see Table 2) indicate that chronic effects of arsenic exposure do not occur in other salmonid lifestages until concentrations are at least about an order of magnitude higher than the levels determined by Birge et al. (1980) to be detrimental to developing embryos. For instance, Spehar et al. (1980, Table 1), found no reductions in the survival of rainbow trout embryos exposed to four different arsenic compounds at concentrations of nearly 1,000 µg/L in 28-day, water-only exposures.

**Dietary Toxicity of Arsenic**

The information discussed below indicates that at environmentally relevant concentrations, arsenic poses significant health risks to salmonids, including reduced growth and survival, organ damage, and behavioral modifications.

Cockell et al. (1991, p. 518) fed inorganic arsenic-contaminated food to rainbow trout under standard laboratory conditions for 12-24 weeks and correlated signs of toxicity with diet and tissue arsenic concentrations. They found that the threshold for the onset of organ damage (gall bladder inflammation and lesions) was between 13 and 33 mg/kg of arsenic in the food. Woodward et al. (1994 51-61, 1995, p. 1998) fed rainbow trout a diet made from invertebrates collected from the metals-contaminated Clark Fork River in Montana; that diet resulted in lower fish growth and survival compared to fish exposed to metals-contaminated water only. However, because these metals-contaminated invertebrates were contaminated with several metals including arsenic, and the effects were equally correlated both with arsenic and copper, these effects could not be attributed to either metal alone. Subsequently, Hansen et al. (2004, pp. 1902-1910) collected metals-contaminated sediments from the Clark Fork River, reared aquatic earthworms (*Lumbriculus*) in them, and fed the *Lumbriculus* to rainbow trout. Fish fed the *Lumbriculus* diet had reduced growth and physiological effects; the effects were strongly correlated with arsenic but not to other elevated metals.

Bull trout and cutthroat trout collected from mining-influenced Gold Creek in northern Idaho showed liver damage with inflammation, necrosis and cellular damage. Arsenic was elevated in the sediments, macroinvertebrates, and fish tissues, and was correlated with the liver damage (Kiser et al. 2010, pp. 301-310). Erickson et al. (2010, pp. 122-123) further implicated arsenic as the causative agent by experimentally mixing arsenic into clean sediments, rearing *Lumbriculus* in them, and feeding the *Lumbriculus* to rainbow trout. The rainbow trout fed the worms that had been raised in arsenic-dosed sediments had reduced growth and disrupted digestion. The study by Erickson et al. (2010) is difficult to directly compare to feeding studies with field-collected invertebrates because Erickson et al. did not report what tissue concentrations bioaccumulated in exposed fish following 30 days on a diet of arsenic-enriched invertebrates. Still, the study results reported by Erickson et al. (2010) produced similar effects to those from field-collected diets with controlled exposures to contaminated field sediments, and strongly implicated arsenic as an important stressor.

Collectively, these studies show that inorganic arsenic in the diet of rainbow trout can be associated with reduced growth, organ damage and other physiological effects starting at concentrations in the diet of about 20 to 30 mg/kg dry weight (dw) (Cockell et al. 1991, p. 518; Hansen et al. 2004, pp. 1902-1910; Erickson et al. 2010, pp. 122,123). Ranges of reported effects in other species are wider. Damage to livers and gall bladders occurred in lake whitefish (*Coregonus clupeaformis*) fed arsenic contaminated diets as low as 1 mg/kg food dw (Pedlar et al. 2002, p. 167). The adverse effects of dietary arsenic to salmonids are summarized in Table 6.

Bioaccumulation of arsenic in salmonid prey organisms to concentrations higher than 30 mg/kg dw has been documented from the Clark Fork River and the Boulder River in Montana, and in the Coeur d'Alene River and Panther Creek in Idaho. Concentrations of arsenic in these streams have been measured at higher than background (< approximately 5µg/L) but were never documented at concentrations even approaching the proposed chronic water quality criterion for arsenic of 150 µg/L (Table 5). Review of waterborne arsenic concentrations collected from the same waters suggests that bioaccumulation of arsenic in invertebrate prey organisms to concentrations harmful to salmonids appears to be able to occur in streams with dissolved arsenic concentrations less than the chronic criterion. These studies focused mostly on the effects of arsenic on organs and growth; however at least one study has shown that arsenic in fish diets can affect reproduction, although the single dietary exposure tested was higher (135 mg/kg dw) than in the studies mentioned with salmonids (Boyle et al. 2008, p. 5356).

While in general, higher concentrations of arsenic in water would be expected to result in higher concentrations of arsenic in tissues of aquatic organisms, simple relationships between water and tissue concentration are elusive. For instance, within a relatively homogenous study area (Gold Creek, Idaho, from Kiser et al. 2010), the arsenic concentrations in water and invertebrate tissues listed in Table 5 were highly correlated ($r^2 = 0.93$, $p$=0.03). However, across different locations the data were not so consistent (Table 5). Reasons for this variability might be related to seasonal variation, food web differences, or differing chemical forms of arsenic, discussed in more detail later in this section. Similarly, in a review of arsenic bioaccumulation in freshwater fishes, Williams et al (2006) found tissue residues tended to increase with increasing concentrations in laboratory studies using the same fish, same water, same chemical form or arsenic, etc., but that in field settings where these sorts of factors were not controlled, no relationship was apparent between arsenic concentrations in water and fish (Williams et al., 2006, their figs 1 and 2).

**Table 6. Effects of arsenic in the diet of salmonids of selected observed and experimental concentrations.**

| Fish Species | Diet source | Effect | Arsenic in diet (mg/kg dw) | Reference |
|---|---|---|---|---|
| Bull Trout and cutthroat trout | Benthic invertebrates (presumed) | Liver damage | 28-97 | (Kiser et al. 2010, p. 301) |
| Cutthroat trout | Metals-contaminated invertebrates collected from the Coeur d'Alene R, ID | Reduced growth, liver damage | 14-51 | (Farag et al. 1999) |
| Cutthroat trout | " " " | None apparent | 2.6-3.5 | Farag et al. (1999) |
| Rainbow trout | Metals-contaminated invertebrates collected from the Clark Fork River, MT | Reduced growth, impaired digestion | 19 – 42 | Woodward et al. (1994,1995) |
| Rainbow trout | " " " | None apparent | 2.8-6.5 | Woodward et al. (1994,1995) |
| Rainbow trout | *Lumbriculus* (aquatic earthworms) contaminated using Clark Fork River sediments | Reduced growth, impaired digestion, liver and gall bladder degeneration | 21 | (Hansen et al. 2004) |
| Rainbow trout | Diet of *Lumbriculus* exposed to arsenic | Reduced growth | 34 | (Erickson et al. 2010) |
| Rainbow trout | Diet (pellets) amended with arsenate | Reduced growth, impaired digestion, gall bladder inflammation | 33 | (Cockell et al. 1991) |
| Rainbow trout, subadult | Diet (pellets) amended with arsenite | Reduced growth | ≥ 51 | (Hoff et al. 2011) |
| Rainbow trout | Diet (live or pellets) amended with inorganic arsenic (arsenite or arsenate) | Reduced growth | >≈ 20 mg/kg | (Erickson et al. 2011a) |
| Rainbow trout | Diet (live or pellets) amended with organic arsenic | Reduced growth | >≈ 100 mg/kg | (Erickson et al. 2011a) |
| Rainbow trout | " " " | None apparent | 13 | Cockell et al. (1991) |
| Lake Whitefish | Diet (pellets) amended with arsenic | Liver and gall bladder damage, no effects on growth | ≥1 | (Pedlar et al. 2002) |

Field studies of resident trout populations in streams influenced by natural geothermal drainage in Yellowstone National Park give indirect evidence of tolerance to elevated arsenic or perhaps density-dependent compensation to low-level toxicity. Goldstein et al. (2001, pp. 2342–2352 ) found that naturalized rainbow and brown trout were at least present in some streams with arsenic concentrations in water that were greatly above typical background concentrations. Arsenic was elevated both in water and invertebrates collected from the Snake River at the southern boundary of Yellowstone National Park (Table 5). Trout and sculpin densities at that location appeared robust in comparison to surveys at other least-disturbed rivers in Idaho and the Pacific Northwest (Maret 1997, p. 49; Mebane et al. 2003, p. 257), so total arsenic concentrations on the order of 30 µg/L in water and 11 mg/kg in insect tissues were causing no obvious harm to resident fish populations. Whether the apparent tolerance of resident fish and invertebrates at this location is related to intrinsic tolerance, pollution-induced community tolerance, or bioavailability cannot be determined from the information at hand.

Most of the fish feeding and field studies discussed above reported total arsenic concentrations, without distinguishing, based on speciation analyses, whether the arsenic is in an inorganic or organic form. Some evidence indicates that organic arsenic in the diet of salmonids is less toxic than inorganic arsenic (Cockell and Hilton 1988, pp. 73-82; Table 1). Whether the arsenic that occurs in salmonid prey items in streams occurs predominately in an inorganic or organic form is unknown, but is assumed here to be primarily in an inorganic form. This assumption is based on a generalization of trophic transfer and biotransformation and of arsenic in the aquatic food chain, as reviewed by Rahman et al. (2012, pp. 118-135). The bulk of dissolved arsenic in freshwater consists of inorganic compounds. In general, arsenic probably enters freshwater food chains in large part because algae actively absorb arsenate, mistaking it for phosphate. Biotransformation of inorganic arsenic by primary consumers of algae appears to be minimal, although once taken up by higher trophic level fish, arsenic is predominantly converted to organic forms. This transformation to organic forms appears to be a detoxification mechanism by fish, although some organic arsenic forms can still be genotoxic to fish (Rahman et al. 2012, pp. 124-126).

Whether dissolved or particulate arsenic contributes more to arsenic risk is also debatable, but the present evidence suggests particulate arsenic may be more of a concern. The proposed water quality criteria are based on dissolved arsenic, the rationale for which is unstated in the description of the proposed action in the Assessment. Arsenic is a metalloid rather than a metal, but apparently for regulatory purposes, arsenic was simply considered another metal like cadmium or zinc without any known analysis. While the information is sparse, field data suggests that dissolved arsenic may be far less important as a source to aquatic food webs than particulate and sediment sorbed (attached) arsenic. This suggests that the dissolved arsenic criterion may be less relevant than a sediment, dietary, or tissue residue-based criterion.

### Tissue Concentrations of Arsenic Associated with Chronic Responses in Fish

McIntyre and Linton (2011) report that regardless of exposure route or form, fish tissue concentrations of arsenic associated with chronic effects were remarkably similar among fish. Adverse effects appear likely to occur when whole-body tissue concentrations reach about 2 to 5 mg/kg wet weight (ww). The critical tissue residue concentrations of arsenic in the liver associated with reduced growth may be somewhat lower, around 0.7 to 1.0 mg/kg ww. This range of critical liver concentrations of arsenic was supported by recent research reported by

Hoff et al. (2011, poster) who showed a change point in the growth of rainbow trout when arsenic levels in liver tissue reached about 6 mg/kg dw, which would be equivalent to about 1 to 1.5 mg/kg ww.

In studies where rainbow trout were fed field-collected invertebrates from the mining-influenced Clark Fork River, Montana, and in which adverse effects occurred, arsenic concentrations in whole-body fish tissues ranged from about 0.6 to 2.5 mg/kg ww (Woodward et al., 1994, p. 61,1995, p. 1998). In a similar study in the Coeur d'Alene River basin, Idaho, Farag et al. (1999, p. 585) fed fish invertebrates collected from mining-influenced reaches and reported reduced growth, liver degeneration, and fish tissue concentrations of arsenic ranging from about 0.5 to 1.2 mg/kg ww. In contrast, arsenic in fish fed a reference diet collected from a minimally polluted reach of the North Fork Coeur d'Alene River ranged from about 0.2 to 0.3 mg/kg ww (Farag et al. 1999, p. 585). Other metals were also elevated in the fish, particularly lead, although results from the Erickson et al. (2010, entire), and Hansen et al. (2004, entire) studies argue that most of the toxicity in Farag's study was probably attributable to arsenic, based upon effects/non-effects or correlation/lack of correlation between arsenic and other metals in Erickson et al.'s (2010) and Hansen et al.'s (2004) studies.

Whole-body arsenic residues associated with reduced growth in fish following feeding studies (>approximately 0.6 mg/kg ww) are difficult to compare to surveys that only sampled edible fillets (muscle). In a probabilistic study of fish captured from 55 randomly selected river sites throughout Idaho, Essig (2010, appendix E) obtained a median arsenic concentration of 0.06 mg/kg ww, ranging from <0.13 to 0.31 mg/kg ww in muscle fillets. The highest value in Essig's (2010) report was from a brown trout collected from a geothermally influenced reach of the Portneuf River. In targeted collections of trout in the Stibnite Mine area, arsenic concentration in fillets were up to 0.96 mg/kg, fresh weight (Woodward-Clyde 2000, Table 8.5.11-12), considerably higher than the maximum value from Essig's (2010) randomized survey. In the Stibnite study, arsenic in muscle fillets was considerably lower than in the remaining trout carcasses (e.g., organs, bone, viscera, skin) after the fillets had been removed. Arsenic in fillets ranged from <0.25 to 0.96 mg/kg fresh weight versus 0.32 to 6.3 mg/kg fresh weight in the remainders (Woodward-Clyde 2000, Table 8.5.11-12).

### Behavioral and Neurotoxic Effects of Arsenic

Despite profound neurotoxic effects of arsenic in mammals, there appears to have been minimal research with behavioral and neurotoxic effects of arsenic in fish. However, the following information suggests that behavioral effects to fish from arsenic exposure may be significant at very low exposure concentrations. Arsenic impaired long-term memory in zebrafish exposed for 96 hours to arsenic concentrations as low as 1 µg/L before avoidance trials (McIntyre and Linton 2011). Measurement of elevated levels of oxidized proteins in brain tissue of fish exposed to 10 µg/L of arsenic suggested that the observed effects may have been related to oxidative stress in brain tissue caused by the exposure to arsenic (McIntyre and Linton 2011, p.297).

### Arsenic Toxicity to Food Organisms

The limited data available suggests that the risk of arsenic toxicity to salmonid food/dietary organisms is lower than the risk of arsenic toxicity to salmonids from eating arsenic-exposed organisms. However, no studies were found that had tested invertebrates using environmentally relevant exposures through arsenic-enriched periphyton or sediments, and none were found that had been conducted through full-life exposures or sensitive life stage exposures.

Norwood et al. (2007, p.266) related bioaccumulation of arsenic in *Hyalella azteca*, a benthic invertebrate common in slow moving rivers and lakes, to mortality in 4-week exposures. Lethal body concentrations associated with 25 and 50 percent mortality of *Hyalella* were about 9 and 10 mg/kg dw, respectively. Burgess et al. (2007) spiked reference sediments with arsenic to allow more definitive cause and effect conclusions. In their tests, arsenic-spiked sediments killed 50 percent of amphipods and mysids at about 81 mg/kg dw in 7-day exposures. At sediment concentrations greater than about 125 mg/kg dw, 100 percent amphipod mortality resulted (Burgess et al. 2007, Figure 1). These experiments were with marine sediments, but unlike cationic metals, the bioavailability in saltwater is not expected to be greatly less than in freshwater.

Irving et al. (2008, pp. 583-590) exposed mayfly nymphs to tri- and pentavalent arsenic in water-only exposures for 12 days. For trivalent arsenic, the threshold of growth effects was about 100 µg/L. However, arsenic levels accumulated by the mayfly nymphs in their study (1.2– 4.6 µg/g dw) were far lower than those reported from stream locations with far lower water concentrations of arsenic but that had elevated arsenic in diet or sediments, suggesting that the water-only exposures may have underrepresented likely environmental exposures to arsenic. Crayfish collected from Australian streams disturbed by mining activities had up to 100 mg/kg dw of arsenic in their tissues. Levels of arsenic in the tissues of the crayfish were similar to those found in the sediment, thus it is highly likely that the primary exposure to arsenic for the crayfish came from the sediment (Williams et al. 2008, pp. 1340-1341).

Canivet et al. (2001, p. 351) similarly found increased mortality of gammarid amphipods and heptagennid mayflies at about 100 µg/L which is lower than the proposed chronic criterion of 150 µg/L.

In addition, the proposed aquatic life criteria for arsenic do not include sediment criteria and, therefore, provide no regulation of sediment contaminant concentrations. Arsenates, one of the common forms of arsenic found in water, sorbs to humic material, iron hydroxides and may coprecipitate with other ions (Eisler 1988, p. 7, Mebane 1994, p. 35; Gray and Eppinger 2012, p. 1060). Elevated arsenic concentrations in sediments may impact early life stages of the bull trout, particularly eggs and juveniles that have a long residence time (approximately 200 days) in channel substrates as discussed in the *Status of the Species* section above (section 2.3.5).

Because aquatic invertebrates are likely to accumulate arsenic from sediments and biofilms, as discussed above, arsenic accumulation in aquatic invertebrates in freshwater food webs has been reasonably implicated as the cause of reduced growth and tissue damage in salmonids. On that basis, we conclude that the proposed chronic criterion for arsenic is likely to cause adverse effects to the bull trout in the form of reduced growth and tissue damage. These effects have been documented in salmonids at concentrations much lower than the proposed chronic arsenic criterion of 150 µg/L. Given that the action area represents 44 percent of bull trout-occupied streams and 34 percent of bull trout-occupied lakes and reservoirs within its range, these adverse effects are considered to be significant. Reduced growth and tissue damage in affected bull trout at that scale are likely to impair or preclude maintaining or increasing the bull trout's current rangewide distribution, abundance, and reproduction.

## 2.5.2.3 Bull Trout Critical Habitat

Of the nine designated PCEs of bull trout critical habitat, two are likely to be adversely affected by the proposed arsenic criteria: PCE 3 (adequate prey base) and PCE 8 (water quality).

Resident and juvenile migratory bull trout prey on small fish, including salmon fry, as well as terrestrial and aquatic insects, and macro-zooplankton (Boag 1987; Goetz 1989; Pratt 1992, p. 6; Donald and Alger 1993). Adult migratory bull trout feed almost exclusively on other fish (Rieman and McIntyre 1993, p. 3); robust bull trout populations may depend on abundant fish prey resources. This relationship is shown by the correlation between declines in bull trout abundance and declines in salmon abundance (Rieman and McIntyre 1993, p. 3).

Bioaccumulation of arsenic in invertebrate organisms (that serve as prey for salmonids like the bull trout) to concentrations harmful to salmonids is likely to occur in streams with dissolved arsenic concentrations below the proposed chronic criterion; inorganic arsenic in the diet of rainbow trout is associated with reduced growth, organ damage and other adverse physiological effects (Cockell et al. 1991, p. 518; Hansen et al. 2004, pp. 1902-1910; Erickson et al. 2010, pp. 122,123). For those reasons, we expect that arsenic concentrations below the proposed chronic criteria are likely to contaminant the prey base within bull trout critical habitat to an extent that precludes it from being adequate to support normal growth and reproduction in the bull trout. For that reason, the proposed chronic criterion for arsenic is likely to significantly impair the capability of bull trout critical habitat to provide an abundant food base (PCE 3) for the bull trout over a significant portion of the range of designated critical habitat. As discussed above, the state of Idaho contains 8,772 miles (44 percent) of streams and 170,217 acres (35 percent) of lakes and reservoirs designated as critical habitat for the bull trout (75 FR 63937).

In addition, due to the continuous interactions between surficial sediment, interstitial water, and overlying water or the water column, the condition or quality of sediment are interrelated with water column concentrations. For these reasons, the Service concludes that the proposed chronic criterion for arsenic is likely to adversely affect PCE 8 (water quality) of bull trout critical habitat. Given that the state of Idaho contains 8,772 miles (44 percent) of streams and 170,217 acres (35 percent) of lakes and reservoirs designated as critical habitat for the bull trout (75 FR 63937), this effect is likely to be significant.

## 2.5.2.4 Kootenai River White Sturgeon

Based on the adverse effects of arsenic to salmonids discussed above that are likely to occur at concentrations below the proposed criteria, it is reasonable to conclude the proposed chronic criterion for arsenic is also likely to adversely affect the Kootenai River white sturgeon. The most appropriate data on the effects of arsenic on fish appears to be related to dietary toxicity, however, no dietary toxicity data specific to the sturgeon are available; such data are also not available for any other species within the order Acipenseriformes. Data on the dietary effects of arsenic to fish are available for the fathead minnow (*Pimephales promelas*), channel catfish (*Ictalurus punctatus*), rainbow trout, and the lake whitefish (*Coregonus clupeaformis*). Based on those data, the rainbow trout, and the lake whitefish are considerably more sensitive to dietary arsenic than are the fathead minnow and the channel catfish (Erickson et al. 2010, Cockell et al. 1991, Pedlar et al. 2002; discussed above). In the absence of specific data related to the sturgeon, the Service is giving the benefit of the doubt to the sturgeon by relying on the more sensitive rainbow trout data as the best available information on the effects of dietary arsenic on the sturgeon. The rainbow trout is a commonly tested species that has previously been used as a surrogate species to evaluate the effects of contaminants on listed species (e.g., Besser et al. 2005a, Dwyer et al. 2005). We also assume that sturgeon sensitivity to arsenic is at least as sensitive as for the rainbow trout. With rainbow trout, dietary arsenic has been linked to reduced growth at about 20 mg/kg dw and higher (see *Dietary Toxicity*, section 2.5.2.2 above), and these concentrations in benthic invertebrates have been measured in field conditions with water concentrations much lower than the proposed 150 µg/L chronic criterion for arsenic (Table 5). The observed effects of arsenic contamination in salmonids include altered feeding behavior, and reduced body weight, reproductive success, and survival.

Absent information specific to the effects of the proposed arsenic criteria on white sturgeon prey species, we are assuming that information on the effects of the proposed arsenic criteria on bull trout prey species also applies to white sturgeon prey species. These potential effects were discussed in section 2.5.2.2 above relative to the bull trout: at environmentally relevant concentrations, arsenic poses significant health risks to salmonids, including reduced growth and survival, organ damage, and behavioral modifications.

In addition, due to the continuous interactions between surficial sediment, interstitial water, and overlying water or water column, the condition or quality of sediment cannot be separated from water quality, and elevated contaminant concentrations, such as arsenic, in sediments are interrelated with water column concentrations.

The Kootenai River white sturgeon DPS is restricted to approximately 270 river kilometers (168 river miles) of the Kootenai River in Idaho, Montana, and British Columbia, Canada. Approximately 39 percent of the range of the DPS occurs within the state of Idaho and would be impacted by the proposed chronic criterion for arsenic. Given that existing data show adverse effects to multiple freshwater fish species (including potential prey species of the white sturgeon) at arsenic concentrations below the proposed chronic criterion, and the probability that arsenic concentrations will be even higher in sediments, which is likely to increase adverse impacts to white sturgeon eggs and juveniles, we conclude the proposed chronic criterion for arsenic is likely to adversely affect the Kootenai River white sturgeon. These effects are likely to be manifested in the form of reduced growth and survival, organ damage, and behavioral modifications. Such effects are likely to impede achievement of the following recovery criteria for the sturgeon: (1) natural reproduction of white sturgeon in at least three different years of a 10-year period; and (2) achieving a stable/increasing sturgeon population in the wild, and ensuring that captive-reared juveniles are available to be added to the wild population every year for a 10-year period (USFWS 1999, p. 38). The nature of these effects (i.e., reduced growth and survival, organ damage, and behavioral modifications) are also likely to reduce the survival of the Kootenai River white sturgeon in the wild. For these reasons, the impacts of the proposed chronic criterion for arsenic on the Kootenai River white sturgeon are considered to be significant.

### 2.5.2.5 Kootenai River White Sturgeon Critical Habitat

As discussed above, implementation of the proposed action is likely to reduce sediment quality and water quality within white sturgeon critical habitat. The nature of these habitat effects are likely to cause reduced growth and survival, organ damage, and behavioral modifications in individual Kootenai River white sturgeon. Sediment quality and water quality were collectively recognized as a PCE of critical habitat for the Kootenai River white sturgeon in the 2001 final rule (66 FR 46551)[15]. Although this PCE was not retained in the 2008 revised critical habitat rule for the Kootenai River white sturgeon (73 FR 39506), adequate sediment and water quality are necessary, in part, for the critical habitat to function in support of Kootenai River white sturgeon recovery.

Because the proposed water quality criteria are implemented statewide, all of the designated white sturgeon critical habitat within Idaho would be subjected to aquatic arsenic chronic concentrations up to 150 µg/L. Thus, the proposed chronic arsenic criterion is likely to adversely affect water and sediment quality in 100 percent of the designated critical habitat and is reasonably certain to impair the capability of the critical habitat to provide for the normal behavior, reproduction, and survival of the Kootenai River white sturgeon. For this reason, this impact is considered to be significant.

## 2.5.2.6 Evaluation of the Protectiveness of the 10 µg/L Human Health Criterion

The information discussed above in section 2.5.2 clearly indicates the potential for adverse effects to be caused to listed species and to the primary constituent elements of their designated critical habitat as a result of exposure to arsenic at the proposed chronic aquatic life concentration of 150 µg/L. While human-health based criteria are not the focus of this Opinion, in instances such as this where the aquatic life criterion is not protective, but a lower, human-health based criterion is both applicable and more stringent, the human health criterion becomes relevant to this analysis (see section 2.1.5.6, *Application of Human Health Criteria*). This leads to the following question: would the human-health based criterion of 10 µg/L, as unfiltered inorganic arsenic, be sufficiently protective to avoid adverse effects to the bull trout, bull trout critical habitat, the Kootenai River white sturgeon, and to Kootenai River white sturgeon critical habitat?

Whether the 10 µg/L human-health based criterion for arsenic is sufficiently protective to the above listed species and critical habitat is not immediately evident. While it is much lower than the proposed chronic criterion, in some field settings, adverse effects to fish, or at least elevated arsenic in prey organisms, were reported from locations where the 10 µg/L criterion was only slightly exceeded, if at all (Table 5).

Two lines of available evidence seem relevant to this question: (1) variability in arsenic concentrations in ambient conditions, and (2) the fraction of highly toxic inorganic arsenic or low toxicity organic arsenic in prey items. Concentration variability is relevant to interpreting field studies of contaminant concentrations or effects because field biological studies are often only able to sample once or a few times, and thus the concentration at the time of sampling is not always representative of the previous conditions that actually exposed the organisms. The forms

[15] PCE 4: Water and sediment quality necessary for normal behavior, including breeding behavior, and viability of all life stages of the Kootenai River white sturgeon, including incubating eggs and yolk sac larvae.

of arsenic present in potential prey organisms is relevant to interpreting the protectiveness of criteria because the laboratory studies testing effects of inorganic arsenic may differ from the forms typically found in the environment. If arsenic accumulating in algae and prey organisms (benthic macroinvertebrates and small fish) is primarily inorganic, this would imply high risk to consumers of concern (e.g. bull trout and white sturgeon), whereas if the organisms are accumulating primarily organic arsenic, this implies much less risk.

### Seasonal Variability of Arsenic in Water

The evaluation of arsenic concentration variability in water was limited, owing to the inability to locate many reliable, time series datasets relevant to the action area. Three highly relevant datasets were examined from the following localities: (1) the East Fork of the South Fork Salmon River (EFSFSR), at Stibnite, Idaho (http://waterdata.usgs.gov/nwis, site 13311000); (2) the Snake River above Jackson Lake, WY 13010065 (http://waterdata.usgs.gov/nwis, site 13010065); and (3) Blackbird and Panther Creeks in central Idaho (Golder 2010). The EFSFSR setting reflects mining-related disturbance, and the Snake River site, immediately downstream of the Yellowstone National Park boundary, reflects natural geothermal sources. The EFSFSR is actually one of five sites with similar datasets. Upon inspection, all five showed similar patterns and only one is shown here. In datasets (1) and (2) the arsenic source was primarily groundwater-based (constant source), while in dataset (3) the arsenic source was derived from mining-related sediments in Blackbird Creek and Panther Creek that were mobilized by runoff events.

Both source scenarios showed that arsenic concentrations tended to be highest at low flows and lowest at high flows (Figure 4). This indicates that in these two settings, arsenic was primarily of groundwater origin, with the lower concentrations during snowmelt indicating dilution. Maximum measured concentrations were about 2X higher than average concentrations in both datasets.

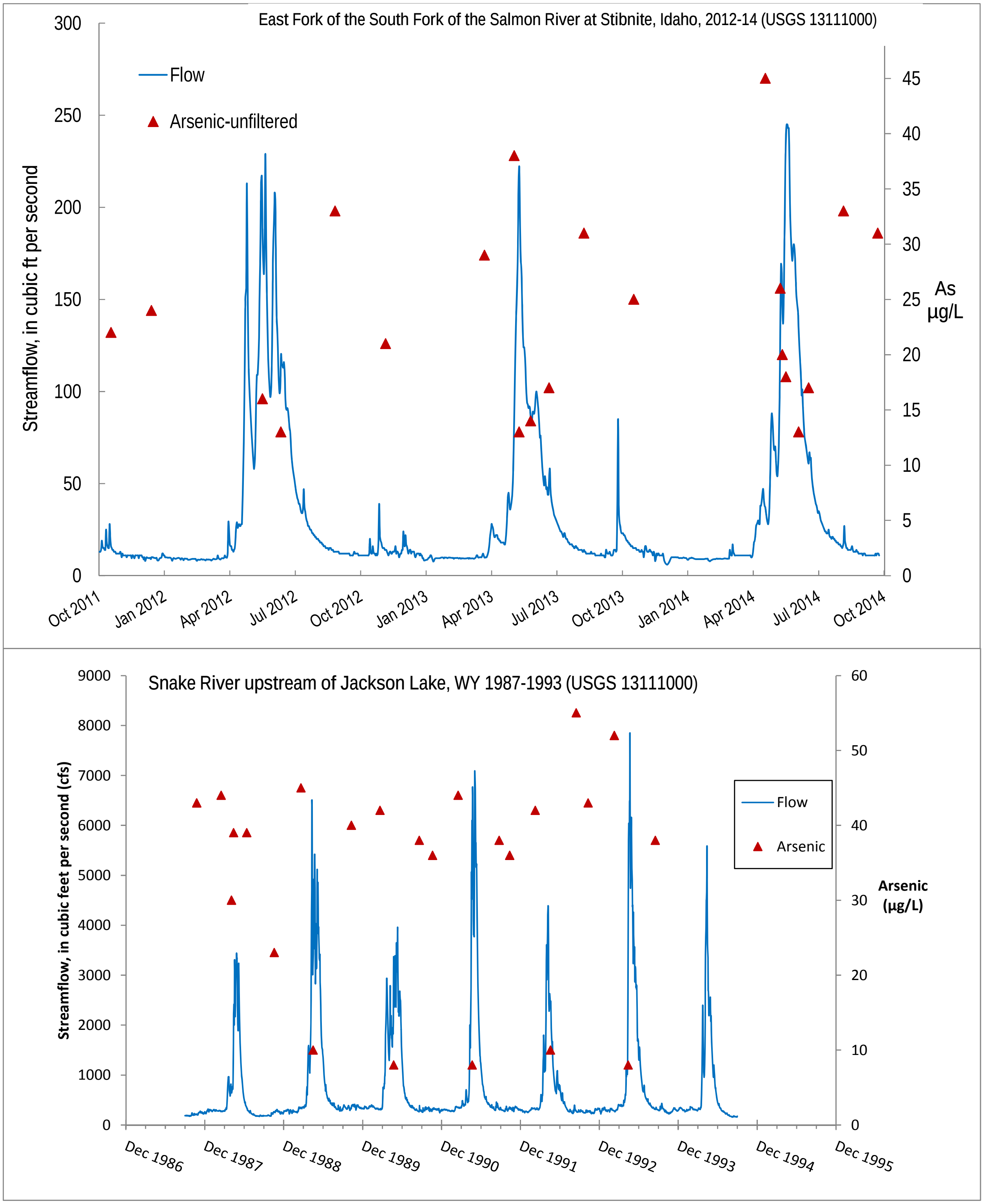


**Figure 4. Arsenic versus streamflow time series from two river settings, in which arsenic sources are from groundwater, and close to 100 percent of the total arsenic is present in a dissolved form.**

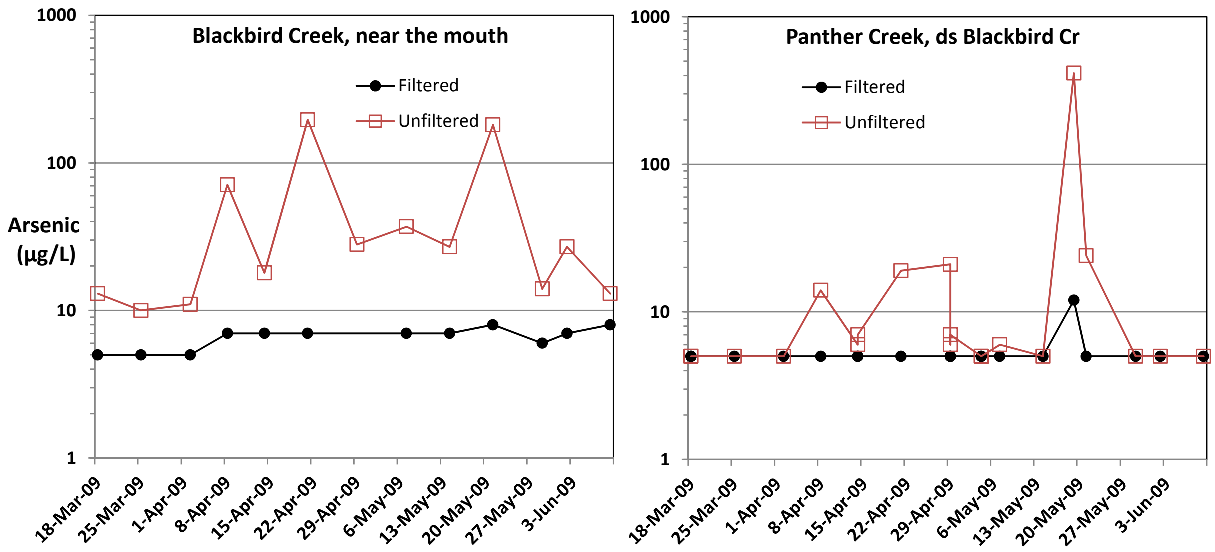


**Figure 5. Unfiltered ("total") and filtered ("dissolved") arsenic concentrations from stream settings in which arsenic sources are mostly derived from snowmelt runoff remobilization of arsenic-laden particles from the streambed and stream banks on Panther Creek and its tributary, Blackbird Creek, in central Idaho. During the runoff period, on average, about 25 percent of detected arsenic values were in a dissolved form, and at peak concentrations, <5 percent of the total arsenic present was in a dissolved form. Non-detects are plotted as the arsenic detection limit of 5 µg/L. Data from Golder (2010).**

An implication of these patterns is that in streams where the arsenic concentration is derived primarily from groundwater, arsenic concentrations from one-time field surveys conducted during mid-summer under dry conditions would reflect groundwater fed conditions with no dilution, and thus represent close to maximum exposure conditions.

A contrasting situation is shown with total (unfiltered) and dissolved arsenic from streams where the arsenic sources appear to result mostly from resuspended sediment and soil particles during spring snowmelt (Figure 5). In this setting, while total and dissolved arsenic concentrations were similar during low-flow conditions, during the runoff season total arsenic reached>100 µg/L, the filtered fraction was much lower. For instance, on 19 May 2009, total-unfiltered arsenic in Panther Creek reached 415 µg/L, whereas the dissolved (filtered) fraction was only 12 µg/L, or 3 percent of the total (Figure 5). Because the dissolved (filtered) form of arsenic is considered more readily taken up through the food web than particulate arsenic (EPA 2004b), this suggests that in settings where mobilized sediments or bank soils are the primary arsenic source, the more readily bioavailable filtered fraction will likely make up a minority of the total (unfiltered) arsenic.

In filtered water samples collected from a randomized survey of Idaho rivers, 73 percent of the total arsenic present consisted of inorganic arsenic (range 25-100%, n=34) (Essig 2010).

### Forms of arsenic in wild aquatic organisms: predominance of inorganic (more toxic) or organic (less toxic) forms

Several studies were reviewed relevant to the form of arsenic likely to occur in the food web utilized by listed species. Arsenic data were located for several taxa groups across trophic levels (algae, grazing invertebrates, predatory invertebrates, and fish). As discussed below, the available data indicate that in most, but not all, settings, the total arsenic in tissue residues of aquatic animals at the second trophic level or higher was dominated by the less toxic organic form.

With algae (the first trophic level), the form of arsenic appears to reflect that present in water: the greater the amount of inorganic arsenic in water, the greater the amount of inorganic arsenic in algae. In natural settings with low-level arsenic concentrations, arsenic was dominated by organic forms, but in a mining-disturbed lake setting with greatly elevated inorganic arsenic, inorganic forms of arsenic were present at elevated concentrations in the algae as well (Phillips 1990; Caumette et al. 2011). In second-trophic level zooplankton feeding on algae that contained only inorganic arsenic, some of the arsenic had been transformed to organic but was still dominated by inorganic arsenic (65% vs 35%) (Caumette et al. 2011; Caumette et al. 2012).

Among non-insect benthic invertebrates, inorganic arsenic made up about 25 percent of the total arsenic present in crayfish (whole bodies) (Devesa et al. 2002). Similar fractions of inorganic arsenic were found in mussels in mining-contaminated estuaries (≈ 33 percent , Whaley-Martin et al. 2012), and amphipods found in association with mining-contaminated lake sediments (≈ 33 to 50 percent, Moriarty et al. 2014).

For aquatic insects, two studies were located that addressed arsenic speciation. Mebane et al. (2015) reported that inorganic arsenic in aquatic insects collected from Panther Creek, Idaho averaged about 50 percent of total arsenic, ranging from 20 – 80 percent, based on speciation data from two species of stonefly and one caddisfly species. In the caddisfly, about 50 percent of the total arsenic was in the less-toxic organic form, and the highest fraction of inorganic arsenic was found in the predatory *Hesperoperla* stonefly (Mebane et al. 2015). Kaise et al. (1997) investigated arsenic speciation in abiotic and biotic components of a river with groundwater arsenic sources, and found that while 93 percent of total river water arsenic was inorganic, only about 10 percent of the total arsenic in green algae and diatoms was present as inorganic arsenic. In the same investigation, two insect species, a caddisfly and a dobsonfly were analyzed, with <10 percent of the total arsenic present as inorganic arsenic (Kaise et al. 1997). Arsenic species in freshwater fish tissue appear to be dominated by organic arsenic, based on two fairly comprehensive studies. From a randomized study of multiple species in Idaho rivers, Essig (2010) reported that >96 percent of arsenic in fish muscle (fillets) was organic (range 86-99 percent, n=55). Similarly, inorganic arsenic made up <3 percent of the total arsenic in 89 composite samples of 10 fish each, representing 21 species of fish, collected from 50 lakes across Idaho (Essig and Kosterman 2008). Kaise et al. (1997) similarly reported organic arsenic contributing >95 percent of the total arsenic in six species of fish. Therefore, arsenic in fish tissue is considered unlikely to pose a risk to predators such as bull trout. In summary, the information located on the different forms of arsenic present in potential prey organisms for the bull trout or the white sturgeon showed that arsenic levels in zooplankton collected from a mining-affected lake was dominated by the more toxic inorganic form. In contrast, the forms of arsenic found in other aquatic invertebrates were mostly (but not always) dominated by the less-toxic organic forms. In forage fish, virtually all arsenic has been reported to be in the organic (less-toxic) form.

**Summary**

Maximum dissolved arsenic concentrations in settings where the arsenic is derived from runoff water appear to be on the order of 2X as high as average arsenic concentrations in settings where the source of arsenic is from groundwater sources. In a setting with arsenic contamination resulting from erosion of bank soils or movement of streambed sediments, most arsenic remained in a particulate form, which is considered to have low bioavailability.

In tissues of aquatic organisms that represent potential prey items for the bull trout or the white sturgeon, the fraction of total arsenic that was made up by the more toxic inorganic form ranged from <3 percent to 80 percent, however, in most of data reviewed, inorganic arsenic made up less than 50 percent of the total arsenic in invertebrates and in fish. It follows that the dietary concentrations of *inorganic* arsenic shown to be harmful in laboratory feeding experiments to the rainbow trout and other species would translate to about 2X or higher *total* arsenic in stream insects, that is >40 mg/kg. In most settings where matched data could be assembled, benthic macroinvertebrate samples with total arsenic >40 mg/kg dw usually were associated with unfiltered arsenic samples in water >10 µg/L.

While the available data were far from comprehensive and were not completely consistent, the seasonal and source (i.e., groundwater v snowmelt) variability of arsenic concentrations in water and the predominance of organic (less toxic) arsenic that was bioaccumulated in the food-web suggest that if unfiltered, the inorganic arsenic concentrations in streams are seldom likely to exceed 10 µg/L, and total arsenic residues in potential bull trout or sturgeon prey items would be expected to be less than ~40 mg/kg dw. Thus, for these reasons, it is concluded that the 10 µg/L unfiltered, inorganic recreational use (human-health) based arsenic criterion is unlikely to cause significant adverse effects to the bull trout, bull trout critical habitat, Kootenai River white sturgeon, and to Kootenai River white sturgeon critical habitat.

# 2.5.3 Copper Aquatic Life Criteria

The proposed acute and chronic criteria values for copper (Cu) are 17 and 11 µg/L, respectively, as calculated from the following equations using a hardness value of 100 mg/L:

$$\text{Acute Cu criterion } (\mu g/L) = e^{(0.9422[\ln(\text{hardness})]-1.464)} * 0.96$$

$$\text{Chronic Cu criterion } (\mu g/L) = e^{(0.8545[\ln(\text{hardness})]-1.465)} * 0.96$$

The proposed acute and chronic criteria values for copper are also referred to as the "criterion maximum concentration" (CMC) and "criterion continuous criterion" (CCC) respectively (EPA 1985c, 1999a). With copper and several other hardness-dependent aquatic life criteria, the actual criteria are defined as an equation, and the table values merely illustrate comparable criteria concentrations, all calculated at a hardness of 100 mg/L. For example, at water hardness values of 10, 25, 50, and 250 mg/L, the acute copper criterion equation produces copper acute values of 4.6, 4.6, 8.9, and 40 µg/L. With the chronic criterion, the same water hardness values of 10, 25, 50, and 250 mg/L produce chronic criterion values of 3.5, 3.5, 6.3, and 25 µg/L. In this example, the values calculated for the hardnesses of 10 and 25 mg/L are the same because of the "hardness floor," a separate part of this action which arbitrarily limits the lowest hardness values used in the equations to 25 mg/L, regardless of actual measurements. Copper occurs naturally in the environment and in waters of the United States away from the immediate influence of discharge; natural copper concentrations typically range from about 0.4 to 4 µg/L (Stephan et al. 1994).

In ecotoxicology, there is a wealth of information related to copper, which is likely a result of copper's importance to society and its potency in the aquatic environment. Features and uses of copper include its high toxicity to some organisms, ubiquity in the environment, important role in manufacturing anti-biofouling and corrosion resistant materials, electrical conductivity, and agricultural uses as a pesticide with low risks to humans (ATSDR 2004, entire). Copper is used by a number of enzymes, which make it an essential element for all aerobic organisms. Copper is also a potent toxicant and, as a result, aquatic organisms have developed delicate homeostatic controls to maintain a balance of copper between deficiency and toxicity levels (Grosell 2011). Copper was recognized as being toxic to aquatic organisms, particularly molluscs and algae, by at least the 1700s when shipbuilders began adding copper cladding to wooden hulls to reduce damage from wood boring molluscs (Dürr and Thomason 2009, p. 217).

In short-term exposures to copper, the risk of copper toxicity appears to be primarily related to ionoregulatory disruption resulting from copper interfering with sodium uptake, and in fish, from copper impairing sensory function through impairment of chemo-olfaction and damage to mechano-reception in olfactory and lateral line cilia (Hecht et al. 2007, Grosell 2011, Wood 2011a). In long-term exposures to copper, the mechanisms of toxicity are not well understood. Sublethal effects can result from exposure to copper at concentrations well below those causing ionoregulatory disruptions, which suggests that in chronic exposures, the fish's health gradually "runs down" owing to the energy costs of dealing with a combined load of many cellular and organ-level disturbances. Though not quantified, sublethal effects that cause an exposed fish to be "run down" may manifest themselves as reduced growth, reproductive output, and swimming performance (Grosell 2011; Wood 2011a).

The proposed action relies on EPA's 1984 version of their copper criteria (EPA 1985b, entire). Although EPA (1985c) noted that organic carbon and other factors were sometimes reported to have more effect on copper toxicity than hardness, consistent with criteria developed for other metals at the time, the criteria were expressed as a function of water-hardness. In the 30 years since EPA's (1985b) copper criteria were developed, much research on the toxicity of copper to aquatic organisms has been conducted. An important outcome of this research is the clear demonstration that copper toxicity is not simply a function of the environmental copper concentrations, but factors such as complexation between positively charged copper and negatively charged organic and inorganic particles or molecules such as clays and dissolved organic carbon (DOC), influence toxicity. Likewise, competition between positively charged copper and other cations such as pH and calcium reduces copper toxicity (Chapman and McRady 1977; Erickson et al. 1996; Grosell 2011).

As the recognition of the importance that factors such as pH and DOC have on copper toxicity through their roles affecting the speciation, competition, and complexation of copper, the old hardness-based criteria approach has come under severe criticism. The nature of the criticisms center in three areas:

(1) Negligible effects of water hardness on copper toxicity and the failure of the hardness-based criteria to track changes in copper toxicity in natural waters (De

Schamphelaere and Janssen 2002; Apte et al. 2005; Hyne et al. 2005; Markich et al. 2006; Wang et al. 2009; NMFS 2014b),

(2) Failure of the hardness-based criteria formulation to protect sensitive organisms (Markich et al. 2005; March et al. 2007; Ingersoll and Mebane 2014; NMFS 2014a), and

(3) Chemosensory toxicity and related maladaptive behaviors, such as the lack of predator avoidance, by aquatic organisms exposed to copper concentrations that may occur at lower than the hardness-based criteria (Hecht et al. 2007; Meyer and Adams 2010; McIntyre et al. 2012; NMFS 2014a).

Research in these three areas has led to more sophisticated (and more complicated) approaches to define the factors that modify copper toxicity to aquatic organisms. These approaches include the "Biotic Ligand Model" (BLM) that was subsequently incorporated into EPA's 2007 updated aquatic life criteria for copper (Di Toro et al. 2001; Santore et al. 2001; EPA 2007b).

#### 2.5.3.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

Several copper toxicity studies have been conducted with snail species from within the same families to which the listed Snake River aquatic snails belong, i.e., the family Hydrobiidae (Bliss Rapids snail and Bruneau hot springsnail), family Physidae (Snake River physa), and the family Lymnaeidae (Banbury Springs lanx). The Banbury Springs lanx is a freshwater limpet that has yet to be formally described as a species and thus the taxonomic classification of this freshwater limpet is not well documented. USFWS (2006b) considered it to be within the family Lymnaeidae although freshwater limpets have also been classified within the family Planorbidae (Pennak 1978).

Air-breathing snails of the subclass Pulmonata (e.g., the families Physidae, Lymnaeidae, and Planorbidae) have been the most widely used snails for laboratory toxicity tests. Their rapid growth, short generation times, and high reproductive output make them easy to use in toxicity tests, including chronic tests with sensitive, sublethal endpoints. Non-pulmonate snails (formerly included in subclass Prosobranchia, which includes the family Hydrobiidae ) are more taxonomically diverse and their physiology (inability to breathe atmospheric oxygen) and life history (slow growth and low reproductive rate) may make them both subject to endangerment and difficult to culture and test in the laboratory (Besser et al. 2009).

The following studies that evaluated the response of Hydrobiidae snails to copper exposure showed toxicity at lower concentrations than the proposed chronic water quality criterion for copper: Besser et al. (2009) tested the response to copper by three Snake River hydrobiidid snail species: the Bliss Rapids snail; the Jackson Lake springsnail (*Pyrgulopsis robusta*, formerly known as the Idaho springsnail (*Prygulopsis idahoensis*)), and a pebblesnail (*Fluminicola* sp.). The Ozark springnail, *Fontigens aldrichi (*Hydrobiidae) was also tested. The tests were conducted in parallel with a potential surrogate species, the easily cultured pulmonate pond snail (*Lymnaea stagnalis*). Tests were conducted for 28 days in moderately hard water with a hardness of about 170 mg/L, for which the corresponding chronic copper criterion concentration is 18 µg/L. The Jackson Lake springsnail was successfully cultured in captivity and was tested for responses to copper exposure by both juveniles and adults. Efforts to culture the Bliss Rapids snail in sufficient numbers to support testing different life stages were unsuccessful.

At the proposed chronic copper criterion concentration calculated for the test waters (18 µg/L), 20 percent of the Bliss Rapids snail were killed and greater than 50 percent of the Jackson Lake springsnails and pebblesnails were killed. The most sensitive hydrobiidid response obtained was a 20 percent reduction in growth of Jackson Lake springsnails exposed to a copper concentration of 7.4 µg/L (Besser et al. 2009). A hydrobiidid snail collected in Oregon was also very sensitive to copper. Nebeker et al. (1986) observed reduced survival in the hydrobiidid snail *Lithoglyphus virens* following a 6-week exposure to 4 µg/L of copper in water with a hardness of 20 mg/L. The corresponding chronic copper criterion value is 3.5 µg/L, which is essentially the same concentration as 4 µg/L, especially considering that Nebeker et al. (1986) only reported their values to one significant digit (0.004 mg/L).

Lymnaeid snails are also very sensitive to copper. Roussel (2005) followed community changes in experimental stream ecosystems that were dosed with copper for 18 months. The most sensitive macroinvertebrate species affected were gastropods such as *Lymnaea* spp., and *Physa* sp. Abundances of *Lymnaea* spp., and *Physa* were reduced in the treatment areas with an average measured copper concentration of 20 µg/L, but the abundance of these species was not affected at a copper concentration of 4 µg/L. The proposed chronic copper criterion concentration calculated for the average test hardness of 342 mg/L is 33 µg/L, indicating that the proposed chronic criterion for copper is not likely to be protective of listed snails. Brix et al. (2011b) reported that *L. stagnalis* was the most sensitive freshwater organism tested to date with copper, with a 20 percent reduction in the growth (EC20) of exposed individuals occurring at a copper concentration of 1.8 µg/L and with 100 percent of exposed individuals killed at a copper concentration of 14 µg/L. For a laboratory test water hardness of about 102 mg/L, the corresponding chronic copper criterion is 12 µg/L, indicating *L. stagnalis* was significantly under-protected by the hardness-based proposed water quality chronic criterion for copper. Besser et al. (2009) also observed adverse effects to aquatic snails at sub- proposed criteria copper concentrations in one of two tests conducted with *L. stagnalis*: Growth was reduced by 20 percent at a copper concentration of 6.2 µg/L in one test and at 22 µg/L in the second test, compared to the proposed chronic criterion concentration for copper of 18 µg/L. A set of particularly comprehensive tests with the closely related Indian pond snail, *L. luteola*, showed a host of adverse effects following exposures to copper concentrations as low as 3.2 µg/L. These adverse effects included loss of chemoreception (so that the snails were no longer attracted to food), feeding inhibition, reduced growth and reduced reproductive output. The survival of exposed snails was reduced at a copper concentration of 10 µg/L (Khangarot and Das 2010; Das and Khangarot 2011). The proposed chronic criterion for copper of 23 µg/L calculated for a test water hardness of 230 mg/L is much higher than copper concentrations sufficient to kill or inhibit the behavior of exposed *L. luteola.*

A long-term copper toxicity test involving a physid snail suggests that snails in the genus *Physa* are almost as sensitive to copper toxicity as the hydrobiidid and lymnaeid snails. Arthur and Leonard (1970) obtained a no observed effects concentration (NOEC) of 8 µg/L for snail reproduction after 6 weeks of snail exposure to copper in water with a hardness of 35-55 mg/L, which is only slightly higher than the proposed chronic criterion for copper of 4.6 to 6.8 µg/L at those hardness values.

Studies of the effects of long-term copper exposure to freshwater molluscs from other families besides the three to which the listed Snake River snails and the Bruneau hot springsnail belong (the Hydrobiidae, Physidae, and Lymnaeidae) also indicate a high sensitivity to copper. For instance, Reed-Judkins et al. (1997) observed 80 percent mortality of the snail *Leptoxis praerosa* (Pleuroceridae) during a 114-day exposure to a copper concentration at ~ 50 percent of the CCC (6.3 µg/L vs. the proposed chronic criterion of 12 µg/L at a water hardness of 110 mg/L) compared to close to 0 percent mortality in the control group. Nebeker et al. (1986) found that the threshold for reduced survival for the pleated juga snail (*Juga plicifins* Semisulcospiridae) was less than 8 µg/L of copper, which is higher than the proposed chronic criterion of 3.5 µg/L. However, because adverse effects were noted at the lowest concentration tested, the true threshold therefore must be lower than any concentration tested. In addition to snails, March et al. (2007) reviewed the protectiveness of hardness-based copper criteria to freshwater mussels and found they were often underprotective. The 1996 version of EPA's hardness-based copper criteria that were evaluated by March et al. (2007), were slightly lower but otherwise similar to the 1985 version used by Idaho and evaluated in this Opinion.

The preceding examples of effects of copper to snails have been with long-term studies relevant to the proposed chronic criterion. Most, but not all, relevant studies reviewed with short-term tests resulted in lethality at concentrations higher than the Final Acute Value used to derive the proposed acute criterion for copper (Arthur and Leonard 1970; Nebeker et al. 1986; Brix et al. 2011b; Das and Khangarot 2011). However, none of the acute toxicity tests involving snails and copper necessarily tested the most sensitive early-life stages (e.g., veligers) of the test snails that are likely the most acutely sensitive stage to metals (Gomot 1998). In freshwater mussels, glochidia are the comparable life stage to snail veligers. Mussel glochidia are sometimes more sensitive to copper than older mussels (March et al. 2007), suggesting that even if the proposed acute copper criterion were protective of older snails it may not always be protective of veligers or other difficult to test the early-life stages of snails.

In summary, best available information indicates that exposure to copper at the concentrations and durations allowed by the proposed acute and chronic copper criteria is likely to cause severe, adverse effects (including mortality) to listed Snake River aquatic snails and to the Bruneau hot springsnail. Since the action area represents the entire ranges of these species, these impacts are likely to appreciably reduce their reproduction, numbers, and distribution at a range-wide scale.

## 2.5.3.2 Bull Trout

Research relevant to evaluating the effects of copper to the bull trout includes copper toxicity testing relative to bull trout growth and survival (Hansen et al. 2002a, b). Other potential effects of copper to the bull trout, such as chemoreception and behavioral alterations, discussed below are based on research with other salmonid species (e.g., Hecht et al. 2007).

Tests with bull trout growth and survival following copper exposures in hard water showed no adverse effects at concentrations lower than the proposed acute and chronic criteria (Hansen et al. 2002a; Hansen et al. 2002b). For instance, no effects on bull trout growth were observed at a copper concentration of 111 µg/L after 2 months of exposure, which is well above the proposed chronic criterion concentration of copper of 22 µg/L for a water hardness value of 220 mg/L. Tests evaluating the effects of acute exposure to copper were conducted in parallel with rainbow trout in moderately-hard and very-hard waters (hardness values of 100 mg/L and 220 mg/L, respectively). The two species had similar sensitivity to copper in water with a hardness value of 100 mg/L, but bull trout were 2.5 to 4 times less sensitive than rainbow trout in water with a hardness value of 220 mg/L (Hansen et al. 2002a, b). In general, hard waters are less common than soft waters in habitats occupied by the bull trout; this matter is further discussed in more detail in section 2.5.8 below.

Tests of the effects of copper exposure on brook trout growth and reproduction in soft water (hardness 45 mg/L, for which the proposed chronic criterion for copper would be 5.7 µg/L) showed considerably more sensitive results relative to the proposed chronic copper criterion than did the hard water tests on the bull trout. McKim and Benoit (1971) found that during the first 23 weeks of life, the growth of brook trout exposed to copper was reduced at all copper concentrations tested (3.2 µg/L and above). However, from week 23 to the end of the study at week 27, the copper-exposed fish caught up in growth, and McKim and Benoit (1971) did not consider the transient growth reductions to be adverse effects. However, others have reasonably argued that such growth delays are not necessarily biologically discountable, if they disadvantage young fish in size-related contests, such as capturing prey, avoiding becoming prey, and obtaining and holding winter shelter (e.g., Metcalfe and Monaghan 2001; Mebane and Arthaud 2010).

Chemoreception, electromechanical function, olfactory function, and dependent critical behaviors such as alarm response, predatory avoidance, rheotaxis, and migratory movements are important in salmonids and probably all fishes (Hecht et al. 2007; Tierney et al. 2010). No studies that specifically tested bull trout and chemoreception related responses to copper are known to exist, but studies with other salmonids and other fishes indicate that neither the proposed acute or chronic hardness-based water quality criteria for copper can be considered protective of chemoreception and related functions (Hecht et al. 2007; Meyer and Adams 2010; McIntyre et al. 2012; NMFS 2014a, 2014b). Owing to the importance of chemoreception and related functions in the life histories of fishes, we conclude that the proposed aquatic life criteria for copper are likely to result in significant adverse effects to the bull trout.

Given that existing data show adverse effects to other salmonids occurring at copper concentrations below the proposed copper criteria, we conclude the proposed acute and chronic criteria for copper are likely to cause significant adverse effects to bull trout growth and behavior. Given that 44 percent of the streams and 34 percent of the lakes and reservoirs occupied by the bull trout rangewide occur within the action area, the proposed copper criteria are likely to cause a reduction in the reproduction, numbers, and distribution of the bull trout within a large portion of its range.

## 2.5.3.3 Bull Trout Critical Habitat

Of the nine PCEs defined for bull trout critical habitat, three are likely to be adversely affected by the proposed acute and chronic criteria for copper: PCE 2 (migration corridors), PCE 3 (adequate prey base), and PCE 8 (water quality).

As discussed above for the bull trout, the proposed criteria for copper are likely to create water conditions in bull trout habitat that adversely affect chemoreception, electromechanical function, olfactory function, and dependent critical behaviors such as alarm response, predatory avoidance, rheotaxis, and migratory movements are important in salmonids and probably all fishes (Hecht et al. 2007, Tierney et al. 2010). As proposed, the acute and chronic concentrations of copper are also likely to create chemical barriers (due to impacts to chemoreception) that would preclude bull trout migration and movement between various types of habitat (e.g., movement to predator and thermal refugia). Bull trout currently exist in numerous small, scattered populations. The ability of the species to move within its range, disperse between populations, and to recolonize former habitat is essential to the survival and recovery of the species. Localized concentrations of copper or areas where copper combines with other chemicals, or where other water quality parameters (pH, temperature, etc.) increase the adverse effects of copper, are likely to create habitat conditions that prevent movement of bull trout within and between populations, precluding the capability of the critical habitat to support normal behavior of the bull trout with respect to migration (PCE 2), breeding, feeding, and sheltering. An avoidance response by affected fish exposed to such habitat conditions, as was observed in studies discussed above and at copper concentrations below the proposed criteria for copper, is indicative of a chemical concentration that represents a chemical barrier to normal movement patterns.

As discussed above for the bull trout, habitat conditions supporting abundant prey species (i.e., -small fish, including salmon fry, as well as terrestrial and aquatic insects, and macro-zooplankton) (Boag 1987; Goetz 1989; Pratt 1992, p. 6; Donald and Alger 1993) are important to creating and maintaining robust bull trout populations. This relationship is shown by the correlation between declines in bull trout abundance and declines in salmon abundance (Rieman and McIntyre 1993, p. 3); as noted above, salmon fry are an important food source for the bull trout. As discussed earlier, avoidance responses have been documented in both salmonids and non-salmonids in response to elevated copper concentrations in the aquatic environment. Should potential prey species avoid areas with elevated copper concentrations, available prey for the bull trout would be reduced in numbers and distribution, thereby adversely affecting the ability of bull trout critical habitat to provide for an abundant food base (PCE 3) for the bull trout.

The proposed action will impair water quality (PCE 8) by allowing aquatic copper concentrations to rise to levels that have been shown to be detrimental and even lethal to other salmonids. Adverse effects to salmonids were observed at dietary concentrations of copper below the proposed criterion; see the discussion above on the bull trout. Assuming bull trout are affected in a similar manner as other salmonids, copper concentrations at the proposed acute and chronic criteria levels are likely to impair the capability of critical habitat to provide habitat conditions supporting normal reproduction, growth, and survival of the bull trout.

Because the proposed water quality criteria for copper would apply statewide, all bull trout critical habitat within the state of Idaho is likely to be subject to harmful aquatic copper concentrations under the proposed acute and chronic criteria. Within the conterminous range of bull trout, a total of 19,729 miles of stream and 488,252 acres of lakes and reservoirs are designated as critical habitat. The state of Idaho contains 8,772 miles of streams and 170,217 acres of lakes and reservoirs designated as critical habitat (75 FR 63937). On that basis, the proposed criteria for copper are likely to impair the capability of approximately 44 percent of the total critical habitat designated for the bull trout along streams and 35 percent of the total critical habitat designated for the bull trout in lakes and reservoirs to adequately provide for bull trout migratory corridors (PCE 2), an adequate prey base (PCE 3), and adequate water quality (PCE 8) essential for bull trout recovery.

# 2.5.3.4 Kootenai River White Sturgeon

Recent research has shown that the Columbia River white sturgeon are highly susceptible to copper toxicity, to the point that the white sturgeon may be the most copper sensitive freshwater fish species tested to date. The age or developmental stage of the white sturgeon is a key factor in its susceptibility to copper, with the youngest fish being the most sensitive (e.g., Little et al. 2012; Vardy et al. 2013; Ingersoll and Mebane 2014; Vardy et al. 2014).

Adverse effects to Columbia River white sturgeon following short-term copper exposures at considerably lower concentrations than the proposed acute copper criterion have been documented for this species. For example, Calfee et al. (2014) reported that for the youngest fish tested (2-days post-hatch (dph)), a 50 percent effects concentration (EC50) of 2.7 µg/L was obtained, in which the adverse effects were defined to include death, loss of equilibrium, or loss of mobility of exposed white sturgeon. For the test water hardness of about 100 mg/L, the acute copper criterion would be 17 µg/L. In the test with the 2-dph fish, 100 percent of the fish were adversely affected at the acute criterion concentration. Few 2-dph fish were killed outright, but all exposed fish exhibited a loss of mobility and a lack of equilibrium. Similarly, with 16-dph and 30-dph white sturgeon, the EC50s of 4.3 and 6.3 µg/L were far lower than the proposed acute criterion, and at the criterion concentration of 17 µg/L, a 100 percent incidence of adverse effects was observed. Additionally, the test with 16-dph and 30-dph fish, 93 percent and 50 percent were killed outright, respectively, by the exposure to copper at an acute criterion concentration of 17 µg/L (Calfee et al. 2014). Additionally, Vardy et al. (2013, 2014) reported close to 50 percent mortalities of 15-dph white sturgeon with LC50s of 9 to 10 µg/L of copper in water with a hardness of about 70 mg/L for which the acute criterion concentration would be about 12 µg/L. Thus, from these tests, greater than 50 percent mortality of exposed young white sturgeon is likely to occur under short-term exposures to the proposed acute copper criterion concentration.

Adverse effects to white sturgeon subject to long-term copper exposure at considerably lower concentrations than the proposed chronic criterion have also been reported. For example, in 14-day exposures starting with 2-dph fish, Wang et al. (2014a) observed 20 percent mortality at 2.2 µg/L and 94 percent mortality at 6.8 µg/L. By comparison, the proposed chronic criterion copper concentration for a test water hardness of about 100 mg/L is 11 µg/L. In a 53-day exposure with 2-dph fish, growth as weight was reduced by 20 percent at a copper concentration of 1.6 µg/L, and 100 percent of the fish were killed at a copper concentration of 7.2 µg/L treatment. Because in the 53-day exposure test with 2-dph fish, only 68 percent of the control fish survived and the test acceptability criterion for a "definitive" test was 70 percent survival of the control group of fish, the test was repeated. The repeat test used older, more robust 27-dph fish and was only conducted for 28 days. In this test, a 20 percent reduction in growth as weight was observed at a copper concentration of 2.7 µg/L, a 20 percent reduction in survival was observed at a copper concentration of 4.2 µg/L, and 87 percent of the fish exposed to the copper concentration of 7.3 µg/L died, compared to a 10 percent mortality rate for the control group of fish. Thus, for the proposed chronic copper criterion concentration of 11 µg/L at a water hardness of 100 mg/L, about a 100 percent kill rate of the early life stage of the white sturgeon would be expected (Wang et al. 2014a).

Additionally, in semi-quantitative 60-day tests with 19-dph white sturgeon, the findings reported by Vardy et al. (2011) indicate that between 20 percent and 50 percent of the tested fish would be killed at a chronic copper criterion concentration of 8.4 µg/L for water with a hardness of 70 mg/L.

The above findings support a conclusion that the proposed copper criteria are likely to create habitat conditions that adversely impact the survival and reproduction of the Kootenai River white sturgeon in Idaho, which represents 39 percent of its range. On that basis, the effects of the proposed copper criteria on this species are considered significant.

### 2.5.3.5 Kootenai River White Sturgeon Critical Habitat

Because the proposed action contains no provision addressing copper concentrations in sediment, sediment concentrations of copper are likely to rise to levels that will adversely affect exposed individuals (particularly eggs and juveniles) of the white sturgeon. Sediment quality is critically important to the health of white sturgeon critical habitat because all life stages of the sturgeon are extensively exposed to sediments, either through dermal contact (all life stages) or through incidental ingestion while feeding (juveniles and adults). An elevated copper concentration in sediment is also likely to influence the concentration of copper in the overlying water because extensive interactions between surficial sediment and the overlying water occur in any waterbody, always with movement toward equilibrium (Rand et al. 1995, pp. 14-15; Walker et al. 1996, p. 47).

Based on the above findings and those reported in the preceding section above, the proposed criteria for copper are likely to create habitat conditions within 100 percent of Kootenai River white sturgeon critical habitat that are likely to impair water quality and sediment to an extent that kills the early life stages of the sturgeon and impairs or compromises its ability to successfully reproduce in the wild.

## 2.5.4 Cyanide Aquatic Life Criteria

The proposed acute and chronic aquatic life criteria for cyanide are 22 µg/L and 5.2 µg/L, respectively, as weak-acid-dissociable cyanide (EPA 1999a).

The cyanide group (CN) includes free cyanide (HCN and $CN^-$), simple cyanide salts (e.g., KCN, NaCN), metal-cyanide complexes, and some organic compounds. The most bioavailable and toxic forms are free cyanide (Gensemer et al. 2007). EPA's (1985c) proposed criteria considered cyanide toxicity to mostly result from HCN. However, because the cyanide ion ($CN^-$) readily converts to HCN at ambient pH values the cyanide criteria proposed in 1985 were stated in terms of free cyanide expressed as $CN^-$. Cyanides can be released into the environment from both natural and anthropogenic sources, including biomass burning, road salts, and ore extraction from gold mining (Barber et al. 2003; USFWS 2010b; Pandolfo et al. 2012).

Free cyanide is extremely toxic and fast acting, and its fast action was one reason for EPA's (1985c) expression of the acute criterion based on 1-hour average concentrations. The EPA recommends measuring free cyanide at the lowest occurring pH and also measuring total cyanide during the monitoring of freshwater systems. In cases where total cyanide concentrations are significantly greater than free cyanide concentrations, EPA recommends evaluating the potential for dissociation of metallocyanide compounds (EPA 1985d). Free cyanide readily degrades under both aerobic and anaerobic conditions in water, and thus is generally not persistent in aquatic sediments, although cyanide can sorb to freshwater sediments with moderate carbon content (Higgins and Dzombak 2006). Free cyanide in water is produced from the dissolution of compounds such as sodium cyanate, potassium cyanide, and hydrogen cyanide. Boening and

Chew (1999) reported that the toxicity and fate of cyanide breakdown products, either during treatment or during natural degradation, was considered poorly understood.

The cyanide criteria proposed for Idaho being evaluated in this Opinion and the cyanide criteria originally proposed for Idaho by EPA in 1992 (EPA 1992) differ in that the most recent cyanide criteria are defined as weak acid dissociable (WAD) cyanide (EPA 1999a). While not explicitly explained, this definition is probably used because although the EPA (1985c) considered free cyanide to be a more scientifically correct basis on which to establish criteria for cyanide, these criteria were to be implemented through regulatory programs, but no EPA-approved methods usable in regulatory programs were available at the time. Until such time as these methods became available, EPA recommended that criteria be applied using the total cyanide method, which "may be overly protective" (EPA 1985c). Weak acid dissociable cyanide analyses are apparently a compromise between free and total cyanide measurements, and WAD cyanide includes metal-cyanide complexes such as zinc-, nickel-, copper-, and cadmium-cyanide that easily dissociate under weakly acidic conditions (i.e., pH 5-6) (American Public Health Association (APHA) 2005; method 4500 CN- I).

**Temperature and cyanide toxicity**

As with metals, water hardness or dissolved organic carbon (DOC) are often important modifiers of toxicity, including for cyanide. Water temperature also has a strong influence on the toxicity of cyanide to salmonids and other fishes. A number of tests with different species indicated a marked positive correlation between resistance to HCN and temperature rather than the negative correlation that might be expected based on applying general stress models. Increased toxicity of cyanide at lower temperatures has been observed in the (*Oncorhynchus mykiss*), brook trout (*Salvelinus fontinalis*), yellow perch (*Perca flavescens*), fathead minnows (*Pimephales promelas*), and bluegills (*Lepomis macrochirus*) (Smith et al. 1978; Kovacs and Leduc 1982b, 1982a). A robust dataset is provided by Kovacs and Leduc (1982a) from which a temperature-cyanide toxicity relationship for the rainbow trout can be estimated as: LC50 (µg/L) = (T°C)*3.167+6, $r^2 = 0.97$. For example, at 6°C (42.8°F) the expected LC50 would reflect about a 25 µg/L concentration of cyanide.

When a water quality parameter, such as temperature, is apparently related to the toxicity of a substance, the EPA guidelines (Stephan et al. 1985a, p. 15) for developing aquatic life criteria provide two approaches to handle this situation: (1) direct incorporation of the parameter into the criteria, or (2) application of a data acceptability approach.

In approach #1, *"…if the acute toxicity of the material to aquatic animals apparently has been shown to be related to a water quality characteristic such as hardness or particulate matter for freshwater animals or salinity or particulate matter for saltwater animals, a Final Acute Equation should be derived based on that water quality characteristic…"* (Stephan et al. 1985a). Examples of this approach include: criteria for ammonia that are based on temperature and pH (EPA 1999b); most metals criteria that are based on hardness; and EPA's 2007 copper criteria that are based on multiple water quality characteristics.

In approach #2, *"…results of acute tests conducted in unusual dilution water. e.g., dilution water in which total organic carbon [TOC] or particulate matter exceeded 5 mg/L, should not be used [in a criterion dataset], unless a relationship is developed between acute toxicity and organic carbon or particulate matter or unless data show that organic carbon, particulate matter, etc.,* *do not affect toxicity…"* (Stephan et al. 1985a, p. 14). While test waters colder than 6°C (42.8°F) could hardly be considered an "unusual" temperature, it clearly affects the toxicity of cyanide, and the criteria guidelines are clear that such characteristics should be incorporated into the criteria. No adjustment for the increased toxicity of cyanide at low temperature is included in the proposed cyanide criteria (EPA 1985d). Why that was not done in the case of cyanide is unexplained in the criteria document.

## 2.5.4.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

The available information on the toxicity of free cyanide to aquatic snails indicates that snails are much less sensitive to free cyanide than are fish. The most sensitive test result found for an aquatic snail was for *Physa heterostropha*, with a 96-hr LC50 of 432 µg/L. When the test was manipulated to also test *Physa heterostropha* under the combined stress of free cyanide and periodic low dissolved oxygen, the exposed snails were more sensitive to cyanide with 96-hr LC50 of 190 µg/L (Gensemer et al. 2007). However, even under these conditions, the 96-hr LC50 for free cyanide was much higher than the proposed acute criterion values of 22 µg/L. Because free cyanide is a fast acting acute poison, effects of chronic exposures to snails (for which no data were found) are expected to be correlated with acute effects. Gensemer et al. (2007) list 9 other acute tests with cyanide to 5 other snail species with LC50s ranging from 1,350 to 760,000 µg/L of cyanide, all well above the proposed acute criterion of 22 µg/L.

For the above reasons the Service concludes that the proposed approval of the cyanide aquatic life criteria is not likely to adversely affect Snake River aquatic snails and the Bruneau hot springsnail; all effects are expected to be insignificant or discountable.

## 2.5.4.2 Bull Trout

No specific information regarding the effects of cyanide on the bull trout was identified during this consultation. However, long- and short-term toxicity tests have been conducted with the closely related brook trout, as well as with other salmonid species. In the absence of data for the bull trout, for purposes of this analysis, the bull trout is presumed to have similar responses to cyanide as other salmonid fishes.

In cold-temperate climates such as the Idaho action area, it follows that if cyanide criteria were not adjusted for temperature, only the coldest test results (6°C (42.8°F)) should be used to assess the protectiveness of the criteria relative to the life history requirements of individual bull trout because of this species association with cold water. The bull trout is an obligate stenotherm (cold-water fish) that spends much of each year at temperatures of 6°C (42.8°F) or less. For example, Baxter and Hauer (2000) found that the bull trout selected spawning and incubation redd locations at "warm" groundwater influenced sites with winter-long water temperatures of about 4°C (39.2°F). In contrast, adult bull trout may overwinter in locations at about 1°C (33.8°F) (Jakober et al. 1998). If data on the effects of cyanide on the incubation and hatching of salmonid eggs are available for temperatures of 6, 12, and 15°C (42.8, 53.6, and 59°F, respectively) (e.g., see Kovacs and Leduc 1982b), only data from the 6°C (42.8°F) exposure should be relied upon for the bull trout. Similarly, since juvenile salmonids from fall-spawning species can be expected to be exposed to near-freezing temperatures for long periods, only the

LC50s obtained from the coldest tests should be used in an analysis for the bull trout. In this case that dataset is for tests conducted around 6°C (42.8°F) or below.

Data on the short-term effects of cyanide on the rainbow trout at 6°C (42.8°F) suggest that substantial mortality of exposed bull trout is likely to occur at the proposed acute criterion of 22 µg/L; a slightly higher concentration (27 µg/L) of cyanide killed 50 percent of the exposed trout (see Table 7).

Data on the long-term effects of cyanide exposure on the brook trout showed adverse effects (18 percent reduction in egg production) at a cyanide concentration of 5.6 µg/L, which is similar to the proposed chronic criterion of 5.2 µg/L. Long-term exposure of rainbow trout to cyanide at cold temperatures also showed reduced growth and swimming performance at concentrations of cyanide less than 4.8 µg/L, which is similar to the proposed chronic criterion concentration for cyanide of 5.2 µg/L (Table 7).

**Table 7. Contrasting effects of cyanide on salmonids at different temperatures. For lethal effects data, if LC50s are greater than the Final Acute Value of 44 µg/L that is assumed to indicate lack of harm at acute criteria concentrations; for sublethal effects, lowest effects concentrations should be greater than 5.2 µg/L.**

| Species | Effect | Exposure duration | T (°C) | Effect statistic | Effect concen-tration (µg/L) | Source/ Notes |
|---|---|---|---|---|---|---|
| | **Lethal effects** | | | | | |
| Rainbow trout | Killed | 4 d | 6 | LC50 | 27 | (Kovacs and Leduc 1982a) |
| " | Killed | 4 d | 12 | LC50 | 40 | (Kovacs and Leduc 1982a) |
| " | Killed | 4 d | 18 | LC50 | 65 | (Kovacs and Leduc 1982a) |
| Rainbow trout | Killed | 4-d | 10 | LC50 | 57 | (Smith et al. 1978) |
| | **Sublethal effects** | | | | | |
| Rainbow trout | Reduced swimming performance | 20 d | 6 | No effect threshold | <4.8 | (Kovacs and Leduc 1982b) |
| " | Reduced swimming performance | 20 d | 12 | No effect threshold | <9.6 | (Kovacs and Leduc 1982b) |
| " | Reduced swimming performance | 20 d | 18 | No effect threshold | 43 | (Kovacs and Leduc 1982b) |
| " | Reduced swimming performance | | | No effect threshold | <10 | (a) |
| " | Reduced growth | 20 d | 6 | No effect threshold | <4.8 | (Kovacs and Leduc 1982b) |
| " | Reduced growth | 20 d | 12 | No effect threshold | <9.6 | (Kovacs and Leduc 1982b) |
| " | Reduced growth | 20 d | 18 | No effect threshold | 24 | (Kovacs and Leduc 1982b) |
| " | Reduced growth in fish forced to exercise | 20d | 10 | LOEC | 9.6 | (b) |
| Brook trout | Reduced egg production | | | 18 percent reduction in spawned eggs/female | 5.6 | (Koenst et al. 1977) |
| Atlantic salmon | Abnormal embryo and larval development | | | LOEC | 9.6 | (Leduc 1978) |

(a) EPA (1985c), citing Broderius 1973; (b) EPA 1985d, citing McCracken and Leduc 1980

Given existing data that show adverse effects to other salmonids occurring at cyanide concentrations similar to the proposed acute and chronic criteria, we conclude the proposed acute and chronic criteria for cyanide are likely to cause significant adverse effects to the bull trout in the form of mortality or a significant disruption of their feeding, breeding, sheltering, and migration behavior. Given that the action area (Idaho) contains 44 percent of the range of bull trout-occupied streams and 34 percent of bull trout-occupied lakes and reservoirs within its range, the effects of the proposed acute and chronic criteria for cyanide are incompatible with and are likely to impede (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations throughout a significant portion of its range.

### 2.5.4.3 Bull Trout Critical Habitat

Of the nine PCEs defined for bull trout critical habitat, the Service has determined that the proposed acute and chronic criteria for cyanide are likely to adversely affect three: PCEs 2 (migration habitats), 3 (adequate prey base), and 8 (water quality), as further discussed below.

The proposed acute and chronic criteria for cyanide are likely to create lethal or sublethal chemical barriers that impair or preclude bull trout migration (PCE 2) and movement between various types of habitats (e.g., bull trout movement to refugia habitat in response to predators and thermal stress). Functional bull trout critical habitat facilitates the capability of the species to move within its range, disperse between populations, and to recolonize formerly occupied habitat. Such movements are essential to the recovery of the species. Localized concentrations of cyanide, or areas where cyanide combines with other chemicals, or where other water quality parameters (e.g., pH, temperature) increase the toxicity of cyanide are likely to create habitat conditions that would prevent movement of bull trout within and between populations, which is also essential for the recovery/conservation of the species.

Resident and juvenile migratory bull trout prey on small fish, including salmon fry, as well as terrestrial and aquatic insects, and macro-zooplankton (Boag 1987; Goetz 1989; Pratt 1992, p. 6; Donald and Alger 1993). Adult migratory bull trout feed almost exclusively on other fish (Rieman and McIntyre 1993, p. 3); robust bull trout populations may depend on abundant fish prey resources. This relationship is shown by the correlation between declines in bull trout abundance and declines in salmon abundance (Rieman and McIntyre 1993, p. 3). Cyanide at concentrations at the proposed criteria has been demonstrated to cause adverse effects to salmonids and presumably other prey species of the bull trout (see Table 7). The likely decline of these prey species in response to the proposed acute and chronic criteria for cyanide is, therefore, likely to adversely affect the capability of bull trout critical habitat to provide an abundant food base (PCE 3) for the bull trout.

The proposed action is likely to impair water quality (PCE 8) by allowing aquatic cyanide concentrations to rise to levels that have been shown to be fatal (at the acute concentration standard) or otherwise detrimental to other salmonids. As noted above, adverse effects from cyanide to salmonids were observed at dietary concentrations below the proposed criteria. Assuming bull trout are affected in a similar manner as other salmonids, cyanide concentrations at the proposed acute and chronic criteria levels in critical habitat are likely to impair the ability of critical habitat to provide for normal reproduction, growth, movement, and survival of the bull trout in surface waters of Idaho.

The state of Idaho contains 8,772 miles (44 percent) of streams and 170,217 acres (35 percent) of lakes and reservoirs designated as critical habitat for the bull trout (75 FR 63937). Therefore, the scale of the above adverse effects is likely to overlay a significant portion of designated bull trout critical habitat.

### 2.5.4.4 Kootenai River White Sturgeon

No data relating to the effects of cyanide on the white sturgeon or any other Acipenser species were located during this consultation. For some organic and inorganic contaminants across differing toxic modes of action, the white sturgeon and other sturgeon species are at least as sensitive as the rainbow trout (Dwyer et al. 2005; Ingersoll and Mebane 2014). On that basis, we assumed for this analysis that the white sturgeon is at least as sensitive to cyanide as are salmonids, and that the nature of temperature-cyanide toxicity relations demonstrated for the rainbow trout also holds for the white sturgeon.

In addition to the effects to salmonids of cyanide at concentrations close to the proposed criteria (Table 7), other fish species have shown serious adverse effects at cyanide concentrations close to the proposed criteria. For instance, Kimball et al. (1978) found that spawning of bluegill was "completely inhibited at a concentration of 5.2 µg/L HCN and presumably, is inhibited to some extent at lower levels" (Kimball et al. 1978). The proposed chronic criterion for cyanide is exactly that concentration, 5.2 µg/L, which clearly indicates that the criteria cannot be considered fully protective of critical life functions in all fish species.

The data available on the effects of cyanide on benthic invertebrates suggests that aquatic invertebrates are considerably more tolerant to free cyanide than are fish. Gensemer et al. (2007) reported that when rank ordering different taxa by their acute sensitivity to free cyanide, benthic invertebrate species were more tolerant to cyanide than fish, with the sensitivity ranks for benthic invertebrates falling in the upper (less sensitive) range of the distribution. Thus, indirect adverse effects to the white sturgeon from reduced invertebrate prey availability from cyanide toxicity are considered unlikely. However as pointed out above and indicated in Table 7, the proposed cyanide criteria may have adverse effects on other fish species, including potential sturgeon prey species.

Based on the effects of the proposed cyanide criteria on the bull trout discussed above in the form of mortality or a significant disruption of their feeding, breeding, sheltering, and migration behavior, we conclude such effects are also likely to occur to the Kootenai River white sturgeon. Given that the action area contains about 39 percent of the range of this species, these adverse effects are considered likely to significantly impact the ability of the Kootenai River white sturgeon to persist in a major portion of its range.

### 2.5.4.5 Kootenai River White Sturgeon Critical Habitat

As free cyanide can sorb to freshwater sediments with moderate carbon content, it is possible that criteria concentrations of free cyanide could result in risk to benthic organisms in some settings (Higgins and Dzomback 2006). Thus sediment-sorbed cyanide could contribute to risk to sediment-associated white sturgeon eggs and early life stage juveniles. Sediment quality is critically important to the health of white sturgeon critical habitat because all life stages of the sturgeon are extensively exposed to sediments, either through dermal contact (all life stages) or through incidental ingestion while feeding (juveniles and adults).

Based on the above findings and those reported in the preceding section above, the proposed criteria for cyanide are likely to create habitat conditions within 100 percent of Kootenai River white sturgeon critical habitat that are likely to impair water and sediment quality to an extent that impairs or compromises the sturgeon's ability to successfully reproduce in the wild.

## 2.5.5 Lead Aquatic Life Criteria

The proposed acute and chronic criteria values for lead are 65 and 2.5 µg/L, respectively, as calculated from the following equations using a hardness value of 100 mg/L:

Acute lead criterion (µg/L) = $e^{(1.273[\ln(\text{hardness})]-1.46)}$*(1.46203-(Ln(hardness)*0.145712)

Chronic lead criterion (µg/L) = $e^{(0.8545[\ln(\text{hardness})]-4.705)}$*(1.46203-(Ln(hardness)*0.145712)

The proposed acute and chronic criteria values are also referred to as the CMC and CCC, respectively (EPA 1985e; Stephan et al. 1985a; EPA 2000). With lead and several other hardness-dependent aquatic life criteria, the actual criteria are defined using an equation. For example, at a water hardness of 10, 25, 50, and 250 mg/L, based on the acute lead criterion equation, the lead acute values are 5, 14, 30, and 172 µg/L, respectively. With the proposed chronic criterion, at water hardness values of 10, 25, 50, and 250 mg/L, the lead chronic criterion values are 0.2, 0.5, 1.2, and 6.7 µg/L, respectively.

In this example, the criterion concentrations were calculated using a range of hardness values that cover most waters within the action area. NMFS (2014b) reported that in data compiled from 324 sites monitored by the USGS from 1979-2004, water hardness values ranged from 4 to 2100 mg/L, but 90% of the values fell between 6 and 248 mg/L ($5^{th}$ and $9^{th}$ percentiles of average site hardnesses). Under the proposed action, the hardness calculations are additionally constrained by assuming the general hardness-toxicity relation only holds between a hardness range of 25 to 400 mg/L. For example, the proposed action presumes that at a hardness value of 10 mg/L, lead is no more toxic than at a hardness of 25 mg/L, and in waters where the hardness values are less than 25 mg/L, the toxic criteria would be calculated using a hardness of 25 mg/L, regardless of the actual hardness (EPA 1999a). We did not find any scientific evidence to support this practice of using a "hardness floor" in the acute and chronic criteria equations, and we did find contrary evidence with respect to lead (Mebane et al. 2012), as well as for other metals (see section 2.5.1.5 *Common Factors* above). For these reasons, we consider the "hardness floor" at a hardness of 25 mg/L to be arbitrary, and we do not rely on it in our analyses in this Opinion (see discussion of hardness cap/floor in section 2.5.1.5 above)[16].

Lead occurs naturally in the environment, commonly in association with zinc. In natural waters, dissolved lead concentrations are usually lower than the proposed criteria values, and in waters of the United States away from the immediate influence of discharge, dissolved lead concentrations typically range from 0.01 to 0.2 µg/L (Stephan et al. 1994). Dissolved lead concentrations may be anthropogenically concentrated through mining, smelting, and processing of myriad products that use lead, such as batteries, paints, electronics, fuel additives, and ammunition. Because of the notoriety of public health concerns about lead relating to brain development in children, many historic uses of lead have been phased out or reduced. At present, over 90 percent of lead is produced for lead-acid battery production (Mager 2011). In

[16] In the National Toxics Rule, EPA described and required minimum and maximum hardness values (25 mg/L and 400 mg/L as $CaCO_3$, respectively) to be used when calculating hardness dependent freshwater metals criteria (EPA 2000, p. 21).

natural waters, lead is usually complexed with particulate matter resulting in much lower dissolved than total concentrations (Mager 2011). For instance, in the lead contaminated Coeur d'Alene River of northern Idaho, dissolved lead concentrations rarely exceed 20 µg/L whereas total concentrations often exceed 100 µg/L (Clark 2002).

Lead is soluble in neutral and acidic freshwaters at pH values less than 7. Solubility decreases with increasing pH, alkalinity, and suspended material (Mager 2011). As solubility decreases, dissolved lead may precipitate or sorb to particles and settle out of the water column, leading to elevated sediment concentrations (Balistrieri et al. 2002).

Lead is not known to have any biological function in plants or animals. Following short-term exposures, acute lead poisoning in fish results from disrupted internal mineral balances. Specifically, hypocalcemia was shown to result from lead interfering with calcium uptake from water through the gills of rainbow trout, and further interfering with enzyme activity, preventing calcium transport to the blood. Sodium and chloride balances were also affected (Rogers et al. 2003; Rogers et al. 2005). Mechanisms of chronic toxicity of lead to aquatic organisms also seem to be related to the ability of lead to mimic calcium in ion transport, which in turn can lead to a plethora or problems, including disruption of intracellular calcium homeostasis leading to injury of neuron cells, degeneration of exposed axons, and interference with neurotransmitters (Mager 2011). In fish, external symptoms of chronic lead exposures include lordoscoliosis (abnormal spinal curvature), reduced growth and death (Mager 2011). In snails, growth reductions appear to be the most sensitive result of lead exposures, which in turn seems to be linked to the inhibition of calcium uptake by lead and to the very high calcium demands of shell growth in juvenile snails (Grosell and Brix 2009).

The toxicity of dissolved lead to aquatic organisms seems to vary primarily with water calcium, organic matter, pH, and ionic strength. These factors appear to influence both the acute and chronic toxicity of lead (Macdonald et al. 2002; Mager et al. 2010; Mager et al. 2011; Mebane et al. 2012).

#### 2.5.5.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

The listed snail species of concern in this Opinion can be grouped as pulmonate or non-pulmonate snails. The Banbury Springs lanx is classified among the pulmonate snails, and while not formally described, is considered to be in the family Lymnaeidae (USFWS 2006b). The Snake River physa (family Physidae) is also a pulmonate snail. The Bliss Rapids snail and the Bruneau Hot Springsnail are non-pulmonate snails in the family Hydrobiidae.

Pulmonate snails in the family Lymnaeidae have been shown to be hypersensitive to chronic lead toxicity (Grosell et al. 2006b; Grosell and Brix 2009). The reasons for this hypersensitivity appear to be related to the high demand for calcium by juvenile pulmonate snails, relative to their body size and the role of lead in mimicking and disrupting calcium uptake. As result, some pulmonate snails appear to be the most sensitive of all known taxa to chronic lead toxicity (Grosell et al. 2006b; Grosell and Brix 2009). A dissolved lead concentration of about 3 µg/L resulted in a 20 percent reduction in growth of juvenile *Lymnaea stagnalis* snails when tested in water with a hardness of about 102 mg/L (Grosell et al. 2006b). The proposed chronic criterion value for lead of 2.5 µg/L is close to that concentration. Older studies with a related species, *Lymnaea palustris*, showed adverse effects to lead exposure at a dissolved lead concentration of 12 µg/L at a hardness value of 139 mg/L. This value is higher than the proposed chronic criterion value for lead of 3.6 µg/L, but is consistent with the view that pulmonate snails are highly sensitive to lead toxicity.

The sensitivity of *Lymnae* spp. to chronic lead toxicity can be assessed by determining the concentration of lead that would result in no effects to the Banbury Springs lanx. To attempt to gain some insight on this, raw data from studies with the surrogate *Lymnaea stagnalis* were analyzed through non-linear, piecewise regression to estimate a no-effect concentration, that is, a 0 percent effects concentration (EC0). Curve fittings suffer from some uncertainties, especially if interpolations are large, or if the best fit effect curves are extrapolated beyond the data. However, some traditional approaches for estimating low- or no-effect concentrations can have worse uncertainities, or even produce misleading “no-observed effects concentrations” that actually may represent fairly large effects even if so-called “statistical significance” was not achieved (Suter 1996; Fox 2008; Newman 2008; NMFS 2014a, Appendix B). The curve fitting software TRAP (Toxicity Response Analysis Program) was used for the regression analyses (Erickson 2010). Piece-wise regressions are also sometimes referred to as “broken-stick” or “jack-knife” regressions in some literature because of their appearance. Summaries of these analyses are presented below.

The high sensitivity of *Lymnaea* spp.to lead is based on several studies (Borgmann et al. 1978; Grosell et al. 2006b; Grosell and Brix 2009; Brix et al. 2012; Esbaugh et al. 2012; Munley et al. 2013). While some of these studies only reported summary statistics, two studies reported exposure-concentration data in sufficient detail to reanalyze to estimated no-effect concentrations (Brix et al. 2012; Munley et al. 2013).

Brix et al. (2012, their figure 5) evaluated the growth of *Lymnaea* with lead exposures for up to 16 days. The focus of their research was to explore physiological or biochemical changes in lead-exposed snails, not to conduct a long-term test to identify the most sensitive life stage or endpoint. The EC0 estimate from piecewise regression (a lead concentration of 2.9 µg/L) was higher than the proposed chronic criterion lead concentration of 1.4 µg/L for a water hardness value of 60 mg/L (Figure 6). However, an important limitation of this estimate is that growth was reduced at the lowest concentration tested (~2 µg/L of lead), and the curve break defining the EC0 does not fit that well.

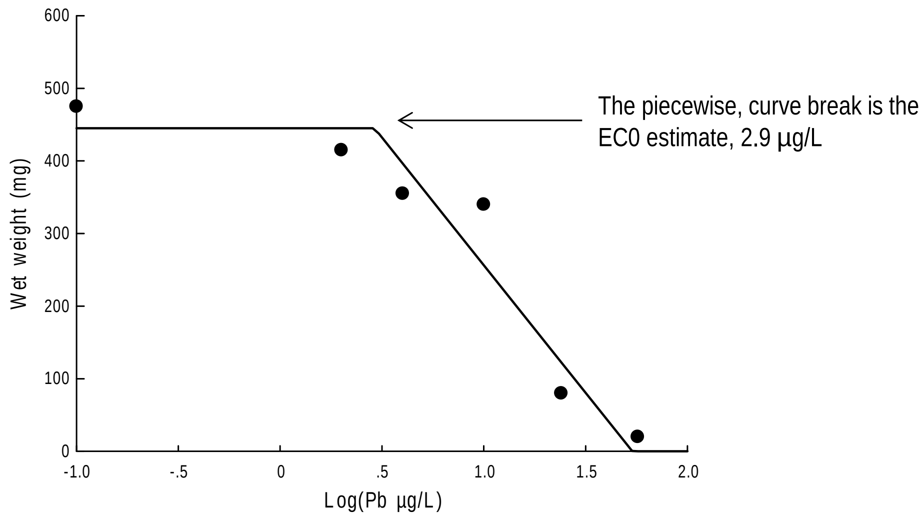


**Figure 6. *Lymnaea stagnalis* growth, under different lead (Pb) exposures. Data taken from Brix et al (2012) curve fitting was done using the nonlinear regression, piecewise regression function in the Toxicity Response Analysis Program (TRAP) (Erickson 2010).**

In contrast to the relatively short (16-day) exposures in the Brix et al. (2012) study, Munley et al. (2013) tracked the growth of *Lymnaea* for 56 days, and tracked *Lymnaea* reproduction in terms of egg production as well. Their tests were conducted in water with a hardness value of 87 mg/L, for which the chronic lead criterion is 2.2 µg/L.

Their research illustrates some of the complexity and possible conflicts of interpretation with ecotoxicological data. After a 28-day exposure, snail growth was reduced in all lead treatments (Figure 7, panel "**A**"). The regression break provides an EC0 estimate of only 0.2 µg/L Pb, although it is uncertain whether the organism threshold of response would match the curve break, since the break is less than the lowest concentrations tested. However, by the end of the test at 56 days, the snails in the lowest treatment concentration (1 µg/L of lead, which is 0 on the log-scale shown in the plots) caught up with the controls in growth (Figure 7, panel "**B**"). Many toxicologists would likely then consider the 1 µg/L lead concentration treatment to be "no-effect" because the snails had recovered from the transient reductions in growth. However, unlike many tests where growth is the only sublethal endpoint, Munley et al. (2013) also tracked reproductive output as egg production (Figure 7, panel "**C**"). The snails in the 1 µg/L lead concentration treatment had reduced egg production, even though they had fully recovered from the growth reductions. The piecewise regression curve break estimate for a no-effect concentration was 0.4 µg/L of lead. From a population viability perspective, survival and reproduction are the only endpoints that directly matter. Growth is only relevant to population viability as a predictor of reproduction. Therefore, the 0.4 µg/L EC0 estimate from the reproductive endpoint from this surrogate species is considered most relevant to estimating the conservation needs of the Banbury Springs lanx, even though it is higher than the lowest EC0 from the growth endpoints (Figure 7, panel "**C**"). The EC0 of 0.4 µg/L is about 0.2X that of the proposed chronic criterion for lead of 2.2 µg/L. Therefore, one approach for making conservative estimates of lead concentrations that are protective of the Banbury Springs Lanx is a 0.2X multiplier to the proposed lead chronic criterion concentration.

 

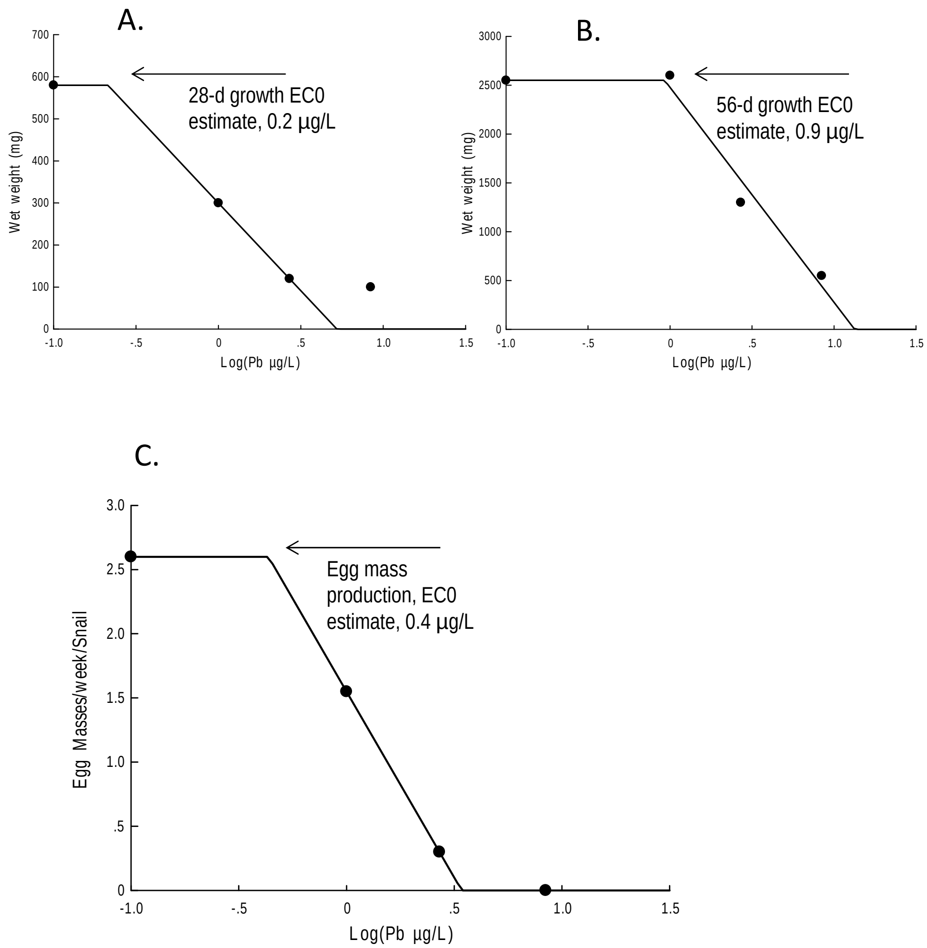


**Figure 7. *Lymnaea stagnalis* growth, under different lead (Pb) exposures. Data taken from Munley et al. (2013), curve fitting was done using the nonlinear regression, piecewise regression function in the Toxicity Response Analysis Program (TRAP) (Erickson 2010).**

Little information was located during this consultation on the sensitivity of snails other than *Lymnaea* to long-term exposures to lead. What information was located suggests that extraordinary sensitivity of "pulmonate" snails to lead (Grosell and Brix 2009) might more accurately be described as the extraordinary sensitivity of snails in the genus *Lymnaea* or family Lymnaeidae. For example, Lefcort et al. (2004) described the pulmonate snail *Physella columbiana* (Physidae) as thriving in lead-contaminated lakes in the Coeur d'Alene, Idaho region. *Physella* was also less sensitive to lead than was *Lymnaea palustris* (Lefcort et al. 2008). Lead had no effect on the survival of the snail *Physa integra* after a 28-day exposure to lead concentrations up to 565 µg/L in 45 mg/L hardness water (Spehar et al. 1978).

No data from a controlled toxicity test on the chronic effects of lead on Hydrobiidae snails were located during this consultation, but a field study indicated that at least some Hydrobiidae snails are resistant to lead. Marqués et al. (2003) found Hydrobiidae snails were abundant in disturbed conditions in lead-zinc mining affected streams with mean lead concentrations of 38 µg/L which is well above the maximum possible Idaho proposed chronic criterion value for lead of 11 µg/L at a water hardness value of 400 mg/L. Marqués et al. (2003) did not report water hardness, but their high reported specific conductance (900-1500 microsiemens/cm) suggested very hard water.

In short-term exposures to lead, the pulmonate snails *Gyraulus* (family Planorbidae) and *Physa* (Physidae) were killed at lead concentrations well above the proposed acute criterion values for lead. EC50 values ranged from 380 to 1,169 µg/L of lead in water with hardness values of 41 mg/L or less (Mebane et al. 2012). The corresponding acute criterion values for these tests were 24 µg/L or less.

Based on the findings discussed above, we conclude that the proposed approval of the acute aquatic life criterion for lead is not likely to adversely affect the three Snake River snails (the Snake River physa snail, Bliss Rapids snail and the Banbury Springs lanx) and the Bruneau hot springsnail; all effects caused by the proposed acute aquatic life criterion for lead are likely to be insignificant or discountable. However, due to extraordinary sensitivity of snails in the genus *Lymnaea* or family Lymnaeidae to lead toxicity, significant adverse effects in the form of reduced growth and egg production are likely to be caused by the approval of the proposed chronic lead criterion to the pulmonate Banbury Springs lanx but not the pulmonate Snake River physa, which is not in the Family Lymnaeidae (see Spehar et al. 1978), or the Bliss Rapids snail and the Bruneau hot springsnail. The effects to the lanx are likely to occur throughout its range and are likely to cause reductions in the reproduction and numbers of this species.

## 2.5.5.2 Bull Trout

No direct toxicity testing information involving the bull trout and lead is known to exist although extensive work with other salmonids has been reported. Of particular note, Holcombe et al. (1976) reported the results of exposing the closely related brook trout to lead for 3 years, including partial exposure of three generations of fish. No effects were detectable at a concentration of 34 µg/L total lead or lower. By contrast, the proposed chronic water quality criterion value for lead is 1.0 µg/L for a test water hardness of 44 mg/L.

With rainbow trout, the lowest thresholds of adverse effects caused by lead exposure have been reported at a concentration of 7-8 µg/L in softwater. Davies et al. (1976) reported that rainbow trout exposed to lead for about 580 days in water with an average hardness of 28 mg/L developed deformities (lordoscoliosis) or blackened tails, a precursor to lordoscoliosis, at a lead concentration of 7.6 µg/L, with no effects detected at a lead concentration of 4.1 µg/L (Davies et al. 1976). Mebane et al. (2008) reported a 10 percent reduction in the growth of rainbow trout exposed to a lead concentration of 7 µg/L for 62 days in water with a hardness value of 29 mg/L (Mebane et al. 2008). The proposed chronic lead criterion at a water hardness value of 29 mg/L is 0.64 µg/L, which is about 10 times lower than the lowest effect concentrations found with salmonids. Other chronic tests of freshwater fish exposed to lead have reported higher thresholds of adverse effects, ranging from lead concentrations of 24-71 µg/L in soft water (Mebane et al. 2008), which are well above the proposed criteria values.

Behavioral alterations in fish exposed to lead have been reported, but in all of the reports reviewed in this consultation, behavioral effects occurred at lead concentrations well above the proposed criteria concentrations. For instance, Mager et al. (2010) found that sustained exposure of fathead minnow fry to a lead concentration of 120 µg/L resulted in feeding impairment and other behavioral alterations, but no effects were apparent at a lead concentration of 35 µg/L. Both of these concentrations are well above the proposed chronic water quality criterion value for lead of 2.3 µg/L at a water hardness value of 93 mg/L. Other behavioral effects to fathead minnows exposed to lead reported in the literature included reduced swimming activity, reduced ability to avoid predation in mummichogs, and altered reproductive behaviors. However, these effects were all documented at high lead concentrations (≥ 100 µg/L) that are well above the proposed criterion concentrations (Mager 2011).

All ages of the bull trout are opportunistic predators that shift their diet towards abundant and easily captured prey at different times and locations. While there is evidence that bull trout can eat armored taxa such as clams and snails (Donald and Alger 1993), in general, we presume that taxa classified as vulnerable to salmonid predation are most important in the diet of the bull trout (Suttle et al. 2004), and that taxa that are generally non-vulnerable to salmonid predation (burrowing or armored taxa) are not critical to the bull trout's diet. Thus, when evaluating reports of adverse effects of chemicals to different potential bull trout prey taxa, adverse effects to taxa considered generally non-vulnerable to salmonid predation are less of a concern than effects to common and vulnerable taxa. For instance in lake populations of the bull trout, amphipods appear to be consistently important to bull trout rearing, as Donald and Alger (1993) showed bull trout weights in lakes were correlated with amphipod abundance. Large zooplankton such as *Daphnia magna* or *Daphnia pulex* may be important food items in lakes, whereas smaller zooplankton such as Ceriodaphnia or copepods are less important (Wilhelm et al. 1999). In streams, bull trout of all ages prey on aquatic insects such as mayflies (ephemeropterans), stoneflies (plecopterans), caddisflies (trichopterans), beetles (coleopterans ) midges (chironomids) and worms (oligochaetes) (Boag 1987; Underwood et al. 1995). In streams, small fish such as sculpin and juvenile salmonids can also be relatively important in the diet of the bull trout (Underwood et al. 1995). Adult migratory bull trout feed on both aquatic invertebrates and other fish depending on prey availability and the size of the bull trout (Donald and Alger 1993; Wilhelm et al. 1999; Beauchamp and Van Tassell 2001).

With lead, sensitive potential bull trout prey taxa include the amphipods *Hyalella* and *Gammarus*, which are important bull trout prey items in lakes. Little evidence is available on the effects of lead exposure to stream-resident aquatic invertebrates such as mayflies and midges, but what information there is suggests these taxa are less sensitive than amphipods to lead exposure, as discussed below.

The most sensitive responses of potential prey species to lead exposure were by amphipods, followed by zooplankton and aquatic insects. Besser et al. (2005b) exposed the amphipod *Hyalella azteca* to lead both through diet and water. Lead was significantly more toxic to *H. azteca* when it was exposed to lead both through diet and through water, than when it was exposed through water alone. *H. azteca* exposed to lead in water only suffered a 25 percent reduction in reproduction at a lead concentration of 2.8 µg/L, which is a lower concentration than the proposed chronic lead criterion concentration of 3.6 µg/L at a water hardness value of 138 mg/L. However, *H. azteca* fed lead-treated diets had significantly increased toxicity across a wide range of dissolved lead concentrations, with a 70 percent reduction in reproductive output at a lead concentration of 3.5 µg/L (Besser et al. 2005b). Similarly low effect lead concentrations were reported for amphipods by Borgmann and Norwood (1999), with 25 percent mortality of amphipods exposed to a lead concentration of 3.3 µg/L at a water hardness value of 130 mg/L. The proposed chronic lead criterion for a water hardness value of 130 mg/L is also 3.3 µg/L. In comparative tests with other aquatic invertebrates reported by Spehar et al. (1978), the amphipod *Gammarus pseudolimnaeus* was the most sensitive taxa tested with a 28-day LC50 of 28.4 µg/L of lead at a water hardness value of 45 mg/L, which indicates the onset of adverse effects would be at lower concentrations.

With other potential prey species of the bull trout, effect threshold concentrations reported in the literature were higher than the allowable proposed chronic criterion concentrations of lead. With the zooplankton *Daphnia magna*, Chapman et al. (1980) reported the onset of adverse effects at lead concentrations between 9 and 16 µg/L at a water hardness value of 51; the corresponding chronic lead criterion was 1.2 µg/L. Mebane et al. (2008) reported chronic effects of lead to two stream-resident insects, the mayfly *Baetis tricaudatus* and the midge *Chironomus dilutus*. Reduced growth in Baetis mayflies and reduced emergence in Chironomus midge occurred at lead concentrations of 37 and 15 µg/L, respectively, in water with hardness values of 20 and 32 mg/L, respectively. The corresponding proposed chronic lead criterion concentrations were much lower for these test conditions: 0.6 to 0.7 µg/L.

In summary, the potential impacts of lead exposure at the proposed criteria concentrations to bull trout prey species appear limited to amphipods, particularly *Hyalella*. Given that bull trout eat a variety of prey items and are known piscivores, a potential reduction in amphipod abundance is not likely to have a significant effect on the available prey base for the bull trout.

Based on the research results referenced above, the Service concludes that EPA's proposed approval of acute and chronic aquatic life criteria for lead is not likely to adversely affect the bull trout; any such effects are expected to be insignificant or discountable.

## 2.5.5.3 Bull Trout Critical Habitat

Of the nine PCEs designated for bull trout critical habitat, the proposed water quality criteria for lead were evaluated for the potential to affect PCE 3 (adequate prey base).

All ages of the bull trout are opportunistic predators that shift their diet towards abundant and easily captured prey at different times and locations. While there is evidence that bull trout can eat armored taxa such as clams and snails (Donald and Alger 1993), in general, we presume that taxa classified as vulnerable to salmonid predation are most important in the diet of the bull trout (Suttle et al. 2004), and that taxa that are generally non-vulnerable to salmonid predation (burrowing or armored taxa) are not critical to the bull trout's diet. Thus, when evaluating reports of adverse effects of chemicals to different potential bull trout prey taxa, adverse effects to taxa considered generally non-vulnerable to salmonid predation are less of a concern than effects to common and vulnerable taxa. For instance in lake populations of the bull trout, amphipods appear to be consistently important to bull trout rearing, as Donald and Alger (1993) showed bull trout weights in lakes were correlated with amphipod abundance. Large zooplankton such as *Daphnia magna* or *Daphnia pulex* may be important food items in lakes, whereas smaller zooplankton such as Ceriodaphnia or copepods are less important (Wilhelm et al. 1999). In streams, bull trout of all ages prey on aquatic insects such as mayflies (ephemeropterans), stoneflies (plecopterans), caddisflies (trichopterans), beetles (coleopterans ) midges (chironomids) and worms (oligochaetes) (Boag 1987; Underwood et al. 1995). In streams, small fish such as sculpin and juvenile salmonids can also be relatively important in the diet of the bull trout (Underwood et al. 1995). Adult migratory bull trout feed on both aquatic invertebrates and other fish depending on prey availability and the size of the bull trout (Donald and Alger 1993; Wilhelm et al. 1999; Beauchamp and Van Tassell 2001).

With lead, sensitive potential bull trout prey taxa include the amphipods *Hyalella* and *Gammarus*, which are important bull trout prey items in lakes. Little evidence is available on the effects of lead exposure to stream-resident aquatic invertebrates such as mayflies and midges, but what information there is suggests these taxa are less sensitive than amphipods to lead exposure, as discussed below.

The most sensitive responses of potential prey species to lead exposure were by amphipods, followed by zooplankton and aquatic insects. Besser et al. (2005b) exposed the amphipod *Hyalella azteca* to lead both through diet and water. Lead was significantly more toxic to *H. azteca* when it was exposed to lead both through diet and through water, than when it was exposed through water alone. *H. azteca* exposed to lead in water only suffered a 25 percent reduction in reproduction at a lead concentration of 2.8 µg/L, which is a lower concentration than the proposed chronic lead criterion concentration of 3.6 µg/L at a water hardness value of 138 mg/L. However, *H. azteca* fed lead-treated diets had significantly increased toxicity across a wide range of dissolved lead concentrations, with a 70 percent reduction in reproductive output at a lead concentration of 3.5 µg/L (Besser et al. 2005b). Similarly low effect lead concentrations were reported for amphipods by Borgmann and Norwood (1999), with 25 percent mortality of amphipods exposed to a lead concentration of 3.3 µg/L at a water hardness value of 130 mg/L. The proposed chronic lead criterion for a water hardness value of 130 mg/L is also 3.3 µg/L. In comparative tests with other aquatic invertebrates reported by Spehar et al. (1978), the amphipod *Gammarus pseudolimnaeus* was the most sensitive taxa tested with a 28-day LC50 of 28.4 µg/L of lead at a water hardness value of 45 mg/L, which indicates the onset of adverse effects would be at lower concentrations.

With other potential prey species of the bull trout, effect threshold concentrations reported in the literature were higher than the allowable proposed chronic criterion concentrations of lead. With the zooplankton *Daphnia magna*, Chapman et al. (1980) reported the onset of adverse effects at lead concentrations between 9 and 16 µg/L at a water hardness value of 51; the corresponding chronic lead criterion was 1.2 µg/L. Mebane et al. (2008) reported chronic effects of lead to two stream-resident insects, the mayfly *Baetis tricaudatus* and the midge *Chironomus dilutus*. Reduced growth in Baetis mayflies and reduced emergence in Chironomus midge occurred at lead concentrations of 37 and 15 µg/L, respectively, in water with hardness values of 20 and 32 mg/L, respectively. The corresponding proposed chronic lead criterion concentrations were much lower for these test conditions: 0.6 to 0.7 µg/L.

In summary, the potential impacts of lead exposure at the proposed criteria concentrations to bull trout prey species appear limited to amphipods, particularly *Hyalella*. Given that bull trout eat a variety of prey items and are known piscivores, a potential reduction in amphipod abundance is not likely to have a significant effect on the available prey base for the bull trout. On that basis, the Service concludes that the proposed water quality criteria for lead are not likely to adversely affect the PCEs of bull trout critical habitat because such effects are likely to be insignificant or discountable.

## 2.5.5.4 Kootenai River White Sturgeon

The sensitivity of juvenile white sturgeon (*Acipenser transmontanus*) to chronic lead exposure was recently investigated in a series of water-only exposures (Wang et al. 2014a). No adverse effects to the white sturgeon were detected at concentrations close to the proposed water quality criteria for lead. The lowest concentration of lead causing adverse effects to the white sturgeon following long-term exposure was a 10 percent reduction in survival at a lead concentration of 26 µg/L, which is 10X greater than the proposed chronic criterion concentration of 2.5 µg/L for a test water hardness value of 100 mg/L.

Potential adverse effects due to lead exposure to white sturgeon food items are likely similar to those discussed above for the bull trout because the diet of these two species overlaps considerably, in that sturgeon are opportunistic feeders with smaller sturgeon feeding predominately on chrionomids and larger sturgeon feeding on fish and crayfish (Scott and Crossman 1973, p. 99, Partridge 1983, pp. 28-35). Additional data on lead effects to mussels were also reviewed in analyzing the effects of the proposed action on the white sturgeon. Wang et al. (2010) reported that while freshwater mussels are among the more sensitive taxa tested with respect to lead exposure, with a 10 percent reductions in mussel growth and survival at a lead concentration of about 6.4 µg/L and 10 µg/L, respectively. The corresponding proposed chronic criterion for lead under the test conditions is 1.1 µg/L at a water hardness value of 46 mg/L water, which is considerably lower.

Based on the above research results, the Service concludes that the proposed acute and chronic lead criteria are not likely to adversely affect the Kootenai River white sturgeon because any such effects are expected to be insignificant or discountable.

### 2.5.5.5 Kootenai River White Sturgeon Critical Habitat

Although not identified as a PCE in the current final rule for Kootenai River white sturgeon critical habitat (73 FR 39506), water quality affects the capability of that habitat to function in support of sturgeon recovery. Based on the above analysis of the likely response of the white sturgeon to habitat conditions that conform to implementation of the proposed water quality criteria for lead, the Service concludes that such implementation is not likely to adversely affect critical habitat for the Kootenai River white sturgeon because such effects are likely to be insignificant or discountable. The proposed approval of water quality criteria for lead will have no effect on the PCEs of sturgeon critical habitat addressing flow regime, water temperature, and rocky substrates.

## 2.5.6 Mercury Aquatic Life Criteria

The proposed acute and chronic aquatic life criteria for dissolved mercury are 2.1 µg/L and 0.012 µg/L (12 ng/L), respectively (EPA 1999a, p.41). The EPA has also developed a human health criterion for mercury, in which fish tissue concentrations are not to exceed 0.3 mg/kg ww (66 FR 1344; EPA 2001). This standard was adopted in Idaho in 2005 and is applicable to all designated critical habitats and waters inhabited by listed aquatic species in Idaho (IDEQ 2005, pp. 141-148).

Mercury is hazardous because of its strong tendency to bioaccumulate in muscle tissue and because it is a potent neurotoxin that causes neurological damage which in turn leads to behavioral effects which in turn lead to reduced growth and reproduction (Wiener et al. 2003; Weis 2009; Sandheinrich and Wiener 2010). Methylmercury is a highly neurotoxic form that readily crosses biological membranes, can be rapidly bioaccumulated through the water, and is taken up primarily through the diet (which accounts for more than 90 percent of the total amount of methylmercury accumulated). Both organic and inorganic mercury bioaccumulate, but methylmercury accumulates at greater rates than inorganic mercury. Methylmercury is more efficiently absorbed, and preferentially retained than inorganic mercury. Methylmercury is biomagnified between trophic levels in aquatic systems and in general proportion to its supply in water (Wiener et al. 2003, entire). In the muscle of predatory fish, accumulated mercury consists almost entirely of methylmercury (Bloom 1992; Hammerschmidt et al. 1999; Harris et al. 2003). In lower trophic level aquatic invertebrates, a much lower proportion of mercury will be present as methylmercury (Lasorsa and Allen-Gil 1995).

The proposed action for mercury is somewhat confusing because the State of Idaho repealed their aquatic life criteria for mercury in 2005, based upon their belief that application of the human health criterion for methylmercury will be protective of aquatic life in most situations (IDEQ 2005; IDEQ NA variously dated, pp. 146). However, EPA did not approve that change, and even though the 12 ng/L chronic standard for mercury does not appear in the Idaho water quality standards, EPA considers it applicable for Clean Water Act purposes (http://www.deq.idaho.gov/epa-actions-on-proposed-standards. accessed July 14, 2014).

Although short-term acute studies have been conducted with mercury and an acute criterion of 2.1 µg/l has been established, because environmental risks to aquatic life from mercury are from long-term, food web exposures, the acute criterion does not have environmental relevance.

## 2.5.6.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

Risks of mercury toxicity to herbivorous invertebrates (such as snails) in any given waterbody are expected to be considerably lower than for predatory (high-trophic level) animals for the following reasons:

1. Because mercury strongly biomagnifies through the food web, the total mercury burden will be lower for herbivorous invertebrates than for fish. For example, Allen-Gil et al.(1997, pp 737-738) reported on the relative bioaccumulation of mercury in snails and lake trout (*Salvelinus namaycush*) in lake food webs. On the average, concentrations in lake trout muscle were about 7 times higher than in snail tissue.
2. The proportion of methyl- to total mercury will be lower in herbivorous invertebrates than for fish. In analyses of various invertebrate and vertebrate species, Lasora and Allen-Gil (1995, p. 913) found that only for predatory fish did the ratio of methymercury to total mercury approach 1.
3. Mercury is a neurotoxin, and the simpler neurologic system of invertebrates (compared to vertebrates) appears to place invertebrates at less risk (Wiener et al. 2003, pp. 14-29).

Becker and Bigham (1995, pp. 566-567) found a significant correlation between mercury concentrations in surficial sediments and mercury tissue concentrations in amphipods and chironomids, indicating that contaminated sediments are a likely source of mercury for benthic macroinvertebrates, including snails.

Concentrations of mercury that result in adverse effects to snails appear to be high relative to those for fish. Benton et al. (2002, entire) reported DNA damage in hornsnails (*Pleurocera* *canaliculatum)* collected from a stream grossly polluted with mercury relative to snails collected upstream of the mercury pollution. In upstream snails, total mercury tissue residues were about 0.1 mg/kg wet weight compared to >0.6 mg/kg in downstream snails. However, despite the DNA damage, there was no evidence of decreased snail density at any downstream, polluted site (Benton et al. 2002, p. 587).

Thain (1984) exposed slipper limpets (*Crepidula fornicate*) to mercury-equilibrated algal suspensions for 16 weeks, and evaluated a variety of lethal and a variety of sublethal effects including adult condition factors and larval swimming behavior, feeding, and settlement. While initially few effects were observed at mercury concentrations <1.0 µg/L, in a third spawning, reductions in settlement success of spat were observed at mercury concentrations ≥0.25 µg/L (Thain 1984, p. 302-303). Mercury tissue concentrations in the snails associated with the lowest tested effects were about 8 mg/kg wet weight. Thain (1984) also tested acute responses that are relevant to the protectiveness of the acute mercury criterion of 2.1 µg/L. While lethal effects ( LC50) to larvae did not occur until a mercury concentration of 60 µg/L was reached, sublethal, short-term effects (cessation of feeding and swimming) occurred at mercury concentrations as low as 6 µg/L, which is higher than both the proposed acute and chronic criteria for mercury.

In the available data, adverse effects to snails associated with mercury in tissues occurred only at much higher tissue concentrations than those for fish, and in waterborne exposures, only at much higher concentrations than allowed by the proposed chronic criterion for mercury. In summary, assuming the sensitivity of listed snails is similar to that of other, tested, snails, the proposed acute and chronic criteria for mercury appear to present minimal risk of adverse effects to Snake River aquatic snails and to the Bruneau hot springsnail. Based on the above analysis, such effects are likely to be insignificant and discountable.

## 2.5.6.2 Bull Trout

Available information on the toxicity to salmonids of short-term exposure to mercury in water indicates that adverse effects at 2.1 µg/L of mercury (the proposed acute criterion) are unlikely. EPA (1985d, Table 1) lists LC50s for salmonids exposed to acute concentrations of mercury in the range 24-84 µg/L, based on tests where the water chemistry was measured. These concentrations are approximately 12 to 40 times higher than the proposed acute criterion; on that basis, the Service concludes that the proposed acute criterion for mercury is unlikely to cause adverse effects to the bull trout.

The proposed chronic criterion for protection of aquatic life relative to mercury is considerably more complex to evaluate. Food chain transfer is by far the most important exposure pathway in aquatic ecosystems (Wiener et al. 2003). Aquatic systems have complex food webs including several trophic levels. Aquatic predators including salmonids are most susceptible to bioaccumulating mercury, and thus their tissue concentrations may best reflect the amount of mercury available to aquatic organisms in the environment. For example, in comparisons of fish and invertebrates across trophic levels McIntyre and Beauchamp (2007, p. 577) determined that the greatest mercury concentrations were found in piscivorous fish species and that mercury content increased with higher trophic levels and the age of the organisms.

Diet is the primary route of methylmercury uptake by fish in natural waters, and contributes more than 90 percent of the amount accumulated (Wiener et al. 2003, p. 17). Sediments are an important reservoir for mercury in freshwater systems. Mercury in sediments can become available for food chain transfer, and instances of elevated mercury in sediment corresponding with elevated mercury in fish have been documented (Scudder et al. 2009, pp. 27-30). One well documented instance was from Onondaga Lake, NY where dissolved mercury in the epiliminion was about 1 ng/L and mercury in the hypolimnium was up to 10 ng/L (Bloom and Effler 1990, p. 260). Mercury in sediments were always above 1 mg/kg dw, often above 5 mg/kg dw, and exceeded 25 mg/kg dw in some samples. Mercury in sediments was strongly correlated with mercury in invertebrate tissues (Becker and Bigham 1995, 563-571).

### Tissue Levels of Concern for Mercury

The following paragraphs provide the information relied upon by the Service for determining if the proposed 12 ng/L aquatic life chronic criterion for mercury is sufficiently protective to avoid harmful tissue bioaccumulation in the bull trout, a predatory salmonid at the top of the aquatic food chain.

Sandheinrich and Wiener (2010) concluded that effects on biochemical processes, damage to cells and tissues, and reduced reproduction in fish have been documented at methylmercury concentrations of about 0.3 to 0.7 mg Hg/kg ww in the whole body and about 0.5 to 1.2 mg Hg/kg ww in axial muscle. NMFS (2014a, p. 152) concluded that mercury tissue concentrations of less than about 0.2 to 0.3 mg/kg were unlikely to be linked to appreciable adverse effects to salmonids. The lowest recommended threshold reviewed was for concentrations of mercury in the diet of fish rather than the tissues of the fish themselves. DePew et al. (2012) concluded that 0.5 mg/kg ww in the diet of fish had been linked to reproductive impairment, and thus thresholds for mercury concentrations to avoid adverse effects to fish need to be lower than 0.5 mg/kg ww (Depew et al. 2012, p. 1542).

Using 0.3 mg/kg ww as an estimate of a low-risk mercury tissue concentration for the bull trout, the next question is whether the proposed 12 ng/L chronic water quality criterion for mercury would be sufficient to avoid tissue concentrations of mercury in the bull trout from exceeding 0.3 mg/kg ww. Available information indicates that mercury would be expected to bioaccumulate to concentrations exceeding 0.3 mg/kg ww in the bull trout and other piscivorous fish in waters with waterborne mercury concentrations much lower than 12 ng/L (Table 8). On its face, the 0.3 mg/kg tissue value might seem a questionable value to use as a low risk screening value for bull trout because adverse effects have been shown at about 0.3 mg/kg (above). However, most data available from bull trout or from other piscivorous fish (surrogates) were of mercury in muscle tissue (filets), rather than whole bodies. Mercury concentrations in muscle will be slightly higher than those measured in the whole. For instance, mercury accumulated in brook trout muscle concentrations averaged about 1.3X those of the whole-body fish (McKim et al. 1976), and Sandheinrich and Wiener (2010) estimated a muscle tissue of 0.5 mg/kg corresponded with a whole-body concentration of 0.3 mg/kg as a low-effect tissue concentration.

**Table 8. Selected examples of mercury concentrations in water, and mercury burdens in muscle tissue of piscivorous fish in relation to the low effect tissue threshold for mercury of 0.3 mg/kg and to the proposed chronic criterion (12 ng/L) for mercury.**

| Location or situation | Hg in unfiltered water (ng/L) | Hg in fish tissue (mg/kg, ww) | Fish species | Source |
|---|---|---|---|---|
| **Lake McDonald, MT** | **0.35 – 2.9** | **0.17 – 0.30** | **Bull Trout (39-50cm)** | **(Watras et al. 1995; Eagles-Smith et al. 2014)** |
| **Alturas Lake, ID** | **~ 0.3 (a)** | **0.11 – 0.16** | **Bull Trout, 39 cm average length** | **(Essig and Kosterman 2008, p. 64; Essig 2010, p. 89)** |
| **Payette Lake, ID** | **0.7** | **0.45** | **Lake Trout** | **(Essig and Kosterman 2008, p. 64; Essig 2010, p. 89)** |
| **Portneuf R., downstream of Lava Hot Springs** | **1.89 – 6.8** | **0.4 – 1.1** | **Brown trout** | **(Essig and Kosterman 2008, p. 64; Essig 2010, p. 89)** |
| **Salmon R. downstream of SF Salmon R** | **0.98 – 1.1** | **0.68** | **N. pikeminnow** | **(Essig 2010, p. 89-92)** |
| **Silver Creek, ID** | **0.15 – 1.45** | **0.5 – 0.67** | **Brown trout** | **(Essig 2010, p. 89-92)** |
| **Willamette River, OR** | **1.2 – 2** | **0.47** | **Piscivores** | **(Hope and Rubin 2005, pp 371,377)** |
| **Cottage Grove Res., OR** | **5.8** | **1.63** | **Piscivores** | **Hope and Rubin 2005, pp 371,377)** |
| **TMDL target for the Willamette R., OR** | **0.92** | **0.3** | **median for higher trophic level fish** | **(Hope et al. 2007, entire)** |

Even waters considered to have significant mercury contamination, as evidenced by fish tissue sample concentrations, seldom exceed the proposed 12 ng/L waterborne criterion. For example, despite concentrations of mercury in the tributaries and water column of Salmon Falls Reservoir, Idaho being within criteria (1.04 to 10.6 ng/L), levels in the tissues of most fish species in the reservoir [walleye (*Sander vitreus vitreus*), yellow perch (*Perca flavescens*), smallmouth bass (*Micropterus dolomieu*), rainbow trout (*Oncorhynchus mykiss*), and the largescale sucker (*Catostomus macrocheilus*)] exceeded the Idaho fish-tissue based water quality standard of 0.3 mg/kg (IDEQ 2007, p. 185). For piscivorous fish in the Willamette River, Oregon, a modeled waterborne mercury concentration of 0.92 ng/L was considered adequate to meet Oregon's 0.3 mg/kg fish tissue criterion (Hope et al. 2007, entire). NMFS (2014a, pp. 144-162) provided many more examples of fish tissue mercury threshold exceedance without even approaching the proposed 12 ng/L chronic criterion for mercury in water.

The common occurrence of mercury tissue concentrations in the tissue of fish exceeding a threshold concentration for reproductive or neurologic harm considered applicable to bull trout (0.3 mg/kg ww) while water concentrations of mercury were considerably less than the proposed 12 ng/L chronic aquatic life criterion indicates that the proposed chronic criterion would not be sufficient to protect all fish species. As no species-specific information were available for bull trout, we consider this general "fish: endpoint to apply to bull trout as well.

Resident and juvenile migratory bull trout prey on small fish, including salmon fry, as well as terrestrial and aquatic insects, and macro-zooplankton (Boag 1987; Goetz 1989; Pratt 1992, p. 6; Donald and Alger 1993). Adult migratory bull trout feed almost exclusively on other fish (Rieman and McIntyre 1993, p. 3); robust bull trout populations may depend on abundant fish prey resources. This relationship is shown by the correlation between declines in bull trout abundance and declines in salmon abundance (Rieman and McIntyre 1993, p. 3). Mercury at elevated concentrations has been demonstrated to cause adverse effects ranging from coughing to neurotoxicity. Long-term dietary exposure to methylmercury can cause incoordination, inability to feed, and diminished responsiveness (Matida et al. 1971, Scherer et al. 1975) in other species, including bull trout prey species. If bull trout prey fish are less available or are available but constitute a lower quality food source, this may adversely impact individual bull trout and ultimately result in reduced weight gain, reduced reproductive success, and reduced survival.

Based on the above information, implementation of the proposed chronic criterion for mercury is likely to adversely affect growth, reproduction, and behavior in the bull trout throughout its distribution in Idaho. Considering that the state of Idaho harbors 44 percent of all streams and 34 percent of all lakes and reservoirs occupied by the bull trout rangewide, these effects are considered to be significant. These effects are likely to impede (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations.

## 2.5.6.3 Bull Trout Critical Habitat

Based on the analysis above regarding the bull trout and the following discussion, the proposed chronic mercury criterion is likely to create habitat conditions that are likely to adversely affect bull trout critical habitat via reductions in prey quality (PCE 3) and reductions in water quality (PCE 8).

Resident and juvenile migratory bull trout prey on small fish, including salmon fry, as well as terrestrial and aquatic insects, and macro-zooplankton (Boag 1987; Goetz 1989; Pratt 1992, p. 6; Donald and Alger 1993). Adult migratory bull trout feed almost exclusively on other fish (Rieman and McIntyre 1993, p. 3); robust bull trout populations may depend on abundant fish prey resources. This relationship is shown by the correlation between declines in bull trout abundance and declines in salmon abundance (Rieman and McIntyre 1993, p. 3). Mercury at elevated concentrations has been demonstrated to cause adverse effects ranging from coughing to neurotoxicity. Long-term dietary exposure to methylmercury can cause incoordination, inability to feed, and diminished responsiveness (Matida et al. 1971, Scherer et al. 1975) in other species, including prey species. Declines in prey species may adversely affect the capability of the bull trout critical habitat to provide an abundant food base (PCE 3) for the bull trout.

In addition, due to the continuous interactions between surficial sediment, interstitial water, and overlying water or water column, the condition or quality of sediment cannot be separated from water quality, and elevated contaminant concentrations, in sediments are interrelated with water column concentrations. Sediments can act as both a sink and source of mercury in streams, lakes and wetlands. While this is a concern with all aquatic contaminants, because of the role of sulfate reducing mercury in sediments causing methylation, this is especially the case with mercury (e.g., Fitzgerald and Lambourgh 2007; Gray and Hines 2009; Marvin-DiPasquale et al. 2009).

The proposed mercury criteria are applied on a statewide basis and are effective in perpetuity; until new numbers are proposed to replace the criteria; or until site-specific exceptions to criteria are made (site-specific criteria generally allow greater concentrations than those allowed under statewide criteria). Because the proposed water quality criteria would be implemented statewide, all bull trout critical habitat within the state of Idaho would be subjected to aquatic mercury concentrations up to 2.1 µg/L (acute) and 0.012µg/L (chronic), in addition to unknown and unregulated concentrations in sediment.

Within the conterminous range of bull trout, a total of 19,729 miles of streams and 488,252 acres of lakes and reservoirs are designated as critical habitat. Of that, the state of Idaho contains 8,772 miles of streams and 170,217 acres of lakes and reservoirs designated as critical habitat (75 FR 63937). Thus, the proposed action would impair the capability of approximately 44 percent of the total designated streams and 35 percent of the total designated lakes and reservoirs (via elevated mercury criteria) to adequately function in support of bull trout recovery with respect to an abundant prey base (PCE 3) and water quality (PCE 8). On that basis, these adverse effects are considered to be significant.

## 2.5.6.4 Kootenai River White Sturgeon

The preceding discussion and cited information regarding the effects of the proposed water quality criteria for mercury on the bull trout are also largely applicable to the evaluation of these criteria on the Kootenai River white sturgeon because for some contaminants, the white sturgeon and other sturgeon species are at least as sensitive as the rainbow trout (Dwyer et al. 2005; Ingersoll and Mebane 2014). On that basis, we assumed for this analysis that the white sturgeon is at least as sensitive to mercury as are salmonids. In addition as discussed below, the greatest mercury concentrations have been found in piscivorous fish species and mercury content increases with higher trophic levels and the age of the organisms. White sturgeon greater than

483 mm (19 in) in length feed primarily on other fish (Scott and Crossman 1973, p. 99) and are, therefore, a high trophic level fish species (trophic level 4) similar to the bull trout.

Available information on the toxicity to salmonids of short-term exposure to mercury in water indicates that adverse effects at 2.1 µg/L of mercury (the proposed acute criterion) are unlikely. EPA (1985g, Table 1) lists LC50s for salmonids exposed to acute concentrations of mercury in the range 24-84 µg/L, based on tests where the water chemistry was measured. Given that these LC50 values are well above (11 to 40 times greater) than the proposed acute criteria, and that rainbow trout is often a reasonably protective surrogate species for avoiding acute toxicity to sturgeon (Dwyer et al. 2005), the Service concludes that the proposed acute criterion for mercury is unlikely to cause adverse effects to the sturgeon.

The proposed chronic criterion for protection of aquatic life relative to mercury is considerably more complex to evaluate. Food chain transfer is by far the most important exposure pathway in aquatic ecosystems (Wiener et al. 2003). Aquatic systems have complex food webs including several trophic levels. Aquatic predators including salmonids are most susceptible to bioaccumulating mercury, and thus their tissue concentrations may best reflect the amount of mercury available to aquatic organisms in the environment. For example, in comparisons of fish and invertebrates across trophic levels McIntyre and Beauchamp (2007, p. 577) determined that the greatest mercury concentrations were found in piscivorous fish species and that mercury content increased with higher trophic levels and the age of the organisms. White sturgeon greater than 483 mm (19 in) in length feed primarily on other fish (Scott and Crossman 1973, p. 99) and are, therefore, a high trophic level fish species (trophic level 4) similar to the bull trout.

Additionally, as a very long-lived species, white sturgeon can be expected to be at considerable risk of mercury bioaccumulation. For instance, a sexually mature female sturgeon of about 41 years of age that was captured in the lower Columbia River had about 1.1 mg/kg mercury in her muscle tissue (Webb et al. 2006, p. 446). In white sturgeon studied in the lower Columbia River, mercury accumulations appeared to result in adverse effects on white sturgeon reproductive potential. Significant negative correlations between testosterone and muscle mercury content; condition factor and relative weight; and, gonad and liver mercury content were found. In addition, immature male sturgeon with increased gonad mercury content had decreased gonad size (Webb et al. 2006, entire). Webb et al (2006, p. 447) also suggested a possible threshold concentration of mercury affecting steroidogenesis at about 0.2 mg/kg muscle tissue. These apparent adverse effects on the white sturgeon are occurring in the Lower Columbia River despite water concentrations of mercury never approaching the proposed 12 ng/L chronic aquatic life criterion. Caton (2012, p. 45) reported a distribution frequency of total mercury concentrations in the Lower Columbia River as of 2009, and found the mean water total mercury concentration was 0.71 ng/L with 100 percent of the samples being less than 2 ng/L. Because the basic premise of this consultation is that all waters in the state of Idaho are at criteria concentrations, the proposed mercury chronic life criterion would allow for water concentrations of mercury in the Kootenai River to be about 16X higher than those already associated with harmful effects to white sturgeon populations (as shown in these Lower Columbia River studies).

Within the Kootenai River White Sturgeon Distinct Population Segment, there are approximately 270 river kilometers (168 river miles), of which more than 39 percent would be impacted by the proposed chronic criterion for mercury (i.e., approximately 39 percent of the distinct population segment is within the state of Idaho). Given that existing data show adverse effects caused by chronic exposure to mercury at concentrations less than the proposed criterion to multiple freshwater fish species, including potential prey species of the white sturgeon, and the likelihood that mercury concentrations will be even higher in sediments - increasing adverse impacts to white sturgeon eggs and juveniles, we conclude the proposed chronic criterion for mercury is likely to cause significant adverse effects to the growth, reproduction, and behavior of the Kootenai River white sturgeon throughout its range in Idaho. On that basis, these adverse effects are considered to be significant, and are likely to impair the capability of the sturgeon population: (1) to achieve natural production of white sturgeon in at least three different years of a 10-year period, and (2) to achieve a stable/increasing population in the wild.

### 2.5.6.5 Kootenai River White Sturgeon Critical Habitat

Based on the preceding information and findings relative to the Kootenai River white sturgeon, the proposed chronic criterion for mercury is likely to impair water quality by allowing aquatic chronic concentrations of mercury to rise to levels that have been shown to be detrimental to the growth, reproduction, and behavior of other freshwater fish. This degradation of water quality is likely to create habitat conditions within sturgeon critical habitat that are likely to impair the capability of the critical habitat to provide for its recovery support function: the normal reproduction, growth, and survival of the white sturgeon.

In addition, due to the continuous interactions between surficial sediment, interstitial water, and overlying water or water column, the condition or quality of sediment cannot be separated from water quality, and elevated contaminant concentrations, in sediments are interrelated with water column concentrations. Sediments can act as both a sink and source of mercury in streams, lakes and wetlands.

For the above reasons, the Service concludes that the proposed chronic criterion for mercury is incompatible with habitat conditions necessary to provide for the normal growth, reproduction, and behavior of the Kootenai River white sturgeon. All designated critical habitat for the sturgeon is likely to be affected in this manner so these adverse effects to habitat conditions are considered to be significant.

## 2.5.7 Selenium Aquatic Life Criteria

The proposed aquatic life criteria for selenium are an acute criterion of 20 µg/L and a chronic criterion of 5 µg/L, both expressed as "total recoverable" selenium (EPA 2000, p. 5). Idaho's chronic aquatic life criterion for selenium of 5 µg/L is unique in that it is based on "other data" (i.e., data from behavioral, biochemical, physiological, microcosm, and field studies) rather than EPA's customary approach that uses the 5$^{th}$ percentile of the species sensitivity distribution (SSD) in conjunction with an acute-chronic toxicity ratio (ACR) (EPA 1985a, p. 28). The "other data" provision in EPA's Guidelines for developing aquatic life criteria serves to allow the use of pertinent information that could not be used directly in the usual ranked species sensitivity approach. Data from any type of adverse effect that has been shown to be biologically important could be used, such as data from behavioral, biochemical, physiological, microcosm, and field studies. If the "other data" show that a lower criterion value should be used instead of the usual final chronic value, then the chronic value would be based on this "other data" (Stephan et al. 1985a, section X).

Selenium occurs naturally in the environment and is an essential micronutrient for all animals that have a nervous system, yet it is toxic at not much higher concentrations (Eisler 1985). Selenium accumulation is modified by water temperature, age of the organism, route of exposure, and other factors (Eisler 1985). Selenium toxicity is primarily manifested as reproductive impairment due to maternal transfer, resulting in embryotoxicity and teratogenicity in egg-laying vertebrates such as birds and fish (Janz et al. 2010, pp. 149-152). The most sensitive toxicity endpoints in fish larvae are teratogenic deformities such as skeletal, craniofacial, and fin deformities, and various forms of edema (Janz et al. 2010, p. 152). Embryo mortality and severe development abnormalities can result in impaired recruitment of individuals into populations (Janz et al. 2010, pp. 209-210).

Diet is the primary pathway of selenium exposure for both invertebrates and vertebrates (Chapman et al. 2009, p. 5). Selenium readily bioaccumulates in aquatic food webs, and biomagnifies (increases with increasing trophic level) (Presser and Luoma 2010, fig. 6). The single largest step in the bioaccumulation of selenium occurs at the base of food webs, characterized by an "enrichment function," with much lower increases at higher trophic levels (Chapman et al 2009, pp. 5-7). However, lower trophic level organisms are less sensitive to selenium toxicity than higher tropic level organisms (Lemly 1993, p. 83). Piscivorous fish accumulate the highest levels of selenium and are generally one of the first organisms affected by selenium exposure, followed by planktivores and omnivores (Lemly 1985).

Short-term (acute) toxicity does not appear to be an issue of concern for any species at concentrations remotely close to the proposed acute criterion for selenium of 20 µg/L. For instance, 96-hr LC50 values for rainbow trout exposed to selenium range from 4,200 to 47,000 µg/L (EPA 1987a, Table 1, pp. 42, 46). For this reason, acute selenium toxicity is not generally considered ecotoxicologically relevant to fish. Dietary exposure of fish to selenium is not considered an acute toxicity hazard, although it is considered a chronic toxicity hazard (Janz 2011, p. 338). EPA (1987a, Table 1, p. 46) reported an LC50 of 193,000 µg/L for the snail *Aplexa hypnorum*. For this reason, the proposed acute criterion for selenium of 20 µg/L appears unlikely to cause adverse effects to listed aquatic snail species or their habitats.

## 2.5.7.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

Although selenium is found in both particulate and dissolved forms in water, selenium found in particulate matter (algae, detritus, and sediment) is the main pathway by which selenium enters into the aquatic food web (EPA 2014, p. 16). Primary producers (bacteria, fungi, algae, and plants) rapidly assimilate and transform inorganic selenium into organic selenium species. (This is the enrichment function described above). By feeding on these selenium "enriched" primary producers, primary consumers including invertebrates like the listed Snake River snails, transfer organic selenium throughout the aquatic food web (Chapman 2009, et al.). However, based on the following references, "most species of invertebrates, which are essential components of aquatic food webs and a key vector for transfer of organic selenium to higher trophic levels are also relatively insensitive to selenium" (Chapman et al. 2009, pp. 22-23).

Lemly (1993, p. 93) noted that "food-chain organisms can build up tissue concentrations of selenium that are toxic to predators while remaining unaffected themselves." In a selenium-contaminated reservoir, Lemly (1985a, p. 314) noted that the abundance and diversity of invertebrate biota, including molluscs, was not affected by selenium concentrations that devastated the fish community. The pattern of selenium accumulation in different taxonomic groups was found to be: fishes > insects > annelids > molluscs > crustaceans > plankton > periphyton. No adverse effects of selenium on the abundance of molluscs (not further identified) were apparent up to 10 µg/L after two years of exposure (Lemly 1985a, p. 314).

Other extended exposures of aquatic snails to selenium have been made in quasi-natural, experimental food webs. In large pond enclosures, Turner and Rudd (1983, pp. 2229, 2232) exposed natural communities to up to 100 µg/L of selenium, as sodium selenite, for at least 40 days. Snails and clams accumulated selenium at up to 4 mg/kg in tissue wet weight (ww) without any obvious adverse effects. Much of the literature on selenium effects relates to tissue residues in dry weight (dw) tissue. Assuming 75 to 80 percent moisture content in tissues (EPA 2014, pp. 250-251), 4 mg/kg wet weight would be about 16 to 20 mg/kg dry weight (dw). Crane et al. (1992, entire) exposed experimental streams for about 9-months with selenium concentrations up to 25 µg/L. In the highest concentration, molluscs accumulated selenium to 56 mg/kg dw with little evidence of reductions in abundance (Crane et al.1992, pp. 444,449). Amweg et al. (2003, entire) monitored invertebrate tissue selenium concentrations for 2 years in selenium-enriched ditches and ponds. Snails (*Physa* sp.) persisted in a ditch with a mean selenium concentration of 384 µg/L, with tissue residues up to about 60 mg/kg dw without obvious adverse effects (Amweg et al. 2003, p. 18). Additional snail taxa (*Heliosoma* and *Lymnaea*) were exposed in microcosms to the same selenium-enriched waters and sediments for 30 days. Snails survived water concentrations up to about 50 µg/L (dominated by selenite) and up to 384 µg/L (dominated by selenate), during which time they accumulated tissue selenium up to about 50 to 60 mg/kg dw (Amweg et al. 2003, pp. 20-22).

Although the populations of listed aquatic snail species are generally fragmented, limited, or isolated, and while the studies cited above were not definitive and were not specific to listed Snake River snails, they do reasonably support a conclusion that there is no evidence that significant effects to molluscs, including the listed Snake River aquatic snails and the Bruneau hot springsnail, are likely to occur at selenium concentrations approaching either the proposed acute aquatic life criterion of 20 µg/L or the proposed chronic aquatic life criterion of 5 µg/L.

## 2.5.7.2 Bull Trout

The recognition of decimated fish populations in selenium-influenced reservoirs and the occurrences of severely deformed aquatic bird embryos in western reservoirs and wetlands that received elevated selenium input in irrigation return water (e.g., Presser 1994, entire; Chapman et al. 2010, Appendix A) led to much research on selenium bioaccumulation and toxicity in aquatic organisms since the 1980s. Thus, a large body of knowledge has become available subsequent to EPA's 1987 selenium criteria document.

Recently, several key areas of consensus in the scientific community have formed regarding selenium risks to fish and water quality criteria to protect them from these risks.

- Diet is the primary pathway of selenium exposure for both invertebrates and vertebrates (Chapman et al. 2009, p. 5).
- Selenium tissue concentrations are more closely related to toxicity in fish than are dissolved selenium concentrations in water (Janz et al., 2010, p. 142).

- Traditional methods for predicting toxicity on the basis of exposure to dissolved concentrations do not work for selenium because the behavior and toxicity of selenium in aquatic systems are highly dependent upon situation-specific factors, including food web structure and hydrology (Chapman et al. 2009, p. 5).
- Selenium toxicity is primarily manifested as reproductive impairment due to maternal transfer, resulting in embryo toxicity and teratogenicity in egg-laying vertebrates (Chapman et al 2009, pp. 5-7; Janz et al 2010, pp. 209-210).

In addition to reproductive failure due to maternal transfer of selenium, growth of juvenile fish can also be impaired from tissue accumulation of selenium (Lemly 1993, p. 85). The NMFS (2014a, p. 180) concluded that a whole-body average fish tissue concentration of 7.6 mg/kg dw would be low risk for appreciable growth reductions in juvenile Chinook salmon or steelhead. Using a food-web model, with water concentrations that were near the proposed chronic selenium criterion concentration of 5 µg/L for an indefinite period, selenium was projected to be transferred through the food web resulting in selenium concentrations in juvenile salmonids greater than twice as high as the 7.6 mg/kg dw concentration estimated to be low risk for appreciable growth reductions in juvenile salmon or steelhead. A water concentration of selenium of about 2 µg/L was derived from the modeling for the selenium value of 7.6 mg/kg dw value estimated to be low risk for appreciable growth reductions in juvenile salmon or steelhead.

Palace et al. (2004, entire) analyzed selenium concentrations in the muscle tissue of bull trout sampled in a river with elevated selenium concentrations caused by coal mining. Selenium residues in bull trout muscle were elevated to the point that the authors considered selenium likely to cause recruitment impairment in a declining bull trout population. However, Palace et al. (2004, entire) reached this conclusion by assuming that selenium toxicity and tissue relations in rainbow trout and brook trout were relevant to bull trout. Although no specific toxicity test data for the bull trout were located during this consultation, reproductive toxicity testing that involved relating the occurrences of unviable or deformed fry to selenium concentrations in eggs has been conducted with rainbow trout and two char species (the brook trout and the Dolly Varden) closely related to the bull trout. These studies are discussed below.

Holm et al. (2005, entire) evaluated patterns in selenium tissue accumulation and incidences of larval deformities in brook trout and rainbow trout collected from coal mining-influenced coldwater streams. While the two species differed in relative tissue values of selenium concentrations between muscle and egg tissues, and in the incidence of deformities, both species showed elevated deformities in fry hatched from adults captured in streams with elevated selenium levels (Holm et al. 2005, p. 230). EPA reanalyzed these data using a consistent approach across multiple studies, and obtained a EC10 (10 percent effects relative to reference) of 20.6 mg/kg dw of selenium in eggs (EPA 2014, pp. 408-421), which approximately translates to a whole-body concentration of 14.9 mg/kg of selenium using a brook trout egg to whole-body selenium tissue ratio of 1.38:1 (EPA 2014, appendix B).

It is important to note that egg to whole-body tissue relationships are variable and different conversion factors have been estimated for different species (EPA 2014, table 11). For instance, the egg to whole body conversion factors for salmonids range from a low of about 1.4 for Dolly Varden and brook trout to 7.39 for the mountain whitefish (EPA 2014, table 11). The implication of these ratios is that when extrapolating egg-ovary tissue concentrations to the more easily measured and more widely reported whole body tissues, a higher ratio will result in a lower whole-body estimate. For example, if the 20.6 mg/kg dw egg-ovary value for the brook trout were extrapolated for the bull trout using a brook trout egg-whole body conversion of 1.38, the estimate of a whole-body tissue, low risk selenium concentration for the bull trout would be 14.9 mg/kg dw. However, if the average egg to whole body ratio for salmonids (2.8) was used to make an estimate for the bull trout instead of the brook trout ratio, then the resulting whole-body tissue, low-risk selenium concentration estimate would be 7.4 mg/kg dw.

EPA (2014, pp. 57-59) also evaluated an unpublished study of selenium exposures to Dolly Varden char. Adult Dolly Varden char were collected from a reference stream, and from streams with high and moderate selenium exposure in a coal mining region of British Columbia. Fertilized eggs obtained from the adult char were taken to a private laboratory for testing, with the survival of the eggs and alevins followed through swim-up, at about 5 months. The prevalence of deformities increased sharply after the selenium egg concentration exceeded 50 mg/kg dw, with no obvious effects at lower concentrations (EPA 2014, pp. 57-59). These highly divergent results from different research groups working in different localities with different species within the genus *Salvelinus* suggests that taxonomic similarity may not always be the most important determinant of a species relative sensitivity to selenium.

In addition to the NMFS (2014a, p.180) conclusion that a selenium concentration of about 2 µg/L would be sufficient to protect listed salmonids from selenium toxicity, other reviews have reached similar conclusions that 5 µg/L of total selenium in water may not always be protective, whereas a concentration of 2 µg/L likely would be (USFWS and NMFS 2000, pp. 132-133; Lemly and Skorupa 2007, entire). EPA (2014, p. 96) proposed criterion concentrations of selenium in water not to exceed 4.8 µg/L in lotic (flowing) waters and 1.3 µg/L in lentic (standing) waters more than once in three years on average. How this might be applied is still uncertain, as the lotic/lentic classification is more of a continuum than a bright line. Some waters, such as the slow moving, highly sinuous, meandering rivers that occur in some alluvial valleys will have hydrologic characteristics intermediate to the classic “lentic” and “lotic” split which is not addressed in EPA (2014, entire) other than the possibility of deriving site-specific criteria (EPA 2014, pp. 100-101).

EPA (2014, p. 96) has recently proposed an updated selenium aquatic life criteria comprised of four elements: (1) fish egg or ovary tissue not to exceed 15.2 mg/kg dw, (2) whole-body tissue of 8.1 mg/kg dw, (3) average selenium concentrations in water (discussed above), and (4) a formula based limit for intermittent selenium concentrations in water. In the absence of bull trout specific data for low-risk whole-body tissue values, the 8.1 mg/kg value provides an estimated tissue residue value that is expected to be protective of most species. Lower tissue residue values have also been recommended. For instance, DeForest et al. (1999, p 1187) suggested a whole-body (wb) value of 6 mg/kg dw would be protective of coldwater fish species, and Lemly (1993, p. 92) suggested a whole-body value of 4 mg/kg dw would be protective of coldwater fish species in general. Presser (2013, p.45) considered a whole-body selenium effect guideline of 5 mg/kg dw would provide protection for adherence to both the Clean Water Act and the Endangered Species Act. This estimate of the threshold guideline is required to provide full protection for individuals of even selenium-sensitive species of threatened or endangered fish. Because the EPA (2014, entire) dataset is more comprehensive than those available to other researchers, it may be more relevant to estimating a low-risk fish tissue value for the bull trout than previous EPA estimates. As stated above, EPA (2014, p. 96) estimated that a water selenium concentration of 1.3 µg/L in lentic habitats (e.g., lakes and slow-moving rivers) would be sufficient to avoid exceeding the 8.1 mg/kg wb dw estimate. The proposed chronic criterion value for selenium of 5 µg/L is considerably higher than the 1.3 µg/L selenium estimated to be protective by EPA.

Using the EPA 8.1 mg/kg whole-body tissue value with the trophic transfer in lotic (stream) food web calculations by NMFS (2014, pp. 175-180), yields a low risk lotic water value of selenium at 2.5 µg/L. This is lower than the proposed chronic criterion value of 5 µg/L and the most recent chronic criterion for selenium proposed by EPA (2014, p. 107) at 4.8 µg/L. The reasons for the differences between the EPA and NMFS calculations could not be determined because while EPA (2014) reported the results of their calculations, they did not provide the actual data used in the calculations.

In addition to possible direct toxicity and adverse effects caused by bull trout exposure to selenium at the proposed chronic criterion level, such a selenium concentration in water may indirectly affect the bull trout through reduced prey availability, or elevated sediment concentrations. Resident and juvenile migratory bull trout prey on small fish, including salmon fry, as well as terrestrial and aquatic insects, and macro-zooplankton (Boag 1987; Goetz 1989; Pratt 1992, p. 6; Donald and Alger 1993). Adult migratory bull trout feed almost exclusively on other fish (Rieman and McIntyre 1993, p. 3); robust bull trout populations may depend on abundant fish prey resources. For example, declines in bull trout abundance have been associated with declines in salmon abundance (Rieman and McIntyre 1993, p. 3). At a selenium concentration of 5 µg/L, researchers observed a collapse (>90 percent) of planktivorous fish biomass (Lemly 1985, Garrett and Inman 1984). These examples have implications for the bull trout. If their prey fish are less available or are available but constitute a lower quality food source, this may adversely impact individual bull trout and ultimately result in reduced weight gain, reduced reproductive success, and reduced survival. Selenium concentrations at the chronic criterion of 5 µg/L proposed for approval in water may result in reproductive failure in exposed bull trout. Lemly (1993) developed toxic effect thresholds for selenium in fish and wildlife that might indicate reproductive failure in fish and wildlife at aquatic selenium concentrations of 2 µg/L of inorganic selenium, or less than 1 µg/L of organic selenium.

Since Idaho contains 44 percent of bull trout-occupied streams and 34 percent of bull trout-occupied lakes and reservoirs within the range of the bull trout, the above adverse effects are considered to be significant, and are likely to impede (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations.

## 2.5.7.3 Bull Trout Critical Habitat

Of the nine PCEs designated for bull trout critical habitat, the proposed chronic criterion for selenium is likely to adversely affect PCE 3 (adequate prey base) and PCE 8 (water quality) for the reasons discussed above in the analysis for the bull trout and further discussed below. Resident and juvenile migratory bull trout prey on small fish, including salmon fry, as well as terrestrial and aquatic insects, and macro-zooplankton (Boag 1987; Goetz 1989; Pratt 1992, p. 6; Donald and Alger 1993). Adult migratory bull trout feed almost exclusively on other fish (Rieman and McIntyre 1993, p. 3); robust bull trout populations may depend on abundant fish prey resources. This relationship is shown by the correlation between declines in bull trout abundance and declines in salmon abundance (Rieman and McIntyre 1993, p. 3). Selenium at concentrations below the proposed chronic criterion has been demonstrated to cause reduced growth, teratogenic deformities, kidney damage, tissue accumulation, and mortality in other species, including bull trout prey species or similar species. The decline of these other species will adversely affect the ability of the bull trout critical habitat to provide an abundant food base (PCE 3) for the bull trout.

The proposed approval action will impair water quality (PCE 8) by allowing aquatic selenium concentrations to rise to levels that have been shown to be detrimental to other salmonids. Adverse effects to salmonids were observed at dietary concentrations below the proposed criterion and were discussed above (see previous *Bull Trout* section). Assuming bull trout are affected in a similar manner as other salmonids, selenium concentrations at the proposed chronic criteria level could impair the ability of critical habitat to provide for the normal reproduction, growth, and survival of bull trout.

In addition, selenium in aquatic environments is tightly linked between sediment and overlying water. Elevated selenium in water can result in sediment loading and subsequently release selenium back into the aqueous environment. Selenium toxicity can result from accumulation of selenium in the sediment, movement into the food chain and resulting dietary uptake. Biogeochemical processes in sediments result in transformation of less-toxic inorganic selenium to more toxic organic selenium (Canton and Van Derveer 1997; Martin et al. 2011).

The proposed selenium criteria are applied on a statewide basis and are effective in perpetuity or until new numbers are proposed to replace the criteria or until site-specific exceptions to criteria are made (site-specific criteria generally allow greater concentrations than those allowed under statewide criteria).

Based on the above findings, the proposed chronic criterion for selenium is likely to impair the capability of approximately 44 percent of the total designation of bull trout critical habitat for streams and 35 percent of the total designation of bull trout critical habitat for lakes and reservoirs to provide an adequate prey base (PCE 3) and water quality (PCE 8) essential for bull trout recovery.

### 2.5.7.4 Kootenai River White Sturgeon

Tashjian et al. (2006, entire) tested the responses of juvenile white sturgeon to dietary exposures of organic selenium using a battery of histopathological, swimming ability and growth measurements. EPA reanalyzed their data and determined a no observed effect concentration (NOEC) for reduced growth of 14.7 mg/kg dw, and a 10 percent reduction in growth resulted at about 15.1 mg/kg (EPA 2014, p. 380-381).

With other fish species, reproductive effects have often been considered the most sensitive endpoint (Chapman et al 2009, pp. 5-7; Janz et al 2010, pp. 209-210). With sturgeon, the available information reported similar effects concentrations resulting from reproductive endpoints as those from the growth test noted above. Linville (2006, p. 144) estimated a EC10 for increased rates of deformed fry (edema or skeletal deformities) at 15 mg/kg dw in eggs or larvae that had been exposed to selenium through maternal transfer.

If EPA's (2014, p. 107) water and whole-body proposed criteria for selenium were applied to the juvenile white sturgeon NOEC as a simple ratio of 1.8 (i.e., 1.8=14.7 mg/kg sturgeon juvenile whole-body tissue value ÷ 8.1 mg/kg generic fish tissue value), multiplied by their lotic and lentic (4.8 and 1.3 µg/L, respectively) low risk concentrations, then that would imply low-risks for reductions in juvenile sturgeon growth at a selenium concentration value of 8.7 µg/L in lotic waters and 2.4 µg/L in lentic waters. This suggests that following the EPA (2014, entire) approach, the 5 µg/L proposed chronic criterion for selenium is likely to be unprotective of the sturgeon in slow-moving lentic environments, but might be in faster-water lotic environments. However, as noted, absent data on the effects of selenium on sturgeon reproduction, a low risk of reducing growth of juvenile white sturgeon may not be the same as a low risk to the normal growth, reproduction, and survival of the white sturgeon over its full life cycle.

Further, it is possible that the generic trophic transfer estimates used by EPA to protect most species might not be protective of white sturgeon. Presser and Luoma (2006, table 12) reported selenium concentrations in the muscle flesh of adult white sturgeon in San Francisco Bay ranged from 7.8 to 15 mg/kg, with the highest value occurring in sturgeon sampled from the North Bay area. Yet the highest water column value of selenium during the study was only 0.44 µg/L, also occurring in the North Bay area. This suggests that depending on the specific dietary pathway, selenium may accumulate in the tissue of white sturgeon to hazardous concentrations even though water column selenium concentrations never exceeded 1.0 µg/L. The characteristics of the dietary pathway in San Francisco Bay may make a given concentration of selenium in the diet relatively more toxic to sturgeon than in river habitats such as the Kootenai River. This is because sturgeon in San Francisco Bay preferentially prey on the Asian clam, *Corbicula sp*., which has a higher trophic transfer factor than would the mixed diet expected in the Kootenai River, such as crayfish, sculpin, mussels, other invertebrates (Scott and Crossman, 1973, p. 99; Muir et al 2000; Stewart et al. 2004). Still, it suggests that selenium concentrations of 5 µg/L could result in harmful trophic transfer in a quasi-lentic, slow moving riverine scenario.

Because acute adverse effects of selenium have only been observed at much higher concentrations than the proposed acute criterion, we conclude the proposed acute aquatic life criterion for selenium is not likely to adversely affect the white sturgeon. However, based on effects observed in juvenile white sturgeon and tissue concentrations occurring in adult white sturgeon, we conclude that the proposed chronic aquatic life criterion for selenium is likely to adversely affect the sturgeon relative to its growth and reproduction.

The proposed criteria levels for selenium are also likely to indirectly affect the white sturgeon through elevated sediment concentrations that affect sturgeon prey and sturgeon growth and reproduction. White sturgeon are known to be opportunistic feeders (Partridge 1983). They are primarily bottom feeders but larger individuals will also take prey in the water column (Scott and Crossman 1973). Smaller sturgeons feed predominantly on chironomids; for larger individuals of sturgeon, fish and crayfish become the predominant foods, although chironomids remain a significant portion of their diet (Scott and Crossman 1973). At a selenium concentration of 5 µg/L, researchers observed a collapse (>90 percent) of planktivorous fish biomass (Lemly 1985, Garret and Inman 1984). These fish species are likely to be prey for white sturgeon. If these fish are less available or are available but constitute a lower quality food source, this is likely to adversely impact white sturgeon and ultimately result in reduced weight gain, reduced reproductive success, and reduced survival. Selenium concentrations at the proposed criteria in water may result in reproductive failure in white sturgeon: Lemly (1993) developed toxic effects thresholds for selenium in fish and wildlife that indicate reproductive failure in fish and wildlife at aquatic concentrations of 2 µg/L of inorganic selenium, or less than 1 µg/L of organic selenium.

The Kootenai River white sturgeon DPS is restricted to approximately 168 river miles of the Kootenai River in Idaho, Montana, and British Columbia, Canada. Approximately 39 percent of the DPS is found within the state of Idaho and would be impacted by the proposed water quality chronic criterion for selenium as described above. Given the nature of the effects and the scale of the effects relative to the range of the Kootenai River white sturgeon, these adverse effects are considered to be significant and will likely impede natural reproduction of the sturgeon in the wild, and the achievement of a stable/increasing population at the rangewide scale.

### 2.5.7.5 Kootenai River White Sturgeon Critical Habitat

The preceding analysis of effects of the proposed selenium criteria on the Kootenai River white sturgeon supports a finding that habitat conditions within critical habitat with selenium at the proposed chronic criterion level are likely to adversely affect water quality by allowing aquatic selenium concentrations to rise to levels that have been shown to be detrimental to other freshwater fish. Adverse effects were observed in numerous freshwater fish species at dietary concentrations resulting from conditions below the proposed criteria. As described earlier, in a slow-moving, quasi-lentic river system, selenium concentrations of 5 µg/L would be expected to load the food web with sufficient selenium to be harmful to white sturgeon.

In addition, selenium in aquatic environments is tightly linked between sediment and overlying water. Elevated selenium in water can result in sediment loading and subsequently release selenium back into the aqueous environment. Selenium toxicity can result from accumulation of selenium in the sediment, movement into the food chain and resulting dietary uptake. Biogeochemical processes in sediments result in transformation of less-toxic inorganic selenium to more toxic organic selenium (Canton and Van Derveer 1997; Martin et al. 2011).

Because the proposed water quality criteria would be implemented statewide, all designated critical habitat for the Kootenai River white sturgeon would be subjected to chronic aquatic selenium criterion concentrations of 5.0 µg/L. Habitats under criterion conditions are likely to cause reduced growth, reproduction, and survival of the sturgeon throughout the area designated as critical habitat. For those reasons, the proposed chronic criterion for selenium is likely to have significant adverse effects on sturgeon critical habitat to an extent that impairs its capability to support recovery of the Kootenai River white sturgeon.

## 2.5.8 Zinc Aquatic Life Criteria

The proposed acute and chronic criteria values for zinc are 117 and 118 µg/L, respectively, as calculated from the following equations using a water hardness value of 100 mg/L:

$$\text{Acute zinc criterion } (\mu g/L) = e^{(0.8473[\ln(\text{hardness})]+0.884)} * 0.978$$

$$\text{Chronic zinc criterion } (\mu g/L) = e^{(0.8473[\ln(\text{hardness})]+0.884)} * 0.986$$

With zinc and several other hardness-dependent aquatic life criteria, the actual criteria are defined as an equation, and the table values merely illustrate comparable criteria concentrations all calculated at a water hardness value of 100 mg/L. For example, applying the above equation for the chronic zinc criterion at water hardness values of 10, 25, 50, and 250 mg/L, the corresponding chronic zinc criterion values are 17, 36, 66, and 257 µg/L, respectively. The only difference between the criteria equations is the constants at the end, which are conversion factors to adjust the criteria from a “total zinc” basis to a “dissolved basis.” While the conversion factors are close to 1.0, the acute conversion factor is slightly lower which results in calculated acute criterion values always being slightly higher than the chronic values. This relationship reflects the presumed reality of zinc being a fast- acting toxicant that is no more toxic in long-term exposures than in short-term exposures.

The proposed aquatic life criteria for zinc are unique because the acute and chronic values are nearly identical. The reason for the nearly identical acute and chronic criteria equations is that in testing with sensitive life stages of acutely sensitive species, effect values with matched short-term and long-term tests sometimes produced about as low of values from the short-term tests as from the long-term tests. Only if the acute to chronic effects ratio (ACR) is greater than 2.0 will the chronic criterion value be less than the acute criterion. This is because the acute criterion is derived by calculating a hypothetical sensitive LC50 called a “Final Acute Value” that is more sensitive than 95 percent of the tested values, and then dividing that LC50 by 2 to extrapolate from a severely lethal value (LC50) to a value expected to kill few organisms (see also the discussion in the *Common Factors* section above related to this topic). Therefore, if the chronic criterion is derived by dividing the Final Acute Value by an ACR obtained from an acutely sensitive species, only if the ACR is greater than 2.0 will the chronic criterion be lower than the acute criterion. The proposed zinc criteria were derived using ACRs <2 (EPA 1987b) which in turn was primarily based upon low ACRs from studies on the Chinook salmon (Chapman 1982). These low ACRs have been supported by more recent testing with sensitive life stages of the cutthroat trout (Brinkman and Hansen 2004), mottled sculpin (Besser et al. 2007), rainbow trout (Brinkman and Hansen 2004; Mebane et al. 2008; Ingersoll and Mebane 2014), and the white sturgeon (Ingersoll and Mebane 2014). However, testing with freshwater crustaceans has sometimes produced more sensitive results for chronic exposures than for acute exposures. DeForest and Van Genderen (2012) obtained a final ACR of 4.1 in their assessment of zinc risks to aquatic organisms following EPA criteria development guidelines.

Zinc is an essential trace element for all living cells, but can be toxic to aquatic life at higher concentrations. Natural concentrations of zinc in unpolluted freshwaters are typically less than 5 µg/L and are sufficient to meet nutritional needs (Hogstrand 2011), but zinc concentrations can exceed 2000 µg/L in mining disturbed areas (Mebane et al. 2012). Zinc is bioconcentrated from water through primary production but biomagnification beyond the primary producers appears to be limited to dietary needs (Cardwell et al. 2013). Zinc concentrations, as with other essential trace elements such as copper and iron, is tightly regulated within tissues. Deficiencies or excesses in fish are counteracted by increased or decreased uptake at the gill (Wood 2011a). The mechanisms of zinc toxicity are best defined for fish. Lethality of waterborne zinc to fish is caused by the free zinc 2+ ion, while calcium, pH, and dissolved organic matter (DOM) in the water are the principal factors modifying zinc toxicity. The principal mode of action for acute zinc toxicity to freshwater fish is inhibition of calcium uptake. Little is known about mechanisms of sublethal toxicity in fish following long-term exposures; however, lethality is often a sensitive endpoint in chronic exposures of freshwater fish (Hogstrand 2011).

In addition to lethal effects, sublethal effects of zinc on fish include behavioral avoidance of water with elevated zinc concentrations. Woodward et al. (1997) tested the avoidance behavior of the cutthroat trout to zinc concentrations and reported that the cutthroat trout avoided zinc concentrations as low as 52 µg/L, which was lower than the proposed acute zinc criterion of 66 µg/L at the (unmeasured) target test water hardness value of 50 mg/L. However, behavioral avoidance tests that were conducted in bare tanks are difficult to extrapolate to the real world where competing habitat behavioral cues are present. For instance, Korver and Sprague (1989) reported that breeding male fathead minnows avoided waters with a zinc concentration of 284 µg/L when zinc concentrations were the only variable in the tank. However, when the fathead minnow was allowed to establish a territory under a shelter within the zinc-contaminated side of the tank, a zinc concentration of 1830 µg/L was required to force the fish from the shelter. Thus, the avoidance threshold of minnows to elevated zinc concentrations was raised by about 6X when the fish had a strong influence (shelter) to remain in the area of the tank with elevated zinc concentrations. Preference for shade, which is a form of shelter, can be a stronger motivation than an avoidance response to metals for some fish. Scherer and McNicol (1998) tested the avoidance of lake whitefish to metals in a countercurrent trough that was either uniformly illuminated, or shaded in one half. Fish preferred the shade when presented with a choice between shaded and illuminated. When metals were injected into the shaded, previously preferred area, avoidance of these ions was strongly suppressed, with the response to zinc reduced by 100X.

The role of calcium as a regulating mechanism for zinc uptake and toxicity is reflected in the criteria-hardness equation, where hardness is a function of the calcium (Ca) and magnesium (Mg) content in the water. Some studies have shown the hardness-zinc toxicity relation to be very strong. For instance, in 5 sets of tests with rainbow and cutthroat trout, where each set was conducted across a range of water hardness values, the coefficient of determination ($r^2$ ) values ranged from 0.90 to 0.99 (Mebane et al. 2012). Brinkman and Johnston (2012) similarly reported an $r^2$ value of 0.94 with tests with different strains of the cutthroat trout. However, calcium is a much more important factor than magnesium as a regulating mechanism for zinc uptake and toxicity. Tests by De Schamphelaere and Janssen (2004) show lesser effects of magnesium than calcium in reducing zinc toxicity and tests by Alsop and Wood (1999) showing no influence of magnesium on zinc toxicity. This has implications for analyzing the effects of the proposed zinc criteria in waters with a high magnesium content (see section 2.5.8.2 below).

Two minor differences were noted between the description of the zinc aquatic life criteria as shown in EPA's (2013d, *in litt.*) revised description of the action and the actual zinc criteria adopted into the Idaho Water Quality Standards. First, the table values in EPA (2013d, *in litt.*) for acute and chronic zinc criteria at a water hardness value of 100 mg/L are given as 120 µg/L, whereas the equations above yield acute and chronic values for zinc of 117 and 118 µg/L, respectively. The published Idaho Water Quality Standards (WQS) contain the same differences with table values for zinc of 120 µg/L yet use the same equation factors as above. However, the Idaho WQS also specify that the equation values take precedence over the table values, at subsection 201.01, footnote (i.), (IDEQ, variously dated, at section 58.01.02.210 of the WQS). The second difference is that the acute and chronic equations for the zinc criteria listed in EPA (2013d, *in litt.*) produce acute and chronic criteria concentration values of 117 and 105 µg/L, respectively. The latter discrepancy appears to be an oversight in updating the revised action table from that originally given in EPA (1999b). The evaluations of the protectiveness of the proposed zinc criteria in this Opinion are based on the criteria equations and values given above. The above differences are minor and do not materially affect the present evaluation, but are identified to allow reconciliation of criteria values in the present Opinion with the description of the action published in the Idaho WQS.

## 2.5.8.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

Several zinc toxicity studies have been conducted with snail species in the same families as the listed Snake River aquatic snails, i.e., the family Hydrobiidae (Bliss Rapids snail and Bruneau Hot springsnail), family Physidae (Snake River Physa), and the family Lymnaeidae (Banbury Springs lanx). The Banbury Springs lanx is a freshwater limpet that has yet to be formally described as a species and thus the taxonomic classification of this freshwater limpet is not well documented. USFWS (2006b) considered it to be within the family Lymnaeidae although other freshwater limpets have been classified within the family Planorbidae (Pennak 1978). To ensure relevant comparisons between European and United States zinc snail research studies, short explanations of assumptions or conversion information are provided below.

In an analysis of field collections across the United Kingdom by Peters et al. (2014), Hydrobiidae snails were the most sensitive of 64 taxa to zinc toxicity. Hydrobiidae snails were not further identified. Peters et al. (2014) determined that a field-based zinc criterion of 11 µg/L was protective of snails at an estimated water hardness value of 21mg/L. The proposed chronic zinc criterion considered herein would be 31 µg/L for a water hardness value of 21 mg/L.[17] On that basis, the proposed chronic criterion for zinc is not likely to be protective of Hydrobiidae snails. To calculate the water hardness value of 21 mg/L, calcium was reported, but magnesium was not and had to be estimated (Peters et al. 2014). Magnesium was estimated by dividing the reported calcium value of 6.4 mg/L by 5.3, which is the median Ca:Mg ratio reported by Bass et al. (2008, table 4-2) for 36 streams surveyed in the UK.

The New Zealand mudsnail, *Potamopyrgus antipodarum*, previously known as *P. jenkinsi*, an invasive species in the same family as the Bliss Rapids snail, appears to be quite sensitive to zinc. An 8-week growth study with zinc in hard water found that a zinc concentration of 72 µg/L was the lowest concentration to significantly suppress growth of P. *jenkinsi*, resulting in a 50 percent reduction in growth at a zinc concentration of about 103 µg/L (Dorgelo et al. 1995). Total hardness was not determined by Dorgelo et al. (1995) but was estimated at 225 mg/L, for which the proposed chronic criterion for zinc would be 235 µg/L, which is substantially higher than the concentrations causing growth effects. We reconfirmed the total hardness estimate reported by Dorgelo et al. (1995) as follows. The tests by Dorgelo et al. were conducted using water from Lake Maarsseveen in the Netherlands with a concentration of 63.6 mg calcium/L (Dorgelo et al. 1995). The magnesium concentration was not reported by Dorgelo et al. We estimated the magnesium concentration to be 15 mg/L based on measurements reported from nearby Lake Markermeer in the Netherlands, which had nearly identical calcium (64.4 mg/L) and 15.7 mg/L of magnesium, (De Schamphelaere and Janssen 2010) and assuming major ion

[17] This comparison ignores the arbitrary constraint that the proposed criteria are limited to hardnesses ≥ 25mg/L.

ratios are often similar across ecoregions with similar soils and geology. This composition of water results in an approximate hardness value of 225 mg/L.

A series of tests with the river limpet *Ancylus fluviatilis* showed that limpets were much more sensitive to zinc toxicity in long-term rather than short-term exposures, and effects varied greatly by exposed lifestage and endpoint (Willis 1988). For instance, a 96-h LC50 of 3,200 µg/L zinc was obtained whereas at 100 days, the LC50 had dropped to 80 µg/L. The estimated zinc chronic criterion for the test conditions was 66 µg/L, which is very close to the 80 µg/L zinc concentration that is lethal to 50 percent of the test population. In reproductive tests, no effects on reproductive rates were observed, and few effects of zinc toxicity on the growth and survival of spats (offspring) were observed for up to 3 months after hatched. Yet, after 6 months of exposure, 100 percent mortality was recorded in all treatments except the controls. The lowest zinc concentration tested, averaged 105 µg/L, and resulted in a 100 percent kill of all tested limpets after 6 months; this zinc concentration is was only moderately greater than the proposed chronic criterion concentration for zinc of about 66 µg/L at a water hardness value of 50 mg/L (Willis1988). A total water hardness value was not reported in Willis (1988), but was estimated from mean calcium concentrations of 15 mg/L, again assuming a Ca:Mg ratio of 5.3 (the median of Bass et al. (2008, table 4-2) surveys of UK streams), which gives an estimated magnesium concentration of 2.8 mg/L, resulting in an estimated total water hardness value of 49 mg/L.

Other tested species of snails in the family Lymnaeidae and Physidae appear to be more resistant to zinc than the New Zealand mudsnail. *Lymnaea stagnalis*, a pulmonate snail that is sensitive to some metals (nickel, copper, and lead) was tested with zinc in 28-day exposures under differing water quality conditions. The most sensitive threshold effect obtained, an EC10 of 200 µg/L in in water with a hardness value of 38 mg/L was four times higher than the proposed chronic criterion of 52 µg/L (De Schamphelaere and Janssen 2010). In acute zinc toxicity testing with *Lymnaea luteola*, a 96-hour LC50 of 1680 µg/L was obtained in water with a hardness value of 195 mg/L (Khangarot and Ray 1988), which is considerably higher than the proposed acute criterion value of 206 µg/L. Nebeker et al. (1986) exposed the snail *Physa gyrina*, a coolwater species from Oregon, to zinc for 21 days and obtained a no-observed effect concentration (NOEC, which is the highest concentration tested without an adverse response) of 570 µg/L of zinc in water with a hardness of 20 mg/L water, which is well above the proposed chronic zinc criterion of 30 µg/L at that hardness.

Freshwater mussels are of conservation concern in part because of their high sensitivity to contaminants, including some metals and thus may be an informative surrogate organism for effects of metals on other sensitive molluscs. Chronic exposure of mussels in the Family Unionidae to zinc showed adverse effects at zinc concentrations at 63 and 68 µg zinc/L, which is close to the proposed chronic water quality criterion for zinc of about 60 µg/L for test water hardness values ranging between 40 and 48 mg/L (Wang et al. 2010).

The most sensitive effects of aquatic organisms to zinc appear to be with algae, including green algae and diatoms. In stream microcosms stressed with zinc, changes in dominant algal taxa were observed at a zinc concentration of 50 µg/L in water with hardness values of about 71-88 mg/L. Zinc-treated stream microcosms tended to shift from diatom dominated surfaces to surfaces dominated by green and blue-green algae (Genter et al. 1987). The proposed chronic criterion values for zinc for this hardness range are 88-100 µg/L. Wong and Chau (1990) tested uptake and effects of zinc on green algae in Lake Ontario, and found reduced primary productivity and reduced cell division at a zinc concentration of 30 µg/L. No water hardness value was reported by Wong and Chau (1990), but other studies have shown Lake Ontario to have water hardness values between 120 and 130 mg/L (Alsop and Wood 1999; Borgmann et al. 2005). The proposed chronic criterion for zinc at a water hardness value of 120 mg/L is 138 µg/L.

Adverse effects of zinc toxicity on algae could have indirect adverse effects on grazing snails, and the algae effects themselves may reflect indirect effects of zinc on phosphate uptake. Because of zinc's close interaction with phosphate uptake, some of the apparent effects of zinc might not be classified as adverse toxic effects, but rather a complex expression of nutrient limitation. Effects to algae in turn may not be directly from zinc toxicity but from zinc interference with phosphate uptake (Paulsson et al. 2000; Paulsson et al. 2002; Kuwabara et al. 2007).

Whether study results (e.g., Genter et al., 1987) showing shifts in algal species composition at zinc concentrations lower than the proposed zinc criteria, support a finding that such shifts are caused by those zinc concentrations and represent indirect, adverse effects on listed snail species depends on species-specific snail feeding requirements. Available information on this matter is sparse and ambiguous. Sheldon and Walker (1997) concluded that changes in response to elevated zinc concentrations lower than the proposed zinc criteria in the composition of attached biofilms from microbial domination to filamentous green algae contributed to snail extinctions in the Murray River, Australia. Although the species of their concern in the Murray River were in different families (Viviparidae and Thiaridae) than those represented by the listed Idaho aquatic snails (Hydrobiidae, Physidae, and Lymnaeidae), the results reported by Sheldon and Walker (1997) suggest that profound changes in primary producers could affect the conservation value of habitats of primary consumers such as snails. The evidence specific to listed snail species is more ambiguous. Mladenka and Minshall (2001) found that Bruneau hot springsnails were less influenced by food resources and water quality than water temperature. Richards (2004) found that Bliss Rapids snail used a non-specific "bulldozer" feeding strategy moving slowly over the biofilm and decimating it within the grazing tracks, apparently consuming all biofilms within its tracks. Bliss Rapids snails were most abundant in stable, spring outlets where the periphyton assemblage was dominated by Oocystis (green algae) and diatoms (Richards 2004).

Based on the preceding discussion, the Service concludes that the proposed acute and chronic criteria for zinc may adversely affect algae that Snake River aquatic snails and the Bruneau hot springsnail feed upon. However, because there is an abundance of algae in the Snake River (EPA 2002a), snails such as the Bliss Rapids snail are indiscriminate biofilm grazers, and Bruneau hot springsnails are less influenced by food resources than water temperature, the Service is not expecting significant adverse effects to the listed Snake River snails and the Bruneau hot springsnail.

## 2.5.8.2 Bull Trout

The toxicity of zinc to the bull trout was extensively investigated by Hansen et al. (1999; 2002c) in laboratory waters of low and high hardness and pH. The tests were conducted in parallel with those of rainbow trout in order to scale the effects to bull trout against a commonly tested, surrogate species. Nine pairs of tests with zinc were conducted with the bull trout and separately with the rainbow trout, including a test pair with a cadmium and zinc mixture. In 8 of the 9 test pairs, the bull trout were less sensitive to zinc than were rainbow trout, and were of similar sensitivity in one pair. In the high hardness tests, zinc toxicity occurred at concentrations well above the proposed zinc criteria, but in some of the low hardness tests, mortalities to bull trout were observed at concentrations below the proposed acute and chronic criteria (Hansen et al. 2002c).

Interpreting the results of Hansen et al. (1999; 2002c) is complicated by two factors in particular: (1) great variability in the results of repeated tests under similar test conditions but using different aged fish, and (2) the laboratory water chemistry. Regarding the first factor, bull trout tested under almost identical water chemistry conditions (target of 30 mg/L hardness at a pH 7.5), LC50s varied by at least a factor of three with the smallest free-swimming (swim-up stage) bull trout and rainbow trout being most resistant. This pattern was interpreted as the smallest fish retaining some of the protective characteristics of the egg and alevin life stages, with juvenile fish become more vulnerable as they develop to a fully exogenous feeding stage (Hansen et al. 2002c). This pattern of increasing sensitivity with increasing size/developmental stage of juveniles has been seen in other studies with salmonids (Hedtke et al. 1982; Mebane et al. 2012) and we interpret the bull trout results by discounting the results obtained with the life stages that are apparently more resistant to zinc and emphasizing the more sensitive life stage results. This apparent pattern of increasing sensitivity to zinc toxicity with increasing fish size likely only holds within the juvenile stage. At some point as fish age and grow, they probably become more resistant to metals. No data supporting this conclusion are available for the bull trout, but with steelhead and Chinook salmon, 96-hr LC50s with smolts of 38 to 68g in weight were >7 times higher than those obtained with fish in the swim-up stage that were <1g in weight (Chapman 1978).

The second factor is more complicated, in that the Ca:Mg ratio of the Red Buttes, Wyoming, well water blend used by Hansen et al. as dilution water was unlike the Ca:Mg ratios expected in natural waters in Idaho and was unlike the majority of laboratory test waters used to develop national criteria. Hansen et al. conducted their studies using a well water blend with an average Ca:Mg ratio of 1.9, in contrast to a statewide Idaho average Ca:Mg ratio of about 4.4 estimated from about 3600 samples collected by the USGS at 324 sites. About 99 percent of the sites had Ca:Mg ratios >1.9 (NMFS 2014b). In the Idaho dataset, Ca:Mg values tended to be lowest in southern Idaho and highest in central Idaho, including the Boise, Salmon, and Clearwater rivers (NMFS 2014b). Similarly, the median Ca:Mg ratio of lab waters from toxicity tests used in EPA's national criteria dataset for copper is 2.7 (range 1.1-4.0) (Welsh et al. 2000, Figure 1, p. 1618). No similar analysis of tests in EPA's national criteria dataset for zinc has been made, but because of overlap between the labs that contributed toxicity data for both the copper and zinc criteria documents, the value from Welsh et al. (2000) is likely representative for zinc. The implication of using high magnesium dilution water is that the zinc toxicity results from Hansen et al. (2002c) are potentially biased low (more sensitive) for comparison to hardness-based Idaho zinc criteria than if they used a Ca:Mg composition representative of Idaho waters. This is because calcium is believed to provide greater protection from zinc toxicity than magnesium, yet Hansen et al's tests had greater magnesium influence than expected in natural waters. The total hardness values reported in Hansen et al. (2002c) were adjusted to a total hardness that we consider more representative of Idaho habitats by dividing their measured calcium values by the average Idaho Ca:Mg ratio to estimate a magnesium value that was more representative of stream water composition in Idaho. Those magnesium values were used to calculate total hardness and to then compare criteria to the toxicity testing results. For example, in the Hansen et al. (2002c) test “9906-2”, calcium was 5.6 mg/L and magnesium was 2.9 mg/L, which gave a total hardness of 26 mg/L. Dividing 5.6 by 4.4 gives an estimated magnesium concentration of 1.3 mg/L and an adjusted Idaho-relevant hardness of 19 mg/L. This gives a relevant Idaho acute criterion concentration value of zinc for this test of 29 µg/L versus 38 µg/L if the criterion was calculated with the unadjusted hardness value of 26 mg/L. The reason for examining these particular test results in detail is that substantial mortality to bull trout resulted in some of these tests at zinc concentrations lower than the proposed acute criterion concentration (Table 9). Given this finding and for the reasons discussed below, the Service concludes that substantial mortality of bull trout is likely to be caused by zinc concentrations lower than the proposed acute criterion concentrations.

**Table 9. Mortality of different sized juvenile bull trout after 96-hrs exposure to zinc in tests with targeted hardness of 30 mg/L and pH 7.5. The acute criterion maximum concentration (CMC, i.e., the acute criterion) was calculated for both the measured test hardnesses and adjusted hardnesses that were intended to better represent Idaho surface waters (see text). LC50s and percent killed at criterion concentrations were calculated from original data presented in Hansen et al. (1999).**

| | Test 1 | Test 2 | Test 3 |
|---|---|---|---|
| Test code from Hansen et al. (2002c) | B, Zn-7.5-30 | E, Zn-7.5-30 | G, Zn-7.5-30 |
| Fish wt (g) | 0.395 | 0.913 | 1.6 |
| Original (measured) hardness (mg/L) | 28.9 | 26 | 28.5 |
| "Idaho" adjusted equivalent hardness | 21.7 | 19.2 | 20.9 |
| 96-hr LC50 (µg/L) | 85 | 36 | 33 |
| Idaho CMC from original hardness | 41 | 38 | 41 |
| Idaho CMC from adjusted hardness | 32 | 29 | 31 |
| Percent killed at Idaho CMC with original hardness (96-hrs) | 0% | 67% | 90% |
| Percent killed at Idaho CMC with adjusted hardness, (96-hours) | 0 | 20% | 50% |

Of the seven acute tests with bull trout and zinc conducted by Hansen et al. (1999, 2002c), five produced no adverse effects at proposed zinc criterion concentrations and two produced substantial mortality at proposed zinc criterion concentrations. The tests with no adverse effects were conducted either at high water hardness values, lower pH (6.5), or used small fish that might have still been transitioning from the alevin life stage. The two tests where substantial mortality of the bull trout occurred, both were conducted at low water hardness values, higher pH (7.5) and with larger (>0.6g) juvenile fish.

In Idaho, many of the waters occupied by bull trout are located in the montane regions of central and northern Idaho where low hardness waters of about 30 mg/L and circumneutral pH values near 7.5 are common (NMFS 2014a, Appendix A; Hardy et al. 2005). For this reason, and because all sizes/life stages need to be protected for fish to complete life cycles, we conclude that the proposed aquatic life criteria for zinc are likely to cause substantial mortality of juvenile bull trout throughout its distribution in Idaho.

The proposed zinc criteria are also likely to adversely affect the bull trout by reducing its prey base. Bull trout are opportunistic feeders with food habits primarily a function of fish size and life history strategy. Resident and juvenile migratory bull trout prey on terrestrial and aquatic insects, macro-zooplankton and small fish (Boag 1987, p. 58; Goetz 1989, pp. 33-34; Donald and Alger 1993, pp. 239-243). Adult migratory bull trout are primarily piscivores and are known to feed on various fish species (Fraley and Shepard 1989, p. 135; Donald and Alger 1993, p. 242). According to Rieman and McIntyre (1993. p. 3) "Vigorous populations [of bull trout] may require abundant fish forage. For example, in several river basins where bull trout evolved with large populations of juvenile salmon, bull trout abundance declined when salmon declined."

The effects of elevated zinc concentrations on aquatic insect populations are complex and some information suggests measureable losses of sensitive, known bull trout prey species could occur at concentrations less than the proposed aquatic life criteria (Schmidt et al. 2011). For forage fish, Besser et al. (2007) found that mottled sculpin were decimated (100 percent killed) in 28-day exposures to a zinc concentration at 150 µg/L at a water hardness value of 103 mg/L, which is only slightly above the proposed chronic criterion concentration for zinc of 121 µg/L. The estimated LC50 (75 µg/L) was less than the proposed chronic criterion concentration for zinc. As concluded above, the proposed zinc aquatic life criteria are likely to cause substantial mortality of juvenile bull trout and other juvenile salmonids as well.[18] The decline of other juvenile salmonids is likely to adversely affect the capability of the bull trout habitat to provide an abundant food base for the bull trout.

For that reason, zinc concentrations at the proposed acute and chronic criteria level are likely to impair the capability of bull trout habitat to provide for the normal reproduction, growth, and survival of bull trout.

Given that the state of Idaho represents 44 percent of streams and 34 percent of lakes and reservoirs occupied by the bull trout within its range, the above effects are considered to be significant and are likely to impede (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations within a significant portion of its range.

## 2.5.8.3 Bull Trout Critical Habitat

Of the nine PCEs identified for bull trout critical habitat, the proposed zinc criteria may affect PCE 3 (adequate prey base) and 8 (water quality), as discussed below.

Bull trout are opportunistic feeders with food habits primarily a function of fish size and life history strategy. Resident and juvenile migratory bull trout prey on terrestrial and aquatic insects, macro-zooplankton and small fish (Boag 1987, p. 58; Goetz 1989, pp. 33-34; Donald and Alger 1993, pp. 239-243). Adult migratory bull trout are primarily piscivores and are known to feed on various fish species (Fraley and Shepard 1989, p. 135; Donald and Alger 1993, p. 242). According to Rieman and McIntyre (1993. p. 3) "Vigorous populations [of bull trout] may require abundant fish forage. For example, in several river basins where bull trout evolved with large populations of juvenile salmon, bull trout abundance declined when salmon declined."

[18] Hansen et al. (2002c, p. 67) concluded that rainbow trout were more sensitive to zinc than bull trout. They also concluded that the water quality criteria may not be protective of sensitive salmonids.

The effects of elevated zinc concentrations on aquatic insect populations are complex and some information suggests measureable losses of sensitive, known bull trout prey species could occur at concentrations less than the proposed aquatic life criteria (Schmidt et al. 2011). For forage fish, Besser et al. (2007) found that mottled sculpin were decimated (100 percent killed) in 28-day exposures to a zinc concentration at 150 µg/L at a water hardness value of 103 mg/L, which is only slightly above the proposed chronic criterion concentration for zinc of 121 µg/L. The estimated LC50 (75 µg/L) was less than the proposed chronic criterion concentration for zinc. As concluded above, the proposed zinc aquatic life criteria are likely to cause substantial mortality of juvenile bull trout and other juvenile salmonids as well[19]. The decline of other juvenile salmonids is likely to adversely affect the capability of the bull trout critical habitat to provide an abundant food base (PCE 3) for the bull trout.

The proposed zinc criteria are likely to impair water quality (PCE 8) by allowing aquatic zinc concentrations to rise to levels that have been shown to be lethal to juvenile bull trout throughout the range of bull trout critical habitat in Idaho. For that reason, zinc concentrations at the proposed acute and chronic criteria level would impair the capability of the critical habitat to provide for the normal reproduction, growth, and survival of bull trout.

In addition, because the proposed action contains no provision for zinc concentrations in sediment, sediment concentrations of zinc are likely to rise to levels that will adversely affect bull trout individuals and due to impaired sediment quality, will also adversely affect bull trout critical habitat. An elevated contaminant concentration in sediment will influence the concentration of the compound in the overlying water. Continuous and dynamic interactions between surficial sediment and overlying water occur in any waterbody (Walker et al. 1996).

Within the conterminous range of the bull trout, a total of 19,729 miles of stream and 488,252 acres of lakes and reservoirs are designated as critical habitat. The state of Idaho contains 8,772 miles of streams and 170,217 acres of lakes and reservoirs designated as bull trout critical habitat (75 FR 63937). Thus, the proposed zinc criteria are likely to significantly impair the capability of approximately 44 percent of the total designated critical habitat in streams and 35 percent of the total designated critical habitat in lakes and reservoirs to adequately support the recovery of the bull trout.

## 2.5.8.4 Kootenai River White Sturgeon

The toxicity of acute and chronic concentrations of zinc to Columbia River white sturgeon was recently reported in a compilation by Ingersoll and Mebane (2014); that publication includes the references identified in the following discussion. The short-term effects that were tested included lethal as well as sublethal effects (e.g., loss of hiding behavior, loss of equilibrium, immobilization, or loss or impairment of swimming behavior by exposed individual sturgeon). Test fish that were subject to loss of equilibrium or immobilization were considered “effective mortalities” in these short-term tests, even if they still had gill movement. The short-term tests were repeated multiple times with different-aged fish. White sturgeon exhibited marked

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[19] Hansen et al. (2002c, p. 67) concluded that rainbow trout were more sensitive to zinc than bull trout. They also concluded that the water quality criteria may not be protective of sensitive salmonids.

differences in metals sensitivity based on age, with the youngest fish tested (at 2 dph) being most sensitive. In the most sensitive test result, the threshold for the onset of effective mortality seemed to occur at the proposed acute zinc criterion concentration. In this test, 10 percent of the fish were affected relative to 2.5 percent in the controls. The corresponding acute zinc criterion concentration for test conditions was almost the same, at 117 µg/L. In next highest concentration tested (225 µg/L zinc), 100 percent of exposed fish suffered effective mortalities (Calfee et al. 2014, Table A-2). While the 10 percent effective mortality rate at the proposed acute criterion concentration for zinc was low, the fact that the apparent threshold for adverse effects of zinc to white sturgeon was the criterion concentration indicates the potential for adverse effects from short-term exposures of zinc to a sensitive life stage of white sturgeon. Calfee et al. (2014) concluded that the proposed acute water quality criterion for zinc may not be protective of sturgeon early life stages.

In long-term tests of zinc toxicity to sturgeon conducted by Wang et al. (2014a), definitive adverse effects of zinc were actually only detected at higher zinc concentrations than those considered to be adverse to the sturgeon in the short-term effects. The most sensitive effect concentration obtained with zinc was a 10 percent reduction in growth at 53 µg/L zinc following a 53-day exposure of white sturgeon to zinc. However, Wang et al. (2014a, Table B-2) cautioned that the result should be used with caution because the control survival (37 percent) was considerably less than the data quality objective of >70 percent for this chronic test (Wang et al. 2014a, Table B-2). The lowest threshold for adverse effects in a long-term zinc exposure that met data quality objectives (e.g., with 97.5 percent control survival) was a zinc concentration of 181 µg/L for reductions in biomass following a 28-day exposure, relative to the proposed chronic aquatic life criterion for zinc of 118 µg/L at a water hardness value of 100 mg/L (Wang et al. 2014a, Table B-3). However, Wang et al. (2014a) concluded that the proposed chronic criterion for zinc may not be protective of sturgeon early life stages.

Zinc toxicity to white sturgeon was also tested by Vardy et al. (2011), who estimated a LC20 (20 percent lethality concentration) of 102 µg/L zinc at a water hardness value of 70 mg/L following a 66-day exposure. This effects concentration is higher than the proposed chronic criterion for zinc of 87 µg/L (at a water hardness value of 70 mg/L). However, these results are not directly comparable to tests by Wang et al. (2014a) principally because sublethal growth measurements were not reported.

In addition to direct toxicity and adverse sublethal effects, the proposed zinc criteria are likely to indirectly affect the Kootenai River white sturgeon through reduced prey availability and elevated sediment concentrations of zinc. As discussed in the Snake River Aquatic Snail section (2.5.8.1) above, the proposed zinc criteria are likely to adversely affect periphyton (algae and diatoms). If the survival and reproduction of algae is impacted at or below the level of the proposed zinc criteria, then it is likely to result in reduced numbers of herbivores (snails and certain fish), which may in turn result in reduced numbers of primary and secondary consumers inclusive of the sturgeon. The proposed zinc criteria are also expected to adversely affect freshwater mussels, a major food item for white sturgeon throughout the Columbia River Basin (Romano et al. 2002). Reduced prey availability would mean reduced sturgeon body weight, increased energy expenditure to procure prey, decreased energy available for reproduction, and generally reduced survival.

Most zinc that is introduced into aquatic environments is partitioned into sediments, where bioavailability is enhanced under conditions of high dissolved oxygen, low salinity, low pH, and high levels of inorganic oxides and humic substances (Eisler 1993). There are no data of which we are aware on the combined toxic effects of zinc in the water column and zinc that is adsorbed to sediment or other particulates. Many aquatic invertebrates and some fish may be adversely affected by ingesting zinc-containing particulates (EPA 1987b). This is particularly important for the white sturgeon, which is a benthic fish. All life stages of the sturgeon have close contact with sediment: eggs are laid in sediment, hatchlings shelter among rocks on the river bottom, and juveniles and adults feed on the bottom. As bottom feeders, juveniles and adults are likely to incidentally ingest significant amounts of sediment. If the majority of the zinc present in an aquatic environment is in the sediment, this may be a substantial route of exposure. White sturgeon are long lived and thus have extended opportunity for exposure, tissue accumulation, and the manifestation of adverse effects in the form of reduced growth and survival.

Given that existing data show adverse effects to multiple freshwater fish species, including potential prey species of the Kootenai River white sturgeon, at zinc concentrations below the proposed criteria, and given the likelihood that zinc concentrations will be even higher in sediments, thus increasing adverse impacts to white sturgeon eggs and juveniles, we conclude the proposed criteria for zinc are likely to have significant adverse effects (in the form of reduced growth and survival) to the Kootenai River white sturgeon throughout its range in Idaho, which represents 39 percent of its range. Such impacts are likely to impede natural reproduction of the Kootenai River white sturgeon and the maintenance or increase of the wild population.

## 2.8.5.5 Kootenai River White Sturgeon Critical Habitat

Given that existing data show adverse effects to the habitat of multiple freshwater fish species, including that of potential prey species of the Kootenai River white sturgeon, at zinc concentrations below the proposed criteria, and given the likelihood that zinc concentrations will be even higher in sediments, thus increasing adverse impacts to white sturgeon eggs and juveniles, we conclude the proposed criteria for zinc are likely to have significant adverse effects to water quality by allowing aquatic zinc concentrations to rise to levels that have been shown to be detrimental and even lethal to other freshwater fish. Adverse effects were observed in rainbow trout, chinook salmon, bluegill, and striped bass at concentrations below the proposed zinc criteria (see the above discussion on zinc effects to the bull trout and the Kootenai River white sturgeon). Zinc concentrations at the proposed acute and chronic criteria levels is likely to create habitat conditions within sturgeon critical habitat in Idaho that are likely to impair the capability of the critical habitat to provide for the normal behavior, reproduction, and survival of the white sturgeon in support of its recovery.

In addition, because the proposed action contains no provision for zinc concentrations in sediment, under the proposed criteria for zinc, sediment concentrations of zinc within the critical habitat are likely to rise to levels that will adversely affect white sturgeon individuals (particularly eggs and juveniles). Sediment quality is critically important to the health of white sturgeon because all life stages are extensively exposed to sediments, either through dermal contact (all life stages) or through incidental ingestion while feeding (juveniles and adults). An elevated contaminant concentration, such as for zinc, in sediment will influence the concentration of the compound in the overlying water. Continuous and dynamic interactions between surficial sediment and overlying water occur in any waterbody (Walker et al. 1996).

Because the proposed water quality criteria would be implemented statewide, all of the designated white sturgeon critical habitat would be subjected to aquatic zinc concentrations up to 117 µg/L (acute) and 118 µg/L (chronic) at a water hardness value of 100 mg/L, in addition to unknown and unregulated concentrations in sediment. Thus, the proposed acute and chronic zinc criteria are likely to adversely affect sediment and water quality in 100 percent of the critical habitat within the distinct population segment and is reasonably certain to impair the ability of critical habitat to provide for the normal behavior, reproduction, and survival of white sturgeon.

## 2.5.9 Chromium (III) and (VI) Aquatic Life Criteria

The definition of aquatic life criteria for chromium is based upon its chemical form (oxidation state). The oxidation state gives the chromium criteria their shorthand names of chromium (III) and chromium (VI). The proposed acute and chronic criteria for chromium (VI) are 16 µg/L and 11 µg/L, respectively, and are not water hardness dependent. The criteria for chromium (III) are water hardness dependent. The proposed acute and chronic criteria for chromium (III) at a water hardness value of 100 mg/L are 570 and 74 µg/L, respectively, and are derived from the following equations:

$$\text{Acute chromium (III) criterion } (\mu g/L) = e^{(0.819[\ln(\text{hardness})]+3.7256)} * (0.316)$$

$$\text{Chronic chromium (III) criterion } (\mu g/L) = e^{(0.819[\ln(\text{hardness})]-0.6848)} * (0.86)$$

For water hardness values of 10, 25, 50, and 250, the proposed acute criterion values for chromium (III) are 86, 183, 323, and 1207 µg/L, respectively. The corresponding proposed chronic values for chromium (III) are 11, 24, 42, and 157 µg/L, respectively.

As with other water hardness-dependent criteria for metals, the above criterion concentrations were calculated using a range of water hardness values that covers most surface waters within the action area. In a compilation provided by NMFS (2014b) of data from 324 sites monitored by the USGS from 1979-2004, water hardness values ranged from 4 to 2100 mg/L, but 90 percent of the values fell between 6 and 248 mg/L (5$^{th}$ and 9$^{th}$ percentiles of average site hardnesses). The proposed action additionally constrains the water hardness value calculations to assume that the general hardness-toxicity relationship only holds between a water hardness range of 25 to 400 mg/L. For example, the proposed action presumes that at a water hardness value of 10 mg/L, chromium is no more toxic than at a water hardness value of 25 mg/L, and in waters where the water hardness values are less than 25 mg/L, the proposed criteria would be calculated using a water hardness value of 25 mg/L, regardless of the actual site-specific water hardness (EPA 1999a). We did not find any scientific evidence to support the practice of using a "hardness floor" in the equations for calculating criteria values.

Chromium can exist in oxidation states from –II to +(VI), but is most frequently found in the oxygenated waters in its hexavalent state, (VI). Chromium (III) is oxidized to chromium (VI) and, under oxygenated conditions, chromium (VI) is the dominant stable species in aquatic systems. Chromium (VI) is highly soluble in water and thus mobile in the aquatic environment (Reid 2011). No single mechanism or impairment has been shown to be responsible for chromium toxicity in fish. Toxicity symptoms include changes in tissue histology, temporary reductions in growth, the production of reactive oxygen species (ROS), and impaired immune function (Reid 2011).

Although weathering processes result in the natural mobilization of chromium, the amounts added by anthropogenic activities are thought to be far greater. Major sources of anthropogenic introductions of chromium into the environment are the industrial production of metal alloys, atmospheric deposition from urban and industrial centers, and large scale wrecking yards and metals recycling and reprocessing centers (Reid 2011).

The few data found on chromium concentrations in Idaho were at very low values. In the Stibnite Mining District in the East Fork and South Fork Salmon River Basin, total chromium concentrations collected under low flow conditions in September 2011 ranged from <0.2 µg/L to 0.24 µg/L (http://waterdata.usgs.gov/nwis, HUC 17060208). In the Blackbird Mining District in the same area, the concentration of chromium in seeps and adits around the Blackbird Mine were not higher than average background filtered surface water concentrations near the Blackbird Site (<2.9 µg/L) (Beltman et al. 1993).

### 2.5.9.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

As noted above, the listed snail species of concern at issue in this Opinion can be grouped as pulmonate or non-pulmonate snails. The Banbury Springs lanx is classified among the pulmonate snails, and while not formally described, is considered to be in the family Lymnaeidae (USFWS 2006b). The Snake River physa (family Physidae) is also a pulmonate snail. The Bliss Rapids snail and the Bruneau Hot Springsnail are non-pulmonate snails in the family Hydrobiidae.

Few data on the toxicity of chromium to freshwater snails are available. EPA (1985f, Table 1) cited 96-hr LC50s for acute chromium(III) exposure to the snail *Amnicola* at concentrations of 9,400 and 12,400 µg/L for adult and embryo stages, respectively, at a water hardness value of 50 mg/L; these values are much higher than the proposed acute criterion of 323 µg/L. Canivet et al. (2001) exposed *Physa fontinalis* (Physidae) to chromium (VI) in three tests with 96- and 240-hr exposures. The resulting LC50s were similar to those reported above by EPA (1985f): 9400 and 9500 µg/L in two 96-hr tests, and 4200 µg/L in a 240-hr test (Canivet et al. 2001, their table 4). With the snail *Lymnaea luteola* (Lymnaidae), Khangarot and Ray (1988) obtained an LC50 for acute exposure to chromium (VI) at a concentration of 3880 µg/L, which again is far above the proposed acute criterion for chromium of 16 µg/L.

No chromium chronic exposure data for freshwater snails were located during this consultation. Chromium chronic exposure data are available for freshwater mussels. Wang et al. (2014b) give preliminary results from acute (96-hr) and longer term (14-d) tests with chromium (VI) and a freshwater mussel native to Idaho, the western pearlshell, *Margaritifera falcata*, in comparison with a more commonly tested freshwater mussel, the fatmucket, *Lampsilis siliquoidea*. The results were similar between the 96-hour and 14-day exposure tests. When tested at 20°C, the 96-hr EC50 concentration for chromium (VI) was 919 µg/L for the pearlshell and 456 µg/L for *Lampsilis* in water with a hardness value of about 70 mg/L. When tested at 27°C, the results were about 3-fold lower (Wang et al. 2014b). The proposed acute water quality criterion for chromium (VI) is much lower, at 16 µg/L.

Cœurdassier et al. (2005) exposed the snail, *Lymnaea palustris,* for four weeks to a complex industrial effluent that included elevated chromium (with an average concentration of 24 µg/L), in addition to elevated Zn, Fe, and total polycyclic aromatic hydrocarbon (PAH) concentrations. The exposed snails accumulated high internal levels of chromium and Zn during the exposures. However, the adverse effects noted (reduced fecundity) were not correlated with high internal concentrations of metals in the snails, suggesting that toxicity resulted from other factors (Cœurdassier et al. 2005). While the chromium concentrations in the effluent were not speciated, chromium was presumed to be predominately chromium (VI) since the oxic conditions would have favored chromium (VI) over chromium (III).

Although no chronic chromium exposure data were located for either chromium (III) or chromium (VI) relative to freshwater snails, the insensitive results in acute exposures and the relatively low acute-to-chronic ratios (<10) for sensitive species with chromium (EPA 1985f) suggests that chronic effects are unlikely to occur at concentrations close to the proposed chronic criteria.

Although the evidence of chromium toxicity to freshwater snails is sparse, based on that information and giving the benefit of the doubt to the listed species, the Service concludes that the proposed criteria for chromium (III) and chromium (VI) are not likely to adversely affect the Snake River snails and the Bruneau hot springsnail; all such effects are expected to be insignificant or discountable.

## 2.5.9.2 Bull Trout

Although no data are available on the toxicity of chromium to the bull trout, Benoit (1976) conducted long-term (chronic) tests with the closely related brook trout. Benoit (1976) observed that growth in terms of body weight was retarded in response to all chromium concentrations during an 8-month exposure of brook trout to chromium (VI) concentrations of 10 µg/L and higher. At a chromium (VI) concentration of 200 µg/L, the exposed brook trout weighed 20 percent less than the control group. The magnitude of growth reductions in chromium (VI) exposures below 200 µg/L was not given beyond that “growth in weight was retarded in all test concentrations” during the 8-month test. However, in a subsequent 22-month test, while brook trout exposed to a concentration of 350 µg/L of chromium (VI) weighed 25 percent less than the control group at 6 months of age, by 12 to 22 months, the chromium-exposed fish were only 10-12 percent lower in body weight than the control group. Benoit (1976) assumed that because the more severe growth effects were overcome after several months, they were not “effects” for the purpose of reporting a summary NOEC for chromium (VI) of 200 µg/L, a “lowest-observed effects concentration”(LOEC) of 350 µg/L, and a “maximum acceptable toxicant concentration” (MATC) for chromium (VI) that split the difference between the NOEC and LOEC (Benoit 1976). Notwithstanding the findings reported by Benoit (1976), temporary growth reductions over the course of several months are not discountable effects. Survival of juvenile salmonids, including juvenile bull trout, in their first year of life is strongly dependent upon their size at the onset of winter, with bigger fish usually surviving better (e.g., Hutchings et al. 1999; Biro et al. 2004; McMahon et al. 2007; Pess et al. 2011).

However, other long-term exposures of salmonids to chromium have produced considerably higher effect concentrations than those reported by Benoit (1976) for the brook trout. Sauter et al. (1976) conducted 60-day chromium exposures of eggs and fry of 7 fish species, including the rainbow trout, *Oncorhynchus mykiss*, and the lake trout, *Salvelinus namaycush.* The rainbow trout was the most sensitive species tested. Sauter et al. (1976) estimated “safe” concentrations of chromium for the rainbow trout between 51 and 105 µg/L in water with a hardness value of 35 mg/L. Sauter et al. (1976) estimated safe concentrations of chromium for the lake trout between 105 and 194 µg/L. Both of these estimates are well above the proposed chronic criterion for chromium (VI) of 11 µg/L. Although chromium was not speciated in the above estimates, we assume it to be in the form of chromium (VI).

Only one study of trivalent chromium toxicity to salmonids was located during this consultation. Stevens and Chapman (1984) designed exposures of steelhead starting as either newly fertilized eggs or eyed eggs, continuing 30-day post swimup where chromium nitrate was dissolved to produce chromium (III); a very rapid water replacement (50 percent every 25 minutes) scheme was used to avoid significant conversion of trivalent to hexavalent chromium. The threshold for adverse effects to the steelhead in this test was found to be caused by chromium concentrations between 30 and 48 µg/L; the adverse effect was a 7 percent reduction in body length at 48 µg/L; no effects to steelhead growth were detected at a chromium (III) concentration of 30 µg/L. A concurrent, acute test with two-month old steelhead yielded a 96-hr LC50 concentration for chromium (III) of 4,400 µg/L (Stevens and Chapman 1984). The corresponding proposed acute and chronic criteria concentrations for chromium (III) are significantly lower: at 187 and 24 µg/L, respectively data test water hardness value of 25 mg/L.

Patton et al. (2007) reported that the survival, development, and growth of early life stage fall Chinook salmon were not adversely affected by extended exposures (i.e., 98 days) to hexavalent chromium ranging in concentration from 0.79 to 260 µg/L.

Short-term exposures of salmonids to chromium have also produced adverse effects at concentrations far higher than the proposed acute criterion under consideration. For instance, Benoit (1976) reported 96-hr LC50s for chromium (VI) of 59,000 µg/L for juvenile brook trout and 69,000 µg/L for juvenile rainbow trout in water with a hardness value of 45 mg/L.

Conflicting results have been obtained from fertilization tests of salmonids under exposures to chromium (VI). Billard and Roubaud (1985) determined that the viability of rainbow trout sperm (but not ova) was adversely affected when exposed directly to a total chromium concentration equal to 5 µg/L, which is below the proposed chronic criterion. Farag et al. (2006) found that a total chromium concentration ranging from 11 to 266 µg/L and a chromium (VI) concentration of 130 µg/L did not affect the fertilization process of Chinook salmon or cutthroat trout. Farag et al. (2006) suggested that the different findings might be accounted for due to different species being tested, but because cutthroat and rainbow trout are so closely related, the differences seem more likely to be based on the different methodologies used. The time allowed for exposure to chromium during fertilization was 1 minute during the Farag et al. (2006) study versus 15 minutes for the study conducted by Billard and Roubard (1985). The shorter time used by Farag et al. (2006) more closely mimicked fertilization events that may occur under river conditions where velocities of the water at the substrate are fast and motility of sperm is short-lived. Also, Farag et al. (2006) reported that the ova were held in the exposure water for 1.5 hours of water hardening after fertilization to more closely mimic natural conditions in which eggs continue to absorb water for approximately 1.5 hours after fertilization. The ova were not exposed to chromium during water hardening in the study performed by Billard and Roubard (1985). Farag et al. (2006) concluded that the instantaneous nature of fertilization likely limits the potential effects of chromium on fertilization success. Neither Billard and Roubard (1985) or

Farag et al. (2006) directly analyzed chromium speciation for most treatments, but in these oxygenated tests the chromium is expected to be present as chromium (VI) (Reid 2011).

Because the Farag et al. (2006) study more closely simulated conditions in the wild, the Service is relying on the results of that study to conclude that the proposed chronic criterion for chromium (VI) of 11 µg/L is most likely protective of (i.e., is not likely to adversely affect) salmonid fertilization including that of the bull trout.

Based on the above information, the Service concludes that the proposed acute and chronic criteria for chromium (III) and the proposed acute criterion for chromium (VI) are not likely to adversely affect the bull trout. Given the information discussed above that long-term exposure to chromium (VI) at the proposed chronic criterion level may cause reduced growth of juvenile bull trout, and depending on the magnitude of the growth reduction, reduced overwinter survival, the Service concludes that individual juvenile bull trout may be adversely affected by the proposed chronic chromium criterion. However, these effects are not likely to occur at a population level given the other above studies involving the chronic exposure effects of chromium that resulted in reduced salmonid growth only at chromium concentrations well above the proposed chronic criterion for chromium (VI) of 11 µg/L.

## 2.5.9.3 Bull Trout Critical Habitat

Based on the above discussion for the bull trout, implementation of the proposed criteria for chromium is not likely to create habitat conditions within bull trout critical habitat that directly affect the bull trout, except with respect to chromium (VI) at the proposed chronic criterion level. Under those circumstances, habitat conditions may cause reduced growth of juvenile bull trout, and depending on the magnitude of the growth reduction, reduced overwinter survival. On that basis, the Service concludes that water quality in bull trout critical habitat may be adversely affected by the proposed chronic chromium (VI) criterion. However, these effects are not likely to compromise the capability of the habitat to support bull trout given the other above studies involving the chronic exposure effects of chromium that resulted in reduced salmonid growth only at chromium concentrations well above the proposed chronic criterion for chromium (VI) of 11 µg/L.

Another possible effect is related to the effects of chromium concentrations at the proposed criterion levels to prey species of the bull trout. As discussed above in this Opinion, the bull trout relies on both invertebrates and smaller fish as its prey base. Among the fish species that serve as bull trout prey, the salmonids appear to be the most sensitive to chromium (EPA 1985f, 1996).

Based on the information discussed above for the bull trout, adverse effects caused by the proposed criteria for chromium to the bull trout at a population level are considered unlikely. Based on the same information, it is considered further unlikely that chromium at the proposed criteria concentrations would substantially reduce bull trout prey fish populations. Relative to aquatic insects, no chronic data for chromium are available; acute data with aquatic insects are available (EPA 1985f), but were discounted as being inherently unreliable owing to the difficulty of performing environmentally relevant toxicity tests with aquatic insects (Brix et al. 2011a). The most sensitive taxa for chromium (VI) appear to be freshwater crustaceans, especially zooplankton. Most taxa were reported as being protected by the criteria concentrations, suggesting that the zooplankton assemblage as a whole would be adequately protected on an assemblage basis by both the chromium(III) and chromium (VI) criteria (EPA 1985f). Based on our review of available information, the Service concludes that it is reasonable to find that the proposed criteria for chromium are not likely to adversely affect the abundance of bull trout prey species (PCE 3).

## 2.5.9.4 Kootenai River White Sturgeon

No specific information on chromium toxicity to the white sturgeon, or any other species within the family Acipenser is available. For that reason, the Service is relying on the above analysis for the bull trout to inform the analysis of effects of the proposed chromium criteria on the sturgeon. Absent direct effects data of chromium to either bull trout or white sturgeon, the bull trout analyses (and by extension, the white sturgeon analyses) rely in large part of evaluations of effects to rainbow trout and Chinook salmon. In light of the limited data, this approach seemed reasonable to us because in comparative chronic testing of sensitive early-life stages, rainbow trout had been shown to be the most sensitive of seven fish species with chromium (rainbow trout, lake trout, channel catfish, bluegill, white sucker, northern pike and walleye) Sauter et al. (1976, pp. 30-42).

Based on the above information, the Service concludes that the proposed acute and chronic criteria for chromium (III) and the proposed acute criterion for chromium (VI) are not likely to adversely affect the Kootenai River white sturgeon.

Given the information discussed above that long-term exposure to chromium (VI) at the proposed chronic criterion levels may cause reduced growth of juvenile bull trout, and depending on the magnitude of the growth reduction, reduced overwinter survival, the Service concludes that individual juvenile Kootenai River white sturgeon may be adversely affected by the proposed chronic criterion for chromium (VI). However, these effects are not likely to occur at a population level given the other above studies involving the chronic exposure effects of chromium that resulted in reduced salmonid growth only at chromium concentrations well above the proposed chronic criterion for chromium (VI) of 11 µg/L.

## 2.5.9.5 Kootenai River White Sturgeon Critical Habitat

As discussed above in this Opinion, sediment and water quality are important factors in providing adequate habitat conditions within Kootenai River white sturgeon critical habitat to support recovery of this species, even though these factors are not considered as PCEs in the 2008 revised rule (73 FR 39506). The PCEs are now limited to factors related to flow regime, temperature requirements during spawning season, and the presence of rocky substrates (see section 2.3.8 above); none of these PCEs are likely to be affected by the proposed action.

Based on the above analyses for the bull trout and bull trout critical habitat, implementation of the proposed criteria for chromium is not likely to create habitat conditions within Kootenai River white sturgeon critical habitat that directly affect the sturgeon, except with respect to chromium (VI) at the proposed chronic criterion level. Under those circumstances, habitat conditions may cause reduced growth of juvenile sturgeon, and depending on the magnitude of the growth reduction, reduced overwinter survival. On that basis, the Service concludes that water quality in sturgeon critical habitat may be adversely affected by the proposed chronic chromium (VI) criterion. However, these effects are not likely to compromise the capability of the habitat to support the sturgeon given the other above studies involving the chronic exposure effects of chromium (VI) that resulted in reduced salmonid growth occurred only at chromium (VI) concentrations well above the proposed chronic criterion for chromium (VI) of 11 µg/L.

Another possible effect is related to the effects of chromium concentrations at the proposed criterion levels to prey species of the sturgeon, which is also an important factor determining the capability of the critical habitat to support recovery of the sturgeon. As discussed above in this Opinion, the sturgeon relies on both invertebrates and fish as its prey base. Among the fish species that serve as bull trout (and presumably sturgeon) prey, the salmonids appear to be the most sensitive to chromium (EPA 1985f, 1996). Based on the information discussed above for the bull trout, adverse effects caused by the proposed criteria for chromium to the bull trout at a population level are considered unlikely. Based on the same information, it is considered further unlikely that chromium at the proposed criteria concentrations would substantially reduce bull trout prey fish populations. Relative to aquatic insects, no chronic data for chromium are available; acute data with aquatic insects are available (EPA 1985f), but were discounted as being inherently unreliable owing to the difficulty of performing environmentally relevant toxicity tests with aquatic insects (Brix et al. 2011a). The most sensitive taxa for chromium (VI) appear to be freshwater crustaceans, especially zooplankton. Most taxa were reported as being protected by the criteria concentrations, suggesting that the zooplankton assemblage as a whole would be adequately protected on an assemblage basis by both the chromium(III) and chromium (VI) criteria (EPA 1985f). Based on our review of available information, the Service concludes that it is reasonable to find that the proposed criteria for chromium are not likely to adversely affect the abundance of sturgeon prey species.

# 2.5.10 Nickel Aquatic Life Criteria

The proposed acute and chronic criteria values for nickel are 468 and 52 µg/L, respectively, as calculated from the following equations using a water hardness value of 100 mg/L:

$$\text{Acute nickel criterion } (\mu g/L) = e^{(0.846[\ln(\text{hardness})]+2.255)} * (0.998)$$

$$\text{Chronic nickel criterion } (\mu g/L) = e^{(0.846[\ln(\text{hardness})]+0.0584)} * (0.997)$$

The acute and chronic criteria values are also referred to as the CMC and CCC, respectively (EPA 1985a; Stephan et al. 1985a; EPA 1999a). Using the above equations, at water hardness values of 10, 25, 50, and 250 mg/L, the acute nickel criterion value is 67, 145, 260, and 1017 µg/L, respectively. At water hardness values of 10, 25, 50, and 250 mg/L, the corresponding chronic criterion values for nickel are 7, 16, 29, and 113 µg/L, respectively.

In the above examples, the criterion concentrations were calculated using a range of water hardness values that cover most waters within the action area. NMFS (2014a) reported water hardness values from 324 sites monitored by the USGS from 1979-2004 that ranged from 4 to 2100 mg/L, but 90 percent of the values fell between 6 and 248 mg/L ($5^{th}$ and $9^{th}$ percentiles of average site hardnesses). In calculating the acute and chronic criteria for nickel, EPA constrains the water hardness calculations to assume the general hardness-toxicity relationship only holds between a water hardness range of 25 to 400 mg/L. For example, the proposed action presumes that at a water hardness of 10 mg/L, nickel is no more toxic than at a water hardness value of 25 mg/L, and in waters where the water hardness values are less than 25 mg/L, the criteria would be calculated using a water hardness value of 25 mg/L, regardless of the actual hardness (EPA 1999a). We did not find any scientific evidence to support this practice of using a "hardness floor" in the criteria equations, and we did find contrary evidence with nickel. Deleebeeck et al. (2007) observed that cladocerans collected from Swedish lakes were more sensitive to nickel in "soft" vs. "moderately hard" or vs. "hard" waters (with water hardness values of about 6.2, 16 mg/L, and 43 mg/L, respectively). Similar findings have been reported for other metals (see section 2.5.1.5, *Common Factors*). Therefore, we consider the "hardness floor" at 25 mg/L to be arbitrary, and we do not rely on it in our analyses.

In natural waters, dissolved nickel concentrations are usually lower than the proposed criteria values, and in waters of the United States away from the immediate influence of discharge, dissolved nickel concentrations typically range from about 1 to 2 µg/L (Stephan et al. 1994). However, more recent data suggests that background nickel concentrations are probably even lower than Stephan's estimates. Targeting a region expected to be high in metals, streams in the Central Colorado Mineral Belt, Church et al (2012) found the median nickel concentration was <0.4 µg/L and 95 percent of the values were <13 µg/L (n=388).

Nickel is rare in the waters of Idaho, even in areas disturbed by mining. In the Blackbird Mine area, Beltman et al. (1993) reported nickel concentrations in mine waters and seeps in excess of 1500 µg/L; however, in the mining-affected streams that were large enough to support fish populations, nickel concentrations ranged from <10 to 60 µg/L. In the mining-affected South Fork of the Coeur d'Alene River, located in northern Idaho, Mebane et al. (2012) reported nickel concentrations ranging from <2 to 8 µg/L.

Nickel is an essential nutrient for plants and terrestrial animals, and while the evidence is sparse, nickel is probably an essential nutrient for fish. At extremely high concentrations, nickel is a respiratory toxicant in fish. However, nickel has generally low toxicity to aquatic organisms in short-term exposures. In longer term exposures, the modes of nickel toxicity are less clear, but probably involve ionoregulatory disruption and cellular damage oxidative stress (Pyle and Couture 2011).

## 2.5.10.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

As noted above, the listed snail species of concern in this Opinion can be grouped as pulmonate or non-pulmonate snails[20]. The Banbury Springs lanx is classified among the pulmonate snails, and while not formally described, is considered to be in the family Lymnaeidae (USFWS 2006b). The Snake River physa (family Physidae) is also a pulmonate snail. The Bliss Rapids snail and the Bruneau Hot Springsnail are non-pulmonate snails in the family Hydrobiidae.

Similar to the situation with lead, the freshwater pulmonate snail, *Lymnaea stagnalis* (Lymnaeidae), has been shown to be the most sensitive aquatic organism tested to date for nickel (Schlekat et al. 2010; Niyogi et al. 2014). Schlekat et al. (2010) reported results of exposing

[20] Freshwater pulmonate snail species such as the Banbury Springs lanx and the Snake River physa do not have gills, but absorb oxygen across the inner surface of the mantle (outer wall of the mollusk's body that encloses the internal organs) (Dillon 2006, p. 252). In contrast the non-pulmonate snails, such as the Bliss Rapids snail and the Bruneau hot springsnail, retain the ancestral condition and breathe through gills (Hershler et al. 2006, p. 167).

*Lymnaea stagnalis* to nickel for 21 days in water from the Calapooia River, Oregon. The EC20 (concentration adversely affecting 20 percent of the test population) for nickel was 1.6 µg/L, in test waters with a hardness value of 212 mg/L. The corresponding chronic water quality criterion for nickel in waters with a hardness value of 212 mg/L is much higher, 98 µg/L, which indicates that the proposed chronic water quality criterion for nickel would be severely underprotective of *Lymnaea*. We assume these underprotective results for *Lymnaea* would also hold for other taxa in the family Lymnaeidae, including the Banbury Springs lanx, absent more direct evidence. In contrast to the observed hypersensitivity of *Lymnaea* to nickel in long-term exposures, in short-term exposures, while *Lymanea* is more sensitive than most taxa, the effect concentrations are close to or only slightly below the proposed criteria for nickel. Leonard and Wood (2013) obtained a 96-hr LC50 for nickel of about 445 µg/L for *Lymnaea stagnalis* in test water with a hardness value of 85 mg/L water, which is similar to the proposed acute criterion for nickel of 408 µg/L calculated at a water hardness value of 85 mg/L water.

While the hypersensitivity of *Lymanea* to nickel was observed across a variety of water chemistry conditions (Schlekat et al. 2010; Niyogi et al. 2014), it is noteworthy that the results reported by Schlekat et al. (2010) for test water from the Calapooia River, Oregon are based on background water chemistry characteristics that affect nickel toxicity that are similar to those occurring in the Snake River springs inhabited by the Banbury Springs lanx. For the Calapooia River, Schlekat et al. (2010) reported alkalinity of 200 mg/L, pH of 8.0, and DOC of 0.7 mg/L. For Box Canyon Springs, Idaho, Mebane et al. (2014) reported alkalinity of 165 mg/L, pH of 8.0, and DOC of 0.4 mg/L, and for Briggs Springs, conditions were similar with alkalinity of 174 mg/L, pH of 7.8, and DOC of 0.6 mg/L (Mebane et al. 2014). Thus, the results of Schlekat et al. (2010) that showed that nickel would be toxic to *Lymnaea* at concentrations more than 60X lower than the proposed chronic criterion are particularly germane to the habitat occupied by the Banbury Springs lanx.

Based on the information discussed above, the proposed chronic criterion concentration for nickel is likely to adversely affect the Banbury Springs lanx. This leads to the question, what concentrations of nickel are likely be adequately protective of the lanx? As with lead (see section 2.5.4 above), we sought to estimate no- or very low-effect nickel concentrations relative to the Banbury Springs lanx by re-examining the published research for within-family surrogate species for the lanx (Lymnaea) with an alternative regression approach that allows estimating no-effect concentrations. However, the results of this approach for nickel were less successful than with lead for two reasons. First, one of the two chronic studies (Schlekat et al. 2010) showing adverse effects to *Lymnaea* at concentrations well below the proposed chronic criterion only reported summary test statistics (e.g., EC10,EC20, and EC50, not the raw data) However, a growth reduction of 10 percent is a low-level effect, and as a practical matter, in our experience working with effect-concentration curve fitting, the 10 percent effect concentration and the 0 percent effect concentration are not very different. The finding in Schlekat et al. (2010) of an EC10 concentration for nickel of 1.1 µg/L in water with a hardness value of 212 mg/L from the Caloopia River is 89X lower than the proposed chronic criterion of 98 µg/L, or rounding off, 1 percent of the proposed chronic criterion. This would suggest a 0.01X multiplier of the chronic criterion for calculating “safe” discharge limits for nickel into the habitats of the Banbury Springs lanx.

The second chronic exposure study showing adverse effects to *Lymnaea* at concentrations well below the proposed chronic criterion was Niyogi et al. (2014) (Figure 8). The 21-day chronic test with nickel and *Lymnaea* reported by Niyogi et al. (2014) could not be reanalyzed to find a threshold of adverse effects. This is because to find a threshold, the data must have some test exposure concentrations without adverse effects, and then a pattern of increasingly severe effects. In the juvenile snail growth experiment reported by Niyogi et al. (2014), following a 21-day exposure, the nickel concentration was reduced by 48 percent relative to the control in the lowest concentration of nickel tested: 1.3 µg/L nickel. At the test water hardness value of 60 mg/L, the corresponding chronic criterion value for nickel is 34 µg/L. Because no threshold for the onset of adverse effects could be estimated from the testing, the data were evaluated for low-effects, comparable to the results of the treatment reported by Schlekat et al. (2010), as discussed above.

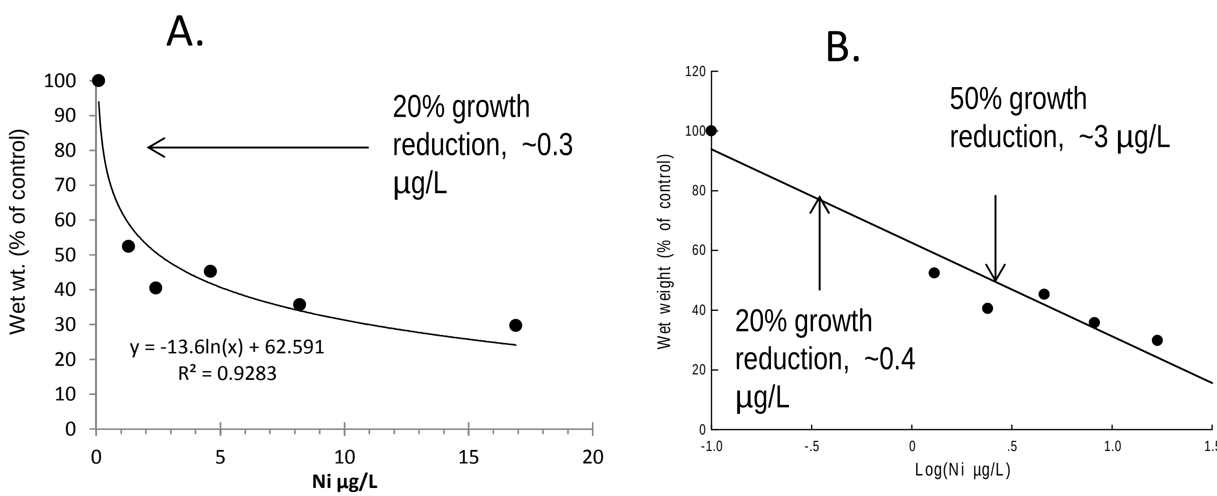


**Figure 8. Lymnaea stagnalis growth, under different nickel (Ni) 21-day exposures. Data taken from Niyogi et al (2014). The data were not amenable to curve fitting to estimate EC values.**

The few toxicity testing reports located for other snail species and molluscs indicated that other tested taxa were more resistant than Lymnaea, but still sensitive to nickel. The pulmonate snail *Physa integra* was collected from ponds in the Willamette Valley and tested in softwater in 4-day exposures. The LC50 of 239 µg/L is relatively sensitive but is higher than the acute water quality criterion of 150 µg/L for hardness 26 mg/L water (Nebeker et al. 1986, p. 807). A non-pulmonate snail collected from Oregon coastal streams, *Juga plicifera* (Pleuroceridae) was more sensitive than *Physa*, with a 4-day LC50 of 237 µg/L in water hardness of 59 mg/L (Nebeker et al. 1986). The acute water quality criterion value for hardness 59 mg/L is 300 µg/L which is only slightly higher than the concentration killing 50 percent of the test population, suggesting some mortality probably would have occurred at the acute criterion. In contrast, in a 30-day exposure, the threshold for lethality of nickel to *Juga plicifera* was higher than the chronic criterion, with no-observed lethal effects at 124 µg/L (Nebeker et al. 1986), relative to the chronic criterion concentration of 33 µg/L for hardness 59 mg/L water.

The fatmucket mussel (*Lampsilis siliquoidea*) is highly sensitive to some substances (such as copper, zinc, and ammonia), which could make it a good surrogate for estimating the effects of nickel toxicity to non-pulmonate snail species such as the Banbury Springs lanx. Besser et al. (2011) reported that fatmuckets were quite sensitive to nickel, with a no-observed effect concentration for nickel of 25 µg/L and a threshold of adverse effects (10 percent reduction in biomass) concentration only slightly higher, at 32 µg/L. Both concentrations are lower than the proposed chronic criterion value for nickel of 54 µg/L at a water hardness value of 104 mg/L.

The previous research mentioned on effects of nickel to snails was all from single-species toxicity tests under tightly controlled experimental conditions. In this usual test methodology, the organisms to be exposed are all carefully selected for similarity, feeding is standardized, and the presence of other species would be considered such a breach of protocol to invalidate the tests. These standard protocols were developed to minimize variability and minimize the influence of confounding factors, but are obviously greatly different than the conditions organisms live in in the wild. Microcosm or "model ecosystem" experiments attempt to straddle the artificiality of standard laboratory toxicity tests and observational field studies which are both more natural and more chaotic. In field studies, contaminant concentrations are variable and uncertain, organisms may move freely in and out of exposures, and other environmental effects such as temperatures and weather effects could confound interpretations. Laboratory microcosm tests allow toxicity testing of complex communities and for indirect effects of the contaminant, such as effects through food webs.

With nickel, Hommen et al (2011) tested the effects of long-term nickel exposures to complex pond-like communities established in 750 L (200gallon) aquaria that had initially been inoculated with natural pond sediments and natural pond plankton. Lymnaea snails were also added. Following 4-months exposure to nickel, no-effects of nickel could be detected in the lowest level tested (12 µg/), and slight declines in snail abundance occurred at 24 µg/L. However, snails were extirpated from the highest nickel treatments of 48 and 96 µg/L. Snail abundance was dominated by *Lymnaea* with *Planorbarius* (family Planorbidae) and undetermined small snails also present. *Planorbarius* was very rare, yet since it was abundant enough for Hommen et al. (2011) to mention but was eliminated from the 48 and 96 µg/L treatments, our interpretation of the results is that *Planorbarius* was also severely affected at 48 µg/L nickel and higher. All other components of microcosms studied were also adversely affected in the 48 and 96 µg/L nickel treatments (i.e., meiobenthos, phytoplankton populations and community structure, periphyton, zooplankton populations and community structure) Hommen et al. (2011). The 48 µg/L treatment with extirpated snails was almost the same nickel concentration as the chronic aquatic life criterion of 52 µg/L (tests waters had mean hardness of 100 mg/L, dissolved organic carbon of 3.8 mg/L, and pH of 8.6.).

The results of this community study indicates that sensitivity of snails to nickel is not solely limited to within the family Lymnaeidae, and that snails in some other families may also suffer adverse effects at chronic criterion concentrations.

Although no chronic nickel effects data were located for species within the families Physidae or Hydrobiidae, based on the above discussion, the Service concludes that approval of the acute and chronic aquatic life criteria for nickel is likely to adversely affect the Snake River physa, the

Bliss Rapids snail, and the Bruneau hot springsnail throughout their ranges via mortality and population reductions, which in turn have adverse implications for long-term survival and reproductive fitness of affected snails.

Based on the above discussion, the Service concludes that the proposed acute and chronic criteria for nickel are likely to adversely affect the Banbury Springs lanx throughout its range via severely retarded growth resulting from the acute criterion, and mortality and population reductions resulting from the chronic criterion, which in turn have adverse implications for long-term survival and reproductive fitness of the lanx.

## 2.5.10.2 Bull Trout

No information is available on nickel toxicity to the bull trout or to any *Salvelinus* species. However, numerous data are available on nickel toxicity to the rainbow trout, which are presumed to be similar in sensitivity as the bull trout and can be used as a reliable surrogate.

In short-term (i.e., acute) exposures, nickel is only toxic to the rainbow trout at environmentally unrealistic concentrations. For instance, LC50 concentrations of nickel in soft water (hardness value of ~23 mg/L) range from 8,100 to 10,900 µg/L (Nebeker et al. 1985). On that basis, the proposed acute criterion for nickel is not likely to adversely affect the bull trout.

In long-term (i.e., chronic) exposures, nickel is toxic to the rainbow trout at much lower and more environmentally relevant concentrations. However, as discussed below, most toxicity data still indicate that direct adverse effects to the bull trout are only likely to occur at concentrations greater than the proposed chronic water quality criterion concentration for nickel.

The lowest effect concentration relative to the nickel proposed chronic criterion was reported by Birge et al. (1978) with a 28-day LC50 from static-renewal exposures of rainbow trout embryos of "0.05 ppm" (i.e., 50 µg/L) at a water hardness value between 93 mg/L and 105 mg/L, for which the corresponding chronic criterion for nickel would essentially be the same, 49 to 54 µg/L. While this low effect concentration is concerning, experimental details were sparse (e.g., actual exposure concentrations and responses were not reported) which lessens the confidence that can be placed on the highly summarized results presented in Birge et al. (1978).

Another low effect concentration with rainbow trout was reported by Nebeker et al. (1985), who found that newly fertilized eggs were the most sensitive life stage. For exposures that began with newly fertilized eggs, reduced growth was observed at the lowest nickel concentration tested (35 µg/L at a water hardness value of 27-39 mg/L, 85 days total exposure), indicating that the threshold for effects was lower. Because this result was unexpected, Nebeker et al. (1985) repeated the test under similar conditions and determined that the threshold for reduced growth fell between 35 and 62 µg/L of nickel. For the test water hardness value of 52 mg/L, the chronic nickel criterion was 30 µg/L, which is slightly lower than the lowest concentration causing reduced growth (35 µg/L).

Other long-term exposures with rainbow trout under similar conditions have produced much higher (less sensitive) effect concentrations for nickel. Brix et al. (2004) tested newly fertilized rainbow trout eggs in 85-day exposures to nickel using a similar test design to the tests reported by Nebeker et al. (1985), using a higher water hardness dilution water of about 89 mg/L. No adverse effects to the rainbow trout were reported from exposures up to a nickel concentration of 466 µg/L, which is 10 times greater than the proposed chronic criterion for nickel of 47 µg/L. In multiple tests of nickel toxicity on the growth and survival of juvenile rainbow trout in 26-day tests under various pH, calcium and magnesium conditions, the NOECs ranged from 5 to >20 times greater than the corresponding proposed chronic criterion concentration of nickel (Deleebeeck et al. 2007).

Limited work by Giattina et al. (1982) with sublethal, behavioral testing of nickel toxicity indicates that behavioral avoidance could potentially occur at nickel concentrations that are slightly higher than the proposed chronic criterion. Giattina et al. (1982) determined that rainbow trout fry avoided a nickel concentration equal to 24 µg/L at a mean water hardness of value of 28 mg/L. This effect concentration is higher than the proposed chronic criterion for nickel of 18 µg/L at a water hardness value of 28 mg/L.

The proposed chronic criteria for nickel may affect bull trout prey species and indirectly affect the bull trout. Bull trout of all ages are opportunistic predators, shifting their diet towards abundant and easily captured prey at different times and locations. While there is evidence that bull trout can eat armored taxa such as clams and snails (Donald and Alger 1993), in general, we presume that taxa that are classified as vulnerable to salmonid predation are the most important in the diet of the bull trout (Suttle et al. 2004), and taxa that are generally non-vulnerable to salmonid predation (burrowing or armored taxa) are not critical to bull trout sustenance. Thus, when evaluating reports of adverse effects of chemicals to different taxa, adverse effects to taxa considered generally non-vulnerable to salmonid predation are less of a concern than effects to common and vulnerable taxa. For instance, in lake populations, amphipods appear to be consistently important to bull trout rearing, as Donald and Alger (1993) showed that bull trout weights in lakes are correlated with amphipod abundance, in the lakes they studied. Large zooplankton, such as *Daphnia magna* or *Daphnia pulex*, may be important food items for the bull trout in lakes, whereas smaller zooplankton such as Ceriodaphnia or copepods are less important (Wilhelm et al. 1999). In streams, bull trout of all ages prey primarily on aquatic insects such as mayflies (ephemeropterans), stoneflies (plecopterans), caddisflies (trichopterans), beetles (coleopterans) midges (chironomids) and worms (oligochaetes) (Boag 1987; Underwood et al. 1995). In streams, small fish such as sculpin and juvenile salmonids can be relatively important in the diet of the bull trout (Underwood et al. 1995). Adult migratory bull trout feed on both aquatic invertebrates and other fish depending on prey availability and the size of the bull trout (Donald and Alger 1993; Wilhelm et al. 1999; Beauchamp and Van Tassell 2001).

With nickel, sensitive taxa include the snail *Lymnaea*, the amphipod *Hyalella*, and the small zooplankton *Ceriodaphnia*, and other small-bodied zooplankton. Less sensitive taxa that could potentially be bull trout prey items include the mayfly *Hexagenia*, the large zooplankton *Daphnia*, *Chironomus* midges, and oligochaete worms (Deleebeeck et al. 2007; Schlekat et al. 2010; Besser et al. 2011). As discussed earlier, the snail *Lymnaea* would not be protected by the proposed chronic nickel criterion concentration, but because of their shell armor, snails are probably not usually important prey items for the bull trout. Reduced growth or survival of the amphipod *Hyalella azteca* at sub-chronic criterion nickel concentrations have been reported. Besser et al. (2011) reported a 20 percent reduction in survival of *Hyalella* at 12 µg/L, which is much lower than the proposed chronic criterion of 54 µg/L calculated for a water hardness value of 104 mg/L. In comparison, Keithly et al. (2004) reported an EC20 for nickel of 61 µg/L at a water hardness value of 98 mg/L, which is at least a little higher than the proposed chronic criterion concentration for nickel of 51 µg/L. Thus, the proposed chronic nickel criterion concentration could have adverse effects to *Hyalella*, which may in turn be an important prey item for subadult bull trout in lakes.

Stream-resident aquatic invertebrates appear less sensitive to nickel toxicity than lake-resident crustaceans. In a life cycle test with a caddisfly, Nebeker et al. (1984) found that the no effect concentration of nickel was 66 µg/L, and that nickel concentrations >250 µg/L prevented the caddisflies from completing their life cycle. The corresponding proposed chronic nickel criterion for a water hardness value of 54 mg/L is 31 µg/L. With the mayfly *Hexagenia*, Besser et al. (2011) found a low threshold for adverse growth effects in a 28-day exposure, with a 10 percent reduction in growth at a nickel concentration of 53 µg/L which is the same as the proposed chronic criterion concentration for nickel at a water hardness value of 104 mg/L, with survival unaffected at extremely high concentrations (>1335 µg/L) of nickel. *Hexagenia* is actually a lake resident mayfly species, but here is assumed to be a reasonable surrogate for estimating the toxicity of nickel to mayflies in general.

Assuming that *Lymnaeid* snails are not an important component of bull trout prey items, potential impacts of the nickel criteria to bull trout prey species appear limited to amphipods, particularly *Hyalella*. Given that bull trout eat a variety of prey items and are known piscivores, the Service does not expect significant adverse effects to the bull trout prey base from any reduction in amphipod abundance.

Based on the research results referenced above, the Service concludes that the proposed approval of the chronic aquatic life criteria for nickel is likely to adversely affect the bull trout via effects to one component (amphipods) of its prey base. Given the variety of prey species in the diet of the bull trout, this adverse effect is not likely to cause a significant adverse effect to the bull trout.

## 2.5.10.3 Bull Trout Critical Habitat

Of the nine PCEs defined for bull trout critical habitat, the Service has determined that the proposed chronic criteria for nickel may affect PCE 3 (adequate prey base), as discussed below. In short-term (i.e., acute) exposures, nickel is only toxic to the rainbow trout at environmentally unrealistic concentrations. For instance, LC50 concentrations of nickel in soft water (hardness value of ~23 mg/L) range from 8,100 to 10,900 µg/L (Nebeker et al. 1985). On that basis, the proposed acute criterion for nickel is not likely to adversely affect water quality within bull trout critical habitat.

Bull trout of all ages are opportunistic predators, shifting their diet towards abundant and easily captured prey at different times and locations. While there is evidence that bull trout can eat armored taxa such as clams and snails (Donald and Alger 1993), in general, we presume that taxa that are classified as vulnerable to salmonid predation are the most important in the diet of the bull trout (Suttle et al. 2004), and taxa that are generally non-vulnerable to salmonid predation (burrowing or armored taxa) are not critical to bull trout sustenance. Thus, when evaluating reports of adverse effects of chemicals to different taxa, adverse effects to taxa considered generally non-vulnerable to salmonid predation are less of a concern than effects to common and vulnerable taxa. For instance, in studied lake populations, amphipods appear to be consistently important to bull trout rearing, as Donald and Alger (1993) showed that bull trout weights in several lakes are correlated with amphipod abundance. Large zooplankton, such as *Daphnia magna* or *Daphnia pulex*, may be important food items for the bull trout in lakes, whereas smaller zooplankton such as Ceriodaphnia or copepods are less important (Wilhelm et al. 1999). In streams, bull trout of all ages prey primarily on aquatic insects such as mayflies (ephemeropterans), stoneflies (plecopterans), caddisflies (trichopterans), beetles (coleopterans) midges (chironomids) and worms (oligochaetes) (Boag 1987; Underwood et al. 1995). In streams, small fish such as sculpin and juvenile salmonids can be relatively important in the diet of the bull trout (Underwood et al. 1995). Adult migratory bull trout feed on both aquatic invertebrates and other fish depending on prey availability and the size of the bull trout (Donald and Alger 1993; Wilhelm et al. 1999; Beauchamp and Van Tassell 2001).

With nickel, sensitive taxa include the snail *Lymnaea*, the amphipod *Hyalella*, and the small zooplankton *Ceriodaphnia*, and other small-bodied zooplankton. Less sensitive taxa that could potentially be bull trout prey items include the mayfly *Hexagenia*, the large zooplankton *Daphnia*, *Chironomus* midges, and oligochaete worms (Deleebeeck et al. 2007; Schlekat et al. 2010; Besser et al. 2011). As discussed earlier, the snail *Lymnaea* would not be protected by the proposed chronic nickel criterion concentration, but because of their shell armor, snails are probably not usually important prey items for the bull trout. Reduced growth or survival of the amphipod *Hyalella azteca* at sub-chronic criterion nickel concentrations have been reported. Besser et al. (2011) reported a 20 percent reduction in survival of *Hyalella* at 12 µg/L, which is much lower than the proposed chronic criterion of 54 µg/L calculated for a water hardness value of 104 mg/L. In comparison, Keithly et al. (2004) reported an EC20 for nickel of 61 µg/L at a water hardness value of 98 mg/L, which is at least a little higher than the proposed chronic criterion concentration for nickel of 51 µg/L. Thus, the proposed chronic nickel criterion concentration could have adverse effects to *Hyalella*, which may in turn be an important prey item for subadult bull trout in lakes.

Stream-resident aquatic invertebrates appear less sensitive to nickel toxicity than lake-resident crustaceans. In a life cycle test with a caddisfly, Nebeker et al. (1984) found that the no effect concentration of nickel was 66 µg/L, and that nickel concentrations >250 µg/L prevented the caddisflies from completing their life cycle. The corresponding proposed chronic nickel criterion for a water hardness value of 54 mg/L is 31 µg/L. With the mayfly *Hexagenia*, Besser et al. (2011) found a low threshold for adverse growth effects in a 28-day exposure, with a 10 percent reduction in growth at a nickel concentration of 53 µg/L which is the same as the proposed chronic criterion concentration for nickel at a water hardness value of 104 mg/L, with survival unaffected at extremely high concentrations (>1335 µg/L) of nickel. *Hexagenia* is actually a lake resident mayfly species, but here is assumed to be a reasonable surrogate for estimating the toxicity of nickel to mayflies in general.

Assuming that *Lymnaeid* snails are not an important component of bull trout prey items, potential impacts of the nickel criteria to bull trout prey species appear limited to amphipods, particularly *Hyalella*. Given that bull trout eat a variety of prey items and are known piscivores, the Service does not expect significant adverse effects to the bull trout prey base from any reduction in amphipod abundance.

Based on the above analysis, the Service concludes that the proposed approval of the chronic aquatic life criterion for nickel is likely to adversely affect PCE 3 of bull trout critical habitat via effects to one component (amphipods) of its prey base. However, given the variety of prey species in the diet of the bull trout, this adverse effect is not likely to cause a significant adverse effect to the capability of bull trout critical habitat in Idaho to provide for an abundant prey base for the bull trout.

## 2.5.10.4 Kootenai River White Sturgeon

No information is available on nickel toxicity to the sturgeon. If the toxicity of nickel to the white sturgeon is roughly similar to that of rainbow trout, then nickel at the proposed acute criterion concentration is not likely to adversely affect the sturgeon. If the toxicity of nickel to the white sturgeon is roughly similar to that of rainbow trout, then nickel at the proposed chronic criterion concentration may adversely affect the prey base for the sturgeon. With cadmium, lead, and zinc, rainbow trout were about, or more sensitive as white sturgeon. However, with copper, the white sturgeon was more sensitive than rainbow trout (Ingersoll and Mebane 2014). This begs the question, would the responses of white sturgeon to the proposed chronic concentration of nickel be expected to follow the patterns of cadmium, lead and zinc, in which the sturgeon was less sensitive than the rainbow trout surrogate, and the criterion thus would be unlikely to adversely affect the white sturgeon? Or would the response of the white sturgeon to nickel exposure at the proposed chronic criterion concentration be expected to be more like that of copper, which would imply the nickel criterion might not be protective? Unfortunately, these questions cannot be answered definitively with existing data, and since the mechanisms of chronic nickel toxicity in fish are not well established (Pyle and Couture 2011), it is not even clear whether the mode of action of nickel is more like the calcium antagonist metals (cadmium, lead and zinc) or like copper, a sodium antagonist.

In the absence of data, an interspecies correlation estimation ("ICE") modeling framework was used to contrast the possible relative acute sensitivity of white sturgeon to the sensitivity of the rainbow trout to untested chemicals (Raimondo et al. 2013). The model outputs were that for acutely toxic effect concentrations of 35 and 62 µg/L of a generic chemical to the rainbow trout; the corresponding effect estimate for the genus *Acipenser* (that of white sturgeon) would be at 21 and 40 µg/L of the generic chemical. These particular concentrations were used because they represent the no- and lowest-observed adverse effects concentrations from a chronic study of effects of nickel on the rainbow trout by Nebeker et al. (1985). If these ICE estimates are considered meaningful, that would suggest that effects of nickel to the sturgeon could occur at concentrations close to the proposed chronic nickel concentration of 30 µg/L used in the test conditions reported by Nebeker et al. (1985). The results reported by Nebeker et al. (1985) may be a worse case comparison, based on the lack of observable effects reported by Brix et al. (2004) to the rainbow trout at nickel concentrations 10 times greater than the proposed chronic criterion.

However, based on the discussion above for bull trout critical habitat, although some adverse effects to sturgeon prey species may occur from their exposure to nickel at the proposed chronic criterion concentration, white sturgeon eat a variety of prey items and are known piscivores. For these reasons, the Service does not expect significant adverse effects to the sturgeon to be caused by the proposed chronic criterion for nickel.

## 2.5.10.5 Kootenai River White Sturgeon Critical Habitat

The designated PCEs of sturgeon critical habitat (73 FR 39506) are limited to factors related to flow regime, temperature requirements during spawning season, and the presence of rocky substrates (see section 2.3.8). However, for the purposes of this Opinion, we still consider sediment and water quality as important factors contributing to the capability of the critical habitat to support the recovery of the sturgeon. The designated PCEs would not be affected by the proposed action.

However, based on the discussion above for bull trout critical habitat, although some adverse effects to sturgeon prey species may occur from their exposure to nickel at the proposed chronic criterion concentration, white sturgeon eat a variety of prey items and are known pisciviores. For these reasons, the Service does not expect significant adverse effects to habitat conditions within sturgeon critical habitat to be caused by the proposed acute criterion for nickel.

# 2.5.11 Silver Aquatic Life Criterion

The proposed action includes only a proposed acute life criterion for silver that therefore limits both acute and chronic exposures. This necessitates a slightly different approach to the effects analyses than was done with substances that have both an acute and a chronic criterion value. For most substances, toxicity from short-term exposures is compared to the short-term (acute) criterion, and toxicity from long-term exposures is compared to long-term (chronic) criterion. However, since only a single criterion value is available for silver, regardless of the length or exposure or type of test, the results are compared against the sole silver criterion.

The aquatic life criterion for silver is defined as a function of water hardness. Using a water hardness value of 100 mg/L, the criterion value for silver is 3.5 µg/L, as calculated from the following equation:

$$\text{Silver criterion } (\mu g/L) = e^{(1.72[\ln(\text{hardness})]-6.52)} * (0.850)$$

At water hardness values of 10, 25, 50, and 250 mg/L, the acute silver criterion values are 0.07, 0.32, 1.05, 2.60, and 16.7 µg/L, respectively. As described for other metals discussed above, the criterion equation for silver also has a hardness “floor” of 25 mg/L, which was not used to calculate the example values above. The EPA (1980a) aquatic life criteria document on which the proposed acute silver criterion is based, did include a chronic criterion value. This chronic criterion value, 0.12 µg/L, did not vary with hardness. The different treatment for silver from all of the other metals criteria considered in this Opinion was not explained in EPA (1992) beyond the statement “*…with this rule, EPA is promulgating its 1980 criteria for silver, because the Agency believes the criteria is protective and within the acceptable range based on uncertainties associated with deriving water quality criteria.*” (EPA 1992, p. 60883).

Silver, in the free ion form, has been noted to be one of the most toxic metals to freshwater organisms and is highly toxic to all life stages of salmonids. Ionic silver is the primary form responsible for causing acute toxicity in freshwater fish (Wood 2011b). Toxicity varies widely depending on the anion present: silver nitrate has a much higher toxicity than silver chloride or silver thiosulfate, by approximately four orders of magnitude (Hogstrand et al. 1996). Documented effects of silver toxicity in fish from silver nitrate include interruption of ionoregulation at the gills, cell damage in the gills, altered blood chemistry, interference with zinc metabolism, premature hatching, and reduced growth rates (Hogstrand and Wood 1998).

Silver is sparingly soluble and rare in aquatic environments. EPA (1987c) provided natural background concentrations of silver ranging from 0.1 to 0.5 µg/L. Wood (2011b), however, noted that values in this range were obtained before the widespread adoption of clean sampling techniques in the 1990s and Wood considered values in this range to be orders of magnitude too high. Wood (2011b) indicated that better estimates of natural background silver concentrations were in the range of 0.1 to 5 ng/L (0.0001 to 0.005 µg/L). Such concentrations are not detectable with the technology used in non-specialty analytical laboratories. Even in highly contaminated areas, silver concentrations rarely exceed 0.1 to 0.3 µg/L. In nature, silver is unlikely to be found in its ionic form. Given the extremely high affinity of silver for reduced sulfur, most silver in the environment is expected to occur as silver sulfides, even in oxygenated waters (Wood 2011b). Even in Idaho's Silver Valley where 100+ years of silver mining resulted in one of the largest Superfund cleanup projects in the nation, silver is not a contaminant of concern (National Research Council (NRC) 2005). Although silver sulfides are the form most likely found in the environment, the form of silver usually used in most toxicity tests is silver nitrate because it dissolved better. Silver nitrate is much more toxic than the sparingly soluble silver sulfides (Wood 2011b). Chronic toxicity to freshwater aquatic life from silver nitrate may occur at concentrations as low as 0.12 µg/L (EPA 1980a) and the published silver criterion ranges from 0.07 to 11 µg/L over a range of water hardness values from 10 to 200 mg/L.

***Hardness as a Predictor of Silver Toxicity***

While the proposed silver criterion relies on a water hardness-dependent equation in the same manner as the other metals, EPA's (1987c) revised criteria document for silver concluded the hardness-toxicity relationship was insufficient to base a national criteria upon. The hardness-toxicity relationship in the proposed criteria is based on EPA's (1980f) older criteria for silver. The acute and chronic toxicities of silver are influenced by water hardness, chloride ion, DOC (i.e., dissolved organic carbon), sulfide, and thiosulfide concentrations, and with pH and alkalinity (Erickson et al. 1998; Hogstrand and Wood 1998). However, in natural waters, hardness is a less important influence on silver toxicity than other water quality constituents, specifically, chloride and DOC concentrations (Hogstrand and Wood 1998; Ratte 1999; Wood 2011b). The presence of the chloride ion is protective because silver chloride precipitates out of solution readily, although under certain conditions it is possible to observe the formation of the dissolved silver(Ag)$Cl^0$ complex (Erickson et al. 1998). Bury et al. (1999) determined that chloride and DOC concentrations ameliorated the silver ion inhibition of $Na^+$ influx and gill $Na^+/K^+$-ATPase activity in rainbow trout. Toxicity of silver was found to change very slowly with hardness, where a hundredfold increase in hardness resulted in reducing toxicity only by roughly 50 percent. In contrast, only a twofold increase in chloride ion was required to produce toxic effects similar to a hundredfold increase in hardness (Wood 2011b). DOC was more important than hardness for predicting the toxicity of ionic silver in natural waters to the rainbow trout, fathead minnow and *Daphnia magna* because DOC greatly reduced gill accumulations of silver through complexation. The presence of the chloride ion did not reduce gill accumulations of silver because it bound with free silver ($Ag^+$) and accumulated in gills as silver chloride, but reduced toxicity because the silver chloride did not enter cells and disrupt ionoregulation (Wood 2011b).

A key point from the environmental chemistry and aquatic toxicology literature for silver is overwhelming differences in toxicity between free ionic silver and complexed silver compounds. Most laboratory toxicity tests with silver used silver nitrate because it readily disassociates into ionic silver which tends to remain in solution (Hogstrand and Wood 1998). In contrast, in rivers, streams, lakes, and effluents, ionic silver tends to be vanishingly low, and measureable silver in natural waters and effluents occurs as either silver sulfide, silver chloride, silver thiosulfate, or as complexes with natural DOC (Adams and Kramer 1999; Kramer et al. 1999). The differences in effects concentrations obtained between tests using silver nitrate and other forms of silver may be on the orders of magnitude. For instance, Hogstrand et al. (1996) obtained a 7-day LC50 with rainbow trout and silver nitrate of 9 µg silver/L, but silver chloride and silver thiosulfate LC50s were >100,000 µg silver/L. Similarly, with the fathead minnow, compared to the free silver ion resulting from silver nitrate additions, silver chloride complexes were about 300 times less toxic and silver sulfide was at least 15,000 times less toxic (Leblanc et al. 1984). When very low and environmentally realistic levels of sulfide were added to a test water (0.0016 mg/L), the LC50 of *Daphnia magna* was increased by a factor of 5.5 (Bianchi et al. 2002).

Because this analysis considers that the forms of silver likely to be found in the environment (sulfide, thiosulfate, chloride) differ from the form used to derive the silver (nitrate), should for some unanticipated reason this assumption be falsified, the analyses would not hold, and re-initiation of consultation would be needed.

## 2.5.11.1 Snake River Aquatic Snails and the Bruneau Hot Springsnail

Few data relevant to the toxicity of silver to listed Snake River aquatic snails and the Bruneau Hot Springsnail are available. Croteau et al. (2011) demonstrated that *Lymnaea* (a snail in the same family, Lymnaidae, as the listed Banbury Springs lanx) could accumulate silver from either waterborne or dietary exposures. When snails ingested manufactured silver nanoparticles mixed with diatoms, digestion was damaged with snails eating less and inefficiently processing food. No similar effects were recorded following a waterborne pulsed exposure up to 60 nmol/L (6.5 µg/L) obtained from diluted silver nitrate into laboratory water with extremely low DOC and a target water hardness value (unmeasured) of about 90 mg/L (Croteau et al. 2011). At a water hardness value of about 90 mg/L, the silver criterion is about 2.9 µg/L which is lower than concentrations that did not cause adverse effects in short-term exposures in Croteau et al.'s (2011) experiment on *Lymnaea*.

In longer-term exposures, silver nitrate caused adverse effects to *Lymnaea* at less than the proposed criterion concentration of silver. Khangarot and Ray (1988) obtained a 96-hr LC50 for silver of 4.2 µg/L with *Lymnaea* in water with a hardness value of 195 mg/L. This adverse effect concentration is considerably lower than the corresponding silver criterion of 10.8 µg/L.

In contrast, with *Aplexa hypnorum* (a snail in the same family, Physidae, as the listed Snake River physa), a 96-hr LC50 for silver of 241 µg/L was obtained by Holcombe et al. (1983) in water with a hardness value of 50 mg/L. This value is much greater than the proposed silver criterion value of 1.1 µg/L at a water hardness value of 50.

No information was found on dissolved silver toxicity to snails in the family Hydrobiidae (i.e., the same family as the listed Bliss Rapids snail and the Bruneau Hot Springsnail). Völker et al. (2014) tested the long-term tolerance of the hydrobiidid snail *Potamopyrgus antipodarum* (New Zealand mudsnail) with nanosilver solutions. The solution was toxic to the snails with a 50 percent reduction in reproductive output at 15 µg/L in hard freshwater. Hardness was not reported, but specific conductivity was. For the specific conductivity of 770 µs/cm, for commonly occurring water types water hardness would be between about 300 ±50 mg/L as $CaCO_3$ (Hardy et al. 2005). The silver criterion for hardness 300 mg/L is 24 µg/L, which is higher than the 15 µg/L concentration causing reduced reproduction. Nanosilver is capable of dissolving in aqueous media, and releasing silver ions to contribute to toxic effects. While nanosilver solutions are made up of particles, because of the small size (15 nm) they would pass the 0.45 µm filter pore size commonly used for defining "dissolved" metals.

However, intracellular nanosilver toxicity is not fully attributable to released ions since coated silver nanoparticles were shown to pass the cell membrane and become localized inside cells. The observed toxic effects could result because the silver nanoparticles continued to dissolve internally within the cells, or become internalized probably by endocytosis. Once internalized, silver can interfere with amino acids, altering protein structures and functions (Völker et al. 2014, Wood 2011b). These mechanisms are different from truly dissolved silver, which is believed to act as a sodium antagonist, causing disrupted mineral balance through gill exposure. Therefore, the literature on the adverse effects of nanosilver is probably not fully appropriate to compare with the dissolved silver criterion.

Based on the studies referenced above, and principally because the form of silver in natural waters is much less toxic than ionic silver used in most laboratory exposures, the Service concludes that the proposed approval of the proposed silver aquatic life criteria is not likely to adversely affect the three listed Snake River aquatic snails and the Bruneau hot springsnail; all effects are expected to be insignificant or discountable.

## 2.5.11.2 Bull Trout

No specific information on the toxicity of silver to the bull trout or other members of the genus *Salvelinus* is available. However, much work on the sensitivity and mechanisms of silver toxicity has been published for the rainbow trout. Because rainbow trout were at least as sensitive as the bull trout to cadmium, copper, and zinc (Hansen et al. 2002a; Hansen et al. 2002c), the Service concludes the silver data related to exposure tests for the rainbow trout are sufficiently informative to evaluate the likely effects of the silver criterion to the bull trout.

Generally, the available data on effects of chronic exposures of the rainbow trout to silver indicate that adverse effects would be expected below the proposed acute criterion. Again, because EPA (1992) inexplicably only issued an acute criterion for silver, and the present action to approval Idaho's toxics criteria is closely linked to EPA's (1992) rulemaking, the effect of this is that the acute criterion becomes the sole criterion for evaluating the effects of acute or chronic silver exposures.

The data reviewed on chronic effects of silver (as silver nitrate) to rainbow trout indicate that the proposed acute criterion, which effectively acts as a chronic criterion, would not avoid chronic toxicity at concentrations below the acute criterion. For example, the work of Davies et al. (1978) suggests that the maximum acceptable silver concentration to prevent chronic mortality in rainbow trout embryos, fry, and juveniles, and avoid premature hatching, is less than 0.17 µg/L for a water hardness equal to 26 mg/L (Davies et al. 1978). The proposed acute criterion at a water hardness value of 26 mg/L is twice that concentration, 0.34 µg/L. Likewise, Nebeker et al. (1983) concluded that the maximum acceptable toxicant concentration of silver to prevent inhibition of growth of steelhead embryos was less than 0.1 µg/L for a water hardness value equal to 36 mg/L. The proposed acute criterion for silver at a water hardness value of 36 mg/L is six times that concentration, 0.6 µg/L.

In contrast to the studies of the aquatic toxicity of silver in exposures using silver nitrate, if fish were exposed from silver sulfate or silver chloride compounds, toxicity was far lower. For example, in 168-h tests with silver as silver nitrate, 50 percent of rainbow trout were killed at 9 µg/L (the LC50), but when exposed as silver as silver thiosulfate the LC50 was 137,000 µg/L, and no mortality was achieved with silver chloride up to 100,000 µg/L (Hogstrand et al. 1996).

Thus, at face value the absence of a chronic silver criterion implies potential mortality at proposed acute criteria concentrations to listed salmonids based on the data and information reviewed above. The internal inconsistencies between the criteria derivation guidelines (Stephan et al. 1985a), published criteria (EPA 1980a, 1987c), and the silver criterion published by EPA (1992) which led to the present proposed action are unexplained and inscrutable. Nevertheless, the more recent information on the forms of silver occurring in natural waters and the finding that the expected toxicity of silver in natural waters is far less than that obtained from laboratory tests using silver nitrate is compelling (Hogstrand and Wood 1998; Adams and Kramer 1999; Wood 2011b).

The proposed acute criterion for silver has the potential to adversely affect the prey base of the bull trout. Bull trout of all ages are opportunistic predators, shifting their diet towards abundant and easily captured prey at different times and locations. While there is evidence that bull trout can eat armored taxa such as clams and snails (Donald and Alger 1993), in general, we presume that taxa that are classified as vulnerable to salmonid predation are most important in bull trout diets (Suttle et al. 2004), and taxa that are generally non-vulnerable to salmonid predation (burrowing or armored taxa) are not critical to bull trout sustenance. On that basis, adverse effects to taxa considered generally non-vulnerable to salmonid predation are less of a concern than effects to common and vulnerable taxa. For instance, in studied lake populations, amphipods appear to be consistently important to bull trout rearing, as Donald and Alger (1993) showed bull trout weights in several lakes were correlated with amphipod abundance. Large zooplankton such as *Daphnia magna* or *Daphnia pulex* may be important food items in lakes, whereas smaller zooplankton such as Ceriodaphnia or copepods are less important (Wilhelm et al. 1999). In streams, bull trout of all ages prey on aquatic insects such as mayflies (ephemeropterans), stoneflies (plecopterans), caddisflies (trichopterans), beetles (coleopterans) midges (chironomids) and worms (oligochaetes) (Boag 1987; Underwood et al. 1995). In streams, small fish such as sculpin and juvenile salmonids can also be relatively important in the diet of the bull trout (Underwood et al. 1995). Adult migratory bull trout feed on both aquatic invertebrates and other fish depending on prey availability and the size of the bull trout (Donald and Alger 1993; Wilhelm et al. 1999; Beauchamp and Van Tassell 2001).

Daphnids appear to be considerably more sensitive to silver than fish, with LC50s reported for cladocerans below the proposed acute criterion (EPA 1987c). The response of *Daphnia magna* to silver exposure tested in the absence of sulfide in water with a hardness value of about 120 mg/L yielded an LC50 concentration of silver at 0.22 µg/L (Bianchi et al. 2002, p. 1294); which was 20 times lower than the proposed acute criterion value of 4.7 µg/L for that water hardness value. When tested in the presence of environmentally realistic levels of sulfide, the LC50 concentration was increased by about 5.5 times (Bianchi et al. 2002, p. 1294). Other invertebrate taxa serving as potential food for juvenile salmonids have been determined to experience mortality only at silver concentrations that are above the proposed acute criterion. Reduced growth in mayfly larvae in response to silver exposure occurred at a silver concentration of 2.2 µg/L in test water with a hardness value of 49 mg/L (Diamond et al. 1992), which is greater than the proposed acute criterion for silver of 1.1 µg/L for that water hardness value.

Although some adverse effects to Daphnids may be expected from exposure to silver at the proposed acute criterion concentration, bull trout eat a variety of prey items and are known piscivores. In addition, the form of silver in natural waters is much less toxic than ionic silver used in most laboratory exposures. For these reasons, the Service does not expect significant adverse effects to the bull trout to be caused by the proposed acute criterion for silver.

## 2.5.11.3 Bull Trout Critical Habitat

Of the nine PCEs defined for bull trout critical habitat, the proposed acute criterion for silver has the potential to adversely affect PCE 3 (adequate prey base), as discussed below.

Bull trout of all ages are opportunistic predators, shifting their diet towards abundant and easily captured prey at different times and locations. While there is evidence that bull trout can eat armored taxa such as clams and snails (Donald and Alger 1993), in general, we presume that taxa that are classified as vulnerable to salmonid predation are most important in bull trout diets (Suttle et al. 2004), and taxa that are generally non-vulnerable to salmonid predation (burrowing or armored taxa) are not critical to bull trout sustenance. On that basis, adverse effects to taxa considered generally non-vulnerable to salmonid predation are less of a concern than effects to common and vulnerable taxa. For instance in studied lake populations, amphipods appear to be consistently important to bull trout rearing, as Donald and Alger (1993) showed bull trout weights in several lakes were correlated with amphipod abundance. Large zooplankton such as *Daphnia magna* or *Daphnia pulex* may be important food items in lakes, whereas smaller zooplankton such as Ceriodaphnia or copepods are less important (Wilhelm et al. 1999). In streams, bull trout of all ages prey on aquatic insects such as mayflies (ephemeropterans), stoneflies (plecopterans), caddisflies (trichopterans), beetles (coleopterans ) midges (chironomids) and worms (oligochaetes) (Boag 1987; Underwood et al. 1995). In streams, small fish such as sculpin and juvenile salmonids can also be relatively important in the diet of the bull trout (Underwood et al. 1995). Adult migratory bull trout feed on both aquatic invertebrates and other fish depending on prey availability and the size of the bull trout (Donald and Alger 1993; Wilhelm et al. 1999; Beauchamp and Van Tassell 2001).

Daphnids appear to be considerably more sensitive to silver than fish, with LC50s reported for cladocerans below the proposed acute criterion (EPA 1987c). The response of *Daphnia magna* to silver exposure tested in the absence of sulfide in water with a hardness value of about 120 mg/L yielded an LC50 concentration of silver at 0.22 µg/L (Bianchi et al. 2002); which was 20 times lower than the proposed acute criterion value of 4.7 for that water hardness value. When tested in the presence of environmentally realistic levels of sulfide, the LC50 concentration was increased by about 5.5 times (Bianchi et al. 2002). Other invertebrate taxa serving as potential food for juvenile salmonids have been determined to experience mortality only at silver concentrations that are above the proposed acute criterion. Reduced growth in mayfly larvae in response to silver exposure occurred at a silver concentration of 2.2 µg/L in test water with a hardness value of 49 mg/L (Diamond et al. 1992), which is greater than the proposed acute criterion for silver of 1.1 µg/L for that water hardness value.

Although some adverse effects to Daphnids may be expected from exposure to silver at the proposed acute criterion concentration, bull trout eat a variety of prey items and are known piscivores. For this reason the Service does not expect significant adverse effects to the bull trout prey base from any reduction in Daphnid abundance. In addition, the form of silver in natural waters is much less toxic than ionic silver used in most laboratory exposures. For these reasons, the Service concludes that the proposed acute criterion for silver is not likely to cause significant adverse effects to PCE 3 of bull trout critical habitat or otherwise create habitat conditions within the critical habitat that are likely to adversely impair the capability of the critical habitat to support bull trout recovery.

### 2.5.11.4 Kootenai River White Sturgeon

No data on silver toxicity to the white sturgeon, or any other Acipenser species were found during this consultation. For that reason, the data reviewed above regarding the effects of silver toxicity to the rainbow trout, which was used as a surrogate for characterizing the effects of the proposed acute criterion for silver on the bull trout, was also relied upon as a surrogate for characterizing the likely effects of the silver criterion to the Kootenai River white sturgeon. The Service concludes this analytic approach is reasonable because for some contaminants, the white sturgeon and other sturgeon species are at least as sensitive as the rainbow trout (Dwyer et al. 2005; Ingersoll and Mebane 2014).

Based on the discussion above for bull trout and bull trout critical habitat, although some adverse effects to sturgeon prey species may occur from their exposure to silver at the proposed acute criterion concentration, white sturgeon eat a variety of prey items. In addition, the form of silver in natural waters is much less toxic than ionic silver used in most laboratory exposures. For these reasons, the Service does not expect significant adverse effects to the sturgeon to be caused by the proposed acute criterion for silver.

### 2.5.11.5 Kootenai River White Sturgeon Critical Habitat

The proposed acute criterion for silver will have no effect on the PCEs (flow regime, water temperature, and rocky substrates) of sturgeon critical habitat. Based on the discussion above for bull trout critical habitat, although some adverse effects to sturgeon prey species may occur from their exposure to silver at the proposed acute criterion concentration, white sturgeon eat a variety of prey items. In addition, the form of silver in natural waters is much less toxic than ionic silver used in most laboratory exposures. For these reasons, the Service does not expect significant adverse effects to habitat conditions within sturgeon critical habitat to be caused by the proposed acute criterion for silver.

## 2.5.12 Organic Toxic Substances

Table10 summarizes the information contained in the following sections of the Opinion addressing organic toxic substances. As shown in this table, the Service has concluded that these substances are not likely to adversely affect (NLAA) the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

**Table 10. Summary of effect analyses for organic toxic substances considered under the proposed action.**

| Organic Toxin | Year Registration Cancelled | Aquatic Life Criteria | Human Health Criteria (applicable to all waters in Idaho) | Effects Determination | Rationale |
|---|---|---|---|---|---|
| Endosulfan | 2012 (all uses in US will end in 2016) | 0.22 µg/L (acute); 0.056 µg/L (chronic) | Less protective than Aquatic Life Criteria (0.93 and 2. | NLAA | Most uses banned, relatively short half-life, mitigation measures to minimize risk of exposure |
| Aldrin/Dieldrin | 1975 | Aldrin – 3.0 µ/L Dieldrin - 2.5 µ/L (acute) and 0.0019 µ/L (chronic) | Aldrin – 0.00013 µg/L and 0.00014 µg/L; Dieldrin – 0.00014 µg/L | NLAA | No new discharges, human health criteria will minimize exposure risk |
| Chlordane | 1983 | 2.4 µ/L (acute), 0.0043 µ/L (chronic) | 0.00057 and 0.00059 µg/L | NLAA | No new discharges permitted and human health criteria will minimize exposure risk |
| DDT | 1972 | 1.1 µ/L (acute), 0.001 µ/L (chronic) | 0.00059 µg/L | NLAA | No new discharges permitted and human health criteria will minimize exposure risk |
| Endrin | 1984 | 0.18 µ/L (acute), 0.0023 µ/L (chronic) | No human health criteria | NLAA | No new discharges, soil half-life reduce exposure risk |
| Heptachlor | 1988 | 0.52 µ/L (acute), 0.0038 µ/L (chronic) | 0.00021µ/L | NLAA | No new discharges, soil half-life of 6 months to 3.5 years, human health criteria |
| Lindane | 2006 | 2.0 µ/L (acute), 0.08 µ/L (chronic) | No human health criteria | NLAA | No new discharges short soil half-life (900 days), |

| Organic Toxin | Year Registration Cancelled | Aquatic Life Criteria | Human Health Criteria (applicable to all waters in Idaho) | Effects Determination | Rationale |
|---|---|---|---|---|---|
| | | | | | meets COE sediment screening guidelines |
| PCB | 1979 (production banned by Congress) | 0.014 µ/L (chronic) - there is no acute criterion | 0.000044 and 0.000045 µg/L | NLAA | No new discharges, human health criteria will minimize exposure risk |
| Toxaphene | 1990 | 0.73 µ/L (acute), 0.0002 µ/L (chronic) | Human health criteria (0.00073 µ/L) less protective than chronic aquatic life criteria | NLAA | No new discharges, soil half-life reduce exposure risk |
| PCP (pentachlorophenol) | 1987 (became a registered use product only available for wood preservation by certified applicators) | 20 µ/L (acute), 13 µ/L (chronic) – at pH 7.8 | 0.28 and 8.2 µg/L | NLAA ? | Double vacuum treatment used for wood preservation, limited paths for exposure (used primarily for utility poles), mitigation measures for use near water, human health criteria will minimize exposure risk |

One of the basic premises of this Opinion, based on the description of the proposed action, is that all waters within the action area are expected to be at proposed criteria levels. This condition requires us to consider the potential exposure of listed species and the PCEs or other essential features of critical habitat to these criteria concentrations. In the case of the inorganic chemicals at issue herein there is a reasonable potential for such levels to occur. In the case of the eleven above organic chemicals, it is highly unlikely that they will reach criteria concentrations in waters within the action area.

The principal rationale for the above conclusion is: (1) exposure of listed aquatic species to these chemicals is unlikely because the majority are banned for any use and their half-lives preclude significant concentrations remaining in the environment; and (2) in many cases there is a human health criterion in place, applicable to all waters in Idaho, that is much more restrictive and protective of listed species and critical habitat than the aquatic life criteria under consultation.

It is important to note that for rationale (1) above, we are deviating from the basic premise underlying the rest of this Opinion in relying on the minimal risk of exposure in our effects analysis for the organic substances. In contrast, for the inorganic substances we are assessing the protectiveness of the proposed aquatic life criteria by assuming all waters in the action area are at criteria levels. This is an unrealistic approach for the organic substances precisely because they are banned or extremely limited in application and future discharges are highly unlikely to occur.

# 2.5.13 Endosulfan Aquatic Life Criteria

The proposed acute criterion for endosulfan is 0.22 μg/L and the chronic criterion is 0.056 μg/L.

Endosulfan is a broad spectrum polychlorinated cyclodiene insecticide that does not occur naturally in the environment and was used for controlling over 100 agricultural pests and 60 food and non-food crops (NMFS 2014a, p. 233). Due to concerns about worker and environmental safety, the EPA (1) cancelled the registration for home and garden use of endosulfan in 2000; and (2) implemented a voluntary cancellation and phase-out of endosulfan that began on July 31, 2012. The phase-out will be implemented in six phases over a 4-year period. During these phases, use of endosulfan on certain types of crops and products are scheduled to end. All uses of endosulfan will end by July 31, 2016 (ATSDR 2013, p. 9).

Endosulfan is mixture of two biologically-active isomers (α and β) in a ratio of 70:30. The chemical is relatively persistent and semi-volatile. The β -isomer is generally more persistent and the α-isomer is more volatile (EPA 2009, p. 12).

After endosulfan is released into the environment, it "undergoes a variety of transformation and transport processes" (ATSDR 2013, p. 9). Endosulfan sulfate, the major degradation product, is more persistent than the parent compound. Neither endosulfan nor its biodegradation products are expected to be mobile in soil. In an aerobic soil metabolism study using five different soils, half-lives of α-endosulfan ranged from 35 to 67 days and half-lives of β-endosulfan ranged from 104 to 265 days with endosulfan sulfate as the major metabolite (ATSDR 2013, p. 219). In water, endosulfan is expected to partition to sediment where the half-life is >329 days (EPA 2010a, Table 5.26). Endosulfan may volatilize from soil, water, or plant surfaces. The half-life in air is 1.3 days (EPA 2010a, Table 5.26). Long-range aerial transport of endosulfan to areas distant from the point of release (e.g., the Arctic) is well documented (EPA 2010a, p. 9 of 11).

EPA (2010a, p. 5 of 11) reports that based on laboratory studies, endosulfan is classified as "very highly toxic to aquatic animals"; LC50 values (i.e., the concentration that is lethal to 50 percent of the test animals) are as low as 0.1 μg/L for freshwater fish and 0.7 μg/L for freshwater invertebrates. Although these acute toxicity LC50s values are similar, the LC50s for fish span approximately 3 orders of magnitude while the values for freshwater invertebrates span approximately 5 orders of magnitude, suggesting that as a "taxonomic group, the assemblage of freshwater fish species are collectively more vulnerable to acute endosulfan exposure compared to the assemblage of freshwater invertebrates" (EPA 2010a, p. 5 of 11). EPA (2010a, p 5 of 11) found that the chronic toxicity of endosulfan to freshwater fish is estimated as low as 0.023 μg/L (estimated No Observed Adverse Effects Concentration (NOAEC)), which is about half the freshwater chronic water quality criterion of 0.056 μg/L. Chronic toxicity of freshwater invertebrates is estimated as low as 0.011 μg/L (estimated NOAEC). Chronic effects associated with NOAECs used to derive these toxicity reference values include impacts on survival, growth and reproduction (EPA 2010a, p. 5 of 11). These data suggest that adverse effects to listed aquatic species are likely to occur from exposure to endosulfan at the acute and chronic criteria levels.

However, EPA (2010a, pp. 70, 78) reported on multi-year (1991 – 2008) monitoring programs which showed no detections of endosulfan in surface waters in the action area and no detections in benthic sediments. NMFS (2014a) also found that there were no known concentrations of endosulfan in waters inhabited by listed salmon and steelhead in the action area. Although some very restricted uses of the endosulfan will continue until July 31, 2016, the extent of use in the action area is unclear but expected to be very limited[21]. In addition the EPA requires the implementation of additional protective measures and BMPs (i.e., mitigation measures) that will reduce the chance of introducing endosulfan to aquatic habitats occupied by listed species. These measures include a 100-foot setback for ground applications between treated areas and water bodies, a 30-foot vegetative buffer between treated areas and water bodies, reductions in single maximum application rates, reductions in maximum seasonal application rates, and reductions in maximum numbers of applications allowed in a single growing season (EPA 2002b, p. vii). As an additional protective measure, EPA banned aerial applications of endosulfan in 2010 (EPA 2010b, Appendices B, C, and D).

Because all uses will be banned after 2016 and exposure of listed species to endosulfan is unlikely given the relatively short half-lives (compared to other organochlorine insecticide, such as DDT) and mitigation measures that are in place to reduce the risk of introducing endosulfan to aquatic habitats, the Service concludes that the proposed acute and chronic endosulfan aquatic life criteria are not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat; all effects will be insignificant or discountable.

## 2.5.14 Aldrin/Dieldrin Aquatic Life Criteria

The proposed acute criterion for aldrin is 3.0 μg/L. EPA (1980b) determined that the available data did not support the determination of a chronic toxicity criteria for aldrin. For dieldrin, the acute and chronic criteria are 2.5 μg/L and 0.0019 μg/L, respectively.

Aldrin and dieldrin are the common names of two structurally similar compounds (synthetic cyclic chlorinated hydrocarbons called cyclodienes) that were extensively used as soil insecticides from the 1950s until 1970. While the U.S. Department of Agriculture canceled all agricultural uses of aldrin and dieldrin in 1970, their use for killing termites was approved by the EPA in 1972 and continued until 1987. With the manufacturer voluntarily canceling the

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[21] The Service found reference to only one remaining use in Idaho – the treatment of alfalfa grown for seed only (www.agri.state.id.us/.../Endosulfan3ECIDAHOSLNOnAlfalfa980003.pdf)

registration for their use in controlling termites in 1987 (ATSDR 2002a, p. 2), aldrin and dieldrin are no longer produced or used in the United States.

Dieldrin is more often detected in the environment because aldrin degrades to dieldrin (EPA 1980b, p. A-2). EPA (1980b, p. A-2) reports that dieldrin is probably the most stable and persistent insecticide among the cyclodienes, requiring from 5 to 25 years for 95 percent of the dieldrin to disappear from soil, depending on the available microbial flora. Given this information, and the fact that aldrin/dieldrin were last lawfully used in 1987, we assume there should be no significant concentrations of dieldrin in action area soils as of the writing of this Opinion.

While no data on the toxicity of aldrin/dieldrin to molluscs was found, available information suggests adverse effects to aquatic life are possible (Peakall 1996, p. 74; EPA 1993b, pp. 3-8; Sanders and Cope 1966) when concentrations of aldrin are at or below acute criterion levels. Thus, the EPA (1999b) determined that the approval of the acute aquatic life criteria for aldrin/dieldrin established by the Idaho Water Quality Standards may have adverse effects on the bull trout and white sturgeon. Similarly, available studies demonstrate that chronic effects of aldrin do occur to freshwater fish at 0.0466 µg/L, and to prey items at 2.5 µg/L, suggesting that the absence of a chronic criterion could result in adverse chronic effects to listed salmonids and their food source (NMFS 2014a). However, given that aldrin and dieldrin have not been allowed for agricultural use for over 40 years, that the 1987 cancellation of registration precludes any lawful releases into the environment, and that the half-life would minimize the potential for any post-use environmental releases, the risks to listed species and critical habitat of exposure to aldrin/dieldrin subject to this action are discountable.

Therefore the Service concludes that the approval of the acute and chronic aldrin and dieldrin criteria established by the Idaho Water Quality Standards is not likely to adversely affect the Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

## 2.5.15 Chlordane Aquatic Life Criteria

The proposed acute and chronic chlordane aquatic life criteria are 2.4 µg/L and 0.0043 µg/L, respectively.

Chlordane is a chlorinated, cyclodiene pesticide (EPA 1999a, p.108) and does not naturally occur in the environment. It has been used extensively as a broad spectrum pesticide, and is a potential carcinogen (Eisler 1990, pp. 3, 4). It bioconcentrates and may bioaccumulate (Eisler 1990, p. 33).

EPA banned the use of chlordane on food crops in 1978, and phased out other uses over the next 5 years (EPA 1980c). From 1983 to 1988, its only approved use was to control termites in homes. When its application for termite control was banned in 1988, all approved use of chlordane in the U.S. stopped.

Chlordane is persistent in the environment with a soil half-life of approximately 4 years (http://extoxnet.orst.edu/pips/chlordan.htm, accessed August 26, 2014). Given this information, and the fact that chlordane was last lawfully used in 1988; we assume there should be no significant concentrations of chlordane in action area soils as of the writing of this Opinion.

However, chlordane does not degrade rapidly in water and readily adsorbs to sediment. Adsorption to sediment is expected to be a major fate process (NMFS 2014a) and the presence of chlordane in sediment core samples suggests that chlordane may be very persistent in the adsorbed (to sediment) state in the aquatic environment (http://water.epa.gov/drink/contaminants/, accessed August 29, 2014). Therefore the major source of this compound will not be through point source discharges into surface water bodies, but from repositories of the contaminant that are persistent in sediments.

The Assessment (EPA 1999a, pp. 108-109) provided no data on the toxicity of chlordane on freshwater snails of any species nor on the more general category of molluscs. Available information for aquatic invertebrates appears to be limited to Daphnia, amphipods, stoneflies, and crayfish and the proposed criteria for freshwater life protection (0.0043 µg/L, 24-h mean; not to exceed 2.4 µg/L) overlap the range of 0.2 to 3.0 µg/L shown to adversely affect certain fish and aquatic invertebrates (Eisler 1990). For freshwater algae, both growth stimulation and inhibition have been reported, with the direction of effects dependent on species tested (Eisler 1990, p. 34).

For listed salmon and steelhead, NMFS (2014a) concluded that the acute or chronic chlordane criteria would not harm or kill these species, but noted that sublethal effects like the ones found in mammals (i.e., neurological damage, altered immune and reproductive function, and increased cancer risk) have not been studied under applicable exposure scenarios (i.e., long-term chlordane exposure at concentrations near or below the criterion). Similarly, few data are available on the sublethal effects of long-term exposure to chlordane on salmonid prey species. Additionally, bioaccumulation can occur in salmonids with chronic exposure to chlordane at levels allowable under the proposed criteria, and exposure is likely to occur not only through the water column but also through diet and particularly contact with sediments. The proposed criteria do not account presently for these other sources of exposure.

Because sediments are likely a primary source of chlordane to the aquatic environment, NMFS (2014a) calculated the chlordane sediment concentration that would result in chlordane concentrations in the water column at or below the proposed chronic criteria (0.0043 µg/L) and found that the chronic aquatic life criterion would be associated with chlordane concentrations in sediment ranging between 12 ng/g to 61 ng/g sediment. This exceeds the sediment screening guideline of 10 ng/g dry weight (dw) established by the U.S. Army Corps of Engineers (USCOE) for in-water disposal of dredged sediment (USCOE 1998, Table 8-1, p. 8-7). These data suggest that chlordane released from sediments (e.g., during in-water disposal of contaminated sediments) could adversely affect the salmonid prey base at concentrations below the proposed chronic criteria, as the USCOE sediment quality criteria are based primarily on tests with benthic invertebrates.

However, NMFS (2014a) noted that the most stringent applicable criterion in the action area, the human health (fish consumption based) water quality criterion of 0.00057 µg/L that is also applicable to waters occupied by listed species and designated critical habitats, is about eight times lower than the chronic criterion of 0.0043 µg/L for chlordane. When extrapolated to predict sediment concentrations in the same fashion as the chronic criterion, the resulting sediment concentration would be about 1.6 ng/g to 8 ng/g dw sediment, which is less than the USCOE screening criteria.

Also, given that chlordane has not been allowed for agricultural use for 36 years, that the 1988 cancellation of registration precludes any lawful releases into the environment, that the soil half-life would minimize the potential for any post-use inputs into aquatic habitats, and potential exposure via inwater disturbance/disposal of chlordane contaminated sediments during USCOE actions (e.g., dredging) are minimized by their sediment screening guideline, the risks to listed species of chlordane subject to this action are discountable.

Therefore the Service concludes that the approval of the acute and chronic chlordane criteria established by the Idaho Water Quality Standards is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

## 2.5.16 Dichlorodiphenyltrichloroethane (DDT) Aquatic Life Criteria

The acute and chronic DDT aquatic life criteria are 1.1 and 0.001 µg/L, respectively. DDT is a chlorinated pesticide that does not naturally occur in the environment. The insecticidal properties of DDT were first discovered in the early 1940s, and the pesticide was used extensively on crops in the United States over the period 1945 to 1972. It was also used as a mosquito larvacide, as a spray for eradication of malaria in dwellings, and as a dust in human delousing programs for typhus control.

The EPA banned the use of DDT in the United States in 1972 (EPA 1999a, p. 111). EPA based this decision on (1) DDT and its metabolites are toxicants with long-term persistence in soil and water, (2) DDT is widely dispersed by erosion, runoff, and volatilization, and (3) the low-water solubility and high lipophilicity of DDT result in concentrated accumulation of DDT in the fat of wildlife and humans which may be hazardous (EPA 1980d, p. A-1).

DDT has a reported half-life in soil of between 2-15 years or longer depending on soil and climate conditions, and is immobile in most soils (http://pmep.cce.cornell.edu/profiles/extoxnet/carbaryl-dicrotophos/ddt-ext.html)[22]. Breakdown products in the soil environment are dichlorodiphenylethylene (DDE) and dichlorodiphenyldichloroethane (DDD), which are also highly persistent and have similar chemical and physical properties. (http://pmep.cce.cornell.edu/profiles/extoxnet/carbaryl-dicrotophos/ddt-ext.html). Given this information, and the fact that DDT was last lawfully used in agriculture in 1972, we assume there should be no significant concentrations of DDT in action area soils as of the writing of this Opinion. However, when released into water DDT that does not volatize will adsorb strongly to particulate matter in the water column and primarily partition into the sediment, which “is the sink for DDT released into water” (ATSDR 2002b, p. 233). Therefore the major source of this compound will not be through point source discharges into surface water bodies, but from repositories of the contaminant that are persistent in sediments.

[22] Accessed September 12, 2014.

Once in bottom sediments DDT can enter aquatic food webs through ingestion by benthic organisms. Because DDT (including DDE and DDD) is highly lipophilic (i.e., lipid soluble) and has a very long half-life it readily bioconcentrates in aquatic organisms (i.e., levels in organisms exceed those levels occurring in the surrounding water) (ATSDR 2002b, p. 235). Reported biocentration factors (BCF) are 51,000-100,000 in fish, 4,550-690,000 in mussels, and 36,000 in snails (ATSDR 2002b, p. 235). Trophic level differences in bioconcentration are largely due to increased lipid content and decreased elimination efficiency among higher level organisms. Organisms also feed on other animals at lower trophic levels. The result is a progressive biomagnification of DDT in organisms at the top of the food chain. Biomagnification is the cumulative increase in the concentration of a persistent contaminant in successively higher trophic levels of the food chain (i.e., from algae to zooplankton to fish to birds) (ATSDR 2002b, p. 235).

The Assessment (EPA 1999a) provided no data on the toxicity of DDT on freshwater snails of any species or on the more general category of molluscs. Available information for aquatic invertebrates appears to be limited to Daphnia, scuds, glass shrimp, and crayfish. NMFS (2014a) reports that DDT is highly toxic to many aquatic invertebrate species based on data from Johnson and Finley (1980) and Lotufo et al. (2000). These results suggest that the acute (1.1 µg/L) criterion is probably not protective of gammarid amphipods and related invertebrates, but the chronic aquatic life (0.001 µg/L) standard would likely be protective if the major source of DDT exposure were through the water column. However, because DDT tends to accumulate in sediment, some reduction in available salmonid invertebrate prey species will likely occur in areas with contaminated sediments.

NMFS (2014a) states that concentrations of DDT in the action area at the proposed action acute criterion could harm listed fish. The chronic criteria have risk of sublethal health effects in salmonids if bioconcentration results in tissue concentrations that are higher than those expected by EPA. The proposed chronic criterion may allow substantial bioaccumulation to occur because DDT is taken up not only from the water column but also from sediments and prey organisms. No reports of direct adverse effects to listed salmonids were located at concentrations lower than the chronic criterion.

Because DDT is no longer in use in the United States, the primary source of this compound will not be through point source discharges into surface water bodies, but rather from repositories of the contaminant that are persistent in sediments; sediments are likely the primary potential source of DDT.

NMFS (2014a) calculated the sediment DDT concentration that would result in DDT concentrations in the water column at or below the proposed criteria and found that the proposed criteria would be associated with sediment DDT concentrations ranging from 12 ng/g to 60 ng/g sediment. This level exceeds the sediment screening guideline of 6.9 ng/g dw established by the USCOE for in-water disposal of dredged sediment (USCOE 1998, Table 8-1, p. 8-7). This suggests the potential for impacts on the salmonid prey base, as these guidelines are based primarily on tests with benthic invertebrates.

However, the most stringent applicable criterion in the action area, the human health (fish consumption based) water quality criterion of 0.00059 µg/L that is also applicable to waters occupied by listed species and designated critical habitats, is about two times lower than the chronic criterion of 0.001 µg/L for DDT and would provide an additional level of protection to listed species.

Also, given that DDT has not been allowed for agricultural use for 42 years, that the 1972 ban precludes any lawful releases into the environment, that the soil half-life would minimize the potential for any post-use inputs into aquatic habitats, and potential exposure via inwater disturbance/disposal of DDT contaminated sediments during USCOE actions (e.g., dredging) are minimized by their sediment screening guideline, the risks to listed species and critical habitat from the proposed DDT criteria are discountable.

Therefore the Service concludes that the approval of the acute and chronic DDT criteria established by the Idaho Water Quality Standards is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

## 2.5.17 Endrin Aquatic Life Criteria

The proposed acute and chronic aquatic life criteria for endrin are 0.18 µg/L and 0.0023 µg/L, respectively.

Endrin is a chlorinated, broad spectrum pesticide that is a stereoisomer of dieldrin. It is no longer manufactured in the United States. Endrin was first used as an insecticide, rodenticide, and avicide in 1951 to control agricultural pests on cotton, apples, sugarcane, tobacco, and grain (ATSDR 1996, p. 96). In part as a result of its observed toxicity to non-target organisms, bioaccumulation potential, and persistence, EPA banned the use of endrin in 1984 (EPA 1993c, p. 1-6). There are no current authorized uses of endrin in the United States (NMFS 2014a).

Endrin is persistent in the environment with a half-life in soil of up to 12 years (WHO 1992). Runoff from agricultural areas where endrin was used was an important route for delivering endrin to aquatic systems (WHO 1992, p. 32). In water, endrin readily adsorbs to sediment and significantly bioconcentrates in aquatic organisms (1,450 to 10,000 times the concentration in water) (EPA 1993c, p. 1-6; ATSDR 1996, p. 100); however biomagnification appears to be limited.

Because endrin is no longer used in the United States, the major source of this contaminant will not be through point source discharges into surface water, but from repositories of the chemical that remain in sediment (NMFS 2014a). According to ATSDR (1996, p.3) exposure to endrin is unlikely except in “areas where it is concentrated, such as a hazardous waste site.”

Invertebrates tend to be more tolerant of endrin than fishes (EPA 1999a, p. 112). Benthic organisms were among both the most sensitive, and most resistant freshwater species to endrin (EPA 1993c, p. 3-2). Effects of endrin toxicity to freshwater organisms include reduced growth, increased time to metamorphosis, reduced disease resistance reduced reproduction (reduced number of eggs), and reduced survival (EPA 1993c, pp. 3-5 - 3-8).

The Assessment stated that LC50s for aquatic snails ranged from 73 to 12,000 µg/L, but did not indicate which species were tested (EPA 1999a, p. 112). This may be important, as a wide variation in 96-hour LC50s was observed in 18 species of invertebrates tested, ranging from 0.007 µg/L for crane flies to 320 µg/L for mature crayfish (Johnson and Finley 1980, p. 38). Most available information for aquatic invertebrates appears to be limited to Daphnids, seed shrimp, sowbugs, scuds, crayfish, glass shrimp, stoneflies, mayflies, damselflies, crane flies, and snipe flies. Lethality was observed in stoneflies at concentrations at or below the proposed acute criteria; lethality was observed in other test species at concentrations above the proposed acute criterion (Johnson and Finley 1980, pp. 37-38).

NMFS (2014a) found that for fish, most reports of mortality following short-term endrin exposures produced LC50s greater than the acute criterion, although some effects occurred at lower concentrations. Evidence indicates that concentrations at the acute criterion will not harm salmonid prey species (NMFS 2014a).

NMFS (2014a) also states that while data are sparse, most reports of adverse effects from chronic exposures to salmonids or other fish occurred at concentrations higher than the chronic criterion.

Because sediments are likely a primary source of endrin to the aquatic environment, NMFS (2014a) calculated the endrin sediment concentration that would result in endrin concentrations in the water column at or below the proposed chronic criteria (0.0023 μg/L) and found that the chronic aquatic life criterion would be associated with endrin concentrations in sediment ranging between 7.36 μg/kg to 36.8 μg/kg dw sediment. These levels are within the range of the interim Canadian freshwater sediment guidelines of 2.67 to 62.4 ng/g dw sediment (NMFS 2014a). The higher of these values is a probable effect level, based on spiked sediment toxicity testing and associations between field data and biological effects (CCME 2001)[23]. This suggests that the proposed criteria are unlikely to reduce the quality or quantity of listed salmonid food items (NMFS 2014a).

Also, given that endrin has not been allowed for agricultural use for 28 years (as of the writing of this Opinion), that the 1986 cancellation of registration precludes any lawful releases into the environment, that the soil half-life would minimize the potential for any post-use inputs into aquatic habitats, the risks to listed species from the proposed endrin criteria are discountable.

Therefore, the Service concludes that approval of the acute and chronic endrin criteria established by the Idaho Water Quality Standards is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

## 2.5.18 Heptachlor Aquatic Life Criteria

The acute and chronic criteria established by the Idaho Water Quality Standards for heptachlor are 0.52 μg/L and 0.003 μg/L, respectively.

Heptachlor is an organochlorine cyclodiene, broad-spectrum insecticide that does not naturally occur in the environment. Until it was banned for home and agricultural use in 1976, heptachlor was commonly used for crop pest control and by exterminators and home owners to kill termites. In 1976, it was prohibited from home and agricultural use, although commercial applications to

[23] For some other organic pesticides in this Opinion the Service referenced the Corps' sediment screening criteria (USCOE 1998). However, the Corps framework document does not contain a screening criterion for endrin; the Service referenced the Canadian guidelines instead.

control insects continued. In 1988, its use for termite control was banned, and currently its only permitted commercial use in the United States is fire ant control in power transformers (ATSDR 2007, p. 2; NMFS 2014a).

Heptachlor and its degradation products or metabolites are still commonly detected in environmental samples due to their stability and persistence in the environment (EPA 1999a, p. 113). The soil half- life of heptachlor is 6 months to 3.5 years, but trace levels have been detected in soil 14 and 16 years after application (http://pmep.cce.cornell.edu/profiles/extoxnet/haloxyfop-methylparathion/heptachlor-ext.html, accessed September 19, 2014). When released into water, it adsorbs strongly to suspended and bottom sediment (ATSDR 2007, p. 95).

Because heptachlor is no longer in use in the United States, except for selected special applications, the primary potential source of this compound will be from repositories of the contaminant that are persistent in sediments, not from point source discharges into surface water bodies (NMFS 2014a).

Heptachlor is lipophilic and bioconcentrates and bioaccumulates (EPA 1999a, p. 113). Toxicity of heptachlor may be altered by a number of factors including temperature, duration of exposure (Johnson and Finley 1980, p. 44), and presence of mixtures. Effects of heptachlor toxicity to freshwater organisms include reduced growth, inhibited ATP-ase activity, and reduced survival (see EPA 1999a, p. 113).

The Assessment (EPA 1999a, pp. 113-114) provided no data on the toxicity of heptachlor on freshwater snails of any species or on the more general category of molluscs. Available information for aquatic invertebrates appears to be limited to Daphnids, scuds, crayfish, and glass shrimp. All LC50s were well above the proposed criteria. Mayer and Ellersieck (1986) reported heptachlor LC50s for a variety of invertebrates in static tests. Daphnids had 48-hour EC50s ranging from 42-80 µg/L. In 24-hour and 96-hour tests, scuds had LC50s ranging from 140-180 µg/L and 29-56 µg/L, respectively; crayfish values were 2.6 µg/L and 0.5 µg/L, respectively; and glass shrimp test results were 30 µg/L and 1.8 µg/L, respectively.

NMFS (2014a) found that available evidence indicates that listed salmon or steelhead experience acute lethal effects at concentrations much higher than the proposed acute criterion. However, all such evidence is derived from static tests with nominal heptachlor concentrations, a methodology that tends to underestimate toxicity. There is a greater likelihood that heptachlor could harm salmon or steelhead through lethal effects on aquatic invertebrates; however, little information is available on the effects on invertebrate prey species (NMFS 2014a).

Data on chronic effects of heptachlor are sparse, but suggest that the risk of adverse effect through water-borne exposure is likely to be low. Some studies suggest that tissue concentrations that are possible under the chronic criterion could have sublethal or lethal effects on alevins or fry. Bioaccumulation can occur in salmonids with chronic exposure to heptachlor, and when exposure occurs, this is could occur through the water column, diet and contact with sediments (NMFS 2014a).

Because sediments are likely the primary source of heptachlor, NMFS (2014a) calculated the sediment heptachlor concentration that would result in heptachlor concentrations in the water column at or below the criteria and found the sediment heptachlor concentrations would range from 54 ng/g to 269 ng/g sediment. These levels are higher than the sediment screening guideline of 10 ng/g dw established by the USCOE for in-water disposal of dredged sediment (USCOE 1998, Table 8-1, p. 8-7). These data suggest that heptachlor released from sediments (e.g., during in-water disposal of contaminated sediments) could adversely affect the salmonid prey base at concentrations below the proposed chronic criteria, as the USCOE sediment quality criteria are based primarily on tests with benthic invertebrates.

However, NMFS (2014) noted that the most stringent applicable heptachlor criterion in the action area is the human health (fish consumption) water quality criterion of 0.00021µ/L that is applicable to all waters occupied by listed species and designated critical habitats. The human health fish consumption based criterion is approximately 14 times more restrictive than the chronic criterion and will provide an additional level of protection to listed aquatic species.

Also, given that heptachlor has not been allowed for agricultural use for 38 years (as of the writing of this Opinion) , that the 1988 ban precludes any lawful releases into the environment, that the soil half-life would minimize the potential for any post-use inputs into aquatic habitats, and potential exposure via inwater disturbance/disposal of heptachlor contaminated sediments during USCOE actions (e.g., dredging) are minimized by their sediment screening guideline, the risks to listed species from the proposed heptachlor criteria are discountable.

Therefore, the Service concludes that the approval of the acute and chronic heptachlor criteria established by the Idaho Water Quality Standards is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

## 2.5.19 Lindane Aquatic Life Criteria

The acute and chronic criteria established by the Idaho Water Quality Standards for lindane are 2.0 µg/L and 0.08 µg/L, respectively.

Lindane is an organochlorine insecticide that was first registered in the 1940's and has been used in the United States on a wide variety of fruit and vegetable crops (including seed treatment), ornamentals, tobacco, and Christmas tree plantations. Uses by homeowners include dog dips, household sprays, and shelf paper. It has also been used on commercial food or feed storage areas and containers, and pharmaceutically for treating scabies and lice (EPA 2004b, p. 2; ATSDR 2005, p. 2). Between 1993 and 1998, lindane registrants voluntarily cancelled a large number of lindane uses as long range transport and environmental concerns increased. By 2001 to 2002, all lindane uses were voluntarily cancelled except six lindane seed treatments (EPA 2006, p. 6). On August 2, 2006, EPA announced that the registrants of lindane requested to voluntarily cancel the six lindane seed treatments, thereby eliminating all remaining uses of lindane in the United States (http://www.epa.gov/oppsrrd1/REDs/factsheets/lindane_fs_addendum.htm, accessed September 24, 2014).

Lindane is relatively persistent in the environment and bioconcentrates to some extent in aquatic organisms (EPA 1999a, p. 114). The typical half-life for lindane in soil is 400 days (http://pmep.cce.cornell.edu/profiles/extoxnet/haloxyfop-methylparathion/lindane-ext.html, accessed September 21, 2014). Once released into the environment, lindane is primarily dissipated by volatilization into the air, followed by long-range aerial transport. Lindane has been detected in air, surface water, groundwater, sediment, soil, ice, snowpack, fish, wildlife, and humans (EPA 2004b, p. 12). It has been detected in ambient air, precipitation, and surface water throughout North America, and also has been detected in areas of non-use (for example, the Arctic). Available data also indicate that lindane is expected to adsorb to suspended solids and sediment in water (EPA 2006, p. 10).

NMFS (2014a) states that because there are no registered uses of lindane in the United States, the only sources of lindane will be from repositories of the contaminant that are persistent in sediments. Organisms will accumulate lindane from the water column as well as from direct contact with sediments, or through the diet. However, compared to compounds like DDT and PCB, lindane is less likely to adsorb to, or accumulate in, sediments because the value of the octanol/water partitioning coefficient of lindane ($\log_{10}(K_{ow}) = 3.3$) is relatively low (NMFS 2014a, p. 250-251).

Toxicity of lindane may be altered by a number of factors including temperature, organism life stage, duration of exposure (Johnson and Finley 1980, p. 47; Maund et al. 1992, p. 76), and presence of other chemicals.

The Assessment (EPA 1999a, p. 114) provided no data on the toxicity of lindane on freshwater snails of any species or on the more general category of molluscs. Available information for aquatic invertebrates appears to be limited to Daphnids, sowbugs, scuds, stoneflies, glass shrimp, and chironomids. There was marked variability among invertebrates in susceptibility to lindane toxicity. Daphnids were relatively resistant, and chironomids seem to be among the most sensitive species tested. Among chironomid tests, variability was observed among tests. Lowest observable effect concentrations (LOECs) for *Chironomus riparius*, a freshwater midge, ranged from 0.2 to 1.0 µg/L; the observed adverse effect was reduced growth rate (Taylor et al. 1991, p. 375; Maund et al. 1992, p. 76; Taylor et al. 1993, pp. 148-149). In contrast, LC50s for this species ranged from 6.5 to 235 µg/L (Peither et al. 1996, p. 54) and demonstrate that basing aquatic life criteria on tests that use lethality as the endpoint is likely to miss sublethal adverse effects. In addition, an additional test with *C. riparius* documented a large difference between the no observed effect concentration (NOEC, 1.1 µg/L) and the LOEC (9.9 µg/L) (Taylor et al. 1993, p. 145). However, NMFS (2014a, p. 250) citing as an example a study by Blockwell et al (1998)[24], found that most studies of the chronic effects of lindane exposure on aquatic invertebrates have reported effects occurring at levels that are more than 25 times the proposed criterion of 0.08 µg/L.

For salmonids, NMFS (2014a, p. 248) reported an LC50 value of 1.7 µg/L for brown trout (from Johnson and Finley 1980), indicating that the acute criterion could allow mortality to salmonids. However, for most salmonids and other fish species LC50 values are more than an order of magnitude greater than the proposed acute criterion of 2 µg/L. Johnson and Finley (1980) reported 96-hour LC50 values of 23 µg/L, 27 µg/L, and 32 µg/L, for coho salmon, rainbow trout, and lake trout, respectively, in static exposure tests.

[24] For the amphipod, *Hyallela azteca*, Blockwell *et al*. (1998) reported 240-hour LC50s of 26.9 µg/L and 9.8 µg/L for adults and neonates, respectively.

NMFS (2014a) reported that water column only exposure at the chronic criterion was not likely to have adverse effects on salmonids (see Macek et al. 1976, Mendiola et al. 1981). However NMFS (2014a) did find that when biocentration factors and fish tissue concentrations were assessed, the chronic criterion could result in fish tissue concentrations associated with adverse effects (i.e., 1.2 mg/kg from Macek et al. 1976).

Because sediments are likely a primary source of lindane to the aquatic environment, NMFS (2014a) calculated the lindane sediment concentration that would result in lindane concentrations in the water column at or below the proposed chronic criteria (0.08 µg/L) and found that the chronic aquatic life criterion would be associated with lindane concentrations in sediment ranging between 1 ng/g to 7 ng/g sediment. These values are about an order of magnitude below the sediment screening guideline of 10 ng/g dry wet established by the COE for in-water disposal of dredged sediment (USCOE 1998, Table 8-1, p. 8-7), This suggests that the proposed criterion is reasonably likely not to harm salmonids or impact their prey items.

Also, given that lindane has not been allowed for any use since 2006, that the 2006 voluntary cancellation of registration precludes any lawful releases into the environment, that the soil half-life (400 days) would minimize the potential for any post-use inputs into aquatic habitats, and potential exposure via inwater disturbance/disposal of lindane contaminated sediments during USCOE actions (e.g., dredging) are minimized by their sediment screening guideline, the risks to listed aquatic species from the proposed lindane criteria are discountable.

Therefore, the Service concludes that the approval of the acute and chronic lindane criteria established by the Idaho Water Quality Standards is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

### 2.5.20 Polychlorinated Biphenyl (PCB) Aquatic Life Criterion

The aquatic life criterion for PCBs is 0.014 µg/L (chronic); there is no acute criterion.

PCBs are halogenated aromatic hydrocarbons that do not naturally occur in the environment. In the past PCBs had a wide range of uses, including consumer products (Washington Department of Ecology 2014, p. 2). Because of their non-flammability and good insulating properties, PCBs were also used as coolants and lubricants in transformers, capacitors, and other electrical equipment (ATSDR 2000, p. 2).

Some commercial PCBs produced by the Monsanto Company are known by the trade name “Aroclor” and are identified by a four digit numbering system. For example, “12” was used as the first 2 digits to indicate a PCB mixture and the last two digits identified the percent chlorine by weight of the mixture (e.g., the PCB mixture Aroclor 1254 contains 54 percent chlorine by weight). Aroclor 1254 is one of the most common PCB mixtures that continues to persist as a global pollutant (NMFS 2014a). Their production, processing, and distribution in commerce were banned in the United States in 1979 because evidence indicated harmful accumulation of PCBs in the environment (EPA 1999a, p. 115; ATSDR 2000, p. 2).

PCBs are lipophilic, bioconcentrate, bioaccumulate, and biomagnify up the food chain (Niimi 1996, pp. 121-122). Because of their persistence, PCBs are still commonly detected in environmental samples, at elevated concentrations at some locations, due to their stability and persistence in the environment (EPA 1999a, p. 115). PCBs also cycle easily between air, water, and soil. Once in the air (by evaporation from soil and water) PCBs can be transported long distances and have been found in soil, snow, sea water, sediments, and animals at all levels of the food web at locations distant from the point of release (e.g. the Arctic) (ATSDR 2000, p. 3). The mean half-life of PCBs in riverine sediments was calculated to be 9.5 (±2.2) years, although this can vary by the type of PCB present (ATSDR 2000, p. 533). PCB levels in soils and sediments have decreased in many areas of the United States since its ban in 1979 (ATSDR 2000, p. 532).

Because PCBs are no longer produced in the United States and they have a high affinity for sediment (NMFS 2014a), the major source of this contaminant will not be through point source discharges into surface water, but from repositories of the chemical that remain in sediment. PCB concentrations of concern are typically not found in the water column (NMFS 2014a).

Toxicity of PCBs may be altered by such factors as the proportion of different congeners (i.e., individual chlorinated biphenyl components) present, suspended sediment, species, organism life stage, organism growth rate, lipid content of the species, dose rate, duration of exposure, and presence of other chemicals (Eisler 1986b, pp. 9, 11).

General effects of PCB toxicity to freshwater organisms include reduced growth, reduced egg survival, increased fry deformities, impaired or failed reproduction, induction of hepatic and extra-hepatic drug-metabolizing enzymes, impaired smoltification, depressed ATP-ase and plasma thyroxine in gills, disrupted metabolism, loss of coordination, anemia, enlarged livers, lowered muscle lipid content, and reduced survival in fish (e.g., see Eisler 1986, pp. 14-15; Niimi 1996, pp. 131-136). Reduced growth resulting from exposure to PCBs has also been documented in algae and invertebrates (Eisler 1986, p. 14).

The Assessment (EPA 1999a, pp. 115-116) provided no data on the toxicity of PCBs on freshwater snails of any species or on the more general category of molluscs. Available information for aquatic invertebrates appears to be limited to Daphnids, amphipods, mysids, and zebra mussels. Zebra mussels accumulated PCB 77 from their diet and from surrounding lake sediments; uptake rate followed the descending order of sediment, food, and water (Brieger and Hunter 1993). Lethality in Daphnids and amphipods (LC50 tests) varied greatly (0.3 µg/L to 710 µg/L) and depended on the PCB congener that was being tested, but concentrations were higher than the 0.014 µg/L proposed chronic criterion.

NMFS (2014a) found from the studies they reviewed that water borne PCB concentrations close to, or below, the proposed chronic criterion, in concert with predicted bioaccumulation rates, were projected to impair thyroid function in coho salmon and result in embryo mortality in lake trout. Even though the proposed chronic criterion may result in some effects to listed species, this appears unlikely to occur because the product is banned and there should be no new discharges. Additionally, in the Snake River basin, NMFS (2014a) found studies showing that sediment concentrations are "likely close to, or below the TEC" or threshold effect concentration which is the concentration below which adverse effects on sediment dwelling organisms are not expected.

If discharges of PCBs were to occur, the most stringent controlling ambient water quality criterion applicable in the action area is the human health fish consumption based criterion (0.000045 µg/L) rather than the chronic aquatic life criteria. The fish consumption based criterion is more than100 times more restrictive than the aquatic life criteria and will provide an additional level of protection for listed aquatic species.

Also, given that PCBs have been banned since 1979 (i.e., 35 years as of the writing of this Opinion), that the 1979 ban precludes any lawful releases into the environment, and that the sediment half-life (9.5 years) would minimize the potential for any post-use inputs into aquatic habitats the risks to listed species from the proposed PCBs criterion are discountable.

Therefore, the Service concludes that the approval of the proposed chronic PCBs criterion established by the Idaho Water Quality Standards is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

## 2.5.21 Toxaphene Aquatic Life Criteria

The acute and chronic aquatic life criteria for toxaphene are 0.73 µg/L and 0.0002 µg/L, respectively.

Toxaphene is a trade name for an organochlorine pesticide that is comprised of a mixture of at least 670 chlorinated camphenes (EPA 1999c). Toxaphene was first introduced in 1947 and used extensively as an insecticide in the 1970s after DDT was banned. The pesticide was used primarily in the southern United States to control insects on cotton and livestock, and to kill undesirable fish in lakes (NMFS 2014a).

Because toxaphene poses a risk of significant adverse impacts on humans and the environment, EPA banned all uses of toxaphene in 1990, after canceling the registrations for all uses (with few exceptions) in 1986 (EPA 2005).

Toxaphene is extremely persistent in soil and water, with documented half-times of 9 to 11 years (Eisler and Jacknow 1985, p. 2). Toxaphene bioconcentrates, bioaccumulates, and biomagnifies through the food chain (Eisler and Jacknow 1985, pp. 2, 7). In water it will not appreciably hydrolyze, undergo photolysis, or biodegrade. Degradation is faster under anaerobic than aerobic conditions. Evaporation from the aqueous phase is a significant process for toxaphene dispersion, with a half-life of approximately 6 hours. Once it has volatilized, toxaphene can be carried far from the original release site (NMFS 2014a). Once deposited in surface waters, toxaphene that does not volatize is eventually deposited in sediments (EPA 1999c).

The Assessment (EPA 1999a, p. 119) provided no data on the toxicity of toxaphene on freshwater snails of any species or on the more general category of molluscs. Available information for aquatic invertebrates appears to be limited to Daphnids, scuds, glass shrimp, stoneflies, midges and a freshwater mussel (*Anodonta imbecilis*).

In 96-hour lab tests, LC50s were less than 10µg/L for the most sensitive species of freshwater insects. Values for stoneflies were 1.3 and 2.3 µg/L for two species, 18µg/L and 30µg/L were reported for cranefly and midge, respectively (Eisler and Jacknow 1985, Table 2, p. 14). LC50s for Daphnids ranged from 10-19µg/L, and values for scuds and glass shrimp were 26µg/L and 28µg/L, respectively (Johnson and Finley 1980, p. 77). The freshwater mussel (*Anodonta imbecilis*) appeared relatively tolerant to direct toxicity from toxaphene, with a 48-hour LC50 of 0.74 mg/L (Keller 1993, Table 1, p. 699). All of these LC50s are above the proposed acute and chronic toxaphene criteria.

NMFS (2014a) reported that based on available literature, it appears that invertebrates are less sensitive to toxaphene exposure than fish. The information NMFS reviewed (2014) showed LC50s that were orders of magnitude higher than the chronic criterion (0.0002 µg/L), indicating that chronic effects from long-term exposure to toxaphene are not expected for salmonid invertebrate prey species.

NMFS (2014a) reviewed studies showing that LC50 values for fish are relatively close to the proposed acute criterion, suggesting that acute criterion may result in fish mortality. For example, Macek and McAllister (1970) listed $LC_{50}$ values (and 95% confidence intervals) of 3 µg/L (2 µg/L to 5 µg/L) and 8 µg/L (6 µg/L to 10 µg/L) for the brown trout and coho salmon, respectively. Johnson and Finley (1980) reported 96-hour $LC_{50}$s for toxaphene to coho salmon, rainbow trout, and brown trout of 8.0 µg/L, 10.6 µg/L, and 3.1 µg/L, respectively.

NMFS (2014a) found no studies that documented chronic effects when toxaphene concentrations were below the chronic criterion of 0.0002 µg/L. Mayer and Mehrle (1977) and Mayer et al. (1975) reported that water concentrations of 0.039 µg/L had significant effects on survival and growth in brook trout fry. Other treatments in these studies (0.068 µg/L, 0.14 µg/L, 0.29 µg/L, and 0.5 µg/L) also caused adverse effects in this species. All of these concentration are orders of magnitude higher than the chronic criterion.

Although there is the potential for adverse effects from the acute toxaphene criterion, NMFS (2014a) concluded that toxaphene, "under most circumstances, appears unlikely to cause lethal or sublethal effects from direct exposure at toxaphene concentrations in water equal to or below the proposed acute or chronic criteria."

Also, given that toxaphene has been banned for 24 years (as of the writing of this Opinion), that the 1990 cancellation of registration precludes any lawful releases into the environment, and the soil half-life of 9 to 11 years would minimize the potential for any post-use inputs into aquatic habitats, the risks to listed aquatic species from the proposed toxaphene criteria are discountable.

Therefore, the Service concludes that the approval of the acute and chronic chlordane criteria established by the Idaho Water Quality Standards is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat.

## 2.5.22 Pentachlorophenol (PCP) Aquatic Life Criteria

The acute and chronic PCP aquatic life criteria are 20.0 µg/L and 13.0 µg/L, at pH of 7.8.

The criteria for PCP established by the EPA are pH dependent. In general, the toxicity of PCP increases with decreasing pH. The acute and chronic criteria values are also referred to as the "criterion maximum concentration" (CMC) and "criterion continuous criterion" (CCC) respectively (EPA 1986, 1999a). The following equations are used to determine the freshwater criteria as a function of pH:

CMC (acute) = $\exp^{(1.005 \text{ x pH} - 4.83)}$ (*in µg/L*)
CCC (chronic) = $\exp^{(1.005 \text{ x pH} - 5.29)}$ (*in µg/L*)

For example, at a pH of 7.0, the corresponding criteria are 9.1 µg/L and 6.7 µg/L for acute and chronic exposures, respectively. At a pH of 8.0, the corresponding criteria are 25 µg/L and 18 µg/L for acute and chronic exposures, respectively (NMFS 2014a, p. 207).

PCP is an organochlorine compound that does not naturally occur in the environment. PCP also contains chlorinated dibenzodioxins and chlorinated dibenzofurans (CDDs and CDFs) and hexachlorobenzene (HCB). These are contaminants formed during the manufacture process and represent an ecological risk because of their toxicity and persistence (EPA 2008c, p. 1).

PCP was one of the most widely used biocides in the U.S. prior to regulatory actions to cancel and restrict certain non-wood preservative uses of pentachlorophenol in 1987. It now has no registered residential uses. Prior to 1987, pentachlorophenol was registered for use as anherbicide, defoliant, mossicide, and as a disinfectant, but now all these uses are cancelled (http://www.epa.gov/pesticides/factsheets/chemicals/pentachlorophenol.htm).

PCP is currently classified as a Restricted Use Product (RUP) when used as a heavy duty wood preservative. Currently, all of the PCP produced in the U.S. is utilized in heavy duty wood preservation (i.e., pressure or thermal treated, mainly used to treat utility poles and pole crossarms. All non-pressure and non-thermal treatment uses (i.e., spray uses) were prohibited as of December 31, 2004 (EPA 2008c, pp. 1-2, 28). As a restricted use pesticide, PCP is for sale and use by certified applicators only.

In 2008, the EPA, after conducting risk assessments on PCP, concluded that PCP is eligible for reregistration, provided that risk mitigation measures are adopted and product labels are amended (EPA 2008c, p. iv). To protect aquatic organisms from PCP treated wood used in aquatic and other sensitive habitats, the treatment must use a double vacuum treatment process. In addition, product labels must contain precautionary statements about not discharging effluent containing PCP into lakes, streams, ponds, estuaries, oceans, and other waters unless in compliance with an NPDES permit (EPA 2008c, Table 7).

Several PCP toxicity studies have been conducted with snail species within the same families as listed Snake River aquatic snails, i.e., the family Hydrobiidae (Bliss Rapids snail and Bruneau hot springsnail), family Physidae (Snake River physa), and family Lymnaeidae (Banbury Springs lanx). The Banbury Springs lanx is a freshwater limpet that has yet to be formally described as a species and thus the taxonomic classification of this freshwater limpet is not well documented. USFWS (2006b) considered it to be within the family Lymnaeidae although other freshwater limpets have been classified within the family Planorbidae (Pennak 1978).

Air-breathing snails of the subclass Pulmonata (e.g., the families Physidae, Lymnaeidae, and Planorbidae) have been most widely used for laboratory toxicity tests, and their rapid growth, short generation times, and high reproductive output make them easy to use in toxicity tests, including chronic tests with sensitive, sublethal endpoints. Non-pulmonate snails (formerly included in subclass Prosobranchia, which includes the family Hydrobiidae ) are more taxonomically diverse and their physiology (inability to breathe atmospheric oxygen) and life history (slow growth and low reproductive rate) may make them both subject to endangerment and difficult to culture and test in the laboratory (Besser et al. 2009).

Studies reviewed testing the responses of Hydrobiidae snails to PCP showed toxicity at lower concentrations than those allowed by the chronic water quality criterion for PCP, as follows:

Besser et al. (2009) tested the responses to PCP to the Jackson Lake springsnail (*Pyrgulopsis robusta*, formerly known as the Idaho springsnail (*Prygulopsis idahoensis*)), in parallel with pulmonate pond snails (*Lymnaea stagnalis*). Two paired tests were conducted for 28-days in moderately-hard water with a pH of about 8.4, for which the corresponding chronic PCP criterion concentration is about 23 µg/L.

Reduced growth in springsnails was documented at concentrations less than the chronic criterion concentration of 27 µg/L. Reduced growth, as a 20 percent reduction in wet weight, occurred with the Lymnaea at 19 µg/L in the first of two tests; in the second test, reduced growth as reduced shell diameter only occurred at 118 µg/L. The Jackson Lake springsnail was successfully tested once with PCP, with the onset of growth reductions occurring at 11 µg/L and a 20 percent reduction in shell diameter (EC20) occurring at about 17 µg/L (Besser et al. 2009, Table 15).

Hedkte et al. (1986) tested the acute and chronic sensitivity of multiple taxa to PCP, including the snail *Physa grina*. Acute mortality only occurred at concentrations well above criteria. In chronic exposures, survival was reduced in concentrations as low as 26 µg/L. For the test conditions, pH 7.2, the criterion concentration was lower, 7 µg/L. In Hedkte et al.'s (1986) battery of tests, the cladoceran *Ceriodaphnia reticulata* was the most sensitive of 11 diverse taxa.

Other primary research relevant to effects of PCP at close to criteria concentrations to listed snails included the possibility of genetic damage at concentrations as low at 10 µg/L in zebra mussel and at 100 µg/L in the ramshorn snail *Planorbarius corneus*. Mortality resulted at concentrations of 450 µg/L and higher (Pavlica et al. 2000). For the pH of these 14-day tests (7.6 to 8.3) chronic criterion concentrations would range from 11 to 21 µg/L. However, the concentrations given are the intended exposure concentrations; actual concentrations were not analytically determined. Other primary literature on the potential effects of PCP to snails that was of limited use for this review included experimental pond and stream ecosystems with direct and indirect effects occurring at tested concentrations higher than criteria (Crossland and Wolff 1985; Zischke et al. 1985), or from food-only exposures (Gomot-De Vaufleury 2000). Secondary sources (i.e., other review compilations) indicated several additional acute studies with snails in which adverse effects were only observed at greater than the acute criterion concentrations (EPA 1986; Eisler 1989; EPA 1999a).

Therefore, adverse effects to the listed Snake River aquatic snails and the Bruneau hot springsnail are possible from the proposed chronic PCP water quality criteria.

For the bull trout, no literature reports of effects of pentachlorophonal were located. For other salmonids in the genus *Salvelinus*, the only data noted was a listing in EPA (1986) of a single test of adult brook trout with an LC50 of 138 µg/L PCP at pH 7.89, for which the acute criterion is 22 µg/L. For other salmonids, the proposed acute PCP criterion is close to the level where some acute toxicity will occur. For example Van Leeuwen et al. (1985) determined the 96-hour LC50 to be 18 µg/L at pH 7.2 for early fry of rainbow trout. The acute PCP criterion at pH 7.2 is 11 µg/L. Most studies of chronic effects reported the onset of adverse effects occurred at least slightly above the chronic criterion. For example, Dominguez and Chapman (1984, at p. 741) tested steelhead trout in a 72-day test and found no adverse effects at PCP concentrations of 11 µg/L, which is just above the chronic criterion of 8.6 µg/L for their average test conditions. The lowest effect concentration found for any species was a threshold for reduced growth and energy conversion in sockeye salmon at 1.7 µg/L following 56-day exposures at pH 6.8 (Webb and Brett 1973). The chronic criterion for PCP at pH 6.8 is 4.7 µg/L. These growth effects were subtle, with about a 10 percent impairment associated with the chronic criterion concentration, as estimated from Webb and Brett's (1973) figure 5.

Potential indirect risks to bull trout from PCP exposures at chronic criterion concentrations through reduced forage base appear slight. Cyprinids and other potential fish prey of bull trout, are probably no more sensitive than the bull trout themselves, based upon comparative testing and sensitivity-rankings (Cleveland et al. 1982; EPA 1986; Besser et al. 2005a; Dwyer et al. 2005). In experimental stream ecosystem tests that exceeded chronic criterion concentrations, no profound reductions in the abundances of micro- or macroinvertebrate fauna were detected, although species composition changes were detected (Zischke et al. 1985). For instance, in their lowest PCP exposure, 34-44 µg/L PCP at pH 7.2-8.6, the chronic criterion would have ranged between 7 and 28 µg/L. Changes in benthic communities were noted, such as differences in the timing of snail egg deposition, although fundamental metrics such as invertebrate responses measured were changes in density, community composition and drift were not obviously affected. No change in microinvertebrate (e.g., zooplankton) communities were detected in the lowest treatment. Some ecosystem effects were detected at all treatment levels with fish being the most sensitive animals (Zischke et al. 1985). Together, these results indicate low risk of indirect effects from PCP exposures to bull trout food webs at chronic criterion conditions or below.

Dwyer et al. (2005) tested the relative acute sensitivity of PCP to three threatened sturgeon species and "standard" surrogate toxicity test species. Two of the tested sturgeon were from the same genus as the listed Kootenai River white sturgeon, the Atlantic sturgeon (*Acipenser oxyrhynchus*) and Shortnose sturgeon (*Acipenser brevirostrum*), in addition to Shovelnose sturgeon (*Scaphirhynchus platorynchus*). Highly divergent results were obtained with the LC50 for Atlantic sturgeon at <40 µg/L, shortnose sturgeon at 50 µg/L, and with Shovelnose sturgeon, no LC50 was obtained, which suggests it was not an acutely sensitive species to PCP. For the tested conditions, pH about 8.0, the acute criterion was 25 µg/L. Dwyer et al. (2005) cautioned that although Atlantic and Shortnose sturgeon were usually the most sensitive and second most sensitive of all 20 species they tested, the results should be interpreted with caution because the static test conditions may have contributed to more severe effects than observed in tests with continuous or frequent water replacement.

No reports on chronic exposures of PCP to sturgeon were located.

The review (above) of potential effects to bull trout prey base from PCP is also germane to the types of sturgeon prey organisms potentially used by sturgeon. Similarly, risks of indirect effects from PCP exposures to bull trout food webs at chronic criterion conditions or below appear low.

While both the acute and chronic PCP criteria will likely have some, potentially adverse, effects on listed species or their food sources, the most stringent applicable criterion in waters occupied by listed species in the action area is the human health (fish consumption based) water quality criterion of 6.2 µg/L. This criterion is about twice as low as the chronic criterion of 13.0 µg/L for PCP (at pH 7.8) and would provide an additional level of protection to listed species. For the aquatic snails, it would also provide a greater level of protection, because the fish consumption based criterion of 6.2 µg/L does not increase with pH, and pH tends to be higher in waters inhabited by the listed aquatic snails than the waters than those occupied by bull trout or sturgeon. For example, for the median pH values of 8.2 and 8.4 reported for the Bruneau River at Hot Springs, and the Snake River at Buhl respectively (Hardy et al. 2005), the chronic PCP criterion would be 19 and 23 µg/L, respectively.

Additionally, PCP is not likely to be a component of NPDES discharges. As mentioned above, the main use of PCP is for treating wooden utility poles. The treatment must use a double vacuum process and any discharges of effluent containing PCP into aquatic systems are prohibited. For these reasons, the Service is not expecting any significant exposure of listed aquatic species to PCP (or the manufacturing impurities CDDs, CDFs, and HCB) .

After conducting an environmental risk assessment for PCP, EPA similarly concluded that typical concentrations of pentachlorophenol in terrestrial and aquatic environments from wood treatment uses are not expected to adversely impact terrestrial or aquatic organisms (EPA 2008c, p. 31).

The Service therefore concludes that approval of the acute and chronic water quality criteria for PCP is not likely to adversely affect the listed Snake River aquatic snails, the Bruneau hot springsnail, the bull trout and its critical habitat, and the Kootenai River white sturgeon and its critical habitat; all effects are expected to be insignificant or discountable.

# 2.5.23 Summary of Effects

A summary of the effects of toxic pollutants and the associated listed species evaluated in this Opinion are presented in Table 11.

**Table 11. Summary of effects by toxic pollutant and criteria to listed species and critical habitats addressed in this Opinion.**

| Toxic Pollutant | Snake River Physa | Bliss Rapids Snail | Banbury Springs Lanx | Bruneau Hot Springsnail | Bull Trout/ Critical Habitat | Kootenai River White Sturgeon/ Critical Habitat |
|---|---|---|---|---|---|---|
| **Arsenic** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | LAA | LAA | LAA | LAA | LAA/LAA | LAA/LAA |
| **Copper** acute | LAA | LAA | LAA | LAA | LAA/LAA | LAA/LAA |
| chronic | LAA | LAA | LAA | LAA | LAA/LAA | LAA/LAA |
| **Cyanide** acute | NLAA | NLAA | NLAA | NLAA | LAA/LAA | LAA/LAA |
| chronic | NLAA | NLAA | NLAA | NLAA | LAA/LAA | LAA/LAA |
| **Lead**: acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | LAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **Mercury**: acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | LAA/LAA | LAA/LAA |
| **Selenium**: acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | LAA/LAA | LAA/LAA |
| **Zinc** acute | LAA | LAA | LAA | LAA | LAA/LAA | LAA/LAA |
| chronic | LAA | LAA | LAA | LAA | LAA/LAA | LAA/LAA |
| **Chromium (III)** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |

| **Toxic Pollutant** | **Snake River Physa** | **Bliss Rapids Snail** | **Banbury Springs Lanx** | **Bruneau Hot Springsnail** | **Bull Trout/ Critical Habitat** | **Kootenai River White Sturgeon/ Critical Habitat** |
|---|---|---|---|---|---|---|
| **Chromium (VI)** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | LAA/LAA | LAA/LAA |
| **Nickel** acute | LAA | LAA | LAA | LAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | LAA | LAA | LAA | LAA | LAA/LAA | LAA/LAA |
| **Silver** acute* | NLAA | NLAA | NLAA | NLAA | LAA/LAA | LAA/LAA |
| **Endosulfan** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **Aldrin/ Dieldrin** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **Chlordane** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **DDT** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **Heptachlor** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **Lindane** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **PCBs** chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |

| Toxic Pollutant | Snake River Physa | Bliss Rapids Snail | Banbury Springs Lanx | Bruneau Hot Springsnail | Bull Trout/ Critical Habitat | Kootenai River White Sturgeon/ Critical Habitat |
|---|---|---|---|---|---|---|
| **Toxaphene** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| **PCP** acute | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |
| chronic | NLAA | NLAA | NLAA | NLAA | NLAA/ NLAA | NLAA/ NLAA |

*EPA only proposed an acute criterion for silver. However, the Service evaluated both short- and long-term silver exposure using the single criterion proposed; this evaluation reflects all likely effects on listed species/critical habitat caused by any temporal exposure at the criterion level.

## 2.6 Cumulative Effects

Cumulative effects include the effects of future State, tribal, local or private actions that are reasonably certain to occur in the action area considered in this Opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act.

According to the most recent census, between 2000 and 2013, the population of Idaho increased by 2.8 percent.[25] The Service therefore assumes that future private and state actions will continue within the action area with a slight increase from their current rate. Seventy-one percent of the action area is Federally-owned, which somewhat limits possible cumulative effects from private and state actions. However, private land is often clustered in valley bottoms, adjacent to occupied habitat for ESA-listed species.

A large number of non-Federal activities take place in Idaho that have effects on the listed species and critical habitat considered in this Opinion, as described in the Environmental Baseline section of this document. The effects associated with these activities will continue to threaten the persistence of the listed species and the function of critical habitat considered in this Opinion. Cumulative effects to listed aquatic species and critical habitat are likely to include: reduced streamflows from water diversions for urban, agricultural and other purposes; increased flow fluctuation due to water management for hydroelectric power generation; destruction or degradation of spawning and rearing habitat from logging, grazing, mining, farming and urban

[25] U.S. Census Bureau, State and County Quickfacts. Available http://quickfacts.census.gov/qfd/states/16000.html

development on private and other non-federal lands; degraded water quality as a result of polluted runoff from agriculture, aquaculture, and urban and rural areas; contaminant spills including spills or leaks of pesticides such as pentachlorophenol stored in rusting containers near occupied aquatic habitats (Fisher 2014, *in litt*.); migration barriers that result from dams on private or other non-federal lands (not regulated by the federal government); introduced diseases, resource competition and gene pool dilution as a result private-, tribal- or State-operated hatcheries; mortality as a result of illegal harvest through incidental catch; habitat degradation associated with non-Federal road building and maintenance; and competition, predation and hybridization problems associated with introduction of nonnative species. In addition, because of the predicted population increases described above, the number and scope of human activities that affect these species are likely to increase in the future, thereby increasing the potential adverse effects on listed snails and fish, and their habitats.

These ongoing and increased human activities in the action area are likely to alter the quality and quantity of surface waters of the state of Idaho, and reduce the quality of spring and river habitats of species and critical habitat addressed in this Opinion. As we have noted elsewhere in this document, there is a paucity of reliable information available to assess population trends for the four listed snail species considered herein. There is, however, a plethora of data indicating that snail habitat quality and quantity is degraded and is trending toward increased degradation, thereby increasing the risk of extinction for the Snake River aquatic snails and the Bruneau hot springsnail. Available information indicates that for the critically endangered white sturgeon and 77 percent of the core area populations of the bull trout within the action area are at risk of extirpation.

Of most concern in terms of future impacts to listed aquatic snails is ground water depletion, the introduction of polluted waters into the aquifer, and the associated decline in the quality of spring habitats. These effects are particularly significant for stronghold habitats of listed snails and other aquatic life that are dependent on spring habitats. There is growing evidence that these strongholds are not protected from anthropogenic and/or natural impacts, and that degradation of water quality, especially, is increasing in severity (as previously discussed in baseline). Banbury Springs lanx is dependent on high quality, cold water and its present confinement to spring habitats, most assuredly is the result of the degraded quality of mainstem habitats. This species is most threatened by the downward trend in the quality of spring habitats thereby reducing the likelihood of its survival and recovery as those habitats are altered. The Bliss Rapids snail is associated to varying degrees with spring habitats and is vulnerable to reduced flows and degraded water quality. At this time, Bliss Rapids snail colonies in spring habitats are denser than those in the main channel of the Snake River, likely because of the relatively higher quality of the habitats in the springs. With the trend toward decreasing water quality in spring habitats, these habitat strongholds for this species can reasonably be expected to be progressively impaired. This impairment will affect the distribution and resilience of the species overall, as these strongholds are likely source habitats for colonies in the main channel of the Snake River.

The Snake River physa occurs only in the main stem of the Snake River. It is rarely encountered and poorly understood, though it seems to be dependent upon deeper water than that inhabited by the other snail species. Changed stream dynamics and continued water quality impairment represent ongoing and likely increasing threats to the species. Likewise, the Bliss Rapids snail is found in unimpounded reaches (as well as spring habitats) and is vulnerable to effects from non-

Federal activities on near-shore, shallow waters. Its depressed numbers in the main stem of the Snake River, compared to springs, is one indicator that they are being adversely affected by impaired river conditions.

Continuing depletion of groundwater, and the associated degradation of hot spring habitats, as well as contamination of the underlying aquifer, are the primary concerns in terms of future impacts to the Bruneau hot springsnail. Pumping to obtain groundwater for agricultural irrigation adversely affects snails by lowering the water table and desiccating springs upon which the hot springsnail depends. This species of springsnail is only found at a limited number of locations in the Bruneau valley. Therefore, if pumping continues to lower the water table in this area, the hot spring habitat required by the Bruneau hot springsnail will be progressively reduced in size. It is logical to expect that the numbers of this species and its likelihood of survival will reduce as its habitat is lost. Additionally, should the aquifer that supports these snails become contaminated, additional limiting factors or stressors will be placed on these snails. Due to their limited distribution and the fact that all individuals are supported by the same aquifer, contamination of the aquifer will impact all individuals of the species and increase the likelihood of its extinction.

The concerns regarding future impacts to the bull trout and its habitat, including designated critical habitat, are varied and difficult to summarize. Because this species occurs throughout much of the state of Idaho, and the fact that Idaho waterways are considered strongholds for the species, continuing impacts such as those previously discussed have the potential to be significant. Land use practices that degrade water quality or stream habitat will continue to contribute to declining trends in bull trout distribution and abundance throughout the state of Idaho.

The concerns regarding future impacts to the Kootenai River white sturgeon and its critical habitat are primarily related to water management associated with Libby Dam. However, as urban and rural development continues in northern Idaho, additional impacts related to these land uses will occur and be detrimental to white sturgeon. Land use practices that contribute excess sediment, contaminants (including nutrients and pesticides), or that remove structure and instream vegetation (channelization or streambank armoring) will continue to degrade habitat quality. This distinct population segment has a limited distribution and is only found in the Kootenai River. Continuing impacts from various land use practices can be expected to continue to degrade white sturgeon habitat, result in reduced numbers, and thereby reduce the likelihood of the continued survival and persistence of this distinct population segment.

Non-Federal actions likely to occur in or near surface waters in the action area may also have beneficial effects on listed species and critical habitat addressed in this opinion. Recovery Plans for each of the listed aquatic species addressed in this Opinion identify recovery actions necessary for the conservation of these species. They include implementation of riparian improvement measures and fish habitat restoration projects, for example. However, habitat quality in many of Idaho’s waterways is not likely to improve measurably in the near future because efforts to address water quality problems are likely to be offset by increased human activity in the region. Available information indicates that sediment, chemical, and nutrient pollution input to the river will continue to degrade habitat quality for listed aquatic species (EPA 2002a). These effects are exacerbated in the impounded reaches of the river, and the longer the retention time of those waters, the more severe are the impacts of input of pollutants because of elevated temperatures and altered trophic processes. Timing and levels of flow in the river are affected by a suite of activities, and, at best, the status quo will be maintained with respect to non-Federal activities. Overall, the suite of threats to the listed aquatic species that inhabit Idaho's waterways will increase during the life of the proposed action under consideration in this Opinion.

Climate change is another important factor impacting listed species in the action area. Air temperatures have been warming more rapidly over the Rocky Mountain West compared to other areas of the coterminous U.S. (Rieman and Isaak 2010, p. 3). Data from stream flow gauges in the Snake River watershed in western Wyoming, and southeast and southwest Idaho indicate that spring runoff is occurring between 1 to 3 weeks earlier compared to the early twentieth century (Rieman and Isaak 2010, p. 7). These changes in flow have been attributed to interactions between increasing temperatures (earlier spring snowmelt) and decreasing precipitation (declining snowpack). Global Climate Models project air temperatures in the western U.S. to further increase by 1 to 3°C (1.8 to 5.4°F) by mid-twenty-first century (Rieman and Isaak 2010, p. 5), and predict significant decreases in precipitation for the interior west. Areas in central and southern Idaho within the Snake River watershed are projected to experience moderate to extreme drought in the future (Rieman and Isaak 2010, p. 5). Other effects, such as increased vulnerability to catastrophic wildfires, may occur as climate change alters the structure and distribution of forest and aquatic systems.

As discussed in the Baseline sections for the Snake River physa, Bliss Rapids snail, Banbury Springs lanx, Bruneau hot springsnail, bull trout, bull trout critical habitat, Kootenai River white sturgeon, and Kootenai River white sturgeon critical habitat, climate change is likely to have significant adverse effects to each of these species and habitats.

Although these factors are ongoing to some extent and likely to continue in the future, past occurrence is not a guarantee of a continuing level of activity. That will depend on whether there are economic, administrative, and legal impediments or safeguards in place. Therefore, although the Service finds it likely that the cumulative effects of these activities will have adverse effects commensurate with or greater than those of similar past activities; it is not possible to quantify these effects (NMFS 2014a).

# 2.7 Conclusion

After reviewing the current status and environmental baseline of the Snake River physa, Bliss Rapids snail, Banbury Springs lanx (collectively Snake River snails), Bruneau hot springsnail, bull trout, bull trout critical habitat, Kootenai River white sturgeon, and Kootenai River white sturgeon critical habitat, the effects of the proposed action and cumulative effects, it is the Service's biological opinion that the action, as proposed, is likely to jeopardize these species, and is likely to adversely modify the above critical habitats (see Table 12 below).

**Table 12. Summary of jeopardy and adverse modification determinations by species/critical habitat and toxic pollutant criteria (NJ =No Jeopardy, NAM = No Adverse Modification, J = Jeopardy, AM = Adverse Modification).**

| Toxic Pollutant | | Snake River Physa | Bliss Rapids Snail | Banbury Springs Lanx | Bruneau Hot Springsnail | Bull Trout/Critical Habitat | Kootenai River White Sturgeon/ Critical Habitat |
|---|---|---|---|---|---|---|---|
| **Arsenic** | acute | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |
| | chronic | J | J | J | J | J/AM | J/AM |
| **Copper** | acute | J | J | J | J | J/AM | J/AM |
| | chronic | J | J | J | J | J/AM | J/AM |
| **Cyanide** | acute | NJ | NJ | NJ | NJ | J/AM | J/AM |
| | chronic | NJ | NJ | NJ | NJ | J/AM | J/AM |
| **Lead**: | acute | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |
| | chronic | NJ | NJ | J | NJ | NJ/NAM | NJ/NAM |
| **Mercury**: | acute | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |
| | chronic | NJ | NJ | NJ | NJ | J/AM | J/AM |
| **Selenium**: | acute | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |
| | chronic | NJ | NJ | NJ | NJ | J/AM | J/AM |
| **Zinc** | acute | NJ | NJ | NJ | NJ | J/AM | J/AM |
| | chronic | NJ | NJ | NJ | NJ | J/AM | J/AM |
| **Chromium (III)** | acute | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |
| | chronic | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |
| **Chromium (VI )** | acute | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |
| | chronic | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |

| Toxic Pollutant | | Snake River Physa | Bliss Rapids Snail | Banbury Springs Lanx | Bruneau Hot Springsnail | Bull Trout/Critical Habitat | Kootenai River White Sturgeon/ Critical Habitat |
|---|---|---|---|---|---|---|---|
| **Nickel** | acute | J | J | J | J | NJ/NAM | NJ/NAM |
| | chronic | J | J | J | J | NJ/NAM | NJ/NAM |
| **Silver** | acute* | NJ | NJ | NJ | NJ | NJ/NAM | NJ/NAM |

*EPA only proposed an acute criterion for silver. However, the Service evaluated both short- and long-term silver exposure using the single criterion proposed; this evaluation reflects all likely effects on listed species/critical habitat caused by any temporal exposure at the criterion level."

### Rationale for No Jeopardy and No Adverse Modification Determinations

With the exception of zinc, all of the no jeopardy determinations for the listed snails listed in Table 12 were made on the basis of NLAA determinations discussed in the *Effects of the Proposed Action* section above. No critical habitat has been designated for the listed snails considered herein, therefore, none will be affected.

With respect to the proposed acute and chronic zinc criteria, although some research indicates that the criteria may adversely affect some algae species that snails feed upon, evidence indicates that (1) there is an abundance of algae in the Snake River, (2) snails such as the Bliss Rapids snail are indiscriminate biofilm grazers, and (3) the Bruneau hot springsnail is less influenced by food resources than water temperature. For these reasons, the Service is not expecting significant adverse effects to be caused by the zinc criteria to the listed Snake River snails and the Bruneau hot springsnail.

With the exceptions discussed below, all of the no jeopardy and no adverse modification determinations for the bull trout, bull trout critical habitat, Kootenai River white sturgeon, and Kootenai River white sturgeon critical habitat listed in Table 12 were made on the basis of NLAA determinations discussed in *Effects of the Proposed Action* section above.

With respect to the proposed chronic criterion for chromium (VI), although some research studies on salmonids indicate that adverse effects to juvenile bull trout and juvenile Kootenai River white sturgeon may occur through reduced growth and potentially reduced overwinter survival, other studies show that such effects may occur only at chromium (VI) concentrations well above the proposed chronic criterion of 11 μg/L. For these reasons, the Service does not expect significant adverse effects to be caused by the proposed chronic criterion for chromium (VI) to the bull trout and its critical habitat and to the Kootenai River white sturgeon and its critical habitat. These findings also align with those made by NMFS (2014a) for the proposed chronic criterion for chromium (VI).

With respect to the proposed chronic criterion for nickel, assuming that *Lymnaeid* snails are not an important component of bull trout and Kootenai River white sturgeon prey items (as discussed above in the *Effects of the Proposed Action* section), the potential impacts of this criterion to bull trout and sturgeon prey species appear limited to amphipods, particularly *Hyalella*. Given that the bull trout and the sturgeon eat a variety of prey items and are known piscivores, the Service does not expect significant adverse effects to the bull trout and sturgeon prey base from any reduction in amphipod abundance. For these reasons, the Service does not expect significant adverse effects to be caused by the proposed chronic criterion for nickel to the bull trout and its critical habitat and to the Kootenai River white sturgeon and its critical habitat.

With respect to the proposed silver criterion, although some adverse effects to prey species of the bull trout and sturgeon may occur from their exposure to silver at the proposed criteria concentrations, bull trout and sturgeon eat a variety of prey items. In addition, the form of silver in natural waters is much less toxic than ionic silver used in most laboratory exposures. For these reasons, the Service does not expect significant adverse effects to the bull trout and its critical habitat and to the Kootenai River white sturgeon and its critical habitat to be caused by the proposed criterion for silver.

**Rationale for the Jeopardy and Adverse Modification Determinations**

See the following discussion.

# 2.7.1 Arsenic

### Snake River Physa, Bliss Rapids Snail, Banbury Springs Lanx, and the Bruneau Hot Springsnail

Arsenic has been shown to adversely affect natural algal communities with profound impairment of photosynthesis (50 percent impairment) at arsenic concentrations as low as 22 µg/L (Knauer et al. 1999); the proposed chronic criterion is 150 µg/L. Because algae are a primary food source for listed snails, the proposed chronic criterion for arsenic is likely to significantly reduce the availability of an important food resource for these listed snails throughout their respective ranges, which, in turn, is likely to reduce their reproduction, numbers, and distribution in the wild to an extent that appreciably reduces the likelihood of both the survival and recovery of these species.

### Bull Trout

At proposed criteria concentrations, arsenic poses significant health risks to salmonids including reduced growth and survival, organ damage, and behavioral modifications. . Bioaccumulation of arsenic in invertebrate prey organisms to concentrations harmful to salmonids appears to occur in streams with dissolved arsenic concentrations below the proposed chronic criterion of 150 µg/L (see Section 2.5.2.2). Inorganic arsenic in the diet of the rainbow trout is associated with reduced growth, organ damage and other physiological effects (Cockell 1991, p. 518; Hansen et al. 2004, pp. 1902-1910; Erickson et al. 2010, pp. 122,123).

Arsenic accumulation in sediments, biofilms, and aquatic insects in freshwater food webs has been implicated as the cause of reduced growth and tissue damage in exposed salmonids. On that basis, the proposed chronic criterion for arsenic is likely to adversely affect the bull trout, as described in the preceding paragraph, within 44 percent of the streams and 34 percent of the lakes and reservoirs within its current range. The scale and magnitude of these effects are likely to impede (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations in a major portion of its range.

### Bull Trout Critical Habitat

The proposed chronic criterion for arsenic is likely to create habitat conditions within 44 percent of the streams and 34 percent of the lakes and reservoirs designated as critical habitat for the bull trout. These habitat conditions are likely to cause reduced growth and survival, organ damage, and behavioral modifications of exposed bull trout. Bioaccumulation of arsenic in invertebrate prey organisms to concentrations harmful to salmonids appears to occur in streams with dissolved arsenic concentrations below the proposed chronic criteria. Inorganic arsenic in the diet of the rainbow trout is associated with reduced growth, organ damage and other physiological effects (Cockell 1991, p. 518; Hansen et al. 2004, pp. 1902-1910; Erickson et al. 2010, pp. 122,123). Arsenic accumulation in sediments, biofilms, and aquatic insects in freshwater food webs has been implicated as the cause of reduced growth and tissue damage in exposed salmonids. The scale and magnitude of these effects are likely to appreciably impair the capability of the critical habitat to provide its intended recovery support function (persistent core area populations of the bull trout) within a major portion of designated critical habitat.

### Kootenai River White Sturgeon

Based on adverse effects observed in salmonids at concentrations below the proposed chronic criteria, we conclude that the proposed chronic aquatic life criterion for arsenic is also likely to adversely affect the white sturgeon by causing altered feeding behavior, and reduced body weight, prey availability, reproductive success, and survival within 39 percent of its range. The scale and magnitude of these effects are likely to impede natural reproduction and achievement of a stable or increasing sturgeon population within a major portion of its range.

### Kootenai River White Sturgeon Critical Habitat

Based on adverse effects observed in salmonids at concentrations below the proposed chronic criteria, we conclude that the proposed chronic aquatic life criterion for arsenic is also likely to create habitat conditions within all designated Kootenai River white sturgeon critical habitat that are likely to cause altered feeding behavior, and reduced body weight, prey availability, reproductive success, and survival of exposed sturgeon. The scale and magnitude of these effects are likely to impede natural reproduction and achievement of a stable or increasing sturgeon population within the entire range of its designated critical habitat. On that basis, implementation of the proposed chronic criterion for arsenic is likely to appreciably impair the recovery support function of the critical habitat.

## 2.7.2 Copper

### Snake River Physa, Bliss Rapids Snail, Banbury Springs Lanx, and the Bruneau Hot Springsnail

Exposure to copper at concentrations and durations allowed by the proposed acute and chronic criteria are likely to have severe adverse effects to the above snail species throughout their respective ranges. These effects include mortality, loss of chemoreception (so that the snails are no longer attracted to food), feeding inhibition, reduced growth and reduced reproductive output. The scale and magnitude of these impacts are likely to appreciably reduce the likelihood of both the survival and recovery of these species by reducing their reproduction, numbers, and distribution in the wild.

### Bull Trout

Implementation of the proposed copper criteria will create habitat conditions within 44 percent of the streams and 35 percent of the lakes and reservoirs occupied by the bull trout rangewide that are likely to create migration barriers, disrupt normal movement behavior, adversely affect growth of juvenile bull trout and impair chemoreception and related functions in all life stages of the bull trout, decrease bull trout prey abundance, and reduce water quality. The scale and magnitude of these effects are likely to appreciably impair the capability of affected bull trout to survive and reproduce. The magnitude of these impacts are likely to impede (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations throughout a major portion of its range.

### Bull Trout Critical Habitat

Implementation of the proposed copper criteria will create habitat conditions within 44 percent of the streams and 35 percent of the lakes and reservoirs designated as bull trout critical habitat that are likely to create migration barriers, disrupt normal movement behavior, adversely affect growth of juvenile bull trout and impair chemoreception and related functions in all life stages of the bull trout, decrease bull trout prey abundance, and reduce water quality. The scale and magnitude of these effects are likely to appreciably impair the capability of the critical habitat to provide its intended recovery support function (persistent core area populations of the bull trout) over a major portion of the range of designated critical habitat for the bull trout.

### Kootenai River White Sturgeon

White sturgeon in the Columbia River, inclusive of the Kootenai River DPS, are highly susceptible to copper toxicity, to the point that white sturgeon may be the most copper sensitive freshwater fish species tested to date. Implementation of the proposed copper criteria are likely to create habitat conditions that kill early-life stages of the sturgeon, and that cause a loss of equilibrium and mobility by other life stages. These impacts are likely to reduce reproduction and numbers of the Kootenai River white sturgeon within 39 percent of its range. Given the scale and magnitude of anticipated effects, implementation of the proposed copper criteria are likely to impede natural reproduction and achievement of a stable or increasing population of the Kootenai River white sturgeon within a major portion of its range.

### Kootenai River White Sturgeon Critical Habitat

Implementation of the proposed copper criteria are likely to create habitat conditions within the entire area designated as critical habitat for the Kootenai River white sturgeon. As discussed above, Columbia River white sturgeon, inclusive of the Kootenai River DPS, are highly susceptible to copper toxicity, to the point that white sturgeon may be the most copper sensitive freshwater fish species tested to date. Implementation of the proposed copper criteria are likely to create habitat conditions within critical habitat that are likely to kill early-life stages of the sturgeon, and that are likely to cause a loss of equilibrium and mobility by other life stages. These impacts are likely to reduce the reproduction and numbers of the Kootenai River white sturgeon within the designated critical habitat. Given the scale and magnitude of anticipated effects, implementation of the proposed copper criteria are likely to appreciably impair the recovery support function (natural reproduction and achievement of a stable or increasing sturgeon population) of designated critical habitat for the Kootenai River white sturgeon.

## 2.7.3 Cyanide

**Bull Trout**

The proposed acute criterion for cyanide (22 µg/L) is likely to cause mortality of exposed bull trout; an only slightly higher concentration of cyanide at 27 µg/L killed 50 percent of exposed brook trout.

Data on the long-term exposure effects of cyanide on the brook trout and the rainbow trout show reduced egg production for the brook trout, and reduced growth and swimming performance for rainbow trout at cyanide concentrations at or below the proposed chronic criterion.

The above effects are likely to occur within 44 percent of the streams and 34 percent of the lakes and reservoirs occupied by the bull trout within its range. The scale and magnitude of these effects are likely to impede or preclude (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations within a significant portion of its range.

**Bull Trout Critical Habitat**

The proposed criteria for cyanide are likely to create habitat conditions that impair or preclude the capability of the critical habitat to provide for the normal reproduction, growth, movement, and survival of the bull trout within approximately 44 percent of the streams and 35 percent of the lakes and reservoirs designated range-wide as critical habitat. On that basis, implementation of the proposed criteria for cyanide are likely to appreciably impair or preclude the recovery support function (persistent core area populations of the bull trout) of critical habitat within a major portion of the designated area.

**Kootenai River White Sturgeon**

Implementation of the proposed criteria for cyanide is likely to cause mortality, reduced swimming performance, reduced growth, and reduced egg production of exposed individuals within 39 percent of the sturgeon's range. Similar effects are expected to exposed individuals of fish species that sturgeon prey on. These impacts are likely to reduce reproduction and numbers of the Kootenai River white sturgeon within 39 percent of its range. Given the scale and magnitude of anticipated effects, implementation of the proposed criteria for cyanide are likely to impede natural reproduction and achievement of a stable or increasing sturgeon population within a major portion of its range.

**Kootenai River White Sturgeon Critical Habitat**

Implementation of the proposed criteria for cyanide is likely to create habitat conditions within the entire area of designated critical habitat for the Kootenai River white sturgeon that cause mortality, reduced swimming performance, reduced growth, and reduced egg production of exposed individuals of the sturgeon. Similar effects are expected to exposed individuals of fish species that sturgeon prey on. The impacts of these altered habitat conditions are likely to reduce the reproduction and numbers of the Kootenai River white sturgeon within the critical habitat.

Given the scale and magnitude of anticipated effects, implementation of the proposed criteria for cyanide are likely to appreciably impair the recovery support function (natural reproduction and achievement of a stable or increasing sturgeon population) of designated critical habitat for the Kootenai River white sturgeon.

## 2.7.4 Lead

**Banbury Springs Lanx**

Due to the extraordinary sensitivity of snails in the genus *Lymnaea* or family Lymnaeidae to lead toxicity, significant adverse effects in the form of reduced growth and egg production are likely to be caused by implementation of the proposed chronic lead criterion to the pulmonate Banbury Springs lanx, but not the pulmonate Snake River physa, which is not in the Family Lymnaeidae. The effects to the lanx are likely to occur throughout its range and are likely to cause reductions in the reproduction and numbers of this species.

## 2.7.5 Mercury

**Bull Trout**

Available information indicates that mercury would be expected to bioaccumulate to concentrations exceeding 0.3 mg/kg ww in the bull trout and other piscivorous fish in waters with waterborne mercury concentrations much lower than the proposed 12 ng/L chronic criterion concentration. Tissue concentrations of mercury near 0.3 mg/kg ww are considered a threshold for reproductive or neurologic harm to the bull trout. On that basis, implementation of the proposed chronic criterion for mercury is likely to cause cell and tissue damage, and reduced growth and reproduction to exposed bull trout within 44 percent of streams and 34 percent of lakes and reservoirs occupied within its range. Such effects are likely to impede or preclude (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations within a significant portion of its range.

**Bull Trout Critical Habitat**

Implementation of the proposed chronic criterion for mercury is likely to create habitat conditions within 44 percent of streams and 34 percent of lakes and reservoirs designated as critical habitat for the bull trout that are likely to impair the normal growth, behavior, and reproduction of the bull trout. Available information indicates that mercury would be expected to bioaccumulate to concentrations exceeding 0.3 mg/kg ww in the bull trout and other piscivorous fish in waters with waterborne mercury concentrations much lower than the proposed 12 ng/L chronic criterion concentration. Tissue concentrations of mercury near 0.3 mg/kg ww are considered a threshold for reproductive or neurologic harm to the bull trout and other salmonid species on which it preys. On that basis, implementation of the proposed chronic criterion for mercury is likely to create habitat conditions that cause cell and tissue damage, reduced growth and reproduction of exposed bull trout, and reduced availability of salmonid prey species within 44 percent of streams and 34 percent of lakes and reservoirs designated as critical habitat for the bull trout. Such effects are likely to impair the recovery support function (persistent core area populations of the bull trout) of bull trout critical habitat throughout a major portion of the designated critical habitat area.

### Kootenai River White Sturgeon

White sturgeon in the lower Columbia River exhibited reduced reproductive potential when exposed to mercury at a mean water concentration of 0.71 ng/L; the proposed chronic criterion for mercury is 12 ng/L. On that basis, the proposed chronic life criterion for mercury would allow water concentrations in the Kootenai River about 16X higher than those known to reduce the reproductive potential of the white sturgeon. This impact would occur within 39 percent of the range of the Kootenai River white sturgeon DPS. On that basis, this impact is likely to appreciably reduce the reproduction and numbers of the sturgeon to an extent that reduces the likelihood of its survival and recovery.

### Kootenai River White Sturgeon Critical Habitat

Implementation of the proposed chronic life criterion for mercury is likely to create habitat conditions within all of the designated critical habitat for the sturgeon that are likely to reduce the reproductive potential of the sturgeon. As discussed above, white sturgeon in the lower Columbia River exhibited reduced reproductive potential when exposed to mercury at a mean water concentration of 0.71 ng/L; the proposed chronic criterion for mercury is 12 ng/L. On that basis, the proposed chronic life criterion for mercury would allow water concentrations in the critical habitat about 16X higher than those known to reduce the reproductive potential of the white sturgeon. On that basis, this impact is likely to appreciably impair the intended recovery support function (natural reproduction and an increased population of the sturgeon) of sturgeon critical habitat.

## 2.7.6 Selenium

### Bull Trout

Implementation of the proposed chronic criterion for selenium (5 µg/L) is likely to create habitat conditions that cause reproductive failure in the bull trout due to maternal transfer of selenium resulting in embryo toxicity and teratogenicity, and reduced bull trout prey abundance within 44 percent of the streams and 34 percent of the lakes and reservoirs occupied by the bull trout within its range. Such effects are likely to impede or preclude (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, and (3) achieving stable/increasing trends in bull trout populations within a significant portion of its range by imparing the normal reproduction, growth, and survival of individual bull trout.

### Bull Trout Critical Habitat

Implementation of the proposed chronic criterion for selenium is likely to create habitat conditions within 44 percent of streams and 34 percent of lakes and reservoirs designated as critical habitat for the bull trout that are likely to impair the normal growth, behavior, and reproduction of the bull trout. Assuming bull trout are affected in a similar manner as other salmonids, selenium concentrations in critical habitat at the proposed chronic criteria level would impair the normal reproduction, growth, and survival of individual bull trout. Such effects are likely to impair the recovery support function (persistent core area populations of the bull trout) of bull trout critical habitat throughout a major portion of the designated critical habitat area.

**Kootenai River White Sturgeon**

Based on effects observed in juvenile white sturgeon and tissue concentrations occurring in adult white sturgeon, the proposed chronic aquatic life criterion for selenium is likely to adversely affect sturgeon growth and reproduction.

The proposed chronic criterion level for selenium is also likely to indirectly affect the white sturgeon through reduced prey availability, or elevated sediment concentrations that affect sturgeon prey, and sturgeon growth and reproduction. Selenium concentrations at the proposed chronic criterion level in water may result in reproductive failure in the white sturgeon. Lemly (1993a) developed toxic effects thresholds for selenium in fish and wildlife that indicate reproductive failure in fish and wildlife is likely to occur at aquatic concentrations of 2 µg/L of inorganic selenium or less than 1 µg/L of organic selenium.

The above impacts would occur within 39 percent of the range of the Kootenai River white sturgeon DPS. On that basis, this impact is likely to appreciably reduce the reproduction and numbers of the sturgeon to an extent that reduces the likelihood of its survival and recovery at the DPS scale.

**Kootenai River White Sturgeon Critical Habitat**

Implementation of the proposed chronic life criterion for selenium is likely to create habitat conditions within all of the designated critical habitat for the sturgeon that are likely to reduce the growth and reproductive potential of the sturgeon. On that basis, this impact is likely to appreciably impair the intended recovery support function (natural reproduction and an increased population of the sturgeon) of sturgeon critical habitat.

# 2.7.7 Zinc

**Bull Trout**

The proposed aquatic life criteria for zinc are likely to cause mortality of juvenile bull trout and reduce bull trout prey abundance within 44 percent of streams and 34 percent of lakes and reservoirs occupied within its range. These effects will impair the capability of 35 bull trout core areas to persist within the action area, or approximately 30 percent of the core areas within the coterminous distribution of the bull trout. The scale and magnitude of these effects will impede or preclude (1) maintaining/increasing the current distribution of the bull trout, (2) maintaining/increasing the current abundance of the bull trout, (3) achieving stable/increasing trends in bull trout populations, and (4) the capability of occupied habitat to provide for the normal reproduction, growth, and survival of individual bull trout within a significant portion of its range.

**Bull Trout Critical Habitat**

The proposed aquatic life criteria for zinc are likely to create habitat conditions within 44 percent of streams and 34 percent of lakes and reservoirs designated as critical habitat for the bull trout that are likely to cause mortality of juvenile bull trout and reduce bull trout prey abundance. These degraded habitat conditions are likely to impair the capability of 35 bull trout core areas to persist within the action area, or approximately 30 percent of the core areas within the coterminous distribution of the bull trout. The scale and magnitude of these impacts is likely to appreciably impair the capability of the critical habitat to provide its intended recovery support function (persistent core area populations of the bull trout) over a major portion of the designated critical habitat for the bull trout by creating habitat conditions likely to impair the normal reproduction, growth, and survival of individual bull trout.

**Kootenai River White Sturgeon**

The proposed aquatic life criteria for zinc are likely to cause mortality of sturgeon, sub-lethal effects that alter normal sturgeon behavior to an extent that reduces the survival and reproduction of individual sturgeon, and are likely to cause reductions in sturgeon prey species. These impacts are likely to occur within 39 percent of the range of the listed DPS of the white sturgeon. The scope and magnitude of these effects are likely to impair the achievement of a stabile or increasing population through natural reproduction in the wild and through the survival of captive-reared juvenile sturgeon released on the Kootenai River in Idaho.

**Kootenai River White Sturgeon Critical Habitat**

The proposed aquatic life criteria for zinc are likely to create habitat conditions within sturgeon critical habitat that are likely to cause mortality of sturgeon, sub-lethal effects that alter normal sturgeon behavior to an extent that reduces the survival and reproduction of individual sturgeon, and are likely to cause reductions in sturgeon prey species. These impacts are likely to occur within all of the designated critical habitat for the sturgeon. On that basis, this impact is likely to appreciably impair the intended recovery support function (natural reproduction and an increased population of the sturgeon) of sturgeon critical habitat.

## 2.7.8 Nickel

**Snake River Physa, Bliss Rapids Snail, Banbury Springs Lanx, and the Bruneau Hot Springsnail**

The proposed acute and chronic aquatic life criterion for nickel are likely to result in mortality to the Snake River physa, the Bliss Rapids snail, and the Bruneau hot springsnail and affect the reproduction, numbers, and distribution of these snails at the rangewide scale.

The proposed acute and chronic aquatic life criteria for nickel are likely to create habitat conditions that cause ionoregulatory disruption and cellular damage oxidative stress (Pyle and Couture 2011) and mortality to the Banbury Springs lanx. These effects are likely to have lethal and sub-lethal impacts affecting the reproduction, numbers, and distribution of the lanx at the rangewide scale.

# 2.8 Reasonable and Prudent Alternatives

Regulations (50 CFR 402.02) implementing section 7 of the Act define reasonable and prudent alternatives (RPAs) as alternative actions, identified during formal consultation, that (1) can be implemented in a manner consistent with the intended purpose of the proposed Federal action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) would, the Service believes, avoid the likelihood of the Federal action jeopardizing the continued existence of listed species or destroying or adversely modifying critical habitat.

The EPA's authorities include the responsibility to review and approve or disapprove state revisions of their water quality standards; states are to review their water quality standards, at least once every 3 years (40 CFR sections 131.20 through 131.21). If EPA disapproves a state's new or revised water quality criteria and the state does not adopt specified changes, the EPA Administrator has the responsibility and authority to promptly propose and promulgate such criteria (40 CFR section 131.22). The water quality standards considered in this action are implemented, in part, through wastewater discharge permits, administered by EPA through the National Pollutant Discharge Elimination System (NPDES). Monitoring, including biological monitoring, may be required of dischargers as part of their permit conditions (40 CFR 122.48). When the ESA is applicable and requires consideration or adoption of particular permit conditions, those requirements must be followed (40 CFR 122.49).

The RPAs described here are expected to be incorporated into NPDES permits when they are issued or renewed. At present, NPDES permits are issued by EPA for Idaho. The regulations for administering NPDES permits (40 CFR 122.49) state that when the ESA is applicable and requires consideration or adoption of particular permit conditions, those requirements must be followed. It is the Service's understanding that the Idaho Department of Environmental Quality (DEQ) intends on submitting an application to EPA to obtain NPDES permitting authority. If DEQ becomes the NPDES permitting authority in Idaho, EPA will retain oversight authority and may review and under certain conditions, object to the issuance of a State administered permit. Respective agency responsibilities pertaining to the Clean Water Act (CWA) and the Endangered Species Act (ESA) are published in an MOA between EPA, the Service, and NMFS (Federal Register Vol. 66, No. 36, Feb 22, 2001, pp. 11202-11217).

We recognize that although the ESA section 7 consultation process is strictly relevant only to federal interagency coordination and cooperation, the national implementation of the Clean Water Act goals and responsibilities involve partnerships between EPA, the states, and authorized tribes. In Idaho, many of EPA's authorities and responsibilities under the CWA and implementing regulations in turn rely on the application and interpretation of the Idaho DEQ's water quality standards. Authoritative interpretations of the Idaho DEQ water quality standards come from the Idaho DEQ. However, our read of the plain language of their regulations is that components of Idaho's water quality standards that relate to their ability to implement the RPA's named here are matters of discretionary judgment on the part of the IDEQ. In particular, some RPA's relate to mixing zone restrictions. Whether mixing zones for point source permits are authorized for discharges is determined by the Idaho DEQ on a case-by-case basis (IDAPA 58.01.02.060, or for shorthand, §060). As protection of threatened or endangered species are not specifically mentioned in Idaho DEQ's mixing zone rules, whether mixing zones can be limited to protect ESA listed species depends upon Idaho DEQ's interpretation of other relevant language.

Two relevant portions of the Idaho Mixing Zone Policy are first, §60.02.h, "*the mixing zone shall not include more than twenty-five percent (25 %) of the low flow design discharge conditions* …;" and second, §60.02.i., where "*the Department may authorize a mixing zone that varies from the limits in Subsection 060.01.h. if it is established that, (i.) smaller mixing zone is needed to avoid an unreasonable interference with, or danger to, beneficial uses*…" Similarly, a mixing zone could exceed the twenty-five percent (25%) of the low flow design discharge conditions constraint, if "*a larger mixing zone is needed by the discharger and does not cause an unreasonable interference with, or danger to, beneficial uses" (§60.02.ii.).* Section 060.01.h coupled with 060.01.i allows varying from starting point restrictions on size in order to "avoid an unreasonable interference with, or danger to, beneficial uses as described in subsection 060.01.d. The latter subsection at romanette i. calls out "*Impairment to the integrity of the aquatic community, including interfering with successful spawning, egg incubation, rearing, or passage of aquatic life.*"

Therefore it is clear that Idaho DEQ has authority to constrain mixing zones if needed to protect beneficial uses. However, protection of ESA listed species is not a beneficial use per se under the Idaho regulations. While the regulatory definition of "*beneficial use*" of water in the Idaho DEQ definitions does not even mention aquatic life (§010.08), elsewhere aquatic life beneficial uses are described by thermal classifications. For instance, the "cold water aquatic life" use designation (§101.01.a), requires "*water quality appropriate for the protection and maintenance of a viable aquatic life community for cold water species.*" A viable aquatic life community is not specifically defined in regulation, although §054 describes factors to consider using the Department's "Water Body Assessment Guidance" to determine "whether a healthy, balanced biological community is present" when assessing beneficial use support status. The most recent Water Body Assessment Guidance ("WBAG," Grafe et al. 2002) describes procedures for considering aquatic benthic macroinvertebrate, fish community, and stream physical habitat features to assess whether community-based aquatic life beneficial uses are met. The procedures are not species-specific, although communities are comprised of interacting populations of species, so protection of communities generally implies protection of the species within. However, in section 1.3 "*How to use this document*," the assessor is advised that the guidance does not cover every eventuality and that judgement and deviation from the strict methods may be needed in some situations. The WBAG provides "*guidance includes information on DEQ policies, assumptions, and analytical methods. However, the document does not present a rigid structure limiting flexibility for unique situations or preclude the use of sound scientific judgment."*

Therefore, because where present, threatened or endangered species are part of the natural structure of aquatic communities, we believe that EPA and DEQ have certain authority and discretion to further constrain authorized discharges to avoid danger to a vulnerable component of the natural aquatic communities, in situations when they believe it to be appropriate.

Five of the following eight RPAs to avoid jeopardy and adverse modification of critical habitat for listed species other than snails are the same as those described by NMFS (2014a) in their Opinion on this action. The other three RPAs are unique to this Opinion but follow a similar approach to that used by NMFS. That approach involves providing a specific timeframe for EPA to develop revised aquatic criteria that are likely to avoid jeopardy or adverse modification of critical habitat. While these criteria are being developed, either the Human Health Criteria would be used for that purpose, as applicable, or discretionary restrictions would be applied on effluent volume-related permit actions. The RPAs for listed snail species also consider their limited mobility and specific protections are included to protect occupied habitat.

For the bull trout and the Kootenai River white sturgeon (and their critical habitats), the Service has identified a separate RPA for each of six inorganic metals: arsenic, copper, cyanide, mercury, selenium, and zinc. The RPAs for arsenic, copper, cyanide, mercury, and selenium are the same as those described by NMFS (2014a) for salmon and steelhead because we determined that the terms of the RPA are likely to avoid jeopardy and adverse modification of critical habitat for our listed species because they were developed in coordination with EPA, and because consistency of content in the RPAs between the Services will facilitate efficient and effective implementation by EPA. For zinc, a RPA addressing the bull trout and the sturgeon is similar to the RPA for copper in the NMFS (2014a) Opinion.

For the listed snails considered herein, the Service has identified RPAs to address arsenic, copper, lead, zinc, and nickel. The RPA for arsenic is the same as that described by NMFS (2014a) for salmon and steelhead in their Opinion on this action because the RPA is likely to avoid jeopardy of listed snails, because it was developed in coordination with EPA, and because consistency in the content of the RPA between the Services will facilitate efficient and effective implementation by EPA. The RPAs described below for copper, lead, zinc, and nickel are new.

In determining the time frame for implementing the RPAs in this Opinion the Service recognizes that, promulgation of rules under either the state or Federal process will require a minimum of 2 years to complete. For most water quality standards the state of Idaho will likely take the lead and promulgate state rules that require approval by the Idaho Board of Environmental Quality. Additionally, before becoming effective the rules will be reviewed by the Idaho Legislature. Finally, EPA approval of the new rules must also occur. Based on this process we have assumed that the soonest new rules can be completed is 2 years and have used 2 years for the implementation time frame for the RPAs that will not require additional analysis to derive new criteria (i.e., hardness floor, 2007 BLM copper criteria) (see Table 13).

For the other RPAs, EPA and/or the State will likely require additional time to conduct the analyses necessary to support new criteria. These RPAs therefore provide a longer implementation period of up to 8 years (see Table 13). The Service recognizes that providing an incremental time approach for addressing the development of new protective chemical criteria by effected agencies is reasonable both in terms of workload, work prioritization, and staffing. For arsenic, copper, cyanide, mercury, and selenium, chemicals for which both the NMFS and the Service identified the need for developing new protective criteria, the Service identified the same incremental dates between 2017 and 2021 for new criteria to become effective. For Service only criteria (lead, zinc, and nickel), the Service identified an additional one to two years beyond 2021 for new criteria to become effective (2022 and 2023).

To ensure that the listed species are not adversely affected during the implementation period, these RPAs include interim protective measures that the Service expects will adequately reduce any interim risk of harm to the species or their critical habitats. In addition, EPA consults with the Service over each new or reissued NPDES permit in Idaho to ensure that it will not cause jeopardy to the species or adverse modification to critical the habitat. These factors, when considered together, will minimize any adverse effects during the implementation period while new criteria are developed and adopted.

# 2.8.1 RPAs for Arsenic

*2.8.1.1 Interim Protection for Listed Snails*

Until a new chronic criterion for arsenic is adopted, EPA shall ensure that the 10 µg/L recreational use standard is applied in all Water Quality Based Effluent Limitations (WQBELs) and Reasonable Potential to Exceed Calculations using the human health criteria and the current methodology for developing WQBELs to protect human health. The recreational use standard is interpreted to apply as inorganic, unfiltered, arsenic.

*2.8.1.2 Interim Protection for the Bull Trout, Bull Trout Critical Habitat, the Kootenai River White Sturgeon, and Kootenai River Critical Habitat*

Data limitations, ambiguities, and resulting uncertainties in the effects analysis include that waterborne arsenic concentrations that have been associated with risks of toxicity via food webs may overlap background concentrations measured at other locations (section 2.5.2.2). Therefore, in order to reduce the joint risks of causing unreasonable constraints or imprudently allowing discharges or releases of arsenic that could be harmful to listed species or habitats, the following monitoring and decision steps are considered appropriate.

Until a new chronic criterion for arsenic is adopted, EPA shall ensure that all effluent discharges from major point sources for which arsenic is a pollutant of concern, located within habitats occupied by the bull trout and/or the Kootenai River white sturgeon and within areas of their designated critical habitat that are regulated under the NPDES program or controls of releases which meet substantive requirements of NPDES permits, shall comply with the following terms.:

1. At discharge locations where at the edge of the mixing zone, unfiltered arsenic concentrations are measured or projected to be higher than natural background for the locale as the result or the suspected result of the point source discharge, and annual geometric mean concentrations are higher than 5 µg/L above background, aquatic insect tissue samples shall be monitored in locations downstream of the discharge and in reference locations. The results shall be reported as an NPDES permit condition.
2. If the above average aquatic invertebrate tissue concentrations exceed 25 mg/kg dw total arsenic[26], and are higher than reference concentrations for that site, then the issuance of an NPDES permit shall include provisions to reduce arsenic loading in order to reduce impairment of aquatic life uses, unless:
3. If arsenic speciation analyses show that the average aquatic invertebrate tissue concentrations are less than 20 mg/kg dw inorganic arsenic, then dietary arsenic will be presumed to represent a low risk to bull trout or sturgeon, and no further reductions are necessary. However, if the aquatic invertebrate tissue concentrations exceeds 20 mg/kg dw inorganic arsenic and are higher than reference concentrations for that site, then the issuance of an NPDES permit shall include provisions to reduce arsenic loading in order

[26] The 25 mg/kg dw total arsenic screening value represents the dietary toxicity risk threshold identified for inorganic arsenic (20 mg/kg dw), divided by 0.8, which was the highest fraction of the more-toxic inorganic form of arsenic reported for an aquatic invertebrate in the literature reviewed (section 2.5.2.)

to reduce impairment of aquatic life uses. Arsenic in benthic invertebrate prey organisms is intended as a representative composite community sample (NMFS 2014a, Appendix E). These provisions are not required if fish population surveys using surrogate species, such as the rainbow trout, show that appreciable adverse effects are not occurring, as defined in Appendix E, Biomonitoring of Effects, of NMFS (2014a).

#### *2.8.1.2 New Chronic Aquatic Life Criterion for Arsenic (based on NMFS 2014a)*

The EPA shall ensure, either through EPA promulgation of a criterion or EPA approval of a state-promulgated criterion, that a new chronic criterion for arsenic is in effect in Idaho by May 7, 2021. The new criterion shall be likely to avoid jeopardy of listed aquatic snails, the bull trout, the Kootenai River white sturgeon, and adverse modification of critical habitat for the bull trout and the white sturgeon, consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criterion, EPA shall provide an adequate biological assessment/evaluation to the Service and initiate consultation at least 135 days in advance of May 7, 2020, unless the Service and EPA mutually agree to a different time-frame.

#### *2.8.1.3 Analysis of the Reasonable and Prudent Alternative for Arsenic*

An interim level of protection of the listed species and critical habitats referenced above relative to arsenic is available through use of the human health criterion, which is 10 µg/L. This criterion is applicable to all waters in the action area. Because it is more stringent than the chronic criterion of 150 µg/L, the criterion for the protection of human health is the controlling criterion for NPDES permitting actions. Based on the best available information from section 2.5.2.1, the application of this lower standard, is likely avoid adverse effects to listed Snake River snails and the Bruneau hot springsnail.

Significant effects are not expected from the interim RPA for the bull trout, bull trout critical habitat, Kootenai River white sturgeon, and Kootenai River white sturgeon critical habitat, because the 5 µg/L interim RPA trigger for initiating monitoring is at or below the low range of concentrations associated with adverse effects or appreciable bioaccumulation (section 2.5.2).

Because any new or reissued NPDES permits will be subject to individual ESA consultation, as appropriate, to ensure they avoid jeopardy or adverse modification of critical habitat, EPA will make adjustments as necessary during the NPDES permitting cycle taking into account local conditions to avoid measureable direct effects caused by arsenic that are likely to cause jeopardy to the above listed species and/or adverse modification of critical habitats. By avoiding such measureable direct effects, use of the human health criterion is likely to provide adequate protection in the interim to avoid jeopardy to listed species and adverse modification of critical habitat. Adoption of a new chronic aquatic life criterion for arsenic by May 7, 2021 will be subject to ESA consultation, as appropriate, to ensure that the new criterion is likely to be adequately protective of listed species and critical habitats in terms of avoiding jeopardy and adverse modification of critical habitat.

For the above reasons, the Service concludes that implementation of the RPA for arsenic is not likely to jeopardize any of the listed species considered in this Opinion or to adversely modify bull trout or Kootenai River white sturgeon critical habitat.

## 2.8.2 RPAs for Copper

### *2.8.2.1 Interim Protection for Listed Snails, the Bull Trout, Bull Trout Critical Habitat, the Kootenai River White Sturgeon, and Kootenai White Sturgeon Critical Habitat*

Listed Snake River Snails

To provide interim protection to Snake River snails and the Bruneau hot springsnail, until new criteria are adopted, the mixing zone for copper for any authorized NPDES discharges of copper into occupied snail habitat must meet Idaho's approved hardness-based acute and chronic copper criterion at the end of pipe, no mixing zone is allowed. However, a mixing zone may be allowed if the acute and chronic effect thresholds based on the copper BLM are not exceeded beyond the acute mixing zone and the chronic mixing zone respectively. The chronic mixing zone is limited to no more than 25 percent of flow.[27]

Bull Trout and Kootenai River White Sturgeon (based on NMFS 2014a)

Until new criteria are adopted, a zone of passage must be maintained around any mixing zone for discharges that include copper that is sufficient to allow unimpeded passage of adult and juvenile bull trout and sturgeon.

Permits for new discharges must ensure a zone of passage for these species that persists under seasonal flow conditions (see Appendix D of NMFS 2014a). If the regulatory mixing zone is limited to less than or equal to 25 percent of the seasonal flow conditions, then a sufficient zone of passage is presumed to be present.

Permits reissued for existing discharges must ensure a zone of passage for adult and juvenile bull trout and sturgeon that persists under seasonal flow conditions. If the regulatory mixing zone is limited to less than or equal to 25 percent of the volume of a stream, then a sufficient zone of passage is presumed to be present. If existing discharges were calculated using greater than 25 percent of the seasonal flow conditions for applying aquatic life criteria, the mixing zone must be reduced to 25 percent unless one of the following conditions exists:

1. An evidence-based "Salmonid Zone of Passage Demonstration" (see Appendix F of NMFS 2014a) indicates that impeding fish movement is unlikely, or;
2. Biological monitoring of aquatic communities in the downstream receiving waters shows no appreciable adverse effects relative to reference conditions as described in Appendix E, Biomonitoring of Effects, in NMFS (2014a), and biological whole-effluent toxicity (WET) testing is consistently negative, as defined below:

[27] The distributions of the listed aquatic snails do not overlap with those of the bull trout and Kootenai River white sturgeon; therefore, the no mixing zone restriction does not apply to discharges into the habitats of bull trout and sturgeon.

a. WET testing shall be required, using at least the 7-day *Ceriodaphnia dubia* 3-brood test and the 7-day fathead minnow growth and survival test. If previous testing of a facility's effluents has demonstrated that one test is more sensitive, at EPA's discretion, it is acceptable to base further testing on only the more sensitive test. Toxicity trigger concentrations for WET tests shall also be established using dilution series based upon no more than 25 percent of the applicable critical flow volume. The dilution series for WET testing (7Q10) shall be designed such that one treatment consists of 100 percent effluent, and at least one treatment is more dilute than the targeted critical flow conditions. Receiving waters upstream of the effluent discharge should be used as dilution water.

   The "critical concentration" is defined here as the condition when the smallest permitted dilution factor occurs, modified by a 25 percent mixing zone fraction. For example, if the minimum effluent dilution occurring at a site is a 1:4 ratio (one part effluent to four parts streamwater), then because only 25 percent of the measured streamflow is authorized for dilution, the dilution factor for effluent testing is likewise reduced to 1:1. The critical concentration would then be 50 percent effluent, i.e., one part each effluent and dilution water.

   WET tests results need to be consistently negative to indicate the absence of appreciable instream toxicity in test conditions that reflect the critical effluent concentration above. A "negative test result" is produced by a test meeting the performance objectives of a passing test according to EPA (2002c) or EPA (2010c). Test results are considered to be consistently negative if the failure rate is less than one in 20.

b. If instream biological monitoring shows adverse effects to reference conditions or if WET tests are not consistently negative, then a toxicity identification evaluation and a toxicity reduction evaluation (TIE/TRE) must be undertaken to identify and remedy the causes of toxicity, which may include reducing effluent limits as warranted. Because considerable judgment may be involved in designing and carrying out a TIE/TRE, and because the results are performance-based (e.g., no detectable toxicity observed), more specific guidance is inappropriate to provide here. See Mount and Hockett (2000) for an example of a TIE/TRE.

#### *2.8.2.2 New Acute and Chronic Aquatic Life Criteria for Copper (based on NMFS 2014a)*

The EPA shall ensure, either through EPA promulgation of criteria or EPA approval of state-promulgated criteria, that new acute and chronic criteria for copper are in effect in Idaho by May 7, 2017. The new criteria shall be as protective or no less stringent than the 2007 CWA section 304(a) national recommended aquatic life criteria (i.e., the Biotic Ligand Model [BLM]) for copper or an alternative criteria such as an updated BLM or similar modeling approach. The Service does not anticipate that additional consultation will be required if the 2007 national recommended aquatic life criteria or other alternative criteria which would be as protective for copper are adopted by EPA.

#### *2.8.2.3 Removal of Low-End Hardness Floor (based on NMFS 2014a)*

The EPA shall recommend that the state of Idaho adopt, and EPA will promulgate, if necessary, the removal of the low end hardness floor on the hardness dependent metals criteria equations by May 7, 2017.

#### *2.8.2.4 Analysis of the Reasonable and Prudent Alternative for Copper*

Limiting mixing zone fractions of the receiving water discharge in flowing waters is effectively similar to reducing the criteria concentration. For example limiting the mixing zone fractions to 25 percent effectively reduces the criteria by about 0.25X (NMFS 2014a). Therefore, not allowing a mixing zone for discharges of copper into listed snail habitat is expected to reduce copper concentrations to levels where adverse effects to the snails are unlikely.

If a mixing zone is allowed, it will be limited to no more than 25 percent of flow. As previously noted, a mixing zone of 25 percent effectively reduces the copper criteria by about 0.25X, which will minimize the risk of adverse effects.

For the bull trout and the sturgeon and their critical habitats, the interim requirement until May 7, 2017, of ensuring an adequate zone of passage under NPDES permits that contain copper discharge limits, as described in the RPA for copper, is likely to minimize adverse effects to the bull trout and to the Kootenai River white sturgeon and their critical habitat. Any new permits will also be subject to individual consultation, as appropriate, to ensure they avoid jeopardy or adverse modification of critical habitat.

NMFS (2014a, Appendix C) analyzed "implementation of the 2007 BLM EPA copper criteria and conclude[d] that they are likely to avoid jeopardy to the listed species or critical habitat considered in this Opinion." The Service agrees with NMFS's reasoning and this conclusion and finds that it is applicable to the bull trout and its critical habitat, and to the Kootenai River white sturgeon and its critical habitat.

Some adverse effects would still be expected if ambient concentrations were at the 2007 chronic aquatic life criterion, but these would be minimized by further limiting mixing zone fractions to 1/4 (25 percent) of the receiving water discharge in flowing waters is effectively similar to reducing the criteria by about 0.25X (NMFS 2014a). Few if any adverse effects to listed species or habitats would be expected at about 0.25X the concentration resulting from the 2007 version of EPA's copper criteria. The 0.25X mixing zone authorization is consistent with IDEQ water quality standards and EPA permitting practices, as described in the introduction to section 2.8.

For the above reasons, the Service concludes that implementation of the RPA for copper is not likely to jeopardize any of the listed species considered in this Opinion or to adversely modify critical habitat for the bull trout and the sturgeon.

### 2.8.3 RPA for Cyanide

#### *2.8.3.1 Interim Protection for the Bull Trout, Bull Trout Critical Habitat, the Kootenai River White Sturgeon, and Kootenai River White Sturgeon Critical Habitat (based on NMFS 2014a)*

Until new criteria are adopted, a zone of passage must be maintained around any mixing zone for discharges that include cyanide that is sufficient to allow unimpeded passage of adult and juvenile bull trout and sturgeon.

Permits reissued for existing discharges must ensure a zone of passage for these species that persists under seasonal flow conditions. If the regulatory mixing zone is limited to less than or equal to 25 percent of the volume of a stream, then a sufficient zone of passage is presumed to be present. If existing discharges were calculated using greater than 25 percent of the seasonal flow conditions for applying aquatic life criteria, the mixing zone must be reduced to 25 percent unless one of the following conditions exists:

1. An evidence-based “Salmonid Zone of Passage Demonstration” (see Appendix F of NMFS 2014a) indicates that impeding fish movements is unlikely, or;
2. Biological monitoring of aquatic communities in the downstream receiving waters shows no appreciable adverse effects relative to reference conditions as described in Appendix E (Biomonitoring of Effects) of NMFS (2014a), and biological WET testing is consistently negative, as defined below:
    a. WET testing shall be required, using at least the 7-day *Ceriodaphnia dubia* 3-brood test and the 7-day fathead minnow growth and survival test. If previous testing of a facility’s effluents have demonstrated that one test is more sensitive, at EPA’s discretion it is acceptable to base further testing on only the more sensitive test. Toxicity trigger concentrations for WET tests shall also be established using dilution series based upon no more than 25 percent of the applicable critical flow volume. The dilution series for WET testing (7Q10) shall be designed such that one treatment consists of 100 percent effluent, and at least one treatment is more dilute than the targeted critical flow conditions. Receiving waters upstream of the effluent discharge should be used as dilution water.

       The “critical concentration” is defined here as the condition when the smallest permitted dilution factor occurs, modified by a 25 percent mixing zone fraction. For example, if the minimum effluent dilution occurring at a site is a 1:4 ratio (one part effluent to four parts streamwater), then because only 25 percent of the measured streamflow is authorized for dilution; then the dilution factor for effluent testing is likewise reduced to 1:1. The critical concentration would then be 50 percent effluent, i.e., one part each effluent and dilution water.

       WET tests results need to be consistently negative to indicate the absence of appreciable instream toxicity in test conditions that reflect the critical effluent concentration above. A “negative test result” is produced by a test meeting the performance objectives of a passing test according to EPA (2002c) or EPA (2010c). Test results are considered to be consistently negative if the failure rate is less than one in 20.

    c. If instream biological monitoring shows adverse effects to reference conditions or if WET tests are not consistently negative, then a toxicity identification evaluation and toxicity reduction evaluation (TIE/TRE) must be undertaken to identify and remedy the causes of toxicity, which may include reducing effluent limits as warranted. Because considerable judgment may be involved in designing and carrying out a TIE/TRE, and because the results are performance-based (e.g., no detectable toxicity observed), more specific guidance is inappropriate to provide here. See Mount and Hockett (2000) for an example of a TIE/TRE.

*2.8.3.2 New Acute and Chronic Aquatic Life Criteria for Cyanide*

The EPA shall ensure, either through EPA promulgation of criteria or EPA approval of a state-promulgated criteria, that new acute and chronic criteria for cyanide are in effect in Idaho by May 7, 2021. The new criteria: shall be calculated using a temperature/toxicity correlation equation; shall provide adequate protection to avoid jeopardizing the bull trout and the Kootenai River white sturgeon, and to avoid adversely modifying the critical habitats of the bull trout and the Kootenai River white sturgeon; and shall be consistent with the discussion and analysis in this Opinion. In the absence of specific data, the Service's best estimate of adequately safe cyanide concentrations for acute and chronic exposures, respectively, is 13 and 2.5 µg/L[28]. If ESA consultation is required for the new criteria, EPA shall provide an adequate biological evaluation to the Service and initiate consultation at least 135 days in advance of May 7, 2020, unless the Service and EPA mutually agree to a different time-frame.

*2.8.3.3 Analysis of the Reasonable and Prudent Alternative for Cyanide*

Implementation of more restrictive practices in developing cyanide discharge limits that are authorized under NPDES permits as described in the RPA for cyanide is likely to sufficiently minimize adverse effects to the bull trout and to the sturgeon as well as to their critical habitats. These practices will ensure that an adequate zone of passage exists for these species under all flow conditions, and will provide for biological monitoring and whole-effluent toxicity testing to ensure that permit limits are protective of the bull trout and the sturgeon and their prey species. This monitoring will be done at each discharge site by taking into account the localized conditions that affect the toxicity of cyanide. Based on development of these site-specific limits and the associated monitoring of discharge levels, combined with the fact that the Service consults, as appropriate, with EPA over each new or reissued NPDES permit, we expect only minor adverse effects to the bull trout, Kootenai River white sturgeon, and to their critical habitats with implementation of the RPA.

Limiting mixing zone fractions to 1/4 (25 percent) of the receiving water discharge in flowing waters is effectively similar to reducing the criteria by about 0.25X (NMFS 2014a). While adverse effects were identified at or below the existing criteria concentrations, few if any adverse effects to listed species or habitats would be expected at about 0.25X the criteria concentrations. The 0.25X mixing zone authorization is consistent with IDEQ water quality standards and EPA permitting practices, as described in the introduction to section 2.8.

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[28] These numbers are derived by halving the lowest and coldest acute LC50 of 27 µg/L and the lowest chronic low-effects concentrations of about 5 µg/L. The rationale of halving an acutely toxic concentration to extrapolate to a concentration that would kill few if any individuals has been incorporated into EPA's criteria derivation guidelines (Stephan et al. 1985), and has more recently been supported by analyses by the USFWS (2010b) and the NMFS (2014a). For long-term exposures, concentration-effect series tested by Kovacs and Leduc (1982b) supported extending this rationale for acute LC50s to chronic cyanide toxicity responses as well.

The adoption of a new chronic aquatic life criteria for cyanide by May 7, 2021 will be subject to ESA consultation, as appropriate, to ensure that the new criteria are adequately protective in terms of avoiding jeopardy and adverse modification of critical habitat.

For the above reasons, the Service concludes that the RPA for cyanide is not likely to jeopardize any of the listed species considered in this Opinion or adversely modify bull trout or Kootenai River white sturgeon critical habitat.

## 2.8.4 RPAs for Lead

### *2.8.4.1 Interim Protection for the Banbury Springs Lanx*

To provide interim protection to the Banbury Springs lanx until a new chronic lead criterion is adopted, any authorized NPDES discharge into occupied lanx habitat must meet the chronic lead criterion at the end of pipe, no mixing zone is allowed.

### *2.8.4.2 New Chronic Aquatic Life Criterion for Lead*

The EPA shall ensure, either through EPA promulgation of a criterion or EPA approval of a state-promulgated criterion, that a new chronic criterion for lead is in effect in Idaho by May 7, 2023. The new criterion shall be likely to avoid jeopardizing listed snails, the bull trout, and the Kootenai River white sturgeon, and adversely modifying the critical habitats of the bull trout and the Kootenai River white sturgeon, consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criterion, EPA shall provide an adequate biological evaluation to the Service and initiate consultation at least 135 days in advance of May 7, 2022, unless the Service and EPA mutually agree to a different time-frame.

### *2.8.4.3 Hardness Floor (based on NMFS 2014a)*

The EPA shall recommend that the state of Idaho adopt, and EPA will promulgate if necessary, the removal of the low end hardness floor on the hardness dependent metals criteria equations by May 7, 2017.

### *2.8.4.4 Analysis of the Reasonable and Prudent Alternative for Lead*

Limiting mixing zone fractions of the receiving water discharge in flowing waters is effectively similar to reducing the criteria concentration. For example limiting the mixing zone fractions to 25 percent is effectively similar to reducing the criteria by about 0.25X (NMFS 2014a). Therefore, contingent on the dilution ratio and the size of the discharge relative to the size and quality of the receiving water, not allowing a mixing zone for discharges of lead into lanx habitat is expected to limit lead concentrations to levels where adverse effects to the lanx are unlikely.

The adoption of a new chronic aquatic life criterion for lead by May 7, 2023 will be subject to ESA consultation, as appropriate, to ensure that the new criterion will be adequately protective in terms of avoiding jeopardy to the Banbury Springs lanx.

For the above reasons, the Service concludes that the RPA for lead is not likely to jeopardize the Banbury Springs lanx.

Note: The proposed criteria for lead are not likely to adversely affect the other listed species and critical habitats at issue in this Opinion.

## 2.8.5 RPAs for Mercury

*2.8.5.1 Interim Protection for the Bull Trout and its Critical Habitat, and for the Kootenai River White Sturgeon and its Critical Habitat (based on NMFS 2014a)*

1. Until a new chronic criterion for mercury is adopted, EPA shall use the 2001 EPA/2005 Idaho human health fish tissue criterion of 0.3 mg/kg wet weight for WQBELs and reasonable potential to exceed criterion calculations using the current methodology for developing WQBELs to protect human health. Implementation of the Idaho methylmercury criterion shall be guided by EPA's methylmercury water quality criteria implementation guidance (EPA 2010a) or IDEQ's methylmercury water quality criteria implementation guidance (IDEQ 2005); or
2. For water bodies for which appropriate fish tissue data are not available, if the geometric mean of measured concentrations of total mercury in the water is less than 2 ng/L, then the water body will be presumed to meet the fish tissue criterion of 0.3 mg/kg wet weight. If the water column concentration is greater than 2 ng/L, fish tissue data shall be collected and analyzed to determine if the fish tissue criterion of 0.3 mg/kg wet weight is met. If not, the provisions of the previous paragraph (2.8.5.1.1) apply to reduce mercury loading in order to reduce impairment of aquatic life uses.

*2.8.5.2 New Chronic Aquatic Life Criterion for Mercury (based on NMFS 2014a)*

The EPA shall ensure, either through EPA promulgation of a criterion or EPA approval of a state-promulgated criterion, that a new chronic criterion for mercury is in effect in Idaho by May 7, 2021. The new criterion shall be likely to avoid jeopardy and adverse modification of the critical habitats of the bull trout and the Kootenai River white sturgeon, consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criterion, EPA shall provide an adequate biological evaluation to the Service and initiate consultation at least 135 days in advance of May 7, 2020, unless the Service and EPA mutually agree to a different time-frame.

*2.8.5.3 Analysis of the Reasonable and Prudent Alternative for Mercury*

The interim requirement of using a human health criterion that consists of a fish tissue-based water quality criterion of 0.3 mg/kg for mercury to determine NPDES permit limits will be followed. Idaho has adopted this criterion, and is implementing it as a 0.24 mg/kg triggering residue concentration for existing dischargers, using an uncertainty (safety factor) of 0.8 times (IDEQ 2007a). This fish tissue-based criterion is close to being a threshold below which adverse effects to listed fish species are unlikely, and is considered sufficient to protect listed fish and their habitats.

The adoption of a new chronic aquatic life criterion for mercury by May 7, 2021 will be subject to ESA consultation, as appropriate, to ensure that the new criterion will be adequately protective in terms of avoiding jeopardy to listed species and adverse modification of critical habitat.

For the above reasons, the Service concludes that the RPA for mercury is not likely to jeopardize any of the listed species considered in this Opinion or to adversely modify critical habitat for the bull trout and the Kootenai River white sturgeon.

# 2.8.6 RPAs for Selenium

*2.8.6.1 Interim Protection for the Bull Trout and its Critical Habitat, and for the Kootenai River White Sturgeon and its Critical Habitat (based on NMFS 2014a)*

Until a new chronic criterion for selenium is adopted, EPA shall ensure that all effluent discharges located within habitats occupied by the bull trout and/or the Kootenai River white sturgeon and within areas of their designated critical habitat that are regulated under the NPDES program shall comply with the following terms:

1. At discharge locations where at the edge of the mixing zone, selenium concentrations are measured or projected to be higher than natural background for the locale and annual geometric mean concentrations are higher than 2 µg/L, whole body fish tissue shall be monitored in locations downstream of the discharge and in reference locations. The results shall be reported as an NPDES permit condition.
2. If the above fish tissue concentrations exceed the screening risk concentration for selenium of 7.6 mg/kg dw and are higher than reference concentrations, then the issuance of an NPDES permit shall include provisions to reduce selenium loading in order to reduce impairment of aquatic life uses. These provisions are not required if fish population surveys using surrogate species, such as the rainbow trout, show that appreciable adverse effects are not occurring, as defined in Appendix E, Biomonitoring of Effects, of NMFS (2014a).

*2.8.6.2 New Chronic Aquatic Life Criterion for Selenium (based on NMFS 2014a)*

The EPA shall ensure, either through EPA promulgation of a criterion or EPA approval of a state-promulgated criterion, that a new chronic criterion for selenium is in effect in Idaho by May 7, 2018. The new criterion shall be likely to avoid jeopardy and adverse modification of critical habitats of the bull trout and the Kootenai River white sturgeon, consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criterion, EPA shall provide an adequate biological evaluation to the Service and initiate consultation at least 135 days in advance of May 7, 2017, unless the Service and EPA mutually agree to a different time-frame.

*2.8.6.3 Analysis of the Reasonable and Prudent Alternative for Selenium*

The interim requirement of monitoring fish tissues and taking corrective action when fish tissues exceed a selenium concentration of 7.6 mg/kg dw or 2 µg/L in the water column is likely to be sufficiently protective of habitat conditions to minimize any adverse effects to the bull trout and to the sturgeon from food web transfer-related concentrations of selenium. Any new permits addressing discharges of selenium will be subject to individual ESA consultation, as appropriate, to ensure that jeopardy or adverse modification of critical habitat is not likely to occur. Based on these protective, interim practices and the low number of likely discharges, the continued use of the existing selenium standard up until May 7, 2018 is likely to result in only minor adverse effects to these species and their critical habitats.

The adoption of a new chronic aquatic life criterion for selenium by May 7, 2018 will be subject to ESA consultation, as appropriate, to ensure that the new criterion is adequately protective in terms of avoiding jeopardy and adverse modification of critical habitat.

For the above reasons, the Service concludes that the RPA for selenium is not likely to jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats.

## 2.8.7 RPAs for Zinc

### *2.8.7.1 Interim Protection for the Bull Trout and the Kootenai River White Sturgeon and Their Critical Habitats (based, in part, on NMFS 2014a)*

Until new criteria are adopted, a zone of passage sufficient to allow unimpeded passage of adult and juvenile bull trout and sturgeon must be maintained around any mixing zone for discharges that include zinc.

NPDES permits for new discharges must ensure a zone of passage (sufficient to allow unimpeded passage of adult and juvenile bull trout and sturgeon) persists under seasonal flow conditions; see Appendix D of NMFS (2014a). If the regulatory mixing zone is limited to less than or equal to 25 percent of seasonal flow conditions, then a sufficient zone of passage is presumed to be present.

NPDES permits reissued for existing discharges must ensure a zone of passage (sufficient to allow unimpeded passage of adult and juvenile bull trout and sturgeon) persists under seasonal flow conditions. If the regulatory mixing zone is limited to less than or equal to 25 percent of the volume of a stream, then sufficient zone of passage is presumed to be present. If existing discharges were calculated using greater than 25 percent of seasonal flow conditions for applying aquatic life criteria, the mixing zone must be reduced to 25 percent unless one of the following conditions exists:

1. An evidence-based "Salmonid Zone of Passage Demonstration" (see NMFS 2014a, Appendix F) indicates that impeding fish movements is unlikely; or

2. Biological monitoring of aquatic communities in the downstream receiving waters shows no appreciable adverse effects relative to reference conditions as described in Appendix E, Biomonitoring of Effects, of NMFS (2014a), and biological WET testing is consistently negative, as defined below:

    a. WET testing shall be required, using at least the 7-day *Ceriodaphnia dubia* 3-brood test and the 7-day fathead minnow growth and survival test. If previous testing of a facility's effluents have demonstrated that one test is more sensitive, at EPA's discretion, it is acceptable to base further testing on only the more sensitive test. Toxicity trigger concentrations for WET tests shall also be established using dilution series based upon no more than 25 percent of the applicable critical flow volume. The dilution series for WET testing (7Q10) shall be designed such that one treatment consists of 100 percent effluent, and at least one treatment is more dilute than the targeted critical flow conditions. Receiving waters upstream of the effluent discharge should be used as dilution water.

    The "critical concentration" is defined here as the condition when the smallest permitted dilution factor occurs, modified by a 25 percent mixing zone fraction. For example, if the minimum effluent dilution occurring at a site is a 1:4 ratio (one part effluent to four parts streamwater), then because only 25 percent of the measured streamflow is authorized for dilution; then the dilution factor for effluent testing is likewise reduced to 1:1. The critical concentration would then be 50 percent effluent, i.e., one part each effluent and dilution water.

WET tests results need to be consistently negative to indicate the absence of appreciable instream toxicity of zinc in test conditions that reflect the critical effluent concentration discussed above. A "negative test result" is produced by a test meeting the performance objectives of a passing test according to EPA (2002c) or EPA (2010c). Test results are considered to be consistently negative if the failure rate is less than one in 20.

b. If instream biological monitoring shows adverse effects to reference conditions or if WET tests are not consistently negative, then a toxicity identification evaluation and toxicity reduction evaluation (TIE/TRE) must be undertaken to identify and remedy the causes of zinc toxicity; such remedies may include reducing effluent limits for zinc as warranted. Because considerable judgment may be involved in designing and carrying out a TIE/TRE, and because the results are performance-based (e.g., no detectable zinc toxicity observed), more specific guidance is inappropriate to provide here. See Mount and Hockett (2000) for an example of a TIE/TRE.

*2.8.2.2 New Acute and Chronic Aquatic Life Criteria for Zinc*

The EPA shall ensure, either through EPA promulgation of criteria or EPA approval of a state-promulgated criteria, that new acute and chronic criteria for zinc are in effect in Idaho by May 7, 2022. The new criteria shall be likely to avoid jeopardizing listed species, and avoid adversely modifying the critical habitats for the bull trout and the sturgeon consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criteria, EPA shall provide an adequate biological evaluation to the Service and initiate consultation at least 135 days in advance of May 7, 2021 unless the Service and EPA mutually agree to a different time-frame.

*2.8.8.3 Analysis of the Reasonable and Prudent Alternative for Zinc*

Implementation of more restrictive practices in developing zinc discharge limits that are authorized under NPDES permits as described in the RPA for zinc is likely to sufficiently minimize adverse effects to the bull trout and to the sturgeon as well as to their critical habitats. These practices will ensure an adequate zone of passage exists for these species under all flow conditions, and provide for biological monitoring and WET testing to ensure that permit limits are protective of the bull trout and the sturgeon and their prey species. This monitoring will be done at each discharge site by taking into account the localized conditions that affect the toxicity of zinc. Based on development of these site-specific limits and the associated monitoring of discharge levels, combined with the fact that the Service consults, as appropriate, with EPA over each new or reissued NPDES permit, we expect only minor effects to the bull trout, Kootenai River white sturgeon, and to their critical habitats.

Furthermore, limiting mixing zone fractions to 1/4 (25 percent) of the receiving water discharge in flowing waters is effectively similar to reducing the criteria by about 0.25X (NMFS 2014a).

While adverse effects were identified at or below the existing criteria concentrations, few if any adverse effects to listed species or habitats would be expected at about 0.25X the criteria concentrations. The 0.25X mixing zone authorization is consistent with IDEQ water quality standards and EPA permitting practices, as described in the introduction to section 2.8.

The adoption of a new acute and chronic aquatic life criteria for zinc by May 7, 2022 will be subject to ESA consultation, as appropriate, to ensure that the new criteria will be adequately protective in terms of avoiding jeopardy and adverse modification of critical habitat.

For the above reasons, the Service concludes that the RPA for zinc is not likely to jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats.

# 2.8.8 RPAs for Nickel

### *2.8.8.1 Interim Protection for Listed Snails*

To provide interim protection to the Banbury Springs lanx until a new chronic lead criterion is adopted, any authorized NPDES discharge into occupied lanx habitat must meet the chronic lead criterion at the end of pipe, no mixing zone is allowed.

To provide interim protection to the Snake River physa, Bliss Rapids snail, and the Bruneau hot springsnail, until new criteria are adopted, the mixing zone for nickel for any authorized NPDES discharges of nickel into occupied snail habitat must be limited to no more than 25 percent of flow.

### *2.8.8.2 New Acute and Chronic Aquatic Life Criteria for Nickel*

The EPA shall ensure, either through EPA promulgation of criteria or EPA approval of a state-promulgated criteria, that new criteria for nickel are in effect in Idaho by May 7, 2022. The new criteria shall be protective in terms of avoiding jeopardy of the Snake River physa, Bliss Rapids snail, Banbury Springs lanx, and Bruneau hot springsnail consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criteria, EPA shall provide an adequate biological evaluation to the Service and initiate consultation at least 135 days in advance of May 7, 2021, unless the Service and EPA mutually agree to a different time-frame.

### *2.8.8.3 Removal of Low-End Hardness Floor (based on NMFS 2014a)*

The EPA shall recommend that the state of Idaho adopt, and EPA will promulgate, if necessary, the removal of the low end hardness floor on the nickel aquatic life criteria equations by May 7, 2017.

### *2.8.8.4 Analysis of the Reasonable and Prudent Alternative for Nickel*

Limiting mixing zone fractions of the receiving water discharge in flowing waters is effectively similar to reducing the criteria concentration. For example, limiting the mixing zone fractions to 25 percent effectively reduces the criteria by about 0.25X (NMFS 2014a). Therefore, contingent on the dilution ratio and the size of the discharge relative to the size and quality of the receiving water, not allowing a mixing zone for discharges of nickel into lanx habitat is expected to limit nickel concentrations to levels where adverse effects to the lanx are unlikely.

Similarly, while adverse effects were identified at or below the existing criteria concentrations, few if any adverse effects to the Snake River physa, Bliss Rapids snail, and the Bruneau hot springsnail were identified at about 0.25X or less of the criteria concentrations (2.5.10). The 0.25X mixing zone authorization is consistent with IDEQ water quality standards and EPA permitting practices, as described in the introduction to section 2.8.

The adoption of new chronic aquatic life criteria for nickel by May 7, 2022 will be subject to ESA consultation, as appropriate, to ensure that the new criteria will be adequately protective in terms of avoiding jeopardy to the listed snails.

For the above reasons, the Service concludes that the RPA for nickel is not likely to jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats.

Note: The proposed criteria for nickel are not likely to adversely affect the other listed species and critical habitats at issue in this Opinion.

## 2.8.9 Summary of the RPAs

See Table 13 below.

**Table 13. Summary of RPAs and implementation schedule.**

| Metal | Interim Protection | New Criteria in effect in Idaho by: | Consultation (if needed) on new criteria initiated by: | Remove Low End Hardness Floor, if applicable, by: | Source of RPA |
|---|---|---|---|---|---|
| Arsenic | Human Health (snails); Monitoring based NPDES permit conditions (fish) | May 7, 2021 | May 7, 2020 | N/A | NMFS (2014a) |
| Copper | No mixing zone (snails); zone of passage (fish) | May 7, 2017 | N/A | May 7, 2017 | NMFS (2014a) |
| Cyanide | Zone of passage | May 7, 2021 | May 7, 2020 | N/A | USFWS; this Opinion |
| Lead | No mixing zone (lanx) | May 7, 2023 | May 7, 2022 | May 7, 2017 | USFWS; this Opinion |
| Mercury | Human Health | May 7, 2021 | May 7, 2020 | N/A | NMFS (2014a) |
| Selenium | Monitoring based NPDES permit conditions | May 7, 2018 | May 7, 2017 | N/A | NMFS (2014a) |
| Zinc | Zone of passage (fish) | May 7, 2022 | May 7, 2021 | N/A | USFWS; this Opinion |
| Nickel | No mixing zone (lanx); mixing zone ≤ 25% (the other listed snails) | May 7, 2022 | May 7, 2021 | May 7, 2017 | USFWS; this opinion |

If new criteria development and associated consultations, as appropriate, are not finalized by the effective dates listed above, all interim measures identified in the individual RPA shall be adopted as final for purposes of establishing aquatic life criteria in association with Idaho's water quality standards.

### 2.8.10 Notification of EPA's Final Decision

In accordance with 50 CFR 402.15(b) of the implementing regulations for ESA section 7, the EPA is required to notify the Service of its final decision regarding implementation of the proposed action.

## 2.9 Incidental Take Statement

For the reasons discussed below and in the findings of the *Reasonable and Prudent Alternatives* section above, no take of listed species under Service jurisdiction is anticipated under EPA's implementation of the interim RPAs for the proposed action. EPA adoption of new criteria for the toxic pollutants addressed by the RPAs will be subject to section 7 consultation to ensure that the new criteria will not jeopardize listed species or result in adverse modification of critical habitats. An Incidental Take Statement addressing that adoption will be prepared in conjunction with a biological opinion on that action, as appropriate.

Implementation of the interim RPAs (i.e., using the human health criteria where applicable, restricting discharges, or requiring zones of passage around mixing zones, as described in the *Reasonable and Prudent Alternatives* section above for arsenic, copper, cyanide, lead, mercury, selenium, zinc, and nickel) are expected to be protective of listed species and will minimize the potential for incidental take to an insignificant level[29]. Therefore, no take of listed species is anticipated to be caused by the implementation of the interim RPAs for those metals. In addition, this finding will be re-affirmed, as appropriate, as a result of individual consultations on a site-specific basis for each NPDES permit issued in the action area under implementation of the interim RPAs. If such individual consultations reveal that take of listed species is reasonably certain to occur under implementation of the interim RPAs, that finding would constitute new information that will warrant re-initiation of consultation on the action as addressed herein.

If new criteria development and associated consultations, as appropriate, are not finalized by the effective dates listed in Table 13, all remaining interim measures identified in the individual RPAs shall be adopted as final for purposes of establishing aquatic life criteria in association

[29] As described in the *Reasonable and Prudent Alternatives* section, the interim RPAs for copper, cyanide, and zinc require restricting the size of mixing zones to 25 percent or less of stream volume and maintaining zones of passage around mixing zones for bull trout and Kootenai River white sturgeon. These practices will ensure that an adequate zone of passage exists for these species under all flow conditions, and will provide for biological monitoring and whole-effluent toxicity testing to ensure that permit limits are protective of the bull trout and the sturgeon and their prey species. This monitoring will be done at each discharge site by taking into account the localized conditions that affect the metal toxicity. Based on development of these site-specific limits and the associated monitoring of discharge levels, combined with the fact that the Service consults, as appropriate, with EPA over each new or reissued NPDES permit. Limiting mixing zone fractions to 1/4 (25 percent) of the receiving water discharge in flowing waters is effectively similar to reducing the criteria by about 0.25X (NMFS 2014a). Few if any adverse effects to listed species or habitats would be expected at about 0.25X the criteria concentrations. If minor adverse effects to the bull trout and the Kootenai River white sturgeon do occur, they would not significantly disrupt their breeding, feeding, or sheltering behavior with implementation of the RPAs; these minor adverse effects would not result in incidental take of the bull trout and white sturgeon.

with Idaho's water quality standards. If that is the case, no take of listed species under Service jurisdiction is anticipated under EPA's implementation of the proposed action under those conditions. In addition, this finding will be re-affirmed, as appropriate, as a result of individual consultations on a site-specific basis for each NPDES permit issued in the action area under permanent implementation of the interim RPAs. If such individual consultations reveal that take of listed species is reasonably certain to occur under implementation of the interim RPAs, that finding would constitute new information that will warrant re-initiation of consultation on the action addressed herein.

The Service anticipates that the chronic criterion for chromium (VI) and nickel, and the acute criterion for silver are likely to adversely affect the bull trout and the Kootenai River white sturgeon; the acute and chronic criteria for zinc are likely to adversely affect the Snake River snails and the Bruneau hot springsnail. However, based on our review of best available information as discussed in the *Effects of the Proposed Action* section above, exposure of these species to these metals at the proposed criteria concentrations is not likely to significantly disrupt their breeding, feeding, or sheltering behavior. In addition, this finding will be re-affirmed, as appropriate, as a result of individual consultations on a site-specific basis for each NPDES permit issued in the action area under implementation of the interim RPAs. If such individual consultations reveal that the chronic criterion for chromium (VI) and/or nickel, and/or the acute criterion for silver, and/or the acute and/or chronic criteria for zinc are reasonably certain to cause take of listed species, that finding would constitute new information that will warrant re-initiation of consultation on the action addressed herein.

The acute criterion for silver is also likely to adversely affect one component of the diets for both the bull trout and the sturgeon, but because these species feed on a variety of prey items and the form of silver found in natural waters is much less toxic than the ionic silver used in most laboratory exposures, the Service does not anticipate any incidental take of the bull trout and the Kootenai River white sturgeon being caused by implementation of the established acute criterion for silver because exposure of these species to this metal at the proposed criterion concentration is not likely to significantly disrupt their breeding, feeding, or sheltering behavior. In addition, this finding will be re-affirmed, as appropriate, as a result of individual consultations on a site-specific basis for each NPDES permit issued in the action area under implementation of the established acute criterion for silver. If such individual consultations reveal that take of listed species is likely to occur under implementation of the acute criterion for silver, that finding would constitute new information that will warrant re-initiation of consultation on the action addressed herein.

In summary, with implementation of the interim RPAs (for the jeopardy conclusions), or established criteria as identified above, the Service is not anticipating any incidental take of the Snake River physa, Bliss Rapids snail, Banbury Springs lanx, Bruneau hot springsnail, the bull trout, and the Kootenai River white sturgeon from EPA approval of the Idaho water quality criteria for toxic pollutants. On that basis, no Reasonable and Prudent Measures or Terms and Conditions are provided herein. This finding will be re-affirmed, as a result of individual consultations on a site-specific basis for each NPDES permit issued in the action area under implementation of the interim RPAs or established criteria, for the proposed action as appropriate. If such individual consultations reveal that take of listed species is likely to occur under implementation of the interim RPAs or the established criteria, that finding would constitute new information that will warrant re-initiation of consultation on the action addressed herein.

### 2.9.1 Effect of the Take

No incidental take is anticipated with implementation of the interim RPAs. EPA development of new criteria will require a new consultation and an Incidental Take Statement shall be provided, as appropriate, under that consultation.

### 2.9.2 Reasonable and Prudent Measures

No incidental take is anticipated, therefore, none are provided herein.

### 2.9.3 Reporting and Monitoring Requirements

As no incidental take is anticipated, the Service is not including Reporting and Monitoring requirements. However, EPA shall promptly notify the Service of any emergency or unanticipated situations arising during implementation of the proposed aquatic life criteria that may be detrimental to listed species.

Upon locating any dead, injured, or sick individuals of listed species as a result of discharges of the toxic pollutants addressed herein, such discharges shall be terminated and notification must be made within 24 hours to the Service's Division of Law Enforcement at (208) 378-5333. Additional protection measures may be developed through discussions with the Service.

## 2.10 Conservation Recommendations

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery programs, or to develop new information on listed species.

1. Conduct studies on appropriate surrogate species, or the species themselves where possible, with test waters having similar chemistry to waters within the action area, which contain both high particulate metal concentrations and dissolved concentrations near criteria concentrations and re-evaluate the proposed metals criteria that are currently based on dissolved concentrations.
2. Monitor and improve techniques (e.g., bioswales, vegetated buffers) to control and reduce chemical exposure resulting from nonpoint source pollution to listed species.
3. Develop and implement procedures to assess effects to habitat conditions for listed Snake River aquatic snails from anthropogenic impacts including aquifer recharge and fish hatchery operations.
4. Seek opportunities to cooperatively participate in the translocation of Banbury Springs lanx from the four known colonies to other suitable and protected coldwater spring habitats to ensure the continued existence of this species. See the 5-year status review for specific information on potential translocation sites (USFWS 2006b).

5. As described in NMFS 2014a, publish updated aquatic life criteria for silver that include a chronic criterion value, using a biotic ligand model (BLM) to account for factors that modify toxicity. Much of the fundamental research into the proof of principal, refinement and validation of the BLM-approaches to define metals bioavailability and toxicity was with silver (Di Toro et al. 2001; Paquin et al. 2002). As result, the BLMs available for silver may be more mature than those for any other metal except for copper (Niyogi and Wood 2004; this Opinion).
6. As described in NMFS 2014a, bioassessment of receiving waters has been required as a monitoring element for receiving waters in NPDES permits issued by EPA in Idaho; however, to our knowledge, the data collected has never been a factor in determining the adequacy of permit limits in renewal applications. The Service recommends that EPA develop an approach to effectively use pre- project bioassessment data in permitting decisions.
7. For recommending water quality criteria the fundamental information used should have the individual research studies and associated parameters identified (e.g. surrogates temperature parameters, water chemistry).

# 2.11 Reinitiation Notice

This concludes formal consultation on EPA's approval of Idaho Water Quality Standards. As provided in 50 CFR §402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been maintained (or is authorized by law) and if:

1. The amount or extent of incidental take is exceeded.
2. New information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this Opinion.
3. The agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in this Opinion.
4. A new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, any operations causing such take must cease pending reinitiation.

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**UNITED STATES DEPARTMENT OF COMMERCE**
**National Oceanic and Atmospheric Administration**
NATIONAL MARINE FISHERIES SERVICE
West Coast Region
7600 Sand Point Way N.E.
Seattle, Washington 98115

May 7, 2014 **Refer to NMFS No: 2000-1484**

Dan Opalski, Director
Office of Water and Watersheds
U.S. Environmental Protection Agency
1200 Sixth Avenue
Seattle, Washington 98101

Re: Final Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for Water Quality Toxics Standards for Idaho

Dear Mr. Opalski:

The enclosed document contains a biological opinion (Opinion) and letters of concurrence prepared by the National Marine Fisheries Service (NMFS) pursuant to section 7(a)(2) of the Endangered Species Act (ESA) on the effects of approving the Idaho Water Quality Standards for toxic substances. In this Opinion, NMFS concludes that the action, as proposed, is likely to jeopardize the continued existence of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon, and Snake River Basin steelhead and result in the destruction or adverse modification of designated critical habitat for Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon, and Snake River Basin steelhead.

As required under the ESA for consultations concluding with jeopardy and adverse modification determinations, NMFS discussed with U.S. Environmental Protection Agency (EPA), the availability of reasonable and prudent alternatives that the EPA can take to avoid violation of the EPA's ESA section 7(a)(2) responsibilities (50 CFR 402.14(g)(5)). Reasonable and prudent alternatives refer to alternative actions identified during formal consultation: (1) That can be implemented in a manner consistent with the intended purpose of the action; (2) that can be implemented consistent with the scope of the Federal agency's legal authority and jurisdiction; (3) that are economically and technologically feasible; and (4) that NMFS believes would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat (50 CFR 402.02). The Opinion includes a reasonable and prudent alternative which NMFS believes can be implemented to avoid jeopardy and adverse modification of critical habitat, while meeting each of the other requirements listed above. Accordingly, NMFS prepared an incidental take statement describing and exempting the extent of incidental take reasonably certain to occur under the reasonable and prudent alternative.

This document also includes the results of our analysis of the action's likely effects on essential fish habitat (EFH) pursuant to section 305(b) of the Magnuson-Stevens Fishery Conservation and

Management Act, and includes three Conservation Recommendations to avoid, minimize, or otherwise offset potential adverse effects on EFH. These Conservation Recommendations are a non-identical set of the ESA terms and conditions.

If you have questions regarding this consultation, please contact David Mabe, Snake Basin Office, (208) 378-5698.

Sincerely,

William W. Stelle, Jr.
Regional Administrator

Enclosure

cc: D. Miller – IOSC
R. Holder – USFWS
J. Martin – DOJ
M. Lopez – NPT
C. Colter – SBT

bcc: SBAO - File copy, Read File,
GCNW – L. Beale

Mebane:Final:2014 May 7 EPA WQS Toxics.docx:5/7/14:NMFS No: 2000-1484

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**Endangered Species Act Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat (EFH) Consultation**

Idaho Water Quality Standards for Toxic Substances

NMFS Consultation Number: 2000-1484

Action Agency: United States Environmental Protection Agency

Affected Species and Determinations:

| ESA-Listed Species | Status | Are Actions Likely to Adversely Affect Species or Critical Habitat? | Are Actions Likely To Jeopardize the Species? | Are Actions Likely To Destroy or Adversely Modify Critical Habitat? |
|---|---|---|---|---|
| Snake River Basin steelhead | Threatened | Yes | Yes | Yes |
| Snake River Spring/Summer Chinook Salmon | Threatened | Yes | Yes | Yes |
| Snake River Fall Run Chinook Salmon | Threatened | Yes | Yes | Yes |
| Snake River Sockeye Salmon | Endangered | Yes | Yes | Yes |

| Fishery Management Plan That Describes EFH in the Project Area | Does Action Have an Adverse Effect on EFH? | Are EFH Conservation Recommendations Provided? |
|---|---|---|
| Pacific Coast Salmon | Yes | Yes |

Consultation Conducted By: National Marine Fisheries Service, West Coast Region

Issued By:

William Stelle

William W. Stelle, Jr.
Regional Administrator

Date: May 7, 2014

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## APPENDICES

# TABLES

## FIGURES

## ACRONYMS

| | |
|---|---|
| ACR | Acute-to-Chronic Ratio |
| ADR | Alternative Dispute Resolution |
| Ag | Silver |
| Ah | Aryl Hydrocarbon |
| ALC | Aquatic Life Criterion |
| As | Arsenic |
| ASTM | American Society for Testing and Materials |
| ATPase | A class of enzymes that catalyze the decomposition of adenosine triphosphate and is essential for metabolism in all known forms of life |
| AWQC | Ambient Water Quality Criteria |
| BA | Biological Assessment |
| BAF | Measured Bioaccumulation Factor |
| BCF | Bioconcentration Factor |
| BLM | Biotic Ligand Model |
| BMC | Benchmark Concentrations |
| BMP | Best Management Practice |
| BSAF | Biota-sediment Accumulation Factors |
| Ca | Calcium |
| CCC | Criterion Continuous Concentration |
| CCU | Cumulative Criterion Units |
| Cd | Cadmium |
| CERLA | Comprehensive Environmental Response, Compensation, and Liability Act (Superfund) |
| CF | Conversion Factor |
| cfs | cubic feet per second |
| Chronic Value | A synonym for MATC in the context of water quality criteria |
| CMC | Criterion Maximum Concentration |
| CN | Cyanide |
| COE | Army Corps of Engineers |
| Cr | Chromium |
| CRB | Columbia River Basin |
| CrIII | Trivalent Chromium |
| CrIV | Hexavalent Chromium |
| Cu | Copper |
| CWA | Clean Water Act |
| DDD | Dichlorodiphenyldichloroethane |
| DDE | Dichlorodiphenylethylene |
| DDT | Dichlorodiphenyltrichloroethane |

| | |
|---|---|
| DOC | Dissolved Organic Carbon |
| DOM | Dissolved Organic Matter |
| DPS | Distinct Population Segment |
| DQA | Data Quality Act |
| dw | dry weight |
| EC | Effects Concentration |
| $EC_{50}$ | Concentration that caused effects to 50% of the test population |
| ECp | Effect Concentration Percentile |
| EEC | extreme effect |
| EFH | Essential Fish Habitat |
| EFSFSR | East Fork of the South Fork Salmon River |
| ELS | Early Life Stages |
| EMMA | Environmental Mercury Mapping, Modeling, and Analysis |
| EPA | U.S. Environmental Protection Agency |
| ESA | Endangered Species Act |
| ESU | Evolutionary Significant Unit |
| FAV | Final Acute Value |
| FDA | Food and Drug Administration |
| FPC | Fish Passage Center |
| GMAV | Genus Mean Acute Value |
| Hg | Mercury |
| HH | Human-Health |
| HOC | hydrophobic organic compounds |
| HUC | Hydrologic Unit Codes |
| ICE | Interspecies Correlation Estimates |
| ICTRT | Interior Columbia River Basin Technical Recovery Team |
| IDEQ | Idaho Department of Environmental Quality |
| IDFG | Idaho Department of Fish and Game |
| ISAB | Independent Scientific Advisory Board |
| ITS | Incidental Take Statement |
| IWQS | Idaho Water Quality Standards |
| LAA | Likely to Adversely Affect |
| $LC_{50}$ | Lethal Concentration of 50% |
| LGD | Lower Granite Dam |
| LOEC | Lowest Observed Effects Concentration |
| ln(X) | Natural logarithim of the number "X" |
| MATC | Maximum Acceptable Toxicant Concentration |
| MEC | Midrange Effect |
| Mg | Magnesium |
| mg/L | milligram per liter, or parts per million |

| | |
|---|---|
| Mixing Zone | an area where an effluent discharge undergoes initial dilution and is extended to cover the secondary mixing in the ambient water body |
| MLDE | Maximum Likelihood Estimate |
| MPG | Major Population Groups |
| MSA | Magnuson-Stevens Fishery Conservation and Management Act |
| ng/L | nanogram per liter, or parts per trillion |
| Ni | Nickel |
| NLAA | Not Likely to Adversely Affect |
| NMFS | National Marine Fisheries Service |
| NOEC | No-Observed Effects Concentration |
| Nominal Concentrations | In toxicity testing, the intended or target concentration is often called the nominal concentration to distinguish it from measured concentrations, i.e., those concentrations that had been analytically verified. For economy, some studies only report nominal concentrations. In this Opinion, study results that were derived solely from "nominal data" were discounted as not being "best available science |
| NPDES | National Pollution Discharge Elimination System |
| NTR | National Toxics Rule (EPA, 57 Fed. Reg. 60848 (Dec. 22, 1992)) |
| NTR Criteria | Water quality criteria for certain toxic pollutants promulgated by EPA for several states. (EPA 1992, 57 Fed. Reg. 60848 (Dec, 22, 1992)) |
| Opinion | Biological Opinion |
| Pb | Lead |
| PCB | Polychlorinated Biphenyl |
| PCE | Primary Constituent Elements |
| PCP | Pentachlorophenol |
| PFMC | Pacific Fishery Management Council |
| RER | Rough Endoplasmic Reticulum |
| RM | River Mile |
| RPA | Reasonable and Prudent Alternative |
| RPM | Reasonable and Prudent Measures |
| RUP | Registered Use Product |
| Se | Selenium |
| SEC | Sediment Effect Concentration |
| SeMe | Selenomethionine |
| SER | Smooth Endoplasmic Reticulum |
| SMAV | Species Mean Acute Values |
| SMI | Stream Macroinvertebrate Index |
| SQG | Sediment Quality Guidelines |
| SSD | Species Sensitivity Distribution |
| TCDD | Tetrachlorodibenzo-p-dioxin |
| TCM | Thompson Creek Mine |
| TEC | Threshold Effect |

| | |
|---|---|
| TEF | Toxicity Equivalence Factor |
| TEQ | Toxicity equivalence Quantity |
| TIE/TRE | Toxicity Identification Evaluation and Toxicity Reduction Evaluation |
| TMDL | Total Maximum Daily Loads |
| TOC | Total Organic Carbon |
| TOST | Test of Significant Toxicity |
| TTF | Trophic Transfer Factor |
| TU | Toxic Unit |
| USFWS | U.S. Fish and Wildlife Service |
| USGS | U.S. Geological Survey |
| VSP | Viable Salmonid Populations |
| WAD | Weak Acid Dissociable |
| WER | Water Effects Ratio |
| WET | Whole- Effluent Toxicity |
| WQBEL | Water Quality Based Effluent Limitation |
| WQS | Water Quality Standards |
| ww | wet weight |
| YOY | Young-Of-year |
| Zn | Zinc |
| µg/L | microgram per liter or parts-per-billion |
| 1Q10 | The lowest 1-day average of streamflows occurring in a 10-year period |
| 7Q10 | The lowest 7-day running average of streamflows occurring in a 10-year period |

# 1. INTRODUCTION

This introduction section provides information relevant to the other sections of this document and is incorporated by reference into Sections 2 and 3 below.

## 1.1. Background

The biological opinion (Opinion) and incidental take statement (ITS) portions of this consultation were prepared by the National Marine Fisheries Service (NMFS) in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531, *et seq*.), and implementing regulations at 50 CFR 402.

NMFS also completed an Essential Fish Habitat (EFH) consultation. It was prepared in accordance with section 305(b)(2) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. 1801, *et seq*.) and implementing regulations at 50 CFR 600.

The Opinion and EFH Conservation Recommendations are both in compliance with section 515 of the Treasury and General Government Appropriations Act of 2001 (Public Law 106-5444) ("Data Quality Act") and underwent pre-dissemination review.

## 1.2. Consultation History

This Opinion is based on information provided originally in U.S. Environmental Protection Agency's (EPA's) July 2000 biological assessment (BA) and modified in a December 2013 letter. In the interim there were many interactions including telephone conversations, meetings and written correspondence and regulatory changes that occurred to arrive at the final action as described in section 1.3 of this Opinion. The following is a summary of those interactions. A complete record of this consultation is on file at the Snake Basin Office in Boise, Idaho.

Section 303 of the Clean Water Act (CWA) mandates that states adopt water quality standards (WQS) to restore and maintain the chemical, physical, and biological integrity of the nation's waters. The WQS consist of beneficial uses to protect both aquatic life communities and recreational and subsistence based uses (i.e. salmonid spawning, cold water biota, primary or secondary contact recreation) designated for specific water bodies and water quality criteria to protect uses. States have primary responsibility for developing appropriate beneficial uses for water bodies in their state. States review and, if appropriate, revise their WQS on a triennial basis in accordance with CWA section 303(c). Also under CWA section 303(c), EPA must review and approve or disapprove any revised or new standards. If EPA disapproves any portion of the state standards the state has 90 days to adopt the changes specified by the EPA, after which time the EPA must propose and promulgate standards for the state.

On June 25, 1996, staff from EPA's Region 10 completed a review of the Idaho Water Quality Standards (IWQS) adopted August 24, 1994. During this review, the EPA disapproved seven elements within the state's WQS. Most of these elements have since been revised by Idaho and approved by the EPA.  These 1996 approvals were included as part of EPA's (2000) BA initiating this consultation.  The elements which EPA disapproved and did not subsequently approve were not included in EPA's BA or the proposed action for this consultation.

As a result of several meetings held in 1999 between the U.S. Fish and Wildlife Service (USFWS), NMFS, EPA, and Idaho Department of Environmental Quality (IDEQ), all agencies agreed that two BAs should be developed.  The first BA would consist of EPA evaluation of Idaho's numeric water quality criteria for 22 toxic constituents (listed below).

In a letter of March 23, 2000, the IDEQ informed the EPA, NMFS, and the USFWS that it wished to be considered an "applicant" to this action for this consultation as defined by 50 CFR § 402.02.

On August 9, 2000, EPA submitted its final BA to USFWS and NMFS and requested initiation of formal consultation under section 7 of the ESA.  The BA concluded that the proposed criteria were not likely to adversely affect (NLAA) Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River Basin steelhead for the following parameters:

> Criteria for aldrin/dieldrin, chlordane, Dichlorodiphenyltrichloroethane (DDT), endrin, heptachlor, lindane, polychlorinated biphenyl (PCB), pentachlorophenol (PCP), toxaphene, trivalent chromium (Cr[III]) and hexavalent chromium (Cr[VI]), nickel(Ni), and silver (Ag);
>
> Acute and chronic criteria for arsenic (As), cadmium (Cd), copper (Cu), cyanide (Cn), endosulfan, mercury (Hg) lead (Pb), and zinc (Zn); and
>
> Acute criteria for mercury (Hg) and selenium (Se).

The BA concluded that Idaho's proposed criteria were likely to adversely affect (LAA) Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River Basin steelhead, for the following parameters:

> Chronic criteria for selenium (Se).

The BA did not include an analysis of effects for southern resident killer whales, which are listed as endangered and rely on listed salmonids as a food source.  NMFS will complete an analysis on southern resident killer whales within 6 months.

On September 4, 2003, NMFS circulated a draft Opinion to EPA, USFWS, and the IDEQ for review.  This was followed by a series of conference calls and meetings.  No formal comments were received.  Instead, EPA representatives proposed that all parties commit to working through the technical and policy issues through a facilitated alternative dispute resolution (ADR) process. The disputes involved effects determinations and the methods used to determine effects.

The EPA, NMFS, USFWS, and IDEQ representatives formed a technical committee and policy committee and participated in a series of facilitated meetings and conference calls, supported by EPA's ADR contractor. The interagency group did not reach final agreement on a set of recommended actions for completing the consultation. A final report was issued by the ADR contractor on September 22, 2005.

In 2005, IDEQ began negotiated rulemaking to revise the criteria values under consultation. On April 11, 2006, Idaho formally amended its water quality criteria. These criteria were subsequently approved by EPA in 2007, subject to ESA consultation.

On September 2, 2010, EPA provided NMFS a revised BA for cadmium criteria only, and asked NMFS to concur with their determination that their approval of Idaho's cadmium criteria was protective of and NLAA Snake River salmon and steelhead. On January 31, 2011, NMFS wrote to EPA concurring with their determination, accompanied by an independent review (NMFS 2011).

On February 6, 2013, NMFS provided a draft Opinion to EPA for comment. The EPA and NMFS met several times in 2013 and worked on modification to the Opinion and changes to the proposed action.

On November 22, 2013, EPA advised NMFS that they were revising their action for several criteria values under consultation to match those updated by IDEQ in 2006 and subsequently approved by EPA, subject to consultation. The revisions consisted of new acute and chronic criteria values for arsenic, chromium, nickel, and zinc. No new technical analyses of effects were included with the revised action letter and EPA's determinations were unchanged for these criteria.

### 1.3. Proposed Action

"Action" means all activities or programs of any kind authorized, funded or carried out, in whole or in part, by federal agencies. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration.

The CWA requires all states to adopt WQS to restore and maintain the physical, chemical, and biological integrity of the Nation's waters. Section 303(c)(2)(E) of the CWA requires states to adopt chemical-specific, numeric criteria for priority toxic pollutants. The criteria must protect state-designated beneficial uses of water bodies. Development of WQS is primarily the responsibility of the states, but adoption of the WQS is subject to approval by the EPA.
Since 1980, the EPA has published numerous criteria development guidelines for states and tribes and recommended national criteria for numerous pollutants. The national criteria include recommended acute and chronic criteria for the protection of aquatic life resources. States and tribes may choose to adopt EPA's recommended criteria, or modify these criteria to account for site-specific or other scientifically defensible factors. The state of Idaho has adopted criteria for toxic pollutants (IDAPA 58.01.02, 250.02 (a)(iv)). As initially adopted, all of the criteria were identical to criteria promulgated by EPA for several in EPA's 1992 National Toxics Rule (NTR)

(57 Fed. Reg. 60848, Dec. 22, 1992) (EPA 2000a). The state of Idaho subsequently revised several criteria, as listed in Table 1.3.1. The EPA has approved Idaho's adoption, subject to consultation for 23 toxic pollutants (Table 1.3.1). The EPA is consulting only on those aquatic life criteria for the chemicals in Table 1.3.1. There are many criteria for additional water quality parameters in the IWQS that are not part of the proposed action. Those primarily affecting fish include temperature, dissolved oxygen and sediment. Any impaired waters are shown in Idaho's 303(d) list are discussed further in the baseline section.

The IWQS for aquatic life contain two expressions of allowable magnitude that are constrained by allowable exposure duration and frequency:

> An acute, or criterion maximum concentration (CMC), to protect against short-term effects, that is not to be exceeded on average for longer than 1 hour and more than once every 3 years.
>
> A chronic, or criterion continuous concentration (CCC), to protect against long-term effects, that is not to be exceeded on average for longer than 4 days and more than once every 3 years.

### 1.3.1. Idaho's Water Quality Standards for Toxic Pollutants

The EPA has approved, subject to this consultation, Idaho's aquatic life criteria for 11 organic chemicals and replacement of existing aquatic life criteria for 11 metals. The proposed aquatic life criteria would apply to all waters in the state that are protected for aquatic life beneficial uses. The proposed numeric criteria are ambient water quality criteria (AWQC), which are concentrations of each pollutant measured in the water column. Under EPA policy, states may choose to adopt metals criteria measured as either dissolved metal or total recoverable metal. Idaho's aquatic life criteria for metals were based on total recoverable metal (dissolved + suspended). The proposed action would change the aquatic life criteria to concentrations based on dissolved metals only, using a conversion factor (CF) to account for the suspended fraction. With the use of dissolved criteria, water samples are filtered to remove suspended solids before analysis.

The proposed IWQS will apply to actions that require National Pollutant Discharge Elimination System (NPDES) permits, to development of total maximum daily loads (TMDLs) in streams with impaired water quality, and in situations where remedial actions are required to clean up spills or contaminated sites. When a TMDL is needed to regulate discharges into an impaired water body, the dissolved metals criteria must be converted or translated back to a total recoverable value so that the TMDL calculations can be performed. The translator can simply be the CF (i.e., divide the dissolved criterion by the CF to get back to the total criterion), or a dissolved-to-total ratio based on site-specific total/dissolved metal concentrations in the receiving water.

For some of the pollutants subject to this consultation, Idaho has also adopted criteria to protect human health from risk from exposure to the substances through eating fish or shellfish or ingestion of water through recreating on water. Although EPA is not consulting on the human health-based criteria, on a practical level, permitted discharges to a given water body would be constrained by the most stringent applicable criteria. In other words, the human health criteria will constrain discharge levels where they are more stringent than the aquatic life criteria. During the pendency of this consultation, Idaho has further revised some of the criteria under consultation. The EPA has updated its action to reflect these revisions and they are being consulted on as shown Table 1.3.1.

The application of AWQC is based on the principle of designated beneficial uses of water. Together, AWQC and use designations are used to meet the primary objective of the CWA – to "restore and maintain the chemical, physical and biological integrity of the Nation's waters." A further goal of the CWA is that wherever attainable, an interim goal of water quality is to provide "for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water." (Clean Water Act, §101(a)).

**Table 1.3.1. Ambient Water Quality Criteria for toxic pollutants submitted for consultation in EPA's 2000 Biological Evaluation. Also shown are AWQC that have subsequently been revised by the State of Idaho (Idaho Department of Environmental Quality 2011). In two parts, inorganic and organic substances:**

**Part 1. Criteria for metals and other inorganic substances**

| **Substance** (except as noted, as 0.45 µm filtered "dissolved" concentrations) | **Criteria evaluated in EPA's 2000 BA (µg/L)** | | **Idaho revised criteria included in EPA's updated action (25 November 2013) (µg/L)** | | **Relevant IWQS human health based criteria also applicable to waters in the action area (IDEQ 2011)** |
|---|---|---|---|---|---|
| | **Acute** | **Chronic** | **Acute** | **Chronic** | |
| Arsenic (As) | 360 | 190 | 340 | 150 | 10 µg/L human health criterion also applies |
| Cadmium (cd) [f] | 3.7 | 1.0 | 1.3 | 0.6 | |
| [Note: Cd was included in the BA but was subsequently consulted on separately. See NMFS (2011)] | | | | | |
| Copper (Cu) [b] | 17 | 11 | 17 | 11 | |
| Cyanide (CN, weak acid dissociable) | 22 | 5.2 | 22 | 5.2 | |
| Lead (Pb) [b, c] | 65 | 2.5 | 65 | 2.5 | |
| Mercury (Hg) | 2.1 | 0.012 (unfiltered) | g | g | 0.3 mg/kg in fish tissue, fresh weight |
| Selenium (Se) | 20 | 5.0 (unfiltered) | 20 | 5. (unfiltered | |
| Zinc (Zn) [b] | 114 | 105 | 120 | 120 | |
| Chromium (Cr) (III) [b] | 550 | 180 | 570 | 74 | |
| Chromium (Cr) (VI) | 15 | 10 | 16 | 11 | |
| Nickel (Ni)[b] | 1,400 | 160 | 470 | 52 | |
| Silver (Ag)[b] | 3.4 | – | 3.4 | – | |

(µg/L: micrograms per liter; Metals criteria are shown for a water hardness of 100 mg/L).

## Part 2. Criteria for organic toxic substances

| Substance | Aquatic life criteria evaluated in EPA's 2000 BA (µg/L) | | Human-health based AWQC that also apply to waters designated to support "cold water biota" or "salmonid spawning" and to critical habitats for listed species in the action area (µg/L) | Idaho criteria that were revised subsequent to EPA's 2000 BA (µg/L) [(h)] |
|---|---|---|---|---|
| | **Acute** | **Chronic** | | **Idaho** |
| Endosulfan (α and β) | 0.22 | 0.056 | 2 | 89 |
| Aldrin | 3 | – | 0.00014 | 0.000050 |
| Chlordane | 2.4 | 0.0043 | 0.00057 | 0.00081 |
| 4,4'-DDT | 1.1 | 0.001 | 0.00059 | 0.00022 |
| Dieldrin | 2.5 | 0.0019 | 0.00014 | 0.000054 |
| Endrin | 0.18 | 0.0023 | 0.81 | 0.060 |
| Heptachlor | 0.52 | 0.0038 | 0.00021 | 0.000079 |
| Lindane (gamma-BHC) | 2 | 0.08 | 0.063 | 1.8 |
| Polychlorinated biphenyls (PCBs) | N/A | 0.014 | 0.000045 | 0.000064 |
| Pentachlorophenol (PCP) | 20[e] | 13[e] | 6.2 | 3 |
| Toxaphene | 0.73 | 0.0002 | 0.00075 | 0.00028 |

– - no applicable criteria

a. Conversion factors for translating between dissolved and total recoverable criteria.

b. For comparison purposes, the values displayed in this table correspond to a total hardness of 100 mg/l $CaCO_3$ and a Water Effects Ratio (WER) of 1.0. Criteria for these metals are actually expressed as a function of total hardness (mg/L as $CaCO_3$), and the following equation:
Acute Criteria = WER exp($m_A$[ln(hardness)]+$b_A$) x Acute Conversion Factor
Chronic Criteria = WER exp($m_C$[ln(hardness)]+$b_C$) x Chronic Conversion Factor
where:

| Metal | $m_A$[f] | $b_A$[f] | $m_C$[f] | $b_C$[f] |
|---|---|---|---|---|
| Chromium (III) | 0.8190 | 3.688 | 0.8190 | 1.561 |
| Copper | 0.9422 | -1.464 | 0.8545 | -1.465 |
| Lead | 1.273 | -1.460 | 1.273 | -4.705 |
| Nickel | 0.8460 | 3.3612 | 0.8460 | 1.1645 |
| Silver | 1.72 | -6.52 | N/A | N/A |
| Zinc | 0.8473 | 0.8604 | 0.8473 | 0.7614 |

The term "exp" represents the base e exponential function. $m_A$ and $m_c$ are the slopes of the relationship for hardness, while $b_A$ and $b_C$ are the Y-intercepts for these relationships.

c. The conversion factor for lead is hardness dependent. The values shown in the table correspond to a hardness of 100 mg/L $CaCO_3$. Conversion factors for lead: Acute and Chronic- CF=1.46203-[(ln(hardness))x(0.145712)]

d. Criteria expressed as Weak Acid Dissociable

e. Criteria for pentachlorophenol increase as pH increases and are calculated as follows:
Acute Criterion = exp(1.005 (pH) - 4.830)
Chronic Criterion = exp(1.005 (pH) - 5.290) Values shown in the table are for pH 7.8

f. Cadmium aquatic life criteria are listed for descriptive purposes only. Cadmium aquatic life criteria were originally part of EPA's action and the consultation package (EPA 2000a). However in 2006, Idaho substantially revised their aquatic life criteria for cadmium, which EPA (2010a) subsequently proposed separate approval of, and initiated consultation on the revised cadmium criteria. EPA's (2010a) determination was that Idaho's 2006 revised cadmium criteria was NLAA listed salmonids, to which NMFS (2011) concurred.

g. The state of Idaho repealed the water column aquatic life criteria for mercury in 2006, based upon IDEQ's (2005) analysis that concluded the available science no longer supported EPA's (1985g) aquatic life criteria, and that a fish tissue based human-health criteria would be better supported by the science, be adequate to protect aquatic life, and would be more stringent than the 1985 chronic aquatic life criterion of 0.012 µg/L. EPA disapproved Idaho's repeal of its water column acute and chronic mercury criteria on policy grounds that, an exception for California notwithstanding, water column based aquatic criteria were required for Idaho, Idaho's criteria did not include a sufficiently detailed implementation for translating the human health tissue criterion to a protective aquatic life criteria that could be used with effluent limits (Gearheard 2008). The disapproval addressed policy interpretations and was silent on IDEQ's arguments that the EPA (1985g) mercury chronic was outdated and that a 0.3 mg/kg fish tissue criterion was more protective. Gearheard (2008) considered the 0.012 µg/L chronic criterion to be effective for NPDES discharge permits and TMDLs issued by EPA, although the criterion remains repealed under state law and nowhere appears in Idaho administrative rules.

h. Although Idaho's revised human health criteria are considerably more stringent than the previous human health criteria, EPA has not approved these revised criteria and EPA does not consider the more stringent criteria to be effective for Clean Water Act purposed.

### 1.3.2. Application of the IWQS for Metals

Per EPA's guidance, states, when adopting criteria for metals, may adopt criteria measured as either dissolved or total recoverable metal. The Idaho metals criteria under consultation are expressed as dissolved metals, meaning that water samples are filtered to remove suspended solids before analysis.

Metals and inorganic toxic substances addressed in this consultation include: As, cyanide, chromium (III), chromium (VI), copper, lead, mercury, nickel, selenium, silver, and zinc. For several of these chemicals, the water quality criteria are equation-based, meaning the criteria applicable to a particular site vary based on site-specific conditions. The equation-based metals are chromium (III), chromium (VI), copper, lead, mercury, nickel, silver, and zinc. To determine criteria for these metals for a given water body, site-specific data must be obtained, input to an equation, and numeric criteria computed. There are three types of site-specific data that may be necessary to determine and/or modify the criteria for these metals at a site: water hardness, CF and translators, and water effect ratios. Following is a brief description of these types of data.

The general equation for a hardness-based acute (CMC) or chronic (CCC) criterion with respect to total metal concentration (dissolved plus particulate) is:

CMC or CCC (total recoverable) = $e^{(m[\ln(\text{hardness})]+b)}$

Note that this is algebraically equivalent to the simpler expression:

CMC or CCC (total recoverable) = $K \cdot (\text{hardness})^m$

where $K = e^b$. When the m-exponent is close to 1.0, the relationship is approximately linear.

Dissolved concentrations are evaluated using a total-to-dissolved CF that is based on the fraction of the metal that was in a dissolved form during the laboratory toxicity tests and that was used to develop the original total recoverable based criteria. The Idaho AWQC as evaluated in the BA are dissolved. The CFs for the metals are in the footnote to Table 1.3.1. The appropriate equation is:

CMC or CCC (dissolved) = $CF \cdot e^{(m[\ln(\text{hardness})]+b)} = CF \cdot K \cdot (\text{hardness})^m$

There is an added level of complexity in the computations of criteria for cadmium and lead, because the CFs for these metals also vary with water hardness. For those metals that are hardness dependent, EPA calculates NPDES permit limits and load allocations for TMDLs using the $5^{th}$ percentile of the available ambient and or effluent hardness values

If a TMDL is needed to regulate discharges into an impaired water body, the dissolved criterion must be converted or translated back to a total recoverable value so that the TMDL calculations can be performed. The translator can simply be the CF (i.e., divide the dissolved criterion by the CF to get back to the total criterion), or site-specific data on total and dissolved metal concentrations in the receiving water are collected and a dissolved-to-total ratio is used as the translator.

Equations for trivalent chromium, copper, lead, nickel, silver, and zinc also include a Water Effects Ratio (WER), a number that acts as a multiplication factor. If no site-specific WER is determined, then the WER is presumed to be 1 and does not modify the equation result. A WER is intended to account for the difference in toxicity of a metal in site water relative to the toxicity of the same metal in reconstituted laboratory water. The reason is that natural waters commonly contain constituents which "synthetic" or "reconstituted" laboratory waters lack, such as dissolved organic compounds, that may act to bind metals and reduce their bioavailability. Where such constituents act to modify the toxicity of a metal in a site water compared to the toxicity of the same metal in laboratory water, a "water effect" is observed. The EPA has provided procedures and requirements for determining "site-specific" WER values, which include extensive comparative toxicity testing with several test organisms and statistical analysis of results (Stephan *et al*. 1994b) (see Section 2.4.1.8 for additional discussion). The example provided below illustrates the basic principle in defining a WER value.

*Example WER calculation:*

Suppose the lethal concentration of 50% of test organisms ($LC_{50}$) of copper in site water is 15 µg/L
Suppose the $LC_{50}$ of copper in laboratory water is 10 µg/L
Assume a site hardness of 100 mg/L
The freshwater CF for copper = 0.96
Acute AWQC for total recoverable copper without the WER = 18 µg/L

A $LC_{50}$15 µg/L in site water and a laboratory water $LC_{50}$10 µg/L yields a WER of 1.5. Then:

Cu Site-Specific CMC=WER x CF x $e^{(m[\ln(40)]+b)}$
=1.5 x 0.96 x 18 µg/L
=24 µg/L

In this hypothetical example, this approach yielded a site-specific criterion that is higher than the concentration killing 50% of a sensitive organism in the same site water, which is one of the logical problems with the WER approach to setting metals criteria. Additional discussion of implementation of WERs is provided in section 2.4.1.8. The "Water-Effect Ratio" Provision.

## 1.4. Action Area

"Action area" means all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR 402.02).

For this project, the action area includes all watersheds within Idaho that contain anadromous species or their habitats (Figure 1.4.1) or upstream areas where discharges occur that may affect listed salmon, steelhead or their habitat. Table 1.4.1 lists all the 4$^{th}$ field hydrologic unit codes (HUCs) that contain listed salmon or steelhead. Each of these HUCs is located within the larger hydrologic unit, Lower Snake subregion (HUC 1706).

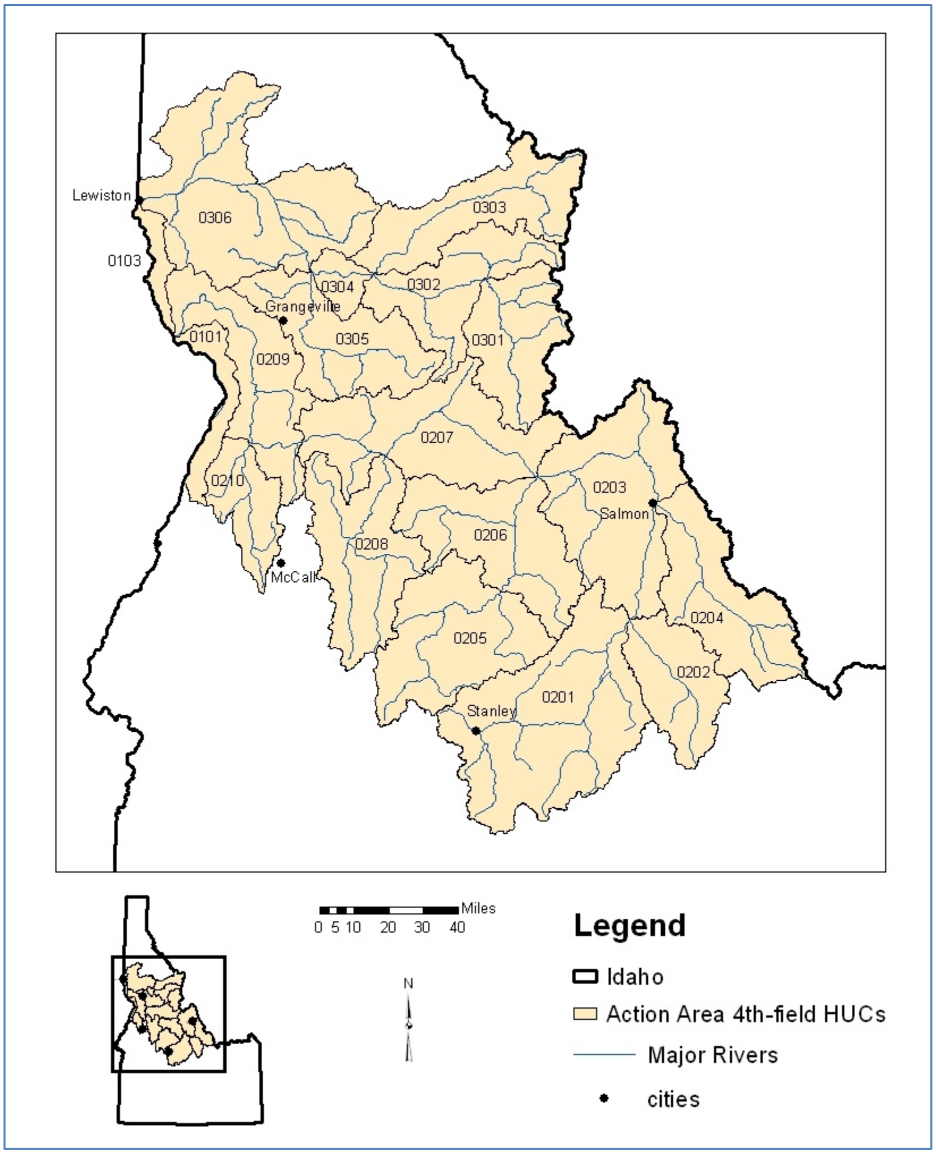



**Figure 1.4.1. Fourth-field hydrologic unit codes (HUCs) containing list salmon or steelhead. Each HUC is labeled with the last 4 digits of the 8-digit HUC code. The first 4 digits are 1706, Lower Snake subregion.**

**Table 1.4.1. Fourth field HUCs containing listed salmon or steelhead.**

| HUC Number | HUC Name | HUC Number | HUC Name |
|---|---|---|---|
| 17060101 | Hells Canyon | 17060209 | Lower Salmon |
| 17060103 | Lower Snake-Asotin | 17060210 | Little Salmon |
| 17060201 | Upper Salmon | 17060301 | Upper Selway |
| 17060202 | Pahsimeroi | 17060302 | Lower Selway |
| 17060203 | Middle Salmon-Panther | 17060303 | Lochsa |
| 17060204 | Lemhi | 17060304 | Middle Fork Clearwater |
| 17060205 | Upper Middle Fork Salmon | 17060305 | South Fork Clearwater |
| 17060206 | Lower Middle Fork Salmon | 17060306 | Clearwater |
| 17060207 | Middle Salmon-Chamberlain | 17060307 | Upper North Fork Clearwater |
| 17060208 | South Fork Salmon | 17060308 | Lower North Fork Clearwater |

The action area is used by all the freshwater life history stages (spawning, rearing, and migration) of threatened Snake River spring/summer and fall Chinook salmon, Snake River sockeye salmon, and Snake River Basin steelhead. Designated critical habitat for fall Chinook includes all reaches of the Snake River from the confluence of the Columbia River, upstream to Hells Canyon Dam; the Palouse River from its confluence with the Snake River upstream to Palouse Falls; the Clearwater River from its confluence with the Snake River upstream to its confluence with Lolo Creek; the North Fork Clearwater River from its confluence with the Clearwater River upstream to Dworshak Dam; and the Salmon River reaches in the lower Salmon hydrologic unit. Designated critical habitat for the Snake River spring/summer Chinook salmon includes all river reaches presently or historically accessible to the species (64 FR 57399; October 25, 1999). Within Idaho, designated critical habitat for sockeye salmon includes the Snake and Salmon Rivers; Alturas Lake Creek; Valley Creek; and Stanley, Redfish, Yellowbelly, Pettit, and Alturas Lakes (including their inlet and outlet creeks). Designated critical habitat for Snake River Basin steelhead includes specific reaches of streams and rivers, as published in the Federal Register (70 FR 52630; September 2, 2005). The action area also contains EFH for Chinook salmon and coho salmon (Pacific Fishery Management Council [PFMC] 1999).

The Snake River below the Idaho border is not considered part of the action area because it is subject to water quality standards in Oregon and Washington and either have been or will be subject to separate consultations. EPA and the state of Idaho are responsible to ensure that downstream standards are attained at the state border (40 CFR 131.10(b)). For example the Potlatch NPDES permit which discharges into the Snake River near the Washington border undergoes a 401 certification review by both states to assure it meets all applicable criteria within both states.

## 2. ENDANGERED SPECIES ACT: BIOLOGICAL OPINION AND INCIDENTAL TAKE STATEMENT

The ESA establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat on which they depend. Section 7(a)(2) of the ESA requires Federal agencies to consult with the United States Fish and Wildlife Service, NMFS, or both, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Section 7(b)(3) requires that at the conclusion of consultation, the Services provide an Opinion stating how the agencies' actions will affect listed species or their critical habitat. If incidental take is expected, Section 7(b)(4) requires the provision of an ITS specifying the impact of any incidental taking, and including reasonable and prudent measures to minimize such impacts.

### 2.1. Introduction to the Biological Opinion

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species, or adversely modify or destroy their designated critical habitat. The jeopardy analysis considers both survival and recovery of the species. The adverse modification analysis considers the impacts to the conservation value of the designated critical habitat.

"To jeopardize the continued existence of a listed species" means to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR 402.02).

This Opinion does not rely on the regulatory definition of 'destruction or adverse modification' of critical habitat at 50 C.F.R. 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat.[1]

NMFS uses the following approach to determine whether the proposed action described in Section 1.3 is likely to jeopardize listed species or destroy or adversely modify critical habitat:

- *Identify the rangewide status of the species and critical habitat likely to be adversely affected by the proposed action*. This section describes the current status of each listed species and its critical habitat relative to the conditions needed for recovery. For listed salmon and steelhead, NMFS has developed specific guidance for analyzing the status of the listed species' component populations in a "viable salmonid populations" paper (VSP; McElhany *et al*. 2000). The VSP approach considers the abundance, productivity, spatial structure, and diversity of each population as part of the overall review of a species' status. For listed salmon and steelhead, the VSP criteria therefore encompass the species' "reproduction, numbers, or distribution" (50 CFR 402.02). In describing the

[1] Memorandum from William T. Hogarth to Regional Administrators, Office of Protected Resources, NMFS (Application of the "Destruction or Adverse Modification" Standard Under Section 7(a)(2) of the Endangered Species Act) (November 7, 2005).

range-wide status of listed species, we rely on viability assessments and criteria in technical recovery team documents and recovery plans, where available, that describe how VSP criteria are applied to specific populations, major population groups (MPG), and species. We determine the rangewide status of critical habitat by examining the condition of its physical or biological features (also called "primary constituent elements" or PCEs in some designations) - which were identified when the critical habitat was designated. Species and critical habitat status are discussed in Section 2.2.

- *Describe the environmental baseline for the proposed action*. The environmental baseline includes the past and present impacts of Federal, state, or private actions and other human activities *in the action area*. It includes the anticipated impacts of proposed Federal projects that have already undergone formal or early section 7 consultation and the impacts of state or private actions that are contemporaneous with the consultation in process. The environmental baseline is discussed in Section 2.3 of this Opinion.

- *Analyze the effects of the proposed actions*. In this step, NMFS considers how the proposed action would affect the species' reproduction, numbers, and distribution or, in the case of salmon and steelhead, their VSP characteristics. NMFS also evaluates the proposed action's effects on critical habitat features. The effects of the action are described in Section 2.4 of this Opinion.

- *Describe any cumulative effects.* Cumulative effects, as defined in NMFS' implementing regulations (50 CFR 402.02), are the effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area. Future Federal actions that are unrelated to the proposed action are not considered because they require separate section 7 consultation. Cumulative effects are considered in Section 2.5 of this Opinion.

- *Integrate and synthesize the above factors to assess the risk that the proposed action poses to species and critical habitat.* In this step, NMFS adds the effects of the action (Section 2.4) to the environmental baseline (Section 2.3) and the cumulative effects (Section 2.5) to assess whether the action could reasonably be expected to: (1) Appreciably reduce the likelihood of both survival and recovery of the species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and critical habitat (Section 2.2). Integration and synthesis occurs in Section 2.6 of this Opinion.

- *Reach jeopardy and adverse modification conclusions*. Conclusions regarding jeopardy and the destruction or adverse modification of critical habitat are presented in Section 2.7. These conclusions flow from the logic and rationale presented in the Integration and Synthesis (Section 2.6) of this Opinion.

- *If necessary, define a reasonable and prudent alternative to the proposed action.* If, in completing the last step in the analysis, NMFS determines that the action under consultation is likely to jeopardize the continued existence of listed species or destroy or adversely modify designated critical habitat, NMFS must identify a reasonable and prudent alternative (RPA) to the action in Section 2.8. The RPA must not be likely to jeopardize the continued existence of ESA-listed species nor adversely modify their designated critical habitat and it must meet other regulatory requirements.

## 2.2. Rangewide Status of the Species and Critical Habitat

This Opinion examines the status of each species that is likely to be affected by the action. The status is the level of risk that the listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. The species status section helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02. The Opinion also examines the condition of critical habitat throughout the designated area, evaluates the conservation value of the various watersheds and coastal and marine environments that make up the designated area, and discusses the current function of the essential physical and biological features that help to form that conservation value. The listed species in the action area and the listing status are shown in Table 2.2.1.

**Table 2.2.1. Federal Register notices for final rules that list threatened and endangered species, designate critical habitats, or apply protective regulations to listed species considered in this consultation.**

| Species | Listing Status | Critical Habitat | Protective Regulations |
|---|---|---|---|
| **Chinook salmon (*Oncorhynchus tshawytscha*)** | | | |
| Snake River spring/summer run | T 6/28/05; 70 FR 37160 | 12/28/93; 58 FR 68543<br>10/25/99; 64 FR 57399 | 6/28/05; 70 FR 37160 |
| Snake River fall run | 12/28/93; 58 FR 68543 | 12/28/93; 58 FR 68543 | 12/28/93; 58 FR 68543 |
| **Sockeye salmon (*O. nerka*)** | | | |
| Snake River | E 6/28/05; 70 FR 37160 | 12/28/93; 58 FR 68543 | ESA Section 9 applies |
| **Steelhead (*O. mykiss*)** | | | |
| Snake River Basin | T 1/05/06; 71 FR 834 | 9/02/05; 70 FR 52630 | 6/28/05; 70 FR 37160 |

### 2.2.1. Status of the Species

For Pacific salmon and steelhead, NMFS commonly uses four parameters to assess the viability of the populations that, together, constitute the species: spatial structure, diversity, abundance, and productivity (McElhany *et al.* 2000). These VSP criteria therefore encompass the species' "reproduction, numbers, or distribution" as described in 50 CFR 402.02. When these parameters are collectively at appropriate levels, they maintain a population's capacity to adapt to various environmental conditions and allow it to sustain itself in the natural environment. These attributes are influenced by survival, behavior, and experiences throughout a species' entire life cycle, and these characteristics, in turn, are influenced by habitat and other environmental conditions.

“Spatial structure” refers both to the spatial distributions of individuals in the population and the processes that generate that distribution. A population’s spatial structure depends fundamentally on habitat quality and spatial configuration and the dynamics and dispersal characteristics of individuals in the population. “Diversity” refers to the distribution of traits within and among populations. These range in scale from DNA sequence variation at single genes to complex life history traits (McElhany *et al.* 2000).

“Abundance” generally refers to the number of naturally-produced adults (i.e., the progeny of naturally-spawning parents) in the natural environment (e.g., on spawning grounds). “Productivity,” as applied to viability factors, refers to the entire life cycle; (i.e., the number of naturally-spawning adults produced per parent). When progeny replace or exceed the number of parents, a population is stable or increasing. When progeny fail to replace the number of parents, the population is declining. McElhany *et al.* (2000) use the terms “population growth rate” and “productivity” interchangeably when referring to production over the entire life cycle. They also refer to “trend in abundance,” which is the manifestation of long-term population growth rate.

For species with multiple populations, once the biological status of a species’ populations has been determined, NMFS assesses the status of the entire species using criteria for groups of populations, as described in recovery plans and guidance documents from technical recovery teams. Considerations for species viability include having multiple populations that are viable, ensuring that populations with unique life histories and phenotypes are viable, and that some viable populations are both widespread to avoid concurrent extinctions from mass catastrophes and spatially close to allow functioning as metapopulations (McElhany *et al.* 2000).

#### *2.2.1.1. Snake River Sockeye Salmon*

The Snake River sockeye salmon, listed as endangered on November 20, 1991 (56 FR 58619), includes all populations of sockeye salmon originating from the Snake River basin, Idaho (extant populations occur only in the Salmon River drainage), as well as sockeye salmon from one artificial propagation program, the Redfish Lake Captive Broodstock program. On August 15, 2011, NMFS completed a 5-year review for the Snake River sockeye salmon ESU and concluded that the species should remain listed as endangered (76 FR 50448).

In Idaho, Snake River sockeye salmon historically spawned and reared in several high mountain lakes (Waples *et al*. 1991a). In the Salmon River basin, sockeye salmon occurred in five lakes (i.e., Alturas, Stanley, Redfish, Yellowbelly, and Pettit Lakes), all of which are near the headwaters of the Salmon River. In the Payette River basin, sockeye salmon historically occurred in the Payette Lakes (Evermann 1895; Fulton 1970); however, access to this basin was blocked upon construction of the Hells Canyon Dam. Thus, spawning and juvenile rearing habitat is currently restricted to the upper portions of the Salmon River Basin. Currently, the Snake River sockeye salmon population is highly dependent on a captive brooding program at the Sawtooth Hatchery (Ford *et al*. 2011).

Since the 1941 completion of the Grand Coulee Dam on the Columbia River that cut off the Arrow Lakes population of sockeye salmon in British Columbia, Snake River sockeye salmon represent the longest inland spawning migration in North America (approximately 930 miles) (Bjornn *et al.* 1968; Behnke and Tomelleri 2002) to the highest elevation (approximately 6,500 feet in elevation) and the most southern destination in the world. Snake River sockeye salmon adults enter the Columbia River primarily during June and July. Arrival at Redfish Lake, which now supports the only remaining run of Snake River sockeye salmon, peaks in August, and spawning occurs primarily in October (Bjornn *et al*. 1968). Eggs hatch in the spring between 80 and 140 days after spawning. Fry remain in the gravel for 3 to 5 weeks, emerge from April through May, and move immediately into the lake. Once there, juveniles feed on plankton for 1 to 3 years before they migrate to the ocean (Bell 1986). Migrants leave Redfish Lake during late April through May (Bjornn *et al*. 1968) and travel to the Pacific Ocean. Smolts reaching the ocean remain inshore or within the influence of the Columbia River plume during the early summer months. Later, they migrate through the northeast Pacific Ocean (Hart 1973; Hartt and Dell 1986). Snake River sockeye salmon usually spend 2 to 3 years in the Pacific Ocean and return in their fourth or fifth year of life.

From 1991 to 1998 a total of 16 natural-origin adult anadromous sockeye salmon returned to Redfish Lake. These natural-origin fish were incorporated into the NMFS/IDFG captive broodstock program that began in 1992. Releases from the NMFS and IDFG captive broodstock programs generated seven returning adults in 1999, 257 adults in 2000, and 1355 adults in 2010 (Table 2.2.2). The 2010 adult return of Snake River sockeye salmon to Redfish Lake reached numbers not seen in decades. For each of the past 3 years for which data is available (2008, 2009, and 2010), the number of returning adults captured in the upper Sawtooth basin was more than the cumulative annual adult return that occurred between the time the fish were listed as endangered in 1991 and 2007.

**Table 2.2.2. Adult returns passing Lower Granite Dam (LGD) and returning to the area of Redfish Lake (Sawtooth Basin, Idaho) (IDFG 2011; Fish Passage Center 2011a; NMFS 2008).**

| Adult Return Year | Number of Adults Passing LGD | #of Adults Returning to Sawtooth Basin | Percent Survival from LGD to Sawtooth Basin |
|---|---|---|---|
| 1995 | 3 | 0 | 0 |
| 1996 | 3 | 1 | 33 |
| 1997 | 11 | 0 | 0 |
| 1998 | 2 | 1 | 25 |
| 1999 | 14 | 7 | 50 |
| 2000 | 299 | 257 | 86 |
| 2001 | 36 | 26 | 72 |
| 2002 | 55 | 22 | 40 |
| 2003 | 11 | 3 | 21 |
| 2004 | 113 | 27 | 24 |
| 2005 | 18 | 6 | 32 |
| 2006 | 17 | 3 | 18 |
| 2007 | 52 | 4 | 8 |
| 2008 | 909 | 650 | 71 |
| 2009 | 1219 | 833 | 68 |
| 2010 | 2201 | 1355 | 62 |

The high return of adult Snake River sockeye salmon is likely due to a combination of factors, including an increased number of fish released from captive broodstock programs, good conditions during downstream and upstream migrations (river flow and temperature, and dam passage conditions), and favorable ocean conditions (Ford 2011). The captive broodstock program has expanded from a starting point of 16 natural-origin adults that returned in the early 1990s to currently releasing hundreds of thousands of juvenile fish each year (Ford *et al*. 2011).

The Snake River sockeye salmon ESU consists of a single MPG. This MPG potentially has five component populations: Redfish Lake (including Little Redfish Lake); Alturas Lake; Pettit Lake; Yellowbelly Lake; and Stanley Lake. Of these, only the Redfish Lake population is currently extant (ICTRT 2007). Assuming there are five populations in this single MPG ESU, three populations would need to achieve viable status for the MPG and ESU to be viable. Since this is a single-MPG ESU, two of the three populations would need to be rated “Highly Viable” based on the four VSP parameters described in McElhany *et al*. (2000), and a third population needs to be rated “Viable.” The latest available Interior Columbia River Basin Technical Recovery Team (ICTRT) recommendation (2007) is to achieve viable populations in three different lakes, with at least at least 1,000 naturally produced spawners per year in each of Redfish and Alturas lakes and at least 500 in Pettit Lake.

The viability status of populations in the ESU and the (single-MPG) ESU as a whole were determined by application of the ICTRT (2007) viability criteria. Viability determinations at the population level were based on extinction risk assessments for the four VSP parameters; abundance, productivity, spatial structure and diversity. A quantitative assessment risk for the VSP abundance/productivity metric was not completed for the populations in the ESU and the single-MPG ESU as a whole because of the lack of abundance and productivity data. Ford (2011) has preliminarily made a qualitative determination that abundance/productivity risk is *High*, based on the current status of the ESU (Endangered) and the recent absence of natural-origin anadromous adults returning to the Stanley Basin. The current average productivity likely is substantially less than the productivity required for any population to be at Low (1% to 5%) extinction risk at the minimum abundance threshold. In addition, the overall spatial structure and diversity has been rated *High* risk for the Redfish Lake population. This rating has been applied to this population because it rated high risk of not being able to maintain: (1) The natural patterns of phenotypic and genotypic expression; (2) natural patterns of gene flow; and (3) the integrity of natural systems. Overall, the Snake River sockeye salmon ESU does not meet the ESU-level viability criteria (non-negligible risk of extinction over 100-year time period) based on current abundance and productivity information.

There have been higher returns in recent years, the annual abundances of natural-origin (or, naturally spawned) sockeye salmon returning to the Stanley basin continue to be extremely low. The captive brood program has been successful in providing substantial numbers of hatchery produced sockeye salmon for use in supplementation efforts, which reduces the risk of immediate loss; yet, substantial increases in survival rates across life history stages must occur in order to re-establish sustainable natural production (Hebdon *et al*. 2004; Keefer *et al*. 2008). Current smolt-to-adult survival of sockeye originating from the Stanley basin lakes is rarely greater than 0.3% (Hebdon *et al*. 2004). Although the risk status of the Snake River sockeye salmon ESU appears to be on an improving trend due to the successes of the captive propagation program, the 5-year review concluded that the ESU remains at high risk (Ford 2011).

#### *2.2.1.2. Snake River Spring/Summer Chinook Salmon*

The Snake River spring/summer Chinook salmon ESU was listed as threatened on April 22, 1992 (57 FR 14653). This ESU occupies the Snake River basin which drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. Environmental conditions are generally drier and warmer in these areas than in areas occupied by other Chinook species. This ESU includes all natural-origin populations in the mainstem Snake River (below Hells Canyon Dam) and the Tucannon, Grande Ronde, Imnaha, and Salmon Rivers. The ESU also includes 15 artificial propagation programs: the Tucannon River (conventional and captive broodstock programs), Lostine River, Catherine Creek, Lookingglass Creek, Upper Grande Ronde River, Imnaha River, and Big Sheep Creek programs in Oregon; and the South Fork Salmon River (McCall Hatchery), Johnson Creek, Lemhi River, Pahsimeroi River, East Fork Salmon River, West Fork Yankee Fork Salmon River, and Upper Salmon River (Sawtooth Hatchery) programs in Idaho (70 FR 37160; June 28, 2005). On August 15, 2011, NMFS completed a 5-year review for the Snake River ESU and concluded that the species should remain listed as threatened (76 FR 50448).

Chinook salmon exhibit a variety of complex life history patterns that include variation in age at seaward migration; length of freshwater, estuarine, and oceanic residence; ocean distribution; ocean migratory patterns; and age and season of spawning migration. Two distinct races of Chinook salmon are generally recognized: "stream-type" and "ocean-type" (Healey 1991; Myers *et al.* 1998). Snake River spring/summer Chinook salmon exhibit stream-type life history characteristics. Adult Snake River spring/summer Chinook salmon enter the Columbia River in late February and early March after 2 or 3 years in the ocean. In high elevation areas, mature fish hold in cool, deep pools until late summer and early fall, when they return to their native streams to begin spawning. They typically spawn in moderate to large-sized streams in shallow gravel bars at the downstream end of pools. Eggs incubate over the winter, and emergence begins in late winter and early spring of the following year. Juveniles rear through the summer, overwinter, and migrate to sea in the spring of their second year of life. During freshwater rearing, juvenile Chinook salmon disperse into tributary streams near their natal streams, and are often concentrated near the mouths of stream confluences. Depending on the tributary and the specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Habitats used by juvenile stream-type Chinook salmon and their feeding habits are similar to those for steelhead. In general, Chinook salmon tend to occupy streams with lower gradients than steelhead, but there is considerable overlap between the distributions of the two species.

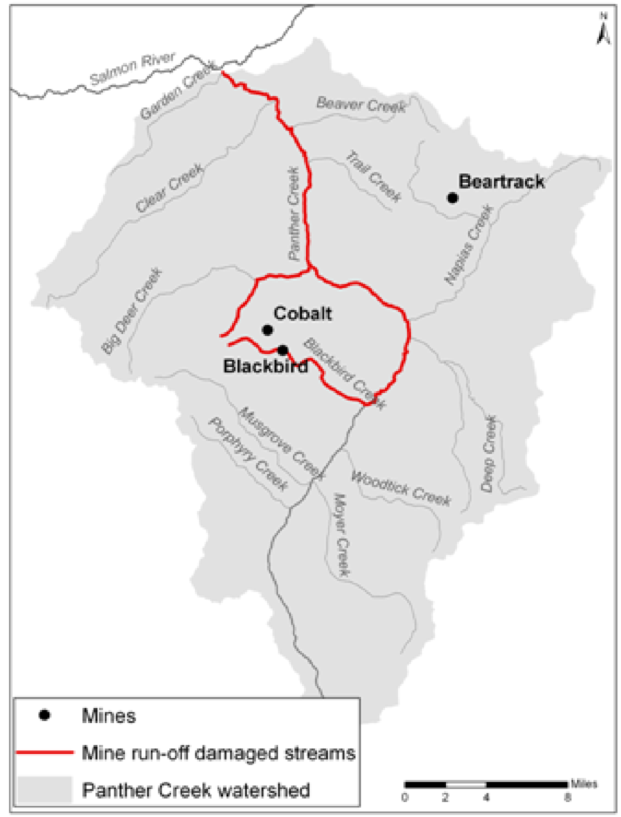
Although direct estimates of historical annual Snake River spring/summer Chinook returns are not available, returns may have declined by as much as 96% between the late 1800s and 2010. According to Matthews and Waples (1991), the Snake River drainage is thought to have produced more than 1.5 million adult spring/ summer Chinook salmon in some years during the late 1800s. By the 1950s the abundance of spring/summer Chinook had declined to an annual average of 125,000 adults and total (natural + hatchery origin) returns fell to roughly 100,000 spawners by the late 1960s (Fulton 1968). Adult returns counted at LGD reached all-time lows in the mid-1990s, although numbers have begun to increase since 1997 (FPC 2011b). The 2001 and 2002 total returns increased to over 185,000 and 97,000 adults, respectively. These large returns are thought to have been a result of favorable ocean conditions (Logerwell *et al.* 2003; Meeings and Lackey 2005) and above average flows in the Columbia River basin (CRB) when the smolts migrated downstream. However, it is important to note that over 80% of the 2001 return and over 60% of the 2002 return originated in hatcheries (Good *et al.* 2005). Furthermore, even these large returns are only a fraction (approximately 5% to 10%) of the estimated returns of the late 1800s. According to the Fish Passage Center (FPC) annual adult passage data (2011b), the 2003 and 2004 runs remained relatively high at 87,031 and 79,509 respectively, and fluctuated over the following years. Adult returns appeared to decline during 2005 to 2007 (average 30,856 total adults), but then increased again from 2008 to 2010. Despite the recent increases in total spring/summer Chinook salmon returns to the basin, natural-origin abundance and productivity are still far below their targets. As such, the Snake River spring/summer Chinook salmon ESU remains likely to become endangered (Good *et al.* 2005; Ford 2011).

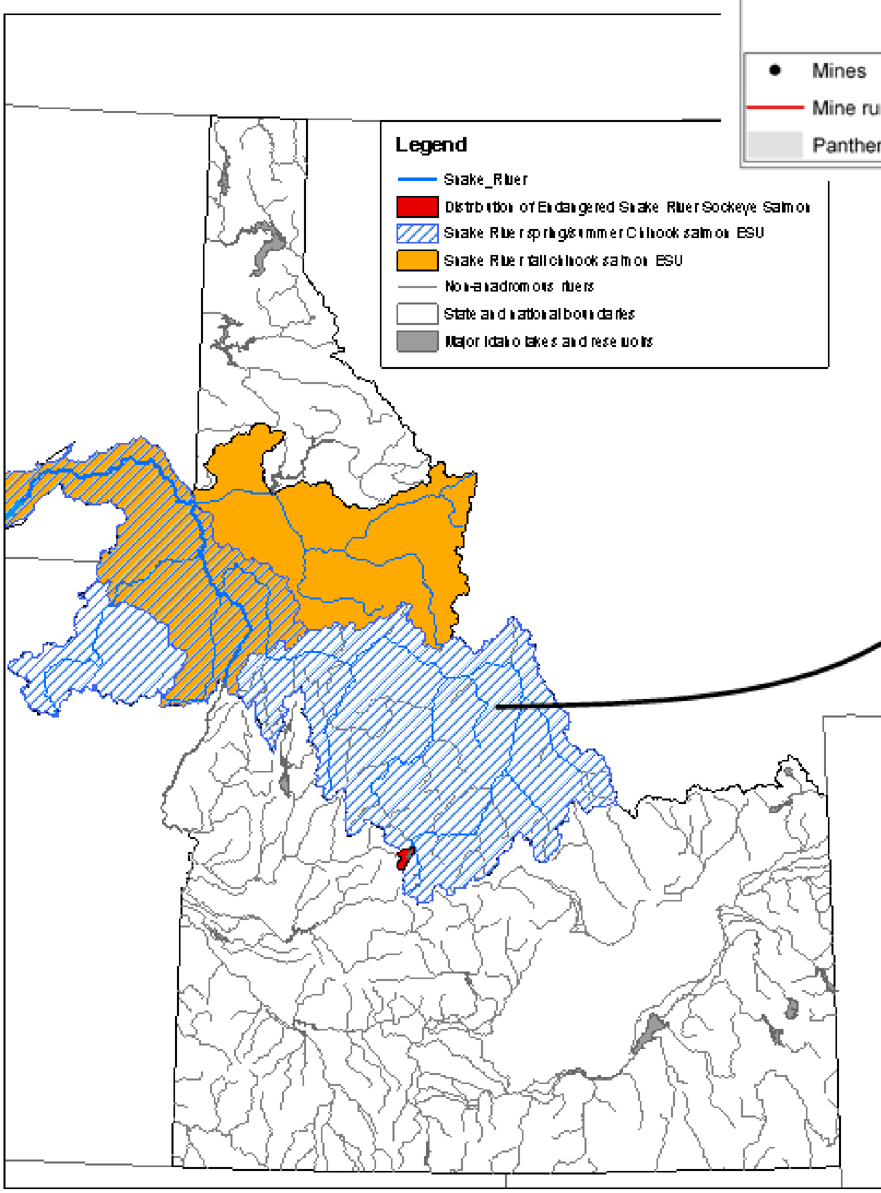
Within the Snake River spring/summer Chinook salmon ESU, independent populations have been grouped into larger aggregates (MPGs) that share similar genetic, geographic, and/or habitat characteristics. This ESU was broken down into five MPGs with 28 extant independent populations and four extirpated or functionally extirpated independent populations (Ford 2011;

ICTRT 2003); McClure *et al*. 2005). Only three of the MPGs (i.e., South Fork Salmon, Middle Fork Salmon, and Upper Salmon) are within the action area. There are 22 independent populations within these three MPGs, one of which (Panther Creek) is considered extirpated by the ICTRT (2003)

In 2005, the ICTRT concluded that the Panther Creek Chinook salmon population was extirpated during the 1960s due to legacy mining and the heavy metal wastes deposited in Lower Panther Creek from the Blackbird Mine operations (ICTRT 2005). The loss of habitat in Panther Creek resulting from water quality degradation from the Blackbird Mine was specifically cited as a contributing factor leading to the decline and subsequent ESA-listing of the Snake River spring/summer Chinook salmon species (NMFS 1991). Once a sizable population, spring/summer Chinook salmon runs declined during the 1940s when extensive mining activity began in the Blackbird Creek Drainage, and was eliminated by the early 1960s. At the time that spring/summer Chinook salmon were being considered for listing under the ESA, the Panther Creek drainage remained largely uninhabitable due to toxic conditions resulting from mine drainage (NMFS 1991).

**Figure 2.2.1. Boundaries of listed anadromous species and the action area (State of Idaho within the range of anadromous species). "Species" are defined as "evolutionarily significant units" or ESUs. The Panther Creek watershed is emphasized because the extirpation of the Panther Creek Chinook salmon population due poor water quality was considered a specific factor for the decline of and ESA listing of the Snake River spring/summer Chinook salmon ESU. Contemporary water quality and biological conditions in Panther Creek are described in the Status of Species section and in the Analyses of Effects sections for arsenic and copper.**



Recovery has been slow. Poor water quality, primarily copper contamination, precluded recolonization through the 1990s, despite supplementation efforts including the release of about 3,383 adult Chinook salmon in 1986. Two Chinook salmon redds each were observed downstream of the Blackbird Mine again in Panther Creek in 1990,1991, and 1992 (Mebane 1994) although no adult or juvenile Chinook salmon could be found despite extensive surveys in 1993 (LeJeune *et al*. 1995). By the early 2000s, extensive mine remediation efforts began to succeed with greater than 90% reductions in copper concentrations in Panther Creek (described more in the Section 2.4.4, Copper)

In the 2000s, Chinook salmon began returning to Panther Creek following improvements in water quality in Panther Creek. The returns and successful reproduction resulted both from natural recolonization and from reproduction following a large release of adult Chinook salmon in 2001. In 2001, as part of "an effort to increase natural production in areas with depressed populations," 1,053 adult Chinook salmon captured from South Fork Salmon River weir were released into Panther Creek (Leth *et al*. 2004). In the fall of 2001, 42 redds were counted and in 2010, 102 redds were counted, both counts were from ground surveys of Panther Creek conducted by the Shoshone Bannock Tribes (EcoMetrix 2011). Aerial counts of Chinook salmon in Panther Creek conducted by the IDFG, which will be lower than ground surveys (e.g., 15 vs. 42 in 2001), ranged from five to 18 from 2001 to 2009 (Figure 2.2.1).

Juvenile Chinook salmon have been found throughout the middle reaches (i.e., downstream of mining influenced Blackbird Creek) and upper reaches of Panther Creek in annual quantitative electrofishing surveys from 2002 through 2010 (Figure 2.2.1). The highest densities were found in 2002, following the large release of adults the previous summer. Peaks in densities in upper reaches of Panther Creek in 2006 (5 years post spawning) and in the middle reaches of Panther Creek in 2005 (4 years after spawning) are consistent with general patterns with inland Chinook populations as well as specific patterns found in the Salmon River drainage, where higher elevation, headwater populations with longer migrations tended to have greater proportions of fish with a 5-year life cycle compared to lower elevation populations where 4-year life cycles are more common with Chinook salmon (Healey 1991; Mebane and Arthaud 2010). This life history pattern, together with the patterns of declining peak densities of juvenile Chinook salmon in the middle reach of Panther Creek, (fish that presumably have a 4-year life cycle) suggests that the juvenile Chinook salmon abundance may be in decline in the middle sections of Panther Creek downstream of Blackbird Creek (Figure 2.2.2). However, no declines are obvious in the Chinook salmon densities in upper Panther Creek, and fish populations may be extremely variable, and with short periods of record, a trend that is apparent 1 year may be gone when the next year's data are added.

Not all the Chinook salmon recently observed or captured in Panther Creek can be attributed to the 2001 release of fish from the South Fork Salmon River. Adult Chinook salmon that were observed in Panther Creek in 2002 and 2004, and young-of-year (YOY) (subyearling) Chinook salmon captured in Panther Creek in 2003 and 2004 cannot be attributed to the artificial release of adult fish in 2001 (Stantec 2004; EcoMetrix 2005). However by 2010, the great majority of the Chinook salmon that continue to return and naturally reproduce in Panther Creek are likely descendants of the 2001 South Fork Salmon River fish (Smith *et al*. 2011).

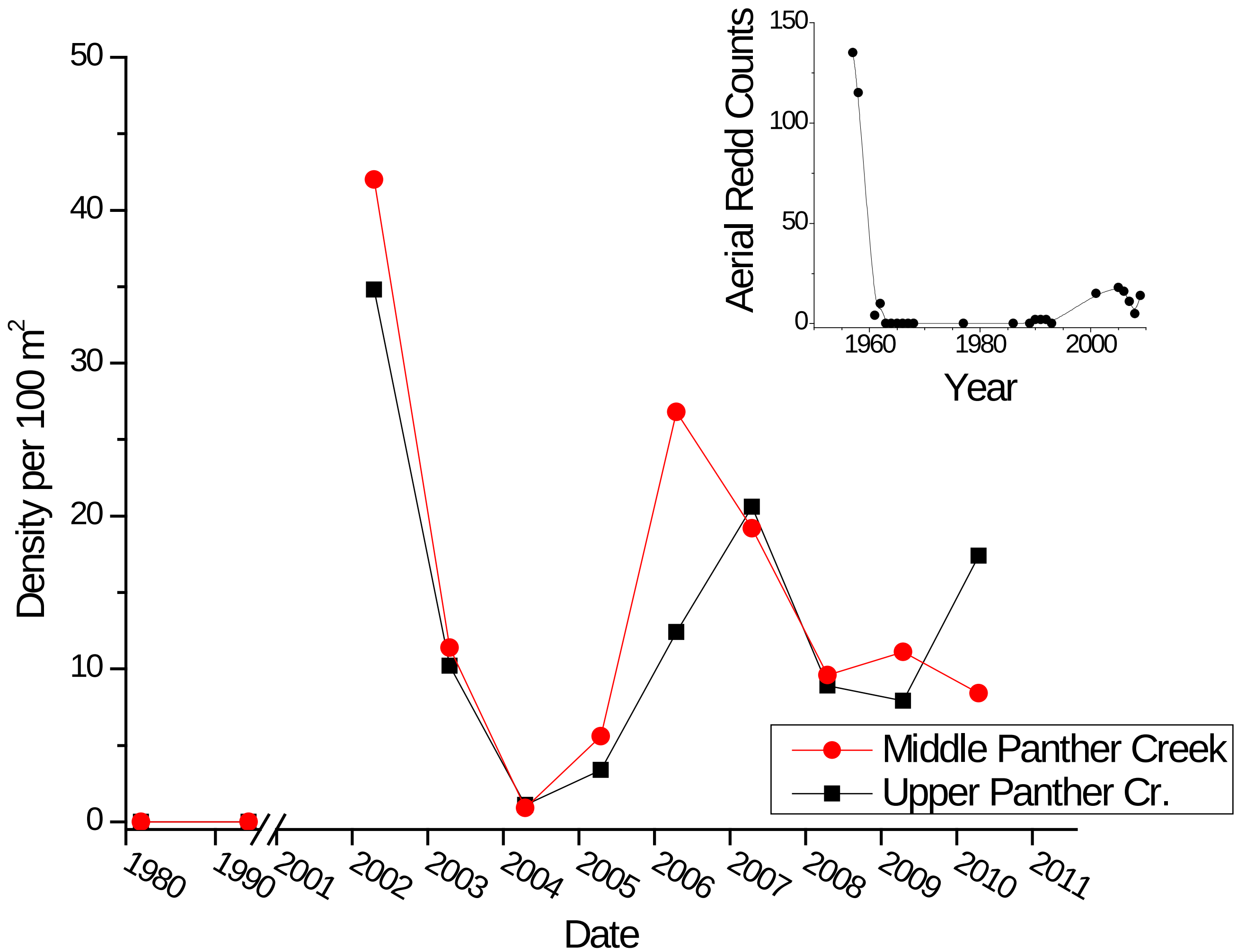



**Figure 2.2.2. Juvenile Chinook salmon abundance in Panther Creek, Idaho, from electrofishing surveys. "Middle" or "Upper" Panther Creek are downstream and upstream of mining influenced Blackbird Creek, respectively. Inset shows trends in aerial redd counts from approximately the same IDFG trend sections. Data from 1992 and earlier were taken from Mebane (1994), 1993 data from LeJeune *et al*. (1995), and subsequent data are from EcoMetrix (2011).**

Under the approach recommended by the ICTRT, the overall rating for an ESU depends upon population level ratings organized by MPG within that ESU (2007). In order for the Snake River spring/summer Chinook salmon ESU to be considered viable, all five MPGs need to achieve viable status. The overall viability ratings for all of the populations in this ESU remain at High risk after the addition of more recent year abundance and productivity data (Ford 2011). Table 2.2.3 summarizes the viability ratings for each population and the overall viability status for each MPG that occurs within the action area. Currently, all of the populations have an overall viability rating of "high risk," and none of the MPGs meet MPG viability criteria (Ford 2011). As such, this ESU does not meet ESU viability criteria (non-negligible risk of extinction over a 100-year time period).

Relatively low natural production rates and spawning levels below minimum abundance thresholds remain a major concern across this ESU. The ability of populations to be self-sustaining through normal periods of relatively low ocean survival remains uncertain. Factors such as habitat modification/degradation, artificial propagation, disease, or predation (NMFS 2011) remain as concerns or key uncertainties for this ESU.

Detailed information on the range wide status of Snake River spring/summer Chinook salmon under the environmental baseline is described in Chinook salmon status reviews (Myers *et al.* 1998; Good *et al*. 2005; Ford *et al*. 2011).

**Table 2.2.3. Summary of VSP parameter risks and viability status for Snake River Spring/Summer Chinook Salmon MPGs and independent populations (Ford 2011; NMFS 2011).**

| MPG | Population Name | Pop. Size & Complexity | VSP Parameter Risk | | Viability Status (Meets Viability Criteria?) | |
|---|---|---|---|---|---|---|
| | | | A/P | SS/D | Population | MPG |
| South Fork Salmon River | Little Salmon River | Intermediate | High | High | Does Not Meet | Does Not Meet |
| | South Fork Salmon River mainstem | Large | High | Moderate | Does Not Meet | |
| | Secesh River | Intermediate | High | Low | Does Not Meet | |
| | East Fork South Fork Salmon River | Large | High | Low | Does Not Meet | |
| Middle Fork Salmon River | Chamberlain Creek | Basic | High | Low | Does Not Meet | Does Not Meet |
| | Middle Fork Salmon River below Indian Creek | Basic | High | Moderate | Does Not Meet | |
| | Big Creek | Large | High | Moderate | Does Not Meet | |
| | Camas Creek | Basic | High | Moderate | Does Not Meet | |
| | Loon Creek | Basic | High | Moderate | Does Not Meet | |
| | Middle Fork Salmon River above Indian Creek | Intermediate | High | Moderate | Does Not Meet | |
| | Sulphur Creek | Basic | High | Moderate | Does Not Meet | |
| | Bear Valley Creek | Intermediate | High | Low | Does Not Meet | |
| | Marsh Creek | Basic | High | Low | Does Not Meet | |
| Upper Salmon River | North Fork Salmon River | Basic | High | Low | Does Not Meet | Does Not Meet |
| | Lemhi River | Very Large | High | High | Does Not Meet | |
| | Salmon River Lower Mainstem | Very Large | High | Low | Does Not Meet | |
| | Pahsimeroi River | Large | High | High | Does Not Meet | |
| | East Fork Salmon River | Large | High | High | Does Not Meet | |
| | Yankee Fork Salmon River | Basic | High | High | Does Not Meet | |
| | Valley Creek | Basic | High | Moderate | Does Not Meet | |
| | Salmon River Upper Mainstem | Large | High | Moderate | Does Not Meet | |
| | Panther Creek | Intermediate | N/A | N/A | Extirpated | |
| Grande Ronde Imnaha | Wenaha River | Intermediate | High | Moderate | Does Not Meet | Does Not Meet |
| | Minam River | Intermediate | High | Moderate | Does Not Meet | |
| | Catherine Creek | Large | High | Moderate | Does Not Meet | |
| | Lostine/Wallowa Rivers | Large | High | Moderate | Does Not Meet | |
| | Upper Grande Ronde River | Large | High | High | Does Not Meet | |
| | Imnaha River | Intermediate | High | Moderate | Does Not Meet | |
| Lower Snake | Tucannon River | Intermediate | High | Moderate | Does Not Meet | Does Not Meet |

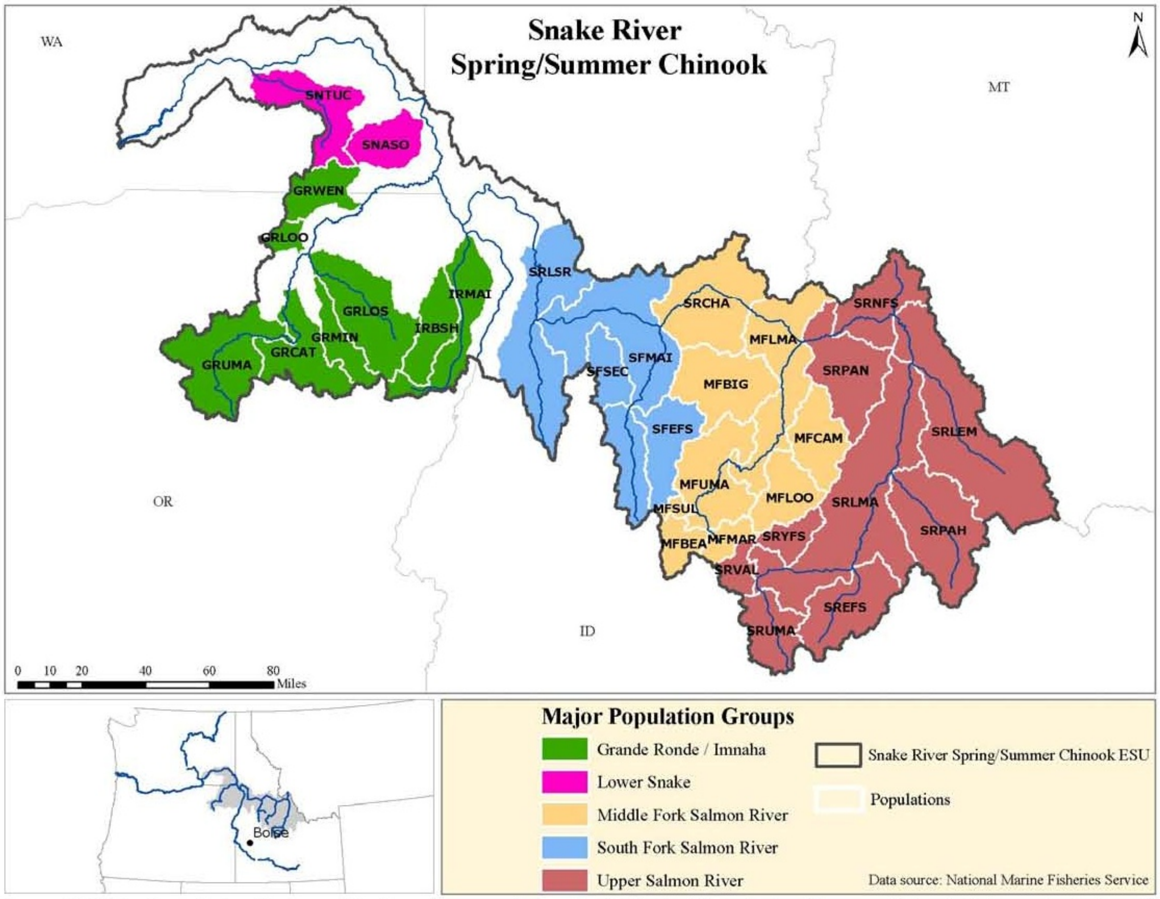



**Figure 2.2.3. Major population groups and independent populations of Snake River spring/summer Chinook salmon. The populations codes are contracted from the above table, for example SRUMA=upper Salmon River, Salmon River mainstem**

*2.2.1.3. Snake River Fall Chinook Salmon*

The Snake River fall Chinook salmon ESU was listed as threatened on April 22, 1992 (57 FR 14653). This ESU includes all natural-origin populations in the mainstem Snake River below Hells Canyon Dam, and the Tucannon, Grande Ronde, Imnaha, Salmon, and Clearwater Rivers. The ESU also includes four artificial propagation programs: the Lyons Ferry Hatchery; fall Chinook acclimation ponds program, Nez Perce Tribal Hatchery, and Oxbow Hatchery (70 FR 37160; June 28, 2005). On August 15, 2011, NMFS completed a 5-year review for the Snake River fall Chinook salmon ESU and concluded that the species should remain listed as threatened (76 FR 50448).

Fall Chinook salmon in the Columbia River generally exhibit an ocean-type life history. In general, fall Chinook salmon are larger than stream-type Chinook salmon and spawn in larger, mainstem rivers and the lower sections of larger tributaries. Adult Snake River fall Chinook salmon return when they are between 2 and 5 years of age, with 4 years being the most common. Adults typically return to fresh water beginning in July, migrate past the lower Snake River mainstem dams from August through November, and spawn from October through early

December.  Juveniles emerge from the gravels in March and April of the following year.  Parr undergo a smolt transformation usually as subyearlings in the spring and summer at which time they migrate to the ocean.  However, in recent years, in both the upper Columbia River basin and in the Snake River basin, some ocean-type Chinook salmon have been utilizing the reservoirs upstream of the mainstem dams and migrating as yearlings the following year.  Subadults and adults forage in coastal and offshore waters of the North Pacific Ocean prior to returning to spawn in their natal streams.

Historically, fall Chinook salmon were widely distributed throughout the Snake River and many of its major tributaries from its confluence with the Columbia River upstream to Shoshone Falls, Idaho (Fulton 1968).  Prior to the 1960s, the Snake River was considered the most important drainage in the Columbia River system for the production of anadromous fishes.  The majority of historic Snake River fall Chinook salmon production was centered on the middle and upper mainstem Snake River in island/channel habitats.  This portion of the Snake River represented approximately 85% of the historically available habitat for this ESU (NMFS 2010a).

Construction of the Swan Falls Dam in 1901 and the Hells Canyon Dam complex between 1956 and 1968 eliminated access to this habitat, reducing the distribution of fall Chinook salmon to mostly remnant areas in the Snake River basin with lower natural production potential than the habitats available in their former range (Connor *et al*. 2002; Dauble *et al*. 2003).  Within Idaho, the current distribution of fall Chinook salmon is located in the Snake River below Hells Canyon Dam; along the lower/middle main Salmon River, from the mouth upstream to approximately its confluence with French Creek; and the lower reaches of the Clearwater River.

Historical abundance of Snake River fall Chinook salmon prior to 1938 is not known.  The estimated annual return for the period 1938 to 1949 was 72,000 fish and had declined to an annual average of 29,000 fish by the 1950s (Bjornn and Horner 1980).  Numbers of fall Chinook salmon continued to decline during the 1960s and 1970s with the construction of numerous dams in the Snake River.  Counts of returning natural-origin fall Chinook salmon at LGD from 1975 through 1980 averaged 610 fish per year (Waples *et al*. 1991b).  The first hatchery-reared Snake River fall Chinook salmon returned to the Snake River in 1981 (Busack 1991), and since then, adult counts represent a mixture of hatchery and natural production.  Since 1983, about 20% to 80% of the total fall Chinook salmon reaching the LGD each year is estimated to have been of hatchery origin (Waples *et al*. 1991b).

Counts of natural-origin[2] adult Snake River fall Chinook salmon at LGD were 1,000 fish in 1975 and declined to an annual low of 78 adults in 1990 (Good *et al*. 2005).  Numbers of natural-origin Snake River fall Chinook salmon began to increase after 1990, with a 5-year geometric mean for 1997 to 2001 of 871 natural-origin fish (Good *et al*. 2005).  The total spawning escapement over LGD has remained relatively high since the rapid increase in the late 1990s. The current 5-year geometric mean (2003 to 2008) of natural-origin fish is 2,291, which is substantially more than the previous estimate.  When considering hatchery-origin fish, the 5-year geometric mean of total adult returns for that same time period exceeded 11,000 (Ford 2011). Clearly, hatchery supplementation continues to play a significant role in the overall abundance of fish, accounting for approximately 78% of the returns during 2003 to 2008 cycle.

[2] Adult fish produced from naturally spawning parents (regardless of the origin of the parents).

There is only one extant[3] population in the Snake River fall Chinook salmon ESU, the Lower Snake River Mainstem population. This population occupies the Snake River from its confluence with the Columbia River to Hells Canyon Dam, and the lower reaches of the Clearwater, Imnaha, Grande Ronde, Salmon, and Tucannon Rivers. The majority of the fish spawn in the mainstem Snake River between the head of Lower Granite Reservoir (River Mile [RM] 146.8) and Hells Canyon Dam (RM 247.6), with the remaining fish distributed among lower sections of the major tributaries. Fall Chinook salmon in the mainstem Snake River appear to be distributed in a series of aggregates from the mouth of Asotin Creek to RM 219, although smaller numbers have been reported spawning in the tailraces of the lower Snake dams. Due to their proximity and the likelihood that individual tributaries did not support separate populations of sufficient size to be self-sustaining, the ICTRT considered these aggregates and the fish in the lower portions of major tributaries to the Snake River to be a single population (McClure *et al*. 2005).

Because there is only one extant population of Snake River fall Chinook salmon, ICTRT criteria indicate that this population should be "Highly Viable" to achieve recovery of this ESU (ICTRT 2007). To be "Highly Viable" under the VSP guidelines, this population must have: (1) A combination of abundance and productivity that create a *very low* risk of extinction (<1% over a 100-year period); and (2) spatial structure and genetic/phenotypic diversity that have no more than a *low* risk of not maintaining key components of spatial structure and diversity described by the ICTRT.

The single extant population of Snake River fall Chinook salmon, the Lower Snake River Mainstem population, is currently not viable. Based upon productivity and escapement estimates, the abundance/productivity metric risk rating is moderate. Similarly, based upon spawner distribution and hatchery composition data, the spatial structure/diversity risk rating is moderate. As such, the overall viability rating for this population is "maintained." To meet the criteria for Highly Viable, the abundance/productivity levels and spatial structure/diversity risk ratings would need to improve.

Detailed information on the range-wide status of Snake River spring/summer Chinook salmon under the environmental baseline is described in Chinook salmon status reviews (Myers *et al*. 1998; Good *et al*. 2005; Ford 2011).

#### *2.2.1.4. Snake River Basin Steelhead*

The Snake River Basin steelhead was listed as a threatened ESU on August 18, 1997 (62 FR 43937), with a revised listing as a distinct population segment (DPS) on January 5, 2006 (71 FR 834). The listed DPS includes all natural-origin populations of anadromous steelhead in the Snake River basin downstream from long-standing barriers in southeast Washington, northeast Oregon, and Idaho. The DPS also includes six artificial production programs: Tucannon River, Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater, East Fork Salmon River,

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[3] The ICTRT also designated two populations of Snake River fall Chinook salmon that are not extant: the Marsing Reach population and the Salmon Falls population (ICTRT May 11, 2005, memorandum regarding updated population delineation in the Interior Columbia Basin).

and the Little Sheep Creek/Imnaha River Hatchery. The Snake River Basin steelhead listing does not include resident forms of *O. mykiss* (rainbow trout) co-occurring with these steelhead.

Steelhead are anadromous fish that spawn in freshwater streams and mature in the ocean. Adult Snake River Basin steelhead return to the Snake River basin from late summer through fall, where they hold in larger rivers for several months before moving upstream into smaller tributaries. Adult dispersal toward spawning areas varies with elevation, with the majority of adults dispersing into tributaries from March through May; earlier dispersal occurs at lower elevations and later dispersal occurs at higher elevations. Spawning begins shortly after fish reach spawning areas, which is typically during a rising hydrograph and prior to peak flows (Thurow 1987). Steelhead generally select spawning areas at the downstream end of pools, in gravels ranging in size from 0.5 to 4.5 inches in diameter (Pauley *et al.* 1986). Juveniles emerge from the gravels in 4 to 8 weeks, depending on temperature. After emergence, fry have poor swimming ability. Steelhead fry initially move from the gravels into shallow, low-velocity areas in side channels and along channel margins to escape high velocities and predators (Everest and Chapman 1972), and progressively move toward deeper water as they grow in size (Bjornn and Rieser 1991). Juveniles typically reside in fresh water for 2 to 3 years (Behnke and Tomelleri 2002). Smolts migrate downstream during spring runoff, which occurs from March to mid-June depending on elevation.

Anadromous Snake River Basin steelhead exhibit two distinct morphological forms, identified as "A-run" and "B-run" fish, which are distinguished by differences in body size, run timing, and length of ocean residence. B-run fish predominantly reside in the ocean for 2 years, while A-run steelhead typically reside in the ocean for 1 year. As a result of differences in ocean residence time, B-run steelhead are typically larger than A-run fish. The smaller size of A-run adults allows them to spawn in smaller headwater streams and tributaries. The differences in the two fish stocks represent an important component of phenotypic and genetic diversity of the Snake River Basin steelhead DPS through the asynchronous timing of ocean residence, segregation of spawning in larger and smaller streams, and possible differences in the habitats of the fish in the ocean.

Although direct historical estimates of production from the Snake River basin are not available, the basin is believed to have produced more than half of the steelhead in the CRB (Mallet 1974). There are some historical estimates of returns to portions of the drainage. Returns to the Clearwater, Grande Ronde, Imnaha, and Tucannon Rivers may have reached or exceeded 62,000 to 82,000 fish in the mid-1950s to early 1960s (Cichosz *et al.* 2003; ODFW 1991; Thompson *et al.* 1958). The Salmon River basin likely supported substantial production as well (Good *et al.* 2005). The longest, consistent indicator of steelhead abundance in the Snake River basin is derived from counts of natural-origin steelhead at the uppermost dam on the LGD. According to these estimates, the abundance of natural-origin steelhead at the uppermost dam on the Snake River has declined from a 4-year average of 58,300 in 1964 to a 4-year average of 8,300 ending in 1998. In general, steelhead abundance declined sharply in the early 1970s, rebuilt modestly from the mid-1970s through the 1980s, and declined again during the 1990s. The 2001 and 2002 total and natural-origin returns of steelhead over LGD (average 240,643 and 52,503, respectively) were substantially higher relative to the low levels seen in the 1990s. The rolling 5-year average abundance of natural-origin returns has generally increased from 2000 (12,090 fish between 1996 and 2000) to 2010 (48,740 fish between 2006 and 2010). Although steelhead numbers have dramatically increased, natural-origin steelhead comprise only 10% to 30% of the total returns since 1994 (FPC 2011c).

The ICTRT identified 29 independent populations (excluding the historically occupied but currently inaccessible habitats upstream of the Hells Canyon Dam complex) in the Snake River Basin steelhead DPS, grouped into six MPGs (McClure *et al*. 2005). Fish from all of these MPGs are found at one time or another migrating through Idaho waters, but only three of the MPGs (i.e., Clearwater River, Salmon River, and Hells Canyon) are located in Idaho. There are 22 independent populations within these three MPGs, of which three are extirpated, one is functionally extirpated, and one (North Fork Clearwater) is blocked from its historic habitat (Table 2.2.4). The three MPGs outside Idaho are Lower Snake MPG (Tucannon River population and Asotin Creek population), the Grande Ronde MPG (Upper and Lower Grande Populations, Joseph Creek population and Wallowa River population) and the Imnaha River MPG (Imnaha River population).

Under the approach recommended by the ICTRT, the overall rating for an ESU depends upon population level ratings organized by MPG within that ESU (2007). In order for the Snake River Basin steelhead DPS to be considered viable, the Clearwater and Salmon MPGs need to achieve viable status. Table 2.2.4 summarizes the viability ratings for each population and the overall viability status for each MPG that occurs within the action area (Ford 2011; NMFS 2011). Currently, none of the MPGs meet MPG viability criteria. As such, this DPS does not meet DPS-level viability criteria (non-negligible risk of extinction over a 100-year time period).

Although recent increases in fish abundances are encouraging, population-level natural-origin abundance and productivity inferred from aggregate data and juvenile indices indicate that many populations in the ESU are likely below the minimum combinations defined by the ICTRT viability criteria. A great deal of uncertainty remains regarding the relative proportion of hatchery fish in natural spawning areas near hatchery release sites. Furthermore, the natural-origin abundance and productivity are still below their targets (Ford 2011).

Detailed information on the range wide status of Snake River Basin steelhead under the environmental baseline is described in status reviews (Myers *et al*. 1998; Good *et al*. 2005; Ford 2011).

**Table 2.2.4. Summary of VSP parameter risks and viability status for Snake River Basin Steelhead MPGs and independent populations (Ford 2011; NMFS 2011).**

| MPG | Population Name | Population Size & Complexity | VSP Parameter Risk | | Status (Meets viability Criteria?) | |
|---|---|---|---|---|---|---|
| | | | A/P | SS/D | Population | MPG |
| Clearwater River | Lower Mainstem | Large | Moderate | Low | Does Not Meet | Does Not Meet |
| | North Fork | | Blocked | Blocked | Extirpated | |
| | Lolo Creek | Basic | High | Moderate | Does Not Meet | |
| | Lochsa River | Intermediate | High | Low | Does Not Meet | |
| | Selway River | Intermediate | High | Low | Does Not Meet | |
| | South Fork | Intermediate | High | Moderate | Does Not Meet | |
| Salmon River | Little Salmon River | Basic | Moderate | Moderate | Does Not Meet | Does Not Meet |
| | Secesh River | Basic | High | Low | Does Not Meet | |
| | South Fork Salmon | Intermediate | High | Low | Does Not Meet | |
| | Chamberlain Creek | Basic | Moderate | Low | Does Not Meet | |
| | Lower Middle Fork | Intermediate | High | Low | Does Not Meet | |
| | Upper Middle Fork | Intermediate | High | Low | Does Not Meet | |
| | Panther Creek | Basic | Moderate | High | Does Not Meet | |
| | North Fork Salmon | Basic | Moderate | Moderate | Does Not Meet | |
| | Lemhi River | Intermediate | Moderate | Moderate | Does Not Meet | |
| | Pahsimeroi River | Intermediate | Moderate | Moderate | Does Not Meet | |
| | East Fork Salmon | Intermediate | Moderate | Moderate | Does Not Meet | |
| | Upper Salmon Mainstem | Intermediate | Moderate | Moderate | Does Not Meet | |
| Grande Ronde | Upper Grande Ronde | Large | Moderate | Moderate | Does Not Meet | Does Not Meet |
| | Lower Grande Ronde | Intermediate | N/A | Moderate | Does Not Meet | |
| | Joseph Creek | Basic | Very Low | Low | Meets | |
| | Wallowa River | Intermediate | High | Low | Does Not Meet | |
| Imnaha | Imnaha | Intermediate | Moderate | Moderate | Doe Not Meet | Does Not Meet |
| Lower Snake River | Tucannon | Intermediate | High | Moderate | Does Not Meet | Does Not Meet |
| | Asotin | Basic | Moderate | Moderate | Does Not Meet | |

## 2.2.2. Status of Critical Habitat

NMFS reviews the status of designated critical habitat affected by the proposed action by examining the condition and trends of essential features for Chinook salmon or PCEs for steelhead throughout the designated area (hereinafter referred to PCEs). The PCEs consist of the physical and biological features identified as essential to the conservation of the listed species because they support one or more of the of the species' life stages (e.g., sites with conditions that support spawning, rearing, migration and foraging).

The ESA-listed species addressed in this Opinion occupy many of the same geographic areas and have similar life history characteristics. The PCEs or essential physical and biological features are also similar and are referred to jointly as PCEs (Table 2.2.5). In general, these PCEs include sites essential to support one or more life stages of the ESA-listed species (i.e., sites for spawning, rearing, migration, and foraging) and contain physical or biological features essential to the conservation of the listed species (e.g., spawning gravels, water quality and quantity, side channels, or food). The PCEs associated with the freshwater spawning, rearing and migratory sites potentially affected by this action include water quality, forage/food, and access/safe passage.

**Table 2.2.5. Types of sites and essential physical and biological features designated as PCEs, and the species life stage each PCE supports.**

| Essential Physical and Biological Features | | ESA-listed Species Life Stage |
|---|---|---|
| **Snake River Basin Steelhead**[a] | | |
| Freshwater spawning | Water quality, water quantity, and substrate | Spawning, incubation, and larval development |
| Freshwater rearing | Water quantity & floodplain connectivity to form and maintain physical habitat conditions | Juvenile growth and mobility |
| | Water quality and forage[b] | Juvenile development |
| | Natural cover[c] | Juvenile mobility and survival |
| Freshwater migration | Free of artificial obstructions, water quality and quantity, and natural cover[c] | Juvenile and adult mobility and survival |
| **Snake River Spring/summer and Fall Chinook Salmon** | | |
| Spawning and Juvenile Rearing | Spawning gravel, water quality and quantity, cover/shelter, food, riparian vegetation, and space | Juvenile and adult. |
| Migration | Substrate, water quality and quantity, water temperature, water velocity, cover/shelter, food[d], riparian vegetation, space, safe passage | Juvenile and adult. |
| **Snake River Sockeye Salmon** | | |
| Spawning and Juvenile Rearing | Spawning gravel, water quality and quantity, water temperature, food, riparian vegetation, and access | Juvenile and adult. |
| Migration | Substrate, water quality and quantity, water temperature, water velocity, cover/shelter, food[d], riparian vegetation, space, safe passage | Juvenile and adult. |

a. Additional PCEs pertaining to estuarine, nearshore, and offshore marine areas have also been described for Snake River Basin steelhead. These PCEs will not be affected by the proposed action and have therefore not been described in this Opinion.
b. Forage includes aquatic invertebrate and fish prey that support growth and maturation.
c. Natural cover includes shade, large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
d. Food applies to juvenile migration only.

Table 2.2.6 provides a brief description of the designated critical habitat for the four ESA-listed species considered in this Opinion. Critical habitat includes the stream channel and water column with the lateral extent defined by the ordinary high-water line, or the bankfull elevation where the ordinary high-water line is not defined. In addition, critical habitat for the three salmon species includes the adjacent riparian zone, which is defined as the area within 300 feet of the line of high water of a stream channel or from the shoreline of standing body of water (58 FR 68543; December 28, 1993). The riparian zone is critical because it provides shade; streambank stability; organic matter input; and sediment, nutrient, and chemical regulation.

**Table 2.2.6. Description of designated critical habitat for ESA-listed species considered in this Opinion.**

| ESU/DPS | Designation | Description of Critical Habitat in Idaho |
|---|---|---|
| Snake River sockeye salmon | 58 FR 68543; December 28, 1993 | Snake and Salmon Rivers; Alturas Lake Creek; Valley Creek, Stanley Lake, Redfish Lake, Yellowbelly Lake, Pettit Lake, Alturas Lake; all inlet/outlet creeks to those lakes |
| Snake River spring/summer Chinook salmon | 58 FR 68543; December 28, 1993 64 FR 57399; October 25, 1999 | All river reaches presently or historically accessible, except river reaches above impassable natural falls and Dworshak and Hells Canyon Dams |
| Snake River fall Chinook salmon | 58 FR 68543; December 28, 1993 | Snake River from state line to Hells Canyon Dam, Clearwater River from its confluence with the Snake River upstream to Lolo Creek, North Fork Clearwater River from its confluence with the Clearwater River upstream to Dworshak Dam, all other river reaches presently or historically accessible within the Clearwater, Lower Clearwater, Lower Snake Asotin, Hells Canyon and Lower Salmon subbasins |
| Snake River Basin steelhead | 70 FR 52630; September 2, 2005 | Specific stream reaches are designated within the Snake, Salmon, and Clearwater basins. Table 21 in the Federal Register details habitat areas within the ESU's geographical range that are excluded from critical habitat designation. |

During all life stages, salmon and steelhead require cool water that is relatively free of contaminants. From a water quality perspective, cool, clean water ensures there is adequate passage conditions for these species to access various habitats required to complete their life cycle. It also contributes to the establishment and maintenance of a healthy, properly functioning ecosystem for prey communities upon which salmon can forage. Water quality degradation within the action area can influence survival and productivity of salmon and steelhead (Regetz 2003).

The PCE for necessary water quality in critical habitats is considered to include the following features. Waters in critical habitats need to be free from substances in concentrations that could cause effects that directly or indirectly, could interfere with important life histories of anadromous salmonids. Potential adverse effects of concern from toxic chemicals include biologically important behaviors and physiological effects to chemoreception, homing, orientation and rheotaxis, downstream migrations, predator avoidance, prey capture, avoidance of habitats or loss of avoidance ability, swimming speed or endurance, altered social status (e.g., dominance and competitive interactions), feeding efficiency, food conversion or growth effects, reproductive impairment, or death, whether resulting from direct exposure or secondary to intermediate effects. The “water quality” PCE also implies waters need to be free from other indirect effects such as effects to invertebrate communities that serve as the prey base for juvenile salmonids, reduced invertebrate diversities, or reduced abundances of preferred prey. Because there are interchanges between the water column and sediments in aquatic habitats, because benthic macroinvertebrate prey are closely linked to sediments, sediments also need to be free from toxic chemicals in concentrations that could cause adverse effects.

Snake River spring/summer Chinook salmon designated critical habitat in the Snake and Columbia Rivers have been altered by: (1) Operation of dams upstream from the migration corridor for water storage and flood control; (2) water diversion for irrigation upstream from the migration corridor; (3) construction of dams, reservoirs, and a navigation channel within the migration corridor; and (4) operation of dams and reservoirs for power generation, flood control, water storage, and navigation within the migration corridor. Use of water, primarily for irrigation, has greatly reduced water quantity available for rearing and migration and construction and operation of storage and flood control reservoirs has further reduced water quantity during spring when juvenile Chinook salmon migrate downstream through the Snake and Columbia Rivers. The eight mainstem dams and their associated reservoirs along the migration route have greatly reduced water velocity and have increased habitat for native and introduced predators, such as pikeminnow, smallmouth bass, and channel catfish. The eight mainstem dams also constitute physical barriers that can substantially decrease migration survival. Impounding water for storage, flood control, and navigation may also increase summer water temperatures, which could adversely affect late migrating juvenile and adult Chinook salmon (NMFS 2014).

Designated critical habitat in the Salmon River drainage has not been affected by mainstem dams and large storage reservoirs, so it is somewhat less altered than habitat in the Snake and Columbia Rivers, but it has been affected by extensive water use, mining, construction and maintenance of water diversion structures, construction and maintenance of roads, conversion of wetlands into agriculture land, and by livestock grazing. Amount of development and condition of habitat varies greatly within the Salmon River drainage. Most of the development, and consequent adverse impacts on habitat, have occurred upstream from the confluence of the Middle Fork Salmon and main Salmon Rivers (RM 199) and within the Little Salmon River drainage. For example: There are approximately 154,000 acres of irrigated agriculture in the Salmon River drainage, the impacts of which deplete flows in the Little Salmon River, North Fork Salmon River, Lemhi River, Pahsimeroi River, portions of the mainstem Salmon River, and numerous smaller Salmon River tributaries; past mining activities have devastated habitat in portions of the Yankee Fork Salmon River drainage and Panther Creek drainages; livestock grazing may also impact riparian habitat throughout this area; and impacts of small cities and towns, which are primarily located on waterways, have cause localized impacts on riparian and instream habitat. In contrast, the Middle Fork Salmon River drainage, large portions of the South Fork Salmon River drainage, and the Chamberlin Creek drainage are largely undeveloped and contain some of the most unimpaired salmonid habitat in the contiguous United States (NMFS 2011).

Spawning and rearing habitat quality in tributary streams in the Snake River varies from excellent in wilderness and roadless areas to poor in areas subject to intensive human land uses (NMFS 2011). Critical habitat throughout much of the Snake River basin has been degraded by intensive agriculture, alteration of stream morphology (*i.e.*, channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer streamflows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in non-wilderness areas. Human land use practices throughout the basin have caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations.

In many stream reaches designated as critical habitat in the Snake River basin, streamflows are substantially reduced by water diversions (NMFS 2011). Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence *et al.* 1996). Reduced tributary streamflow has been identified as a major limiting factor for Snake River spring/summer Chinook and Snake River Basin steelhead in particular (NMFS 2011).

Many stream reaches designated as critical habitat are listed on the state of Idaho's CWA section 303(d) list for impaired water quality, such as elevated water temperature (IDEQ 2010). Some areas that were historically suitable rearing and spawning habitat are now unsuitable due to high summer stream temperatures. Removal of riparian vegetation, alteration of natural stream morphology, and withdrawal of water for agricultural or municipal use all contribute to elevated stream temperatures (Poole *et al.* 2001; Arthaud *et al.* 2010). Water quality in spawning and rearing areas has also been impaired by high levels of sedimentation and by heavy metal contamination from mine waste (e.g., Nelson *et al*. 1991).

Migration habitat quality for Snake River salmon and steelhead has also been severely degraded, primarily by the development and operation of dams and reservoirs on the mainstem Columbia and Snake Rivers (Ford 2011). Hydroelectric development has modified natural flow regimes in the migration corridor—causing in higher water temperatures and changes in fish community structure that have led to increased rates of piscivorous and avian predation on juvenile salmon and steelhead, and delayed migration for both adult and juveniles. Physical features of dams such as turbines also kill migrating fish.

### 2.2.3. Climate Change

Climate change is likely to have negative implications for the conservation value of designated critical habitats in the Pacific Northwest (CIG 2004; Scheuerell and Williams 2005; Zabel *et al*. 2006; Independent Scientific Advisory Board [ISAB] 2007). Average annual Northwest air temperatures have increased by approximately 1ºC since 1900, or about 50% more than the global average warming over the same period (ISAB 2007). The latest climate models project a warming of 0.1ºC to 0.6ºC per decade over the next century. According to the ISAB, these effects may have the following physical impacts within the next 40 or so years:

- Warmer air temperatures will result in a shift to more winter/spring rain and runoff, rather than snow that is stored until the spring/summer melt season.

- With a shift to more rain and less snow, the snowpacks will diminish in those areas that typically accumulate and store water until the spring freshet.

- With a smaller snowpack, these watersheds will see their runoff diminished and exhausted earlier in the season, resulting in lower streamflows in the June through September period.

- River flows in general and peak river flows are likely to increase during the winter due to more precipitation falling as rain rather than snow.

- Water temperatures will continue to rise, especially during the summer months when lower streamflow and warmer air temperatures will contribute to the warming regional waters.

These changes will not be spatially homogenous. Areas with elevations high enough to maintain temperatures well below freezing for most of the winter and early spring would be less affected. Low-lying areas that historically have received scant precipitation and contribute little to total streamflow are likely to be more affected. These long-term effects may include, but are not limited to, depletion of cold water habitat, variation in quality and quantity of tributary rearing habitat, alterations to migration patterns, accelerated embryo development, premature emergence of fry, and increased competition among species.

## 2.3. Environmental Baseline

The environmental baseline includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02).

In general, the environment for ESA-listed species has been dramatically affected by the development and operation of the Federal Columbia River Power System (FCRPS). Storage dams have eliminated mainstem spawning and rearing habitat, and have altered the natural flow regime of the Snake and Columbia Rivers, decreasing spring and summer flows, increasing fall and winter flow, and altering natural thermal patterns. Slowed water velocity and increased temperatures in reservoirs delays smolt migration timing and increases predation in the migratory corridor (NMFS 2014; Independent Scientific Group 1996; National Research Council 1996). Formerly complex mainstem habitats have been reduced to predominantly single channels, with reduced floodplains and off-channel habitats eliminated or disconnected from the main channel (Sedell and Froggatt 2000; Coutant 1999). The amount of large woody debris in these rivers has declined, reducing habitat complexity and altering the rivers' food webs (Maser and Sedell 1994).

Other anthropogenic activities that have degraded aquatic habitats or affected native fish populations in the Snake River basin include stream channelization, elimination of wetlands, construction of flood-control dams and levees, construction of roads (many with impassable culverts), timber harvest, splash dams, mining, water withdrawals, unscreened water diversions, agriculture, livestock grazing, urbanization, outdoor recreation, fire exclusion/suppression, artificial fish propagation, fish harvest, and introduction of non-native species (Henjum *et al.* 1994; Rhodes *et al*. 1994; National Research Council 1996; Spence *et al*. 1996; Lee *et al*. 1997; NMFS 2004). In many watersheds, land management and development activities have:

- Reduced connectivity (i.e., the flow of energy, organisms, and materials) between streams, riparian areas, floodplains, and uplands;

- Elevated fine sediment yields, degrading spawning and rearing habitat;

- Reduced large woody material that traps sediment, stabilizes streambanks, and helps form pools;

- Reduced vegetative canopy that minimizes solar heating of streams;

- Caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations;

- Altered peak flow volume and timing, leading to channel changes and potentially altering fish migration behavior; and,

- Altered floodplain function, water tables and base flows (Henjum *et al*. 1994; McIntosh *et al*. 1994; Rhodes *et al*. 1994; Wissmar *et al*. 1994; National Research Council 1996; Spence *et al*. 1996; and Lee *et al*. 1997).

### 2.3.1. Basins in Action Area

The action area encompasses all areas potentially affected directly or indirectly by this consultation. Because of the potential for downstream effects and additive effects within watersheds, the action area encompasses entire subbasins where ESA-listed species and designated critical habitat occur. A general review of the environmental baseline has been divided up into the three major basins within the action area: (1) The Clearwater River basin; (2) the Salmon River basin; and (3) the Snake River basin.

#### *2.3.1.1. Clearwater River Basin*

The Clearwater River basin is located in north-central Idaho between the $46^{th}$ and $47^{th}$ latitudes in the northwestern portion of the continental United States. It is a region of mountains, plateaus, and deep canyons within the Northern Rocky Mountain geographic province. The basin is bracketed by the Salmon River basin to the south and St. Joe River subbasin to the north.

The Clearwater River drains approximately a 9,645-$mi^2$ area. The basin extends approximately 100 miles north to south and 120 miles east to west. There are four major tributaries that drain into the mainstem of the Clearwater River: the Lochsa, Selway, South Fork Clearwater, and North Fork Clearwater Rivers. The Idaho–Montana border follows the upper watershed boundaries of the Lochsa and Selway Rivers, and the eastern portion of the North Fork Clearwater River in the Bitterroot Mountains. The North Fork Clearwater River then drains the Clearwater Mountains to the north, while the South Fork Clearwater River drains the divide along the Selway and Salmon Rivers. Dworshak Dam, located 2 miles above the mouth of the North Fork Clearwater River, is the only major water regulating facility in the basin. Dworshak Dam was completed in 1972 and eliminated access to one of the most productive systems for anadromous fish in the basin. The mouth of the Clearwater is located on the Washington–Idaho border at the town of Lewiston, Idaho, where it enters the Snake River 139 river miles upstream of the Columbia River (NPCC 2004).

More than two-thirds of the total acreage of the Clearwater River basin is evergreen forests (over 4 million acres), largely in the mountainous eastern portion of the basin. The western third of the basin is part of the Columbia plateau and is composed almost entirely of crop and pastureland. Most of the forested land within the Clearwater basin is owned by the Federal government and managed by the USFS (over 3.5 million acres), but the State of Idaho and Potlatch Corporation also own extensive forested tracts. The western half of the basin is primarily in the private ownership of small forest landowners and timber companies, as well as farming and ranching families and companies. There are some small private in-holdings within the boundaries of USFS lands in the eastern portion of the basin. Nez Perce Tribe lands are located primarily within or adjacent to Lewis, Nez Perce, and Idaho Counties within the current boundaries of the Nez Perce Indian Reservation. These properties consist of both Fee lands owned and managed by the Nez Perce Tribe, and properties placed in trust status with the Bureau of Indian Affairs. Other agencies managing relatively small land areas in the Clearwater basin include the National Park Service, the BLM, Idaho Transportation Department, and IDFG (Ecovista 2004a).

Water quality limited segments are streams or lakes which are listed under section 303(d) of the CWA for either failing to meet their designated beneficial uses, or for exceeding state water quality criteria. The current list of 303(d) listed segments was compiled by the Idaho Department of Environmental Quality (IDEQ) in 2010, and includes many stream reaches within the Clearwater River basin (IDEQ 2010). Individual stream reaches are listed for parameters such as water temperature, sedimentation/siltation, fecal coliform, ammonia, oil and grease, dissolved oxygen, etc. Please refer to the following website for reach-specific 303(d) listed stream segments: http://www.deq.idaho.gov/water-quality/surface-water/monitoring-assessment/integrated-report.aspx.

Small-scale irrigation, primarily using removable instream pumps, is relatively common for hay and pasture lands scattered throughout the lower elevation portions of the subbasin, but the amounts withdrawn have not been quantified. The only large-scale irrigation/diversion system within the Clearwater basin is operated by the Lewiston Orchards Irrigation District within the Lower Clearwater subbasin.

Seventy dams currently exist within the boundaries of the Clearwater Basin. The vast majority of existing dams exist within the Lower Clearwater (56), although dams also currently exist in the Lower North Fork (3), Lolo/Middle Fork (5), and South Fork (6) watersheds (NPPC 2004).

The seven largest reservoirs in the basin provide recreational and other beneficial uses. Dworshak, Reservoir A, Soldiers Meadows, Winchester, Spring Valley, Elk River, and Moose Creek Reservoirs all provide recreational fishing opportunities. Reservoir A and Soldiers Meadows Reservoir are also part of the Lewiston Orchards Irrigation District irrigation system. Capacity of other reservoirs within the Clearwater basin is limited to 65 acre-feet or less, and in most cases is less than 15 acre-feet, limiting their recreational capacity (NPPC 2004).

Agriculture primarily affects the western third of the basin on lands below 2,500 feet in elevation, primarily on the Camas Prairie both south and north of the mainstem Clearwater and the Palouse. Additional agriculture is found on benches along the main Clearwater and its lower tributaries such as Lapwai, Potlatch, and Big Canyon Creeks. Hay production in the meadow areas of the Red River and Big Elk Creek in the American River watershed accounts for most of the agriculture in the South Fork Clearwater. Total cropland and pasture in the subbasin exceeds 760,000 acres. Agriculture is a particularly large part of the economy in Nez Perce, Latah, Lewis, and Idaho Counties, which all have large areas of gentle terrain west of the Clearwater Mountains. Small grains are the major crop, primarily wheat and barley. Landscape dynamics, hydrology, and erosion in these areas are primarily determined by agricultural practices (NPPC 2004).

Subwatersheds with the highest proportion of grazeable area (less than 50%) within the Clearwater basin are typically associated with USFS grazing allotments in lower-elevation portions of their ownership areas. However, the majority of lands managed by the USFS within the Clearwater basin are not subjected to grazing by cattle or sheep, including all or nearly all of the Upper Selway, Lochsa, and Upper and Lower North Fork watersheds. Subwatersheds outside of the USFS boundaries typically have less than 25% of the land area defined as grazeable, although this is as much as 75% for some. Privately owned property within the basin typically contains a high percentage of agricultural use, with grazeable lands found only in uncultivated areas. In contrast, grazing allotments on USFS lands are typically large, often encompassing multiple HUCs, resulting in higher proportions of grazeable area than those contained in primarily privately owned lands (NPPC 2004).

Mines are distributed throughout all eight watersheds in the Clearwater Basin, with the lowest number of occurrences in the upper and lower Selway. Ecological hazard ratings for mines (delineated by the Interior Columbian Basin Ecosystem Management Project) indicate that the vast majority of mines throughout the subbasin pose a low relative degree of environmental risk. However, clusters of mines with relatively high ecological hazard ratings are located in the South Fork Clearwater River and in the Orofino Creek drainage (Lolo/Middle Fork) (NPPC 2004).

*2.3.1.2. Salmon River Basin*

The Salmon River flows 410 miles north and west through central Idaho to join the Snake River. The Salmon River is the largest subbasin in the Columbia River drainage, excluding the Snake River, and has the most stream miles of habitat available to anadromous fish. The total subbasin is approximately 14,000 square miles in size. Major tributaries include the Little Salmon River, South Fork Salmon River, Middle Fork Salmon River, Panther Creek, Lemhi River, Pahsimeroi River, and East Fork Salmon River (IDFG 1990).

Public lands account for approximately 91% of the Salmon River Basin, with most of this being in Federal ownership and managed by seven National Forests or the BLM. Public lands within the basin are managed to produce wood products, domestic livestock forage, and mineral commodities; and to provide recreation, wilderness, and terrestrial and aquatic habitats. Approximately 9% of the basin is privately owned. Private lands are primarily in agricultural cultivation, and are concentrated in valley bottom areas within the upper and lower portions of the basin.

Land management practices within the basin vary among landowners. The greatest proportion of National Forest lands are Federally designated wilderness area or areas with low resource commodity suitability. One-third of the National Forest lands in the basin are managed intensively for forest, mineral, or range resource commodity production. The BLM lands in the basin are managed to provide domestic livestock rangeland and habitats for native species. State of Idaho endowment lands within the basin are managed for forest, mineral, or range resource commodity production. Near-stream or in-channel activities of relevance to fish and wildlife conservation include efforts by landowners, private or otherwise, to modify stream channels in order to protect property. Examination of the geographic distribution of permitted channel alterations during the past 30 years suggests that the long-term frequency of these activities was relatively consistent across much of the Salmon River Basin, but less common in the Upper Middle Fork Salmon, Lower Middle Fork Salmon, Middle Salmon-Chamberlain, and Pahsimeroi subbasins. It is unclear to what degree channel-modifying activities completed without permits may have had on the observed pattern. Stream channels in the basin are also altered, albeit on a smaller scale, by recreational dredging activities (NPCC 2004).

Water quality in many areas of the basin is affected to varying degrees by land uses that include livestock grazing, road construction, logging and mining (Ecovista 2004b). Water quality limited segments are streams or lakes which are listed under section 303(d) of the CWA for either failing to meet their designated beneficial uses, or for exceeding state water quality criteria. The current list of 303(d) listed segments was compiled by the IDEQ in 2010, and includes numerous defined stream reaches within the Salmon River Basin. Individual stream reaches are listed for parameters such as water temperature, escherichia coli, sedimentation/siltation, fecal coliform, ammonia, copper, etc. Please refer to the following website for reach-specific 303(d) listed stream segments: http://www.deq.idaho.gov/water-quality/surface-water/monitoring-assessment/integrated-report.aspx.

In the Lemhi, Upper Salmon, Pahsimeroi, and Middle Salmon-Panther subbasins, less than 20% of the larger streams meet all designated uses (i.e., specific uses identified for each water body through state and tribal cooperation, such as support of salmonid fishes, drinking water supplies, maintenance of aquatic life, consumption of fish, recreational contact with water, and agriculture) (NPCC 2004).

Partial and seasonal barriers have been created on a few of these streams. Partial to complete barriers to anadromous fish exist on the Lemhi, Pahsimeroi and upper Salmon Rivers at water diversions for irrigation. Twenty minor tributaries contain dams that are used for numerous purposes such as irrigation, recreation, and fish propagation (IDFG 1990).

The diversion of water, primarily for agricultural use within the Salmon River Basin, has a major impact on developed areas – particularly the Lemhi, Pahsimeroi, the mainstem Salmon, and several tributaries of the Salmon River. Although many diversions are screened, many need repair and upgrading. A major problem is localized stream dewatering. In addition to water diversions, numerous small pumping operations for private use occur throughout the subbasin. Impacts of water withdrawal on fish production are greatest during the summer months, when streamflows are critically low (IDFG 1990).

The Salmon River Basin encompasses portions of five USFS wilderness areas. The Frank Church River of No Return Wilderness area, one of the five within the subbasin, is the largest wilderness area in the contiguous United States. Specific management guidelines for wilderness areas generally prohibit motorized activities and allow natural processes to function in an undisturbed manner.

Mining, though no longer a major land use as it was historically, it is still very prevalent in parts of the Salmon River Basin. Impacts from mining include severe stream alterations in substrate composition, channel displacement, bank and riparian destruction, and loss of instream cover and pool-forming structures. All of these impacts are typical of large-scale dredging and occur with other types of mining. Natural stream channels within the Yankee Fork, East Fork South Fork, and Bear Valley Creek, have all had documented spawning and rearing habitat destroyed by dredge mining. Furthermore, heavy metal pollution from mine wastes and drainage can eliminate all aquatic life and block access to valuable habitat as seen in Panther Creek (IDFG 1990).

#### *2.3.1.3. Snake River Basin*

The Snake River originates at 9,500 feet, along the continental divide in the Wyoming portion of Yellowstone National Park. The Snake River flows 1,038 miles westward toward the Idaho-Oregon border, northwest to its confluence with Henry's Fork near Rexburg, and then to Pasco, Washington, where it flows into the Columbia River. The Snake River is a large river that is one of the most important water resources in the State of Idaho. The Boise, Payette, and Weiser Rivers in Idaho, and the Owyhee, Malheur, Burnt, and Powder Rivers in Oregon, join the Snake River in this Idaho-Oregon border reach. The Snake River passes through Hells Canyon and Idaho Power Company's Hells Canyon Complex. Brownlee Dam, near River Mile 285, is the uppermost facility, with Oxbow and Hells Canyon dams downstream. The basin includes agriculture, and private and Federal irrigation.

The Snake River basin upstream from Brownlee Dam includes 31 dams and reservoirs with at least 20,000 acre-feet of storage each. The Bureau of Reclamation (BOR), Idaho Power Company, and a host of other organizations own and operate various facilities. These facilities have substantial influence on water resources, supplies, and the movement of surface and groundwater through the region. The total storage capacity of these reservoirs is more than 9.7 million acre-feet. In addition, there are numerous smaller state, local, and privately owned and operated dams and reservoirs throughout the upper Snake River Basin.

Within the action area, water quality limited segments are streams or lakes which are listed under section 303(d) of the CWA for either failing to meet their designated beneficial uses, or for exceeding state water quality criteria. The current list of 303(d) listed segments was compiled by the IDEQ in 2010, and includes 7 defined stream reaches within the Hells Canyon and Lower Snake River Asotin 4th-field HUCs. Individual stream reaches are listed for parameters such as water temperature, sedimentation/siltation, escherichia coli, dissolved oxygen, pH, and nutrient/eutrophication biological indicators. Please refer to the following website for reach-specific 303(d) listed stream segments: http://www.deq.idaho.gov/water-quality/surface-water/monitoring-assessment/integrated-report.aspx.

2.3.2. Baseline for Metals

Because of their wide variety of uses, metals enter the environment through many pathways. The most direct routes are through acid mine drainage from active and abandoned mines and point-source discharges from industrial activities such as plating, textile, tanning, and steel industries. Municipal waste water treatment plants and urban runoff are also significant source of metals to the environment. Arsenic, copper, and zinc used as pesticides and wood preservatives enter the environment via drift, erosion, surface runoff, and leaching. Copper is applied directly to the water as an aquatic herbicide. Particulate metals from combustion and dust can be transported through the air.

Metals can enter the aquatic environment in a dissolved form or be attached to organic and inorganic particulate matter. The amount of metal in the dissolved versus particulate form in natural waters can vary greatly, but the particulate form is usually found in greater concentrations. Metals can flux between different states and forms in an aquatic environment due to changes in pH, temperature, oxygen, presence of other compounds, and biological activity. These transformations can occur within and between water, sediment, and biota as the cycles of nature change. Dredging and disposal operations can result in substantial suspension and re-suspension of particulates in the water column, including those contaminated with metals.

Most metals addressed in this Opinion can enter the environment through natural and anthropogenic pathways, and many of these metals naturally occur in the region in low background concentrations. Most elevated concentrations of toxic metals in critical habitat have been associated with hard rock mining operations, particularly in the Salmon River basin. There has been extensive degradation of critical habitat in many streams, some of which had been associated with complete extirpation of salmon and steelhead populations because of poor water quality (e.g., Panther Creek).

#### *2.3.2.1. Baseline for Arsenic in Action Area*

Concentrations of arsenic in river waters are usually low, typically in the range 0.1 to 2.0 µg/L worldwide. However, relatively high concentrations of naturally occurring arsenic in rivers can occur as a result of geothermal activity or the influx of high-arsenic groundwaters. Arsenic in surface water is strongly associated with sediments and is highest in the toxic zones near the surface water interface (Mok and Wai 1989; Nicholas and others 2003).

Arsenic concentrations of 10 to 70 µg/L have been reported in river waters from geothermal areas, including the western USA (Plant *et al*. 2007; McIntyre and Linton 2011; Table 2.4.3.1). In a probabilistic study of arsenic in 55 Idaho rivers, the median total concentration was 2.0 µg/L, ranging from 0.06 to 17 µg/L, from unfiltered samples (Essig 2010). In the Stibnite Mining District located in the East Fork of the South Fork Salmon River (EFSFSR), arsenic is naturally elevated in groundwater (up to 1000 µg/L), which then has been mobilized by mining and milling. Arsenic concentrations up to 96 µg/L in filtered samples and 109 µg/L in unfiltered have been measured in the EFSFSR downstream of the Stibnite Mining District (Woodward-Clyde 2000).

Arsenic is greatly elevated above background levels in the Panther Creek watershed, downstream of the Blackbird Mine. The loss of the Panther Creek population of Chinook salmon from Blackbird Mine contamination was one of the factors leading to the decline and ESA listing of Snake River spring/summer Chinook salmon (NMFS 1991). High arsenic in whole (unfiltered) surface waters (>100 µg/L) has been detected, although dissolved arsenic in filtered samples has been very low (<2 µg/L) in all samples (Table 2.4.3.2). Based on their relative toxicities and ambient concentrations, copper was probably the biggest factor causing the loss of the Panther Creek Chinook population, although arsenic contributes to aquatic risk (Section 2.4.3; NMFS 2007).

Arsenic, cobalt, and copper were greatly elevated in sediments, periphyton, and in the tissues of aquatic insects in Panther Creek at the time of Chinook listing (Figures 2.3.1.1 to 2.3.1.3). Ongoing remedial efforts that began in 1995 have led to some reductions in arsenic concentrations in Panther Creek sediments and in the foodweb, although concentrations in both remain elevated above upstream reference concentrations as of 2010 (Figures 2.3.1.1 and 2.3.1.3). Arsenic in tissues of aquatic insects declined with initial remedial efforts, but from 2006 to 2010, there have been no further decreases in arsenic in insect tissues. In contrast to marked reductions in copper in Panther Creek (Section 2.3.3.), arsenic in periphyton has yet to decline in Panther Creek. This suggests the presence of a persistent reservoir of arsenic in sediments and floodplain soils.

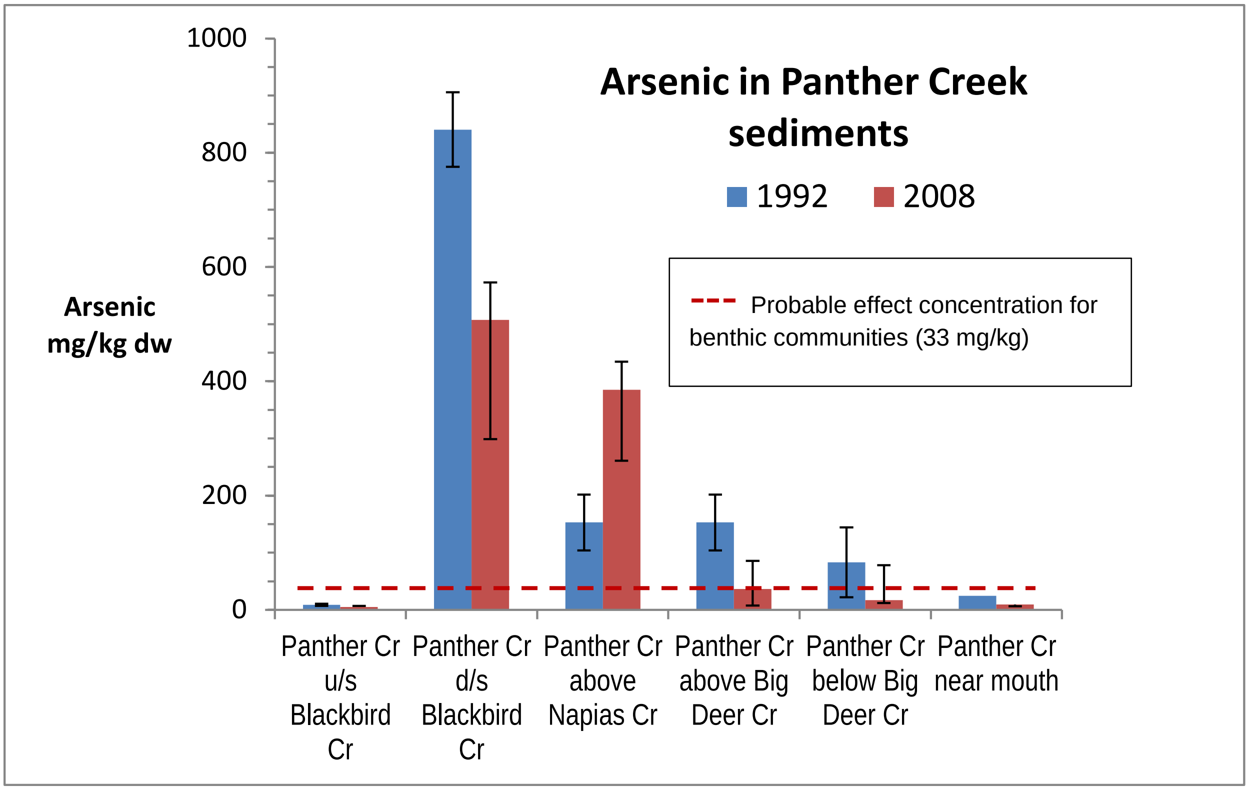



**Figure 2.3.1.1. Arsenic in Panther Creek sediments sampled in similar stream reaches before and after remediation efforts. In both surveys arsenic declined with increasing distance downstream from Blackbird Creek. Arsenic appears to have generally declined over time, although arsenic is still greatly elevated until the diluting flows of Napias Creek, a large tributary, enter. This suggests a reservoir of arsenic may persist in sediments or riparian soils that may be difficult to further control. As of 2011, EPA is evaluating the feasibility of additional remediation to further reduce arsenic releases from Blackbird Creek. Data from Mebane (1994) and Golder (2009), probable effect concentration from MacDonald *et al*. (2000a).**

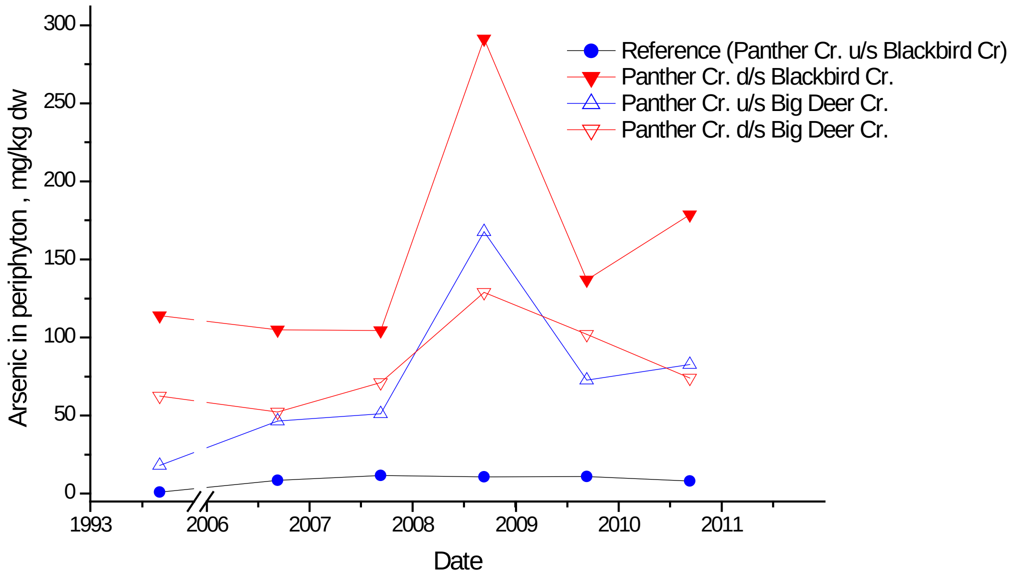



**Figure 2.3.1.2. Arsenic in periphyton (algae and other organic material collected from stream rocks) in Panther Creek sampled in similar stream reaches before and after remediation efforts. Periphyton is the primary food source for many aquatic insects. Data from Beltman *et al*. (1994) and EcoMetrix (2011).**

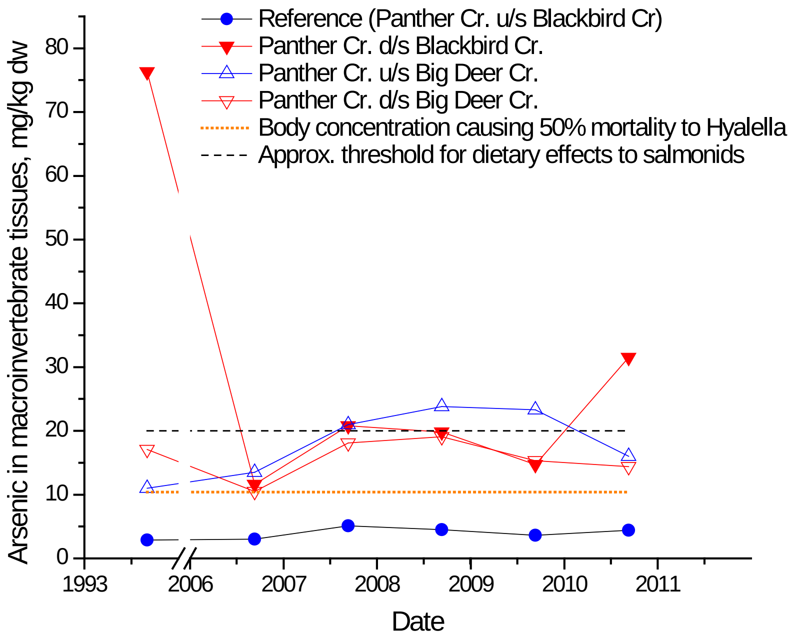



**Figure 2.3.1.3. Arsenic in macroinvertebrate tissues of Panther Creek sampled in similar stream reaches before and after remediation efforts. In both time periods arsenic declined with increasing distance downstream from Blackbird Creek. At the uppermost mining-affected sites, Panther Creek downstream of Blackbird Creek, arsenic initially declined markedly following remediation, but has not further declined from 2006 through 2010. The 2008 spike in arsenic concentrations apparent in sediment and periphyton graphs was not apparent in macroinvertebrates, suggesting limited bioavailability of arsenic in that event. Data from Beltman *et al*. (1994) and EcoMetrix (2011).**

Arsenic is a suspected carcinogen in fish. It is associated with necrotic and fibrous tissues and cell damage, especially in the liver. Arsenic can result in immediate death through increased mucus production and suffocation. Other effects include anemia and gallbladder inflammation. The toxicity of arsenic is influenced by a number of factors including fish size, water temperature, pH, redox potential, organic matter, phosphate content, suspended solids, presence of other toxicants, speciation of the chemical itself, and the duration of exposure (Dabrowski 1976; Eisler 1988a; McGeachy and Dixon 1989; Sorensen 1991; Cockell *et al*. 1992; Rankin and Dixon 1994; McIntyre and Linton 2011). Juvenile salmonids have been found to be more sensitive to arsenic toxicity than alevins (Buhl and Hamilton 1990, 1991). Trivalent arsenic (arsenite) tends to be more toxic than other forms, and inorganic forms of arsenic (including pentavalent) are typically more toxic than organic forms (EPA 1985a; Eisler 1988a; Sorensen 1991). Chronic toxicity in fish appears to be inversely proportional to water temperature under certain experimental conditions (McGeachy and Dixon 1990).

*2.3.2.2. Baseline for Chromium*

Although weathering processes result in the natural mobilization of chromium, the amounts added by anthropogenic activities are thought to be far greater. Major sources are the industrial production of metal alloys, atmospheric deposition from urban and industrial centers, and large scale wrecking yards and metals recycling and reprocessing centers (Reid 2011). Few, if any, of these major urban or industrial sources are expected in the largely rural action area in Idaho.

Few data on chromium concentrations in Idaho were located. In the Stibnite Mining District in the EFSFSR basin, total chromium concentrations collected under low flow conditions in September 2011 ranged from <0.2 µg/L to 0.24 µg/L (*http://waterdata.usgs.gov/nwis*, HUC 17060208). In the Blackbird Mining District, concentration of chromium in seeps and adits around the Blackbird Mine were not higher than average background filtered surface water concentrations near the Blackbird Site (<2.9 µg/L) (Beltman and others 1993)

*2.3.2.3. Baseline for Copper*

Copper concentrations of about 0.4 to 4 µg/L have been considered typical of major river waters in the United States, not directly influenced by industrial or urban activities (Stephan and others 1994). Specific data reviewed within the Idaho action area mostly fell within that range. Whenever available, data given here were limited to the data collected in 1993 or later using "clean" sampling and analyses and quality control measures. This is because prior to the implementation of "clean" procedures, contamination of metals samples during collection and analyses was nearly ubiquitous (Shiller and Boyle 1987; Windom and others 1991; Stephan and others 1994).

In the Salmon River basin, reliable copper data are available for several locations. With the exception of the Panther Creek drainage, discussed separately, almost all other locations had low copper concentrations relative to Stephan *et al.*'s (1994) range. In the Salmon River upstream of Panther Creek, dissolved copper ranged from 1.4 to 1.6 µg/L in six samples collected during high and low flows in 1993. Yet, in the Salmon River sampled a few miles downstream of Panther Creek at the same time, copper ranged from 5.3 to 25.9 µg/L (Maest and others 1994). In the Stibnite Mining District in the EFSFSR basin, copper concentrations collected under low flow conditions in September 2011 ranged from <0.5 µg/L to 4 µg/L which is almost the same as the range given by Stephan *et al*. (1994) (*http://waterdata.usgs.gov/nwis*, HUC 17060208). In the mainstem upper Salmon River in the vicinity of the Thompson Creek Mine (TCM), copper concentrations in 1998 to 2000 ranged from <0.2 to 1 µg/L, in 17 of 18 samples, with a single much higher value of 5.5 µg/L during October 1998. That single high value may not have been reliable, since the tributaries to the Salmon River that directly receive Thompson Creek effluent, and thus should have had higher copper concentrations had the high copper value originated from the mine, showed consistently lower copper concentrations, <0.2 to 2 µg/L (Mebane 2000).

Copper has been monitored in the vicinity of the Hecla Grouse Creek Mine, which discharges to the Yankee Fork River via a pipeline and diffuser, and also to Jordan Creek, a smaller stream. Copper in Jordon Creek downstream of mine discharges in 2010 was very low, ranging from <0.5 µg/L to 1 µg/L, and in the Yankee Fork River, downstream of mine effluents similarly ranged from 0.5 to 0.9 µg/L (Hecla Mining Company data).

In wilderness regions of the Middle Fork Salmon River, Idaho, filtered copper concentrations in Loon Creek and Big Creek ranged from 0.6 to 0.93 µg/L (Maest and others 1994). Other locations in the Salmon River, Idaho drainage with copper data included the Pahsimeroi River at Ellis, Lemhi River near Lemhi, Salmon River near Salmon, Salmon River near White Bird, and Johnson Creek, a tributary to the South Fork Salmon River. Copper concentrations at all these sites ranged from <1 to 4 µg/L from 1991 to 1995 (Hardy and others 2005).

In the Clearwater River basin, much less information is available, which is probably because there is less recent mining activity and associated monitoring in the Salmon River basin. The available copper data located had low values. For instance, in Lapwai Creek and in the South Fork Clearwater River at Stites, copper in filtered samples collected between 1991 and 1995 ranged from <1 µg/L to 2 µg/L, n=8 each (Hardy and others 2005).

In the Hells Canyon reach of the lower Snake River, near Anatone, Washington, copper concentrations from the same time period were a little higher than those usually reported from the Clearwater or Salmon River drainages, ranging from 1 to 4 µg/L, n=18 (Hardy and others 2005).

Other than the Panther Creek drainage, the highest copper concentrations from the state of Idaho's statewide monitoring project was from the Clark Fork River, at Cabinet Gorge, Idaho with values up to 38 µg/L from 1992 to 1995 (Hardy and others 2005). The Clark Fork has been subject to large scale mining disturbances and copper contamination, although these disturbances are >200 miles upstream of Cabinet Gorge.

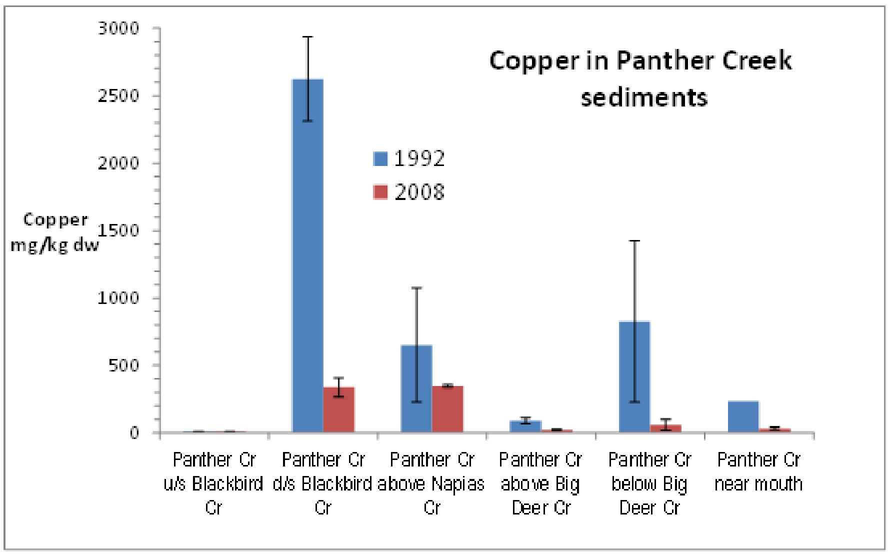
Natural background concentrations of copper and other metals can occur; however, these seem to be rare and limited to very small streams or springs. Areas of Panther Creek, Idaho, that had no evidence of mining disturbances but were near mining prospects did sometimes have much higher copper concentrations than noted above. For example, Mebane (1994) reported 312 µg/L in a spring in the headwaters of Little Deer Creek, and 10.7 µg/L in Little Deer Creek at its mouth.

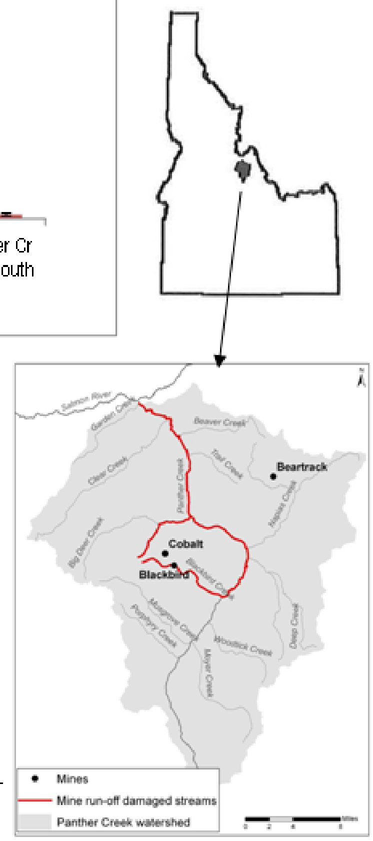
In summary, other than Panther Creek and the Salmon River shortly downstream, copper concentrations measured throughout the action area are usually in the range of <0.5 to 4 µg/L.

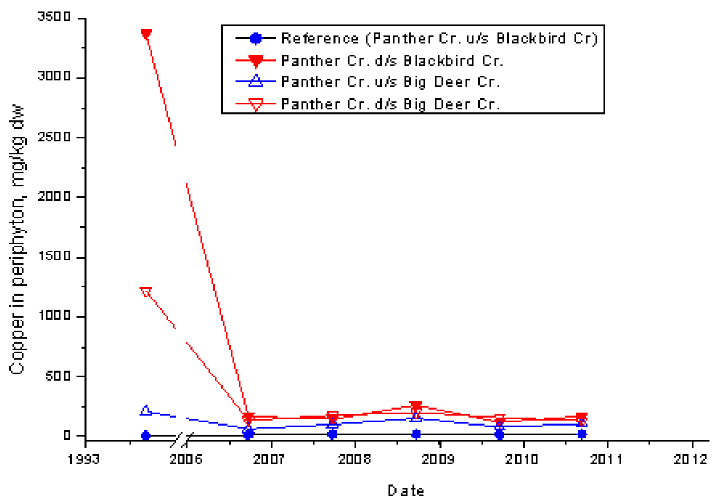
***Baseline for Copper Concentrations in the Panther Creek Watershed.*** Baseline conditions are described separately for the Panther Creek watershed, because copper contamination and the resulting loss of the Panther Creek Chinook salmon population was one specific factor leading to the decline of the species, and listing of spring/summer Chinook salmon under the ESA (NMFS 1991). Because of this, copper concentrations and associated biological conditions in Panther Creek at the time of listing and contemporary conditions are considered here in detail. Concerted site remediation efforts began in 1995 and have been sustained to date. The objectives of these remedial efforts are specifically intended to restore water quality to restore lost anadromous fish populations. To wit, the remedial action objective for Panther Creek is to "*restore and maintain water quality and aquatic biota conditions capable of supporting all life stages of resident and anadromous salmonids and other fishes in Panther Creek*" (EPA 2003d; 2008). As follows, the effectiveness of these efforts is evaluated through comparisons with upstream reference concentrations over time. The following information and series of figures were prepared by compiling available data that had been collected before and after the onset of remedial efforts.

Copper was greatly elevated above background levels in the Panther Creek watershed, downstream of the Blackbird Mine from the 1950s through 1990s. The loss of the Panther Creek population of Chinook salmon was attributed to Blackbird Mine contamination, rather than copper specifically (NMFS 1991). Blackbird Mine contamination to Panther Creek consisted mostly of copper, cobalt, arsenic, and iron (Maest and others 1994; Mebane 1994). However, based on their relative toxicities and ambient concentrations, copper was probably the biggest factor causing the loss of the Panther Creek Chinook population, although arsenic continues to contribute to aquatic risk (Section 2.4.3; NMFS 2007).

Copper was greatly elevated in the Panther Creek stream food webs, that is, sediments, periphyton, and in the tissues of aquatic insects in Panther Creek at the time of Chinook listing (Figure 2.3.1.4). The magnitude of contamination at that time was extreme, with values in sediment, periphyton, and aquatic insects hundreds of times higher than upstream background concentrations. Following initial remedial efforts, copper concentrations in Panther Creek downstream of Blackbird Mine influences dropped markedly by the mid-2000s. These efforts have led to reductions in copper concentrations in Panther Creek water, sediments and in the foodweb on the order of 90%, and are approaching upstream reference concentrations (Figure 2.3.1.4). Sediment, periphyton, and aquatic insect copper values obtained upstream of mine influences have been very consistent over time, even across different studies. This indicates that the more recent, lower copper values obtained downstream of mine influences are likely real, and cannot be attributed to methods differences.



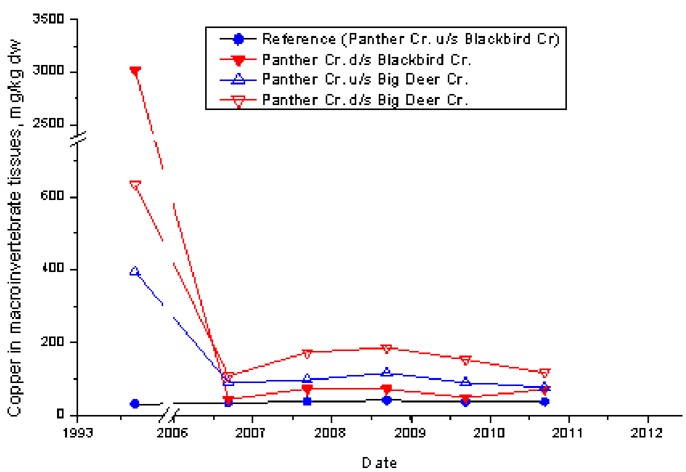




**Figure 2.3.1.4. Copper in Panther Creek foodwebs has greatly declined following Blackbird Mine remediation efforts that have been ongoing from 1995 to date. (Top) copper in sediments in 1992 and 2008 (Mebane, 1994; Golder, 2009); (Middle), copper in periphyton; and (Bottom), copper in aquatic insect tissues (Beltman and others, 1994; EcoMetrix, 2011).**

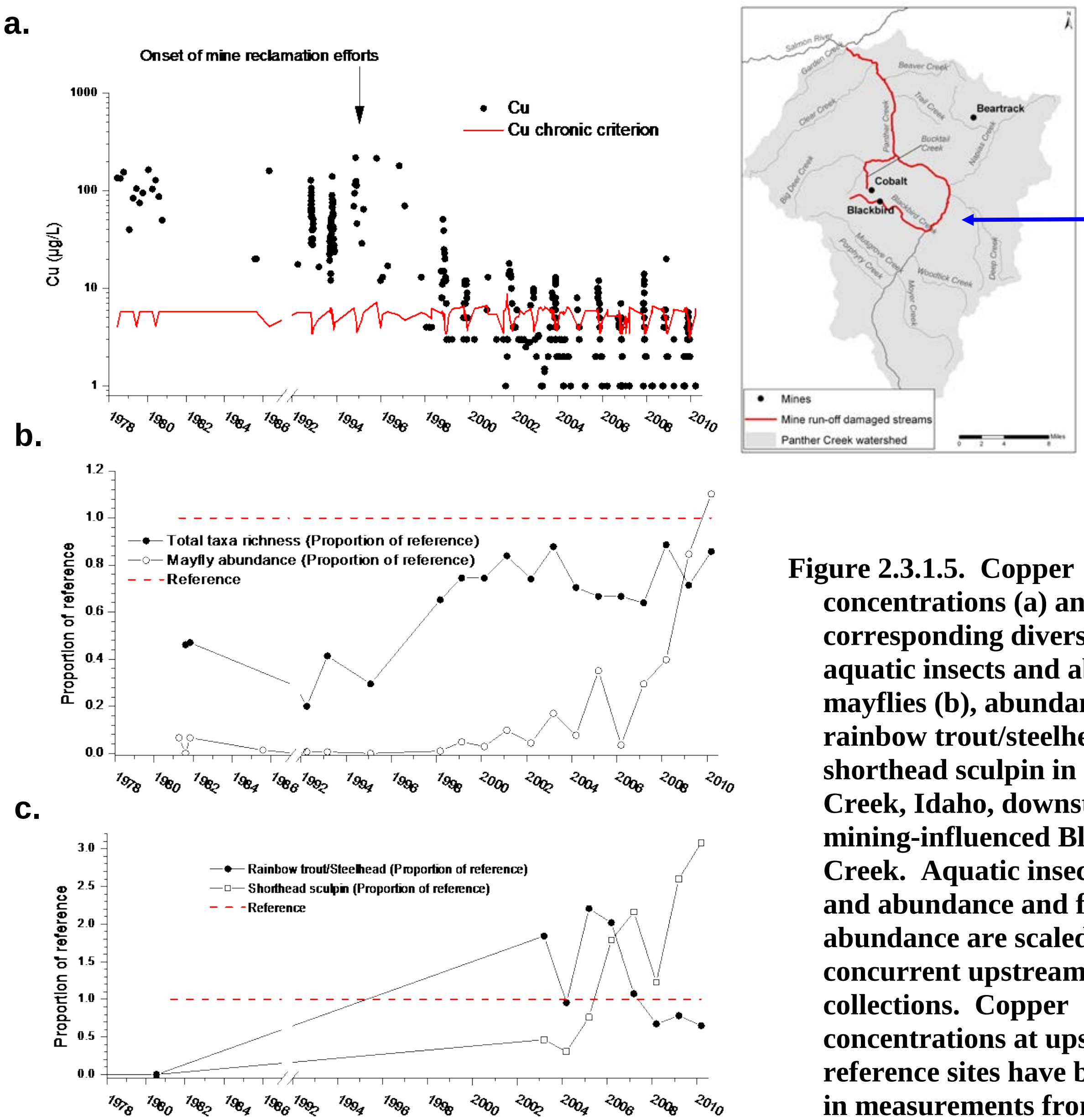



**Figure 2.3.1.5. Copper concentrations (a) and corresponding diversity of all aquatic insects and abundance of mayflies (b), abundance of rainbow trout/steelhead and shorthead sculpin in (c) in Panther Creek, Idaho, downstream of mining-influenced Blackbird Creek. Aquatic insect diversity and abundance and fish abundance are scaled against concurrent upstream reference collections. Copper concentrations at upstream reference sites have been <3µg/L in measurements from 1993 to date, since the routine use of appropriate low detection levels and "clean" field sampling and laboratory techniques.**

Copper data sources: (Davies 1982; Wai and Mok 1986; Beltman *et al.* 1993; Maest *et al*. 1994; Maest *et al*. 1995; Golder 2009); Biological data sources: (Beltman *et al*. 1994; Mebane 1994; Golder 2003; Stantec 2004; EcoMetrix 2005, 2006, 2007, 2008, 2009, 2010, 2011).

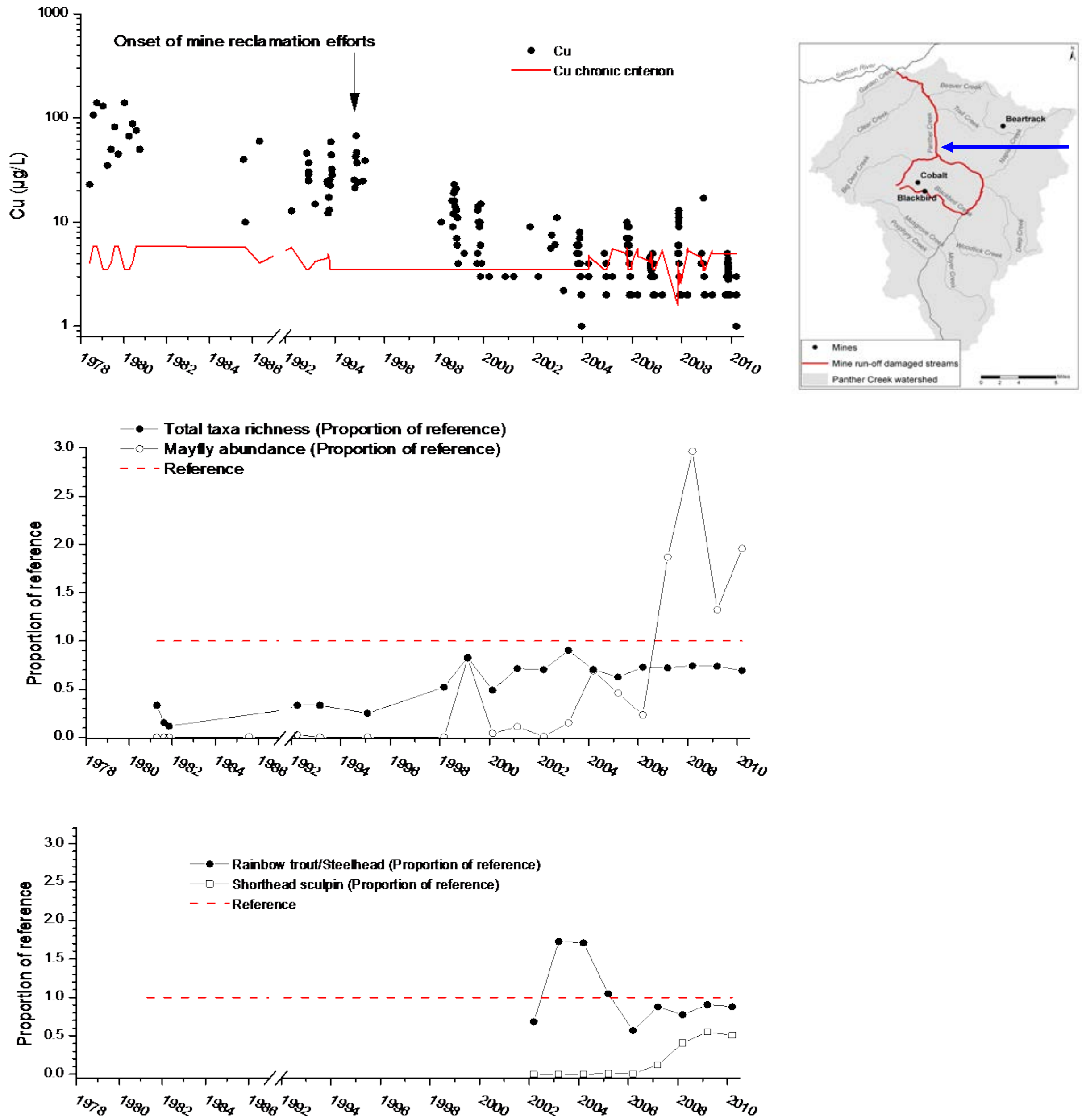



**Figure 2.3.1.6. Copper concentrations (a) and corresponding diversity of all aquatic insects and mayflies (b), and abundance of mayflies (c), in Panther Creek, Idaho, downstream of mining-influenced Big Deer Creek. (See Figure 2.3.1.5 for data sources)**

Data for copper in water and associated biological data were compiled and evaluated for four key locations: Panther Creek downstream of the upstream mining influenced tributary, Blackbird Creek (Figure 2.3.1.5), and Panther Creek downstream of the downstream mining influenced tributary, Big Deer Creek (Figure 2.3.1.6). These locations are particularly data rich with a remarkable 30-year period of record for copper and stream invertebrates. To make the invertebrate data comparable between years and between different studies, all of the results from the mining-influenced locations are scaled as a proportion of the upstream reference locations that were collected concurrently for each sampling event shown.

The ecology of Panther Creek, as measured by the abundance and diversity of aquatic insects and fish populations began to rebound as copper declined. In Panther Creek downstream of Blackbird Creek, prior to about 1998, aquatic invertebrate communities were extremely impoverished with species richness less than half that of upstream samples. Mayflies were absent or scarce. After 1998, mayflies began to appear in the samples and by 2009 were about as abundant as upstream reference (Figure 2.3.1.5, middle). Insect species richness reached about 80% of upstream reference station counts by about 2002 and seems to have plateaued. Quantitative fish data are fewer than for insects. In electrofishing surveys in 1967 and 1980, no fish of any species were captured from Panther Creek downstream of Blackbird Creek. By 2002, when the recent program of biomonitoring started, rainbow trout were more abundant than at the upstream reference. Sculpin were present but were about half the density of the nearby upstream reference stations. By 2006, the sculpin were more abundant than at upstream reference, and as the sculpins became increasingly abundant, rainbow trout densities declined (Figure 2.3.1.5, bottom).

Sculpin are emphasized in these comparisons because they may be a useful indicator species in biomonitoring of potential pollution effects. Sculpin have been observed to decline or disappear from streams with elevated metals from mining, may be more sensitive or at least as sensitive as listed salmonids, and decline with increasing proportions of fine sediments on the stream bottoms (Mebane, 2001; Maret and MacCoy, 2002; Mebane and others, 2003; Besser and others, 2007).

The insect and fish communities in Panther Creek downstream of Big Deer Creek have shown a similar recovery pattern. Prior to the mine reclamation work, insect diversity was even lower than at Panther Creek downstream of Blackbird Creek, and sculpins were completely absent until about 2006. By 2010, sculpin densities had recovered to the point where they were about half as abundant as upstream of Blackbird Creek (Figure 2.3.1.6). This does not necessarily indicate that copper concentrations are still limiting sculpin densities for two reasons. First, in Idaho, there are natural transitions in fish communities from headwaters downstream. Higher elevation headwater streams tend to be steeper and colder than lower elevation streams. Often, trout are the only fish found in perennial headwater streams. As streams drop in elevation they tend to become less steep, warmer, and larger. These mid-sized streams, such as upper Panther Creek tend to be dominated by sculpins and salmonids. As streams transition into larger rivers, the sculpin become less abundant and minnows and suckers appear (Mebane 2002b; Mebane and others 2003). Therefore, sculpin densities would be expected to decline in lower Panther Creek relative to upstream monitoring sites.

Big Deer Creek had higher copper concentrations than did Panther Creek, and biological impairment was so severe that almost all aquatic life had been extirpated. In 1992, total number of aquatic insects in Big Deer Creek upstream of mine influences ranged from 1,938 to 4,995 insects/$m^2$ compared to 0 to 68 insects/$m^2$ downstream of mine influences (Mebane 1994). No fish could be found downstream of the Blackbird Mine influences. Recovery has been slower in Big Deer Creek than Panther Creek, but by 2010 the aquatic insect communities were as diverse as upstream reference, and by 2009 rainbow trout populations had recovered to reference conditions. Sculpins do not occur in Big Deer Creek even upstream of mine pollution.

In summary, in comparison to conditions at the time that Snake River spring/summer Chinook salmon were listed, copper concentrations in Panther Creek have declined and associated biological communities have largely recovered. Aquatic insect diversity is still lower than in reference conditions. Current copper criteria are not consistently met in Panther Creek, particularly during spring runoff. However, whether these spring copper criteria exceedences are likely related to residual effects on aquatic insect communities cannot be determined from the available data. A given relatively low copper concentration such as 3 µg/L would likely be more toxic in Panther Creek during baseflow conditions from late summer to early spring than during high spring flows when more organic carbon is also present (Appendix C).

#### *2.3.2.4. Baseline for Cyanide*

NMFS located few cyanide data that were specific to Idaho. The most likely sources of cyanide in waters are probably forest fires, gold mining operations that use cyanide leaching, and perhaps road salting. The most comprehensive monitoring data were associated with the Grouse Creek Mine, located in the Yankee Fork of the Salmon River near Custer, Idaho. The Grouse Creek Mine is an inactive gold mine that operated from about 1995 to 1997, and used a cyanide vat leach process. When operating, up to 110 µg/L weak acid dissociable (WAD) cyanide was present in effluent discharged to either Jordan Creek (a tributary to the Yankee Fork), or was discharged directly to the Yankee Fork. Subsequently, cyanide levels in the effluent declined to mostly undetectable levels. In 2003 maximum effluent WAD cyanide was 3 µg/L; from 2004 through 2010 all ambient values in Jordon Creek or the Yankee Fork River were less than the detection limit of 2 µg/L (D. Landres, Hecla Mining Company, letter of 31 March 2011 to Michael Gearheard, EPA, Seattle, Washington).

While no Idaho specific data were located, the major current risk of cyanide toxicity in the action area is probably from forest fires or other biomass burning (e.g. burning waste biomass for energy conversion, crop burning, prescribed forest fires and wildfires) (Barber and others 2003; Pilliod and others 2003). Barber *et al*. (2003) examined releases of cyanides from biomass burning and their effect on surface runoff water. In laboratory test burns, available cyanide concentrations in leachate from residual ash were much higher than in leachate from partially burned and unburned fuel and were similar to or higher than a 96-h median lethal concentration ($LC_{50}$) for rainbow trout (45 µg/L). Free cyanide concentrations in stormwater runoff collected after a wildfire in North Carolina averaged 49 µg/L, again similar to the rainbow trout $LC_{50}$ and an order of magnitude higher than in samples from an adjacent unburned area (Barber and others 2003).

In other areas, greatly elevated cyanide had been shown to occur in snow exposed to urban traffic and highway deicing.  Deicing salts contain cyanide compounds as anticaking agents.  In the Cincinnati area, cyanide in snow around urban highways averaged 154 µg/L compared to 20 µg/L in urban areas that were not close to major highways (Glenn and Sansalone 2002).  Similar results could be expected in Idaho if similar deicing compounds are used.

### *2.3.2.5.  Baseline for Lead*

In natural waters, lead is usually complexed with particulate matter resulting in much lower dissolved than total concentrations (Mager 2011).  For instance, in the pervasively lead contaminated Coeur d'Alene River of northern Idaho, dissolved lead concentrations rarely exceed 20 µg/L whereas total concentrations often exceed 100 µg/L.  A maximum dissolved lead concentration of 420 µg/L was reported for this location (Clark 2002; Balistrieri and Blank 2008).  The Coeur d'Alene River is north of occupied habitat, as is the Clark Fork River, Idaho, where up to 60 µg/L dissolved lead has been reported (Hardy and others 2005).  Within the action area, reliable lead data are sparse but the available data are quite low.  The highest lead concentration obtained by the Idaho IDEQ/U.S. Geological Survey (USGS) statewide monitoring program within the action area was from the Hells Canyon reach of Snake River near Anatone, Washington  (7 µg/L).  All other measurements from within the Clearwater and Salmon River basins and the Snake River downstream of Hells Canyon dam were <1 µg/L (Hardy and others 2005).  Mebane (2000) reported lead concentrations in the upper Salmon River near the TCM as high as 2 µg/L, but most values were <0.2 µg/L.

### *2.3.2.6.  Baseline for Mercury*

Mercury is distinguished from other contaminants with natural sources (metals[4]) considered in this Opinion for several reasons, one of which is that ambient concentrations in water as well as concentrations of concern are two to four orders of magnitude lower than for other metals.  As explained in the "*Species Effects of Mercury Criteria*" (Section 2.4.6.1), there are no species effects of concern, only habitat effects through food chain exposure.  Thus the baseline concentrations of mercury are described in the context of the subsection "*Factors influencing mercury tissue concentrations in fish.*"  Generally, mercury concentrations measured in salmonids in Idaho streams and lakes ranged from <0.05 to 1.1 mg/kg ww (Table 2.4.6.2*)* Baseline concentrations of mercury in Idaho waters ranged from <0.2 to 6.8 ng/L (Table 2.4.6.2*).*

### *2.3.2.7.  Baseline for Nickel*

Nickel is rare in the waters of Idaho, even in areas disturbed by mining.  In the Blackbird Mine area, Beltman *et al*. (1993) reported Ni concentrations in mine waters and seeps in excess of 1500 µg/L; however, in the streams that were large enough to support fish populations and that were affected by mining (Blackbird and Big Deer Creeks), nickel ranged from <10 to 60 µg/L.

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[4] i.e., naturally occurring elements as opposed to invented, purely synthetic compounds such as PCBs and most pesticides

In the samples with high nickel concentrations, copper concentrations were greater than 900 µg/L which is sufficient to kill all the aquatic life without any contribution from nickel. These two streams are upstream of critical habitats. In designated critical habitats (Panther Creek and lower Big Deer Creeks) nickel was <10 µg/L. Although few other data were located, what was found indicates nickel concentrations may be assumed to be low in the action area. In the mining affected SF Coeur d'Alene River, located in northern Idaho, Mebane *et al*. (2012) reported nickel concentrations ranging from <2 to 8 µg/L.

### *2.3.2.8. Baseline for Selenium*

In Idaho rivers, the median selenium concentration determined from a probabilistic sampling of 55 river sites was 0.13 µg/L in water (range <0.09 to 1.75 µg/L) and in fish, median muscle selenium residues were 1.28 mg/kg dw (range 0.22 to 14.7 mg/kg dw) (Essig 2010). Essig's study used a randomized design, that is, each sampling site was selected from a random draw of feasible sampling sites, rather than targeting areas of interest because of potentially elevated selenium concentrations. Within the range of listed anadromous salmon and steelhead in Idaho, an area of the upper Salmon River basin was identified as having anomalously high selenium in soils, aquatic habitats, and food webs. These are evaluated further in Section 2.4.8 in the subsection *"Bioaccumulation of selenium through stream food web trophic transfer*."

### *2.3.2.9. Baseline for Silver*

Silver is sparingly soluble and rare in aquatic environments. The EPA (1987b) give natural background silver concentrations as being in the 0.1 to 0.5 µg/L. Wood (2011) however, noted that values in this range were obtained before the widespread adoption of clean sampling techniques in the 1990s and considered values in this range to be orders of magnitude too high. Instead better estimates of natural background silver concentrations were in the range of 0.1 to 5 ng/L (0.0001 to 0.005 µg/L). Such concentrations are not detectable with the technology used in non-specialty analytical laboratories. Even in highly contaminated areas, silver concentrations rarely exceed 0.1 to 0.3 µg/L. In nature, silver is unlikely to be found in its ionic form. Given the extremely high affinity of silver for reduced sulfur, most silver in the environment is expected to occur as silver sulfides, even in oxygenated waters (Wood 2011). Even in Idaho's Silver Valley where 100-plus years of silver mining resulted in one of the largest superfund cleanup projects in the nation, silver is not a contaminant of concern (NRC 2005). No data specific to the action area were located.

Although silver sulfides are the form most likely found in the environment, the form of silver usually used in toxicity tests is silver nitrate, which is much more toxic (Wood,2011). Chronic toxicity to freshwater aquatic life from silver nitrate may occur at concentrations as low as 0.12 µg/L (EPA 1980o) and the literature reviewed for silver criterion ranges from 0.3 to 11 µg/L over a hardness range of 25 to 200 mg/L.

*2.3.2.10. Baseline for Zinc*

Median baseline concentrations of zinc in large rivers, not directly influenced by mining, urban, or industrial activities are usually in the neighborhood of 0.5 to 4 µg/L (Gaillardet and others 2007). In contrast, streams with extensive mining disturbances such as the Coeur d'Alene River basin in north Idaho, sometimes have very high zinc concentrations, in excess of 2000 µg/L. Such ambient Zinc concentrations killed juvenile salmonids in hours to a few days, and fish and aquatic insect populations are depressed. (Maret and MacCoy 2002; Maret and other, 2003; Mebane and others 2012).

In mineralized areas in Idaho with naturally high zinc concentrations in watershed rock and soils, but that have not been highly disturbed, average zinc concentrations may be up to 10X higher than typical large river concentrations. In Jordan Creek, a tributary to the Yankee Fork River in the upper Salmon River subbasin, average zinc concentrations in monthly sampling from 2004 through 2009 were about 12 µg/L, with a maximum measurement of 40 µg/L. This maximum measurement is higher than Idaho's proposed acute criterion of 32 µg/L, calculated assuming the hardness was 25 mg/L, per IDEQ policy (Table 1.3.1). If the criterion were calculated using the actual measured hardness of 15 mg/L, the applicable criterion under Idaho's proposed standard would be about 24 µg/L. This sampling site is located upstream of the Grouse Creek Mine, and presumably mostly natural. Zinc concentrations measured directly in the tailings pond effluent from the Grouse Creek Mine were similar, with a 2010 mean of 11 µg/L and a maximum of 31 µg/L. In the Yankee Fork River, upstream of Jordan Creek and upstream of the tailings pond effluent outfall, the average zinc concentrations were a little lower than they were in Jordan Creek. Average 2004 to 2009 zinc concentrations were 9 µg/L with a maximum of 30 µg/L. If calculated using the sample hardness of 18 mg/L, the zinc acute criterion would be about the same, 28 µg/L (Hecla Mining Company data, sites "S-6" and "S-9," Cindy Gross, Hecla Mining Company, personal communication).

Zinc concentrations measured in the Salmon River near Clayton, in the vicinity of the TCM from 1998 to 2000 ranged from about 2 to 6 µg/L. In Thompson Creek itself, just downstream of a permitted mine effluent discharge, zinc was noticeably higher during that time period, averaging about 7 µg/L, with a maximum concentration of about 30 µg/L. Based on the minimum hardness of Thompson Creek during that period, about 50 mg/L, the acute zinc criteria as calculated under Idaho's proposed standard would be about 65 µg/L, well above measured ambient zinc concentrations downstream of the mining discharges. Upstream background zinc concentrations in Thompson Creek are about 2 µg/L (Mebane 2000).

Zinc has been elevated in a third watershed in the close vicinity of the Yankee Fork and Thompson Creek areas. Kinnikinic Creek, is a small tributary to the upper Salmon River, near Clayton, Idaho, and is the home of the Clayton Silver Mine. In 1999, zinc concentrations ranged from <5 upstream of the Clayton Silver Mine to 224 µg/L downstream of the tailings pile that was encroaching into the stream. Following a 2001 EPA removal action, IDEQ monitoring in Kinnikinic Creek yielded zinc concentrations ranging from 2 µg/L upstream of the mine to 64 µg/L just above the confluence with the Salmon River. In the latter sampling, water hardness was about 100 mg/L, which would yield zinc criteria of about 106 µg/L (IDEQ 2003).

Elsewhere in the Idaho action area, available zinc concentrations were low, with some noticeable exceptions. In USGS monitoring in the mid-1990s in the Lapwai Creek near Lapwai, Pahsimeroi River at Ellis, the Little Salmon River near Riggins, and the Snake River in Hells Canyon near Anatone, Washington, the maximum zinc concentrations were 7 µg/L (n=8). The Lemhi River near Lemhi was a noticeable exception with a maximum zinc concentration of 210 µg/L during this time period, although the median was much lower, 5 µg/L. Other streams that occasionally had anomalously high zinc measurements were Johnson Creek near Yellow Pine, the Salmon River near Salmon, and the Salmon River near White Bird, with maximum zinc measurements of 20, 16, and 24 µg/L respectively (Hardy and others 2005).

### *2.3.2.11. Baseline for Organic Pollutants*

There has not been a comprehensive water quality study conducted of organic pollutant levels in the action area, and little information concerning the occurrence of most organic pollutants is available. There are reports of measurable concentrations of PCBs, DDTs, and organochlorine pesticides (lindane, chlordane, and heptachlor) at specific sites within Idaho (Munn and Gruber 1997; Pinza *et al.* 1992; EPA 1992b; Wegner and Campbell 1991; Apperson and Anders 1990), but contamination does not appear to be extensive. Data collected as part of the National Water Quality Assessment Program in the nearby Central Columbia Plateau suggests that elevated levels of toxic organic pollutants of concern in the action area are most likely to be found in areas influenced by urbanization and agriculture (Williamson *et al.* 1998).

Because of the low usage of these compounds, water column concentrations are expected to be negligible. Water column concentration data from the Snake River, Oregon/Idaho within the Hells Canyon Dam complex are the most relevant environmental concentration data located (Table 2.3.1). The complex is just above the Hells Canyon Dam on the Snake River. Sediment and fish tissue residue data for most of the organic chemicals of concern in this Opinion were available from the lower Snake River downstream of Hells Canyon and the lower Salmon River (Clark and Maret 1998). Clark and Maret (1998) also report data from within Brownlee Reservoir and many sites in the Snake River basin upstream of Brownlee. For the most part, the highest concentrations of organic chemicals of concern within the state of Idaho occurred within Brownlee Reservoir. However, the available concentration data in water, sediment, and fish were generally close to or below the levels of detection (Table 2.3.1). The “true” concentrations from Brownlee Reservoir have some uncertainty because the analytical reporting limits for the available data were sometimes close to, and in the case of PCBs, greater than the most stringent applicable water quality criteria.

**Table 2.3.1. Baseline concentrations of organic pollutants in sediments and fish tissue measured in waters within the action area, or upstream waters that drain into the action area.**

| Substance | Most stringent water criteria from Table 1.3.1 | Water - measured values (range) | Sediment (range) | Fish tissue, any species (range) |
|---|---|---|---|---|
| | **µg/L** | **µg/L** | **mg/kg dry weight** | **mg/kg wet weight** |
| Endosulfan (α and β) | 0.056 | <0.0007 | <0.001 | No data |
| Aldrin | 0.00014 | <0.0005 | <0.001 | <0.005 |
| Chlordane | 0.00057 | 0.00082 | <0.002 | 0.020 |
| 4,4'-DDT (note 1) | 0.00059 | <0.00066 | 0.0081 | 0.072 |
| Any DDE/DDT metabolite | None | 0.00015 | 0.011 | 3.3 |
| Dieldrin | 0.00014 | 0.00093 | 0.0007 | 0.037 |
| Endrin | 0.0023 | <0.00017 | <0.002 | <0.005 |
| Heptachlor | 0.00021 | <0.00097 | <0.001 | <0.005 |
| Lindane (gamma-BHC) | 0.063 | No data | <0.001 | <0.005 |
| Polychlorinated biphenyls (PCBs) | 0.000045 | <0.1 | <0.05 | 0.160 |
| Pentachlorphenol (PCP) (note 2) | 6.2 | 0.00047 | <0.001 | <0.005 |
| Toxaphene | 0.0002 | No data | <0.2 | 0.26 |

Data sources: Water data from Brownlee Reservoir, 2011, Idaho Power Co., unpublished data; Sediment and Fish tissue, various locations in Idaho although highest values tended to be from Brownlee Reservoir (Clark and Maret 1998). Note 1: Sediment and tissue DDT samples are as p,p'-DDT; Note 2: as pentachloroanisole, the principal degradation product of PCP.

## 2.4. Effects of the action on the species and its Designated Critical Habitat

"Effects of the action" means the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur.

This analysis identifies potential effects of each of the criteria that would be expected to occur if water concentrations were equal to the proposed criteria.

NMFS' general analytical approach for evaluating effects for the various chemical criteria under consideration was to first consider general issues related to EPA's methodology for deriving the criteria, which affect all or multiple criteria. We then evaluated the individual constituent criteria for potential species or habitat effects on listed salmon and steelhead. Consistent with the two part structure of EPA's aquatic life criteria, on which the proposed Idaho criteria are based, with CMC to protect against short-term effects of exposures to criteria chemicals, and a CCC to protect against long or indefinite term exposures, the protectiveness of the CMCs were evaluated against data on effects in short-term exposures (≤ 96 hours) and CCCs were evaluated against data on effects in longer-term exposures.

In most instances, direct testing evidence for the listed salmon species was not available, and test data obtained with other fish species was used as surrogate estimates of potential effects to listed salmon. Steelhead were an exception, since they and rainbow trout are different forms of the same species (Behnke and Tomelleri 2002; Quinn 2005). In most cases, rainbow trout data were available since rainbow trout are commonly tested in ecotoxicology. Rainbow trout are often used as a surrogate for all listed *Oncorhynchus*, using geometric means. At least with several metals, rainbow trout are probably similar in sensitivity to Chinook salmon and probably considerably more sensitive than sockeye salmon. Few direct data with sockeye salmon were located, which may be related to Chapman's (1975) recommendation against testing sockeye salmon following his observations that they were much less sensitive to metals than were Chinook or coho salmon or rainbow/steelheads (Chapman 1975).

In addition to Idaho's aquatic life criteria, EPA has also approved Idaho criteria designed to protect human health from recreational, fish consumption, and drinking water uses which are also applicable to the waters in the action area. In practice, when multiple criteria are applicable to the same water body, the most stringent criteria will drive discharge limits and other pollution management efforts (IDEQ 2007a; subsection 70.1, "Applicability of standards, multiple criteria"). For our analysis, if review of the aquatic life CCC indicated that adverse effects to listed species or their habitats were likely, then we reviewed the human health-based ambient water quality criteria concentrations for the same substance to see if the human-health concentrations would be protective of the listed steelhead and salmon.

### 2.4.1. Evaluation of issues that are common to multiple aquatic life criteria

All criteria being evaluated as part of this action were developed by EPA following EPA's guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. For short, these are referred as the "Guidelines" (Stephan *et al*. 1985). Thus it is important to consider the structure of the Guidelines in regard to protection of listed salmon, steelhead, and their critical habitats to evaluate whether criteria derived following them would likely be protective.

The EPA's Guidelines for criteria development represent the best judgments of a committee of EPA scientists as of the mid-1980s. As the title states, the objectives of the criteria development was the "protection of aquatic organisms and their uses." Because the Guidelines are quite detailed and have much explicit guidance, their use has tended to make criteria documents (the supporting documents prepared by EPA in deriving national recommended water quality criteria) objective, transparent, and reproducible. However, the Guidelines recognize that ecotoxicology and criteria derivation cannot be reduced to a series of decision rules, and many judgments are required to produce an individual criteria document. Because the Guidelines are fundamental to criteria, they are fundamental to the evaluation of the protectiveness of criteria for ESA-listed species and habitats. The fundamental assumptions and procedures in the Guidelines are inherent to their degree of protectiveness for listed salmon and steelhead. Thus some of key criteria derivation steps are briefly described here and the underlying assumptions are critically examined.

The Guidelines include some fundamental assumptions:

- Effects which occur on a species in appropriate laboratory tests will generally occur on the same species in comparable field situations.

- For a given substance, if average species sensitivities are rank ordered, the species sensitivity distributes itself in a rather consistent way for most chemicals. Thus, each species tested is not representative of any other species but is one estimate of the general species sensitivity (i.e. a point along the distribution).

- The goal of aquatic life criteria is to protect aquatic communities and socially valued species within those communities. Aquatic organisms may have ecologically redundant functions in communities. The loss of some species might not be important if other species would fill the same ecological function. Thus it is not necessary to protect all of the species all of the time.

- If 95% of the species in acceptable datasets were protected, that would be sufficient to protect aquatic ecosystems in general. In the ecological risk assessment literature, this is often referred to as the $5^{th}$ percentile of a species sensitivity distribution (SSD) or shortened to the HC5 approach, for the hazardous chemical concentration adversely affecting no more than 5% of the species in a natural community.

- To estimate a criterion protective of 95% of the species, it is acceptable to extrapolate from compilations of severely toxic effects from short-term, “acute” tests to less severe effects in long-term, “chronic” exposures.

- If one or more water quality characteristics such as temperature, pH, or water hardness affect the acute toxicity of a substance in a predictable way, then the acute criterion for that substance should be expressed as a function of that characteristic. It is acceptable to assume that toxicity relationships established with short-term exposure data, such as those between water-hardness and metals toxicity, would be the same in long-term exposures. Thus acute-toxicity and hardness or other relations may be applied equally to chronic criteria (Stephan *et al.* 1985; Stephan 1985; Stephan 2002)

Relying on these assumptions, the EPA Guidelines are derived with the following general steps (Stephan *et al.* 1985):

- First, datasets of acute (short-term) responses of aquatic organisms to the substance of interest are compiled and screened for data sufficiency, relevance and quality.

- If a water quality characteristic is considered to affect the toxicity of the substance, then a relation is developed and the acute data are normalized to a common water condition. For example, with several metals, hardness-toxicity regressions were developed and used to adjust acute toxicity values to a common hardness of 50 mg/L.

- The adjusted acute data are averaged to obtain species mean acute values (SMAVs), and SMAVs are averaged to obtain genus mean acute values (GMAVs). The GMAVs are rank ordered, and value close to the $5^{th}$ percentile most sensitive genus is calculated, called the final acute value (FAV). The FAV is divided by 2 to extrapolate from a lethal concentration for sensitive taxa to a concentration expected to kill few sensitive taxa. The FAV/2 value becomes the CMC, which is commonly referred to as the acute criterion.

  [In this procedure, if multiple values for a species were available, with differing sensitivities, a geometric mean of all values was taken to calculate the SMAV. If different SMAVs were available, a geometric mean was similarly calculated. For example, with EPA's 1984 copper criteria, the SMAVs for Chinook, Coho and Sockeye salmon were calculated as 42, 70, and 233 µg/L, and a GMAV of 89 µg/L was calculated to represent all *Oncorhynchus*. In that era, steelhead and rainbow trout were considered in a different genus, *Salmo*.]

- Chronic (long-term) data are compiled, and acute-to-chronic ratios (ACRs) are calculated for at least 3 species. These are calculated by matching acceptable acute and chronic tests and dividing the acute $LC_{50}$ by the "Chronic Value" from the chronic test. The chronic value in turn is calculated as the geometric mean of the highest tested concentration in which selected responses were not statistically significantly different from the controls, called the no observed effect concentration (NOEC), and the lowest concentration that was statistically different from the controls, called the lowest observed effect concentration (LOEC). The selected responses considered are survival, growth, and reproduction, data on other sublethal effects such as swimming performance, or altered behaviors are put aside. The available ACRs are then selectively averaged, for a Final ACR for the substance. The continuous criterion concentration (CCC), commonly called the chronic criterion then becomes the FAV divided by the final ACR (Stephan *et al.* 1985).

This synopsis reflects the most common way the Guidelines were used with the criteria evaluated in the Opinion, but obviously doesn't reflect all the details of Stephan *et al*.'s (1985) 98 page document.

These steps and other key judgments and practices from the EPA Guidelines for developing aquatic life criteria are critically evaluated in the following parts of this section.

*2.4.1.1. The assumption that not harming more than 5% of the species tested in laboratories is sufficient protection of ESA-listed species and critical habitats*

The EPA's fundamental approach to setting criteria involves compiling reports of laboratory tests for species and genus mean values, rank ordering the genus mean values, and basing criteria on the $5^{th}$ percentile of a distribution of the rank ordered values. This approach has been the subject of much criticism and controversy in the ecotoxicology literature. Many arguments relate to further inherent assumptions required of the approach that may not be met, are untested, or are untestable. Published concerns include:

- Whether haphazard collections of data from single-species laboratory toxicity tests can be considered relevant to natural ecosystems;

- Small datasets can be significantly biased toward more or less sensitive species than would be expected in natural ecosystems;

- Whether any species loss from a community due to a toxin is acceptable. Reducing community integrity to a simple proportion of species could discount keystone or dominant species if they were in the lower $5^{th}$ percentile of sensitivity;

- Whether the $5^{th}$ percentile of the SSD as the appropriate level of protection is a scientifically sound number or just a familiar number;

- Because the approach depends on comparable data, it is biased toward mortality data (which are most abundant) and biased against less abundant data on abnormal behavior or other sublethal data that may be as important for maintaining biological integrity and more relevant at low, ambient concentrations;

- The few species for which multiple tests results are available sometimes show high variability in sensitivity, yet this variability is often omitted from SSD presentations, which implies greater precision than is the case. Thus apparent differences between species' ranks on a SSD may not be meaningful, especially for species with only single or few datapoints; and

- Uncertainties in the statistical properties of the distributions and appropriate models.

(Cairns 1986; Forbes and Forbes 1993; Hopkin 1993; Smith and Cairns 1993; Underwood 1995; Power and McCarty 1997; Aldenberg and Jaworska 2000; Newman *et al*. 2000; Forbes and Calow 2002; Suter *et al*. 2002; Duboudin *et al*. 2004; Brix *et al*. 2005; Maltby *et al*. 2005; Forbes *et al*. 2008)

In contrast to these many criticisms, other studies or reviews have found reasonably good agreement between effects in laboratory and field tests (Geckler *et al*. 1976; de Vlaming and Norberg-King 1999), and lack of pronounced adverse effects in ecosystem tests at criteria-like concentrations below the $5^{th}$ percentiles of SSDs (Versteeg *et al*. 1999; Mebane 2010).

No explicit consideration of protection of exceptionally vulnerable populations of threatened or endangered species was included in the criteria guidelines. However, it is clear from contemporaneous and subsequent writings by the authors that they thought criteria should specifically protect or be adjusted to protect socially valued special status species, including threatened and endangered species. For instance, the introduction to the Guidelines states that "*to be acceptable to the public and useful in field situations, protection of aquatic organisms and their uses should be defined as prevention of unacceptable long-term and short-term effects on (1) commercially, recreationally, and other important species….*" as well as fish and invertebrate assemblages (Stephan *et al*. 1985). Other writings and guidance are more explicit about the need to consider protection of species listed under the ESA; suggesting a review of whether the 95% of protected species included listed species and adequate prey for them (Stephan 1985, 1986; EPA 1994). If not, the criteria should be adjusted to protect these "critical" species. Such reviews and adjustments were recommended to be done on a site-specific basis, where a "site" may be a state, region, watershed, water body, or segment of a water body (EPA 1994). The recommendation to consider listed species at the "site" rather than national level was not stated but presumably related to complexity and the fact that imperiled species often have limited distributions.

*2.4.1.2. The assumption that effects in laboratory tests are reasonable predictors of effects in field situations*

The preceding discussion concerned whether compilations of laboratory test values were appropriate to treat as surrogates of the diversity of natural systems. A related but even more fundamental question is, whether tests of chemicals in laboratory aquaria with "domesticated" cultures of test animals are likely to produce similar effects as would exposure to the same substance on the same or closely related species in the wild? If the responses between animals in laboratory aquaria or the wild are different, is there likely a bias in the sensitivity of responses from either the lab or wild settings? That is, are the effects of chemical contamination more likely to be more or less severe in the laboratory or wild settings? This question is important because water quality criteria are designed to apply to and protect ambient waters, that is, streams, rivers, and lakes, yet the data used to develop them are invariably compiled from laboratory testing under tightly controlled and thus quite artificial environments.

While by definition, laboratory toxicity testing is conducted in controlled, artificial condition rather than in the wild under uncontrolled conditions, some laboratory tests are designed such that they are of questionable environmental relevance. By "environmentally relevant" in the context of interpreting laboratory toxicity tests we mean whether the test conditions were designed in a way to be relevant to conditions that might occur in the environment. Whether or not test data were environmentally relevant include the questions such as: Were fish or other organisms exposed to chemicals in concentrations ranges and ratios that actually occur in the environment? Or were organisms exposed to conditions contrived to produce effects, such as massive doses over short time periods? Were organisms exposed in a manner similar to that in the wild such as by water across the gills or diet? Or were organisms exposed in a manner designed to produce effects but wouldn't occur outside of laboratories, such as injection or a bollus in feed? In feeding studies, were chemicals in a form similar to that that might be encountered in ambient conditions? In water studies, was the dilution water a natural water type, rather than a preparation with mineral content unlike that that would occur in nature? "Environmental relevance" cannot be a hard and fast test, because studies would then be limited to field studies, which have the converse problem of being uncontrolled and difficult to unambiguously attribute apparent effects to causes. However, some studies clearly have little direct environmental relevance, and these studies are given less reliance in this opinion than "environmentally relevant" studies. For instance, in vitro tests using excised tissues, or cell lines bathed in a dosed solution are often valuable for investigations comparative biochemistry orphysiology, or on mechanisms of toxicity, but standing alone, have little direct relevance responses of a whole, living organism under conditions experienced in the wild.

There are myriad of factors that may influence the effects of a chemical stressor on aquatic organisms, and this complexity makes the question of bias in sensitivity difficult or even impossible to answer with any certainty. A number of reasons why the effects of a chemical could be more- or less-severe on listed steelhead and salmon in laboratory or in wild settings were considered and are summarized in table 2.4.1.1.

**Table 2.4.1.1. Reasons why the effects of a chemical substance could be more- or less-severe on listed steelhead and salmon in laboratory or in wild settings**

| Factor | Are effects likely more severe in typical lab settings or in the wild? |
|---|---|
| Environmental Conditions | |
| Nutritional state - acute test exposures | In the wild. In acute toxicity tests with fish fry, fish are selected for uniform size, and unusually skinny fish that might be weakened from being in poor nutritional state are culled from tests. For instance, if <90% of control fish survive the 4 days starvation of an acute toxicity test, the test may be rejected from inclusion in the criteria dataset. In the wild, not all fish can be assumed to be in optimal nutritional state. While perhaps counterintuitive, starvation can protect fish against waterborne copper exposure (Kunwar *et al*. 2009). Fish are routinely starved during acute laboratory tests of the type used in criteria development. |
| Nutritional state – chronic test exposures | In the wild. Fish in the wild must compete for prey and if chemicals impair fish's ability to detect and capture prey because of subtle neurological impairment, this could cause feeding shifts and reduce their competitive fitness (Riddell *et al*. 2005). Fish in chronic lab tests with waterborne chemical exposures are often fed to satiation and food pellets don't actively evade capture like live prey. Perhaps these factors dampen responses in lab settings. |
| Temperature | In the wild. In lab test protocols, nearly optimal test temperatures are recommended, e.g., 12°C for rainbow trout, the most commonly tested salmonid. Fish may be most resistant to chemical insults when at optimal temperatures. At temperatures well above optimal ranges, increased toxicity from chemicals often results from increased metabolic rates (Sprague 1985). Under colder temperatures fish have been shown to be more susceptible to at least Cu, Zn, Se and cyanide, although the mechanisms of toxicity are unclear (Hodson and Sprague 1975; Kovacs and Leduc 1982b; Dixon and Hilton 1985; Erickson *et al*. 1987; Lemly 1993b; Hansen *et al*. 2002a). |
| Flow | In the wild. Fish expend energy to hold their position in streams and to compete for and defend preferred positions that provide optimal feeding opportunity from the drift for the energy expended. Subordinate fish are forced to less profitable positions and become disadvantaged. Subordinate fish in lab settings still get adequate nutrition from feeding. Chemical exposure can reduce swimming stamina or speeds, as can exposure to soft water. Chemical exposures in soft water can be expected to exacerbate effects (Adams 1975; Kovacs and Leduc 1982b; McGeer *et al*. 2000; De Boeck *et al*. 2006). |
| Disease and parasites | In the wild. Disease and parasite burden are common in wild fish, but toxicity tests that used diseased fish are likely to be considered compromised and results would not be used in criteria compilations. Chemical exposure may weaken immune responses and increase morbidity or deaths (Stevens 1977; Arkoosh *et al*. 1998a,b). |
| Predation | In the wild. Fish use chemical cues to detect and evade predators; these can be compromised by some chemical exposures (Berejikian *et al*. 1999; Phillips 2003; Scott *et al*. 2003; Labenia *et al*. 2007). |

| Factor | Are effects likely more severe in typical lab settings or in the wild? |
|---|---|
| **Exposure** | |
| Variable exposures | In the lab. Most toxicity tests used to develop criteria are conducted at nearly constant exposures. Criteria are expressed not just as a concentration but also with an allowed frequency and duration of allowed exceedences. In field settings, most point or non-point pollution scenarios that rarely if ever exceed the criteria concentration (i.e., no more than for one four day interval per 3 years), will have an average concentration that is less than the criteria concentration. For some chemicals, such as copper, fish might detect and avoid harmful concentrations if clean-water refugia were readily available. |
| Metal form and bioavailability | Uncertain. Metals other than Hg and some organics are commonly assumed to be more bioavailable in the lab because dissolved organic carbon (DOC), which reduces the bioavailability and toxicity of several metals, is low in laboratory tests that are eligible for use in criteria. The Guidelines call for <5 mg/L TOC (total organic carbon) in order to be used in criteria (Stephan *et al*. 1985), but probably more often TOC is <2 mg/L in laboratory studies However, in mountainous streams in Idaho, TOC is often as low (≈1-2 mg/L) during baseflow conditions (Appendix C), so differences in bioavailability between streams and laboratory waters that both have low TOC are not necessarily large. (Organic carbon is more often discussed as DOC in this Opinion. TOC includes particulates, which other than during runoff conditions in streams will tend to be low and thus TOC and DOC would be similar during conditions without runoff). |
| Chemical equilibria | Uncertain. While results conflict, metals are usually considered less toxic when in equilibrium with other constituents in water, such as organic carbon, calcium, carbonates and other minerals. In the wild, daily pH cycles prevent full equilibria from being reached (Meyer *et al*. 2007a). Likewise, in conventional laboratory flow-through test designs chemicals may not have long enough contact time to reach equilibria. Static-renewal tests are probably nearly in chemical equilibria although organic carbon accretion can lessen toxicity which may not reflect natural settings (Santore *et al*. 2001; Welsh *et al*. 2008). |
| Prior exposure | Uncertain. If fish are exposed to sublethal concentration of a chemical, they could potentially either become weakened or become more tolerant of future exposures. With some metals, normally sensitive life stages of fish may become acclimated and less sensitive during the course of a chronic test if the exposure was started during the resistant egg stage (Chapman 1983, 1985; Sprague 1985; Brinkman and Hansen 2007). (further discussion follows in the text). |
| Life stages exposed | In the wild. Most lab studies are short term; realistically testing all life stages of anadromous fish is probably infeasible. Reproduction is often the most sensitive life stage with fish but most "chronic" studies are much shorter and just test early life stage survival and growth (Suter *et al*. 1987). At different life stages and sizes, salmonids can have very different susceptibility to some chemicals; even when limited to a narrow window of YOY fry, sensitivity can vary substantially (this review). Unless the most sensitive life stages are tested, lab tests could provide misleadingly high toxicity values for listed species (further discussion follows in the text). |

| Factor | Are effects likely more severe in typical lab settings or in the wild? |
|---|---|
| Chemical mixtures | In the wild. In field conditions, organisms never experience exposure to a single pollutant; rather, ambient waters typically have low concentrations of numerous chemicals. The toxic effects of chemicals in mixture can be less than those of the same chemicals singly, greater than, or have no appreciable difference. The best known case of one toxicant reducing the effects of another is probably Se and Hg (e.g., Belzile *et al*. 2006). However, strongly antagonistic responses are probably uncommon, and much more common are situations where chemical mixtures have greater toxicity than each singly or little obvious interaction (e.g., Norwood *et al*. 2003; Borgert 2004; Playle 2004; Scholz *et al*. 2006; Laetz *et al*. 2009). In general, it seems prudent to assume that if more than one toxicant were jointly elevated it is likely that lower concentrations of chemicals would be required to produce a given magnitude of effect than would be predicted from their actions separately. However, the magnitude or increased effects at environmentally relevant concentrations is uncertain and for some combinations may be slight or imperceptible. |
| Dietary exposures | In the wild. Toxicity test data used in criteria development have been mostly based solely on waterborne exposures, yet in the wild, organisms would be exposed to contaminants both through dietary and water exposures. With at least some organics (e.g., dioxins, PCBs) dietary exposures are more important than water exposures as is the case for some inorganics (As, Hg, Se). For some other metals (Cd, Cu, Ni, Pb, Zn), at environmentally relevant concentrations that would be expected when waterborne concentrations are close to criteria, dietary exposures have not been shown to directly result in appreciable adverse effects to fish (Hansen *et al*. 2004; Schlekat *et al*. 2005; Erickson *et al*. 2010). However, while dietary exposures of metals have not yet been implicated in adverse effect to fish at or below criteria concentrations, they may in fact be both the primary route of exposure and an important source of toxicity for benthic invertebrates (Irving *et al*. 2003; Poteat and Buchwalter 2014). For instance Besser *et al*. (2005a) found that the effects threshold for Pb to the benthic crustacean *Hyalella* was well above the chronic criterion in water exposures, but when Pb was added to the diet, effects threshold dropped to near criteria concentrations. Ball *et al*. (2006) found that feeding Cd contaminated green algae to the benthic crustacean *Hyalella* caused a 50% growth reduction at about the NTR chronic criteria. |
| **Population dynamics** | |
| Density effects | In the lab. Salmonid fishes are highly fecund (~500 to 5000 eggs per spawning female). When abundant, overcrowding and competition for food and shelter may result in relatively high death rates for some life stages, particularly YOY during their first winter. After many fish die in a density-dependent bottleneck, the survivors have greater resources and improved growth and survival. Conceptually, if an acute contamination episode killed off a significant portion of YOY fish prior to their entering a resource bottleneck, then assuming no residual contaminant effects, the losses to later life stages and to adult spawners would be buffered. |
| Meta-population dynamics | In the lab. If habitats are interconnected, as is the case in intact stream networks, then if pervasive contamination from discharges to a stream were to impair only some endpoints or life-stages, such as reproductive failure or YOY mortalities, immigration from source populations may make detection of population reductions in the affected sink population difficult (Ball *et al*. 2006; Palace *et al*. 2007). If an episodic contamination pulse were to kill a large proportion of fish in a stream, the proximity of refugia and donors from source populations affect recovery rates (Detenbeck *et al*. 1992). |

Considering all the reasons why the effects of a given chemical concentration could have more or less severe effects in laboratory settings or the wild, general conclusions are elusive. It may be that the best overall conclusion is the same as that reached by Chapman (1983) that "*when appropriate test parameters are chosen, the response of laboratory organisms is a reasonable index of the response of naturally occurring organisms.*" His conclusion in turn contributed to one the most fundamental assumptions of EPA Guidelines, that is, "*these National Guidelines have been developed on the theory that effects which occur on a species in appropriate laboratory tests will generally occur on the same species in comparable field situations.*"

Summary: Based on this analysis, the assumption that effects in laboratory tests are reasonable predictors of effects to species in the wild is dependent upon the specific factor being considered. While it is generally reasonable to interpret effects from laboratory tests as being applicable to field situations where criteria are applied, there is some risk that laboratory tests may underpredict effects in the wild.

### *2.4.1.3. Susceptibility of Salmonids to Chemicals at Different Life Stages*

Since a species can only be considered protected from acute toxicity if all life stages are protected, EPA's Guidelines recommend that if the available data indicate that some life stages are at least a factor of two more resistant than other life stages, the data for the more resistant life stages should not be used to calculate species mean acute values (Stephan *et al*. 1985). Smaller, juvenile life stages of fish are commonly expected to be more vulnerable to metals toxicity than larger, older life stages of the same species. For instance, a standard guide for testing the acute toxicity of fish recommends that tests should be conducted with juvenile fish, that is, post-larval or older and actively feeding, usually in the size range from 0.1 and 5.0g in weight (ASTM 1997).

A review of several data sets in which salmonids of different sizes were similarly tested shows that even among juvenile fish in the 0.1 to 5.0g size range, differences in sensitivity can approach a factor of 10. This emphasizes the importance of EPA's guidance not to use the more resistant life stages. However, the data sets analyzed indicated that in practice, there were sometimes greater influences of life stage on the sensitivity of salmonids to some substances than was apparent to the authors of the individual criteria documents using the datasets available to them at the time. Some of the SMAVs and GMAVs which were used to rank species sensitivity and set criteria were considerably higher than $EC_{50}$s with salmonids that were tested at the most sensitive life stages (Figures 2.4.1.1 to 2.4.1.4).

For three Pacific salmonid species for which comparable test data were available for different life stages; coho salmon (*Oncorhynchus kisutch*), rainbow trout (*O. mykiss*) and cutthroat trout (*O. clarki*), the data suggest that swim-up fish weighing around 0.5g to about 1g may be the most sensitive life stage. None of the data sets examined in detail or other published studies reviewed had sufficient resolution to truly define at what weight fish became most sensitive to metals, but along with other data they suggest that larger fish may be less sensitive than fish at 0.4 to 0.5g. For instance with zinc, rainbow trout in the size range of about 0.1 to about 1.5 g consistently became more sensitive to zinc in two studies with multiple tests in that size range (Figure 2.4.1.2

and Figure 2.4.1.3). The paucity of data with salmonids in the size range of about 0.5 to 2g prevents definitive statements of a most sensitive size across species or even tests. All data located for early swim-up stage *Oncorhynchus* in the 0.1 to 0.5g range were consistent with increasing sensitivity with size. With Hansen *et al's.* (2002c) rainbow trout studies, this relationship continued with fish up to about 1.5g. However, with cutthroat trout, the few data available suggests that fish larger than about 0.5g become less sensitive with increasing size (Figure 2.1.4.2).

Some studies with older and larger rainbow trout have found that the fish became more resistant to zinc and copper (Chapman 1978b; Chapman and Stevens 1978; Howarth and Sprague 1978; Chakoumakos *et al*. 1979). Studies with copper all showed this trend, but the strength of size-sensitivity relations varied across studies. Chakoumakos *et al*. (1979) found that fish between about 1 and 25g in weight varied in their sensitivity to copper by about eight times (Figure 2.4.1.4), but steelhead (*O. mykiss*) that were tested with copper at sizes of 0.2, 7, 70, and 2700g showed little pattern of sensitivity with size (Chapman 1978b; Chapman and Stevens 1978). However, the large differences in sizes may have missed changes at intermediate sizes in the ranges compared at Figures 2.4.1.1 to 2.4.1.4. Similarly, with copper and rainbow trout, Anderson and Spear (1980) found that three sizes of rainbow trout (3.9, 29 and 176g) had similar sensitivities.

NMFS reviewed several data sets that indicated increasing susceptibility of salmonids to at least metals with increasing size and age as fish progressed from the resistant alevin stage. The "U" shaped size-sensitivity response with the most sensitive life stage for salmonids fish around 0.5g in weight seems a reasonable interpretation of the available data, but few data were available in the size range of 0.5 to 2g, so it is possible the most sensitive stage is larger. Hedtke *et al*. (1982) tested coho salmon for the influence of body size and developmental stage with copper, zinc, nickel, and PCP. Fish were exposed as alevins, swim-up fry, and juveniles, and within these developmental stages smaller fish were tested against larger fish. For copper, zinc, and PCP, the swim-up fry stage was most susceptible, and within the swim-up stage, the larger fish were more susceptible to copper and zinc than smaller fish (~0.25g vs. 0.7g fish, wet weight). For PCP, there was no difference for size of fish within the sensitive alevin to swim-up stage, and with Ni all fish were very resistant (Hedtke *et al*. 1982). In three test pairs with rainbow trout exposed to cadmium and zinc under similar hardness, pH, and temperature, the fish tended to become more sensitive with increasing size from 0.4 to 0.9g for rainbow trout and zinc, and 0.26 to 0.66g with Cd. Further growth in juvenile rainbow up to 1.1 and 1.6g for cadmium and zinc had little effect on sensitivity (Figure 2.4.1.3). In parallel tests with bull trout (*Salvelinus confluentus)*, size had little effect on sensitivity over a range of 0.08 to 0.22g for cadmium although with zinc; however, the smallest fish (0.1g) were also least sensitive (Hansen *et al*. 2002c). Similar tests with copper and rainbow and bull trout showed roughly similar patterns. Three tests with rainbow trout at the same hardness and using fish from the same source had the most sensitive results for 0.43g fish ($LC_{50}$s of 36, 54, and 93 µg/L for rainbows weighing 0.43, 0.3, and 0.68g, respectively). Bull trout tested at constant temperature of 8°C tended to become more sensitive with increasing size up to ~1g (Hansen *et al*. 2002a). Besser *et al*. (2007) similarly found that 0.5g rainbow trout were more sensitive than 0.13g fish to copper and zinc, but not for cadmium.

These patterns do not seem to hold for all species. Contrary to the patterns with the salmonids, newly hatched sculpins were more sensitive to cadmium, copper, and zinc than were older juveniles (Besser *et al*. 2007). Similar to the sculpin results but contrary to all the other salmonid results, Carney *et al*. (2008) found that the brown trout (*Salmo trutta*) became less sensitive to copper with increasing size. Guppies exposed to toxicants with different modes of action tended to become more susceptible with increasing size and age (dieldrin, PCP, cyanide, copper, zinc, and nickel) (Anderson and Weber 1975).

<u>Summary:</u> Salmonids can have profound differences in susceptibility to chemicals at different life stages, and in some instances, species mean acute values used in criteria may be skewed high because insensitive life stages were included. A "U" shaped pattern of sensitivity with life stage was suggested for several datasets with Pacific salmon or trout species (i.e., *Oncorhynchus)* and some metals. Across several good datasets, the most vulnerable life stage and size appeared to be swim-up fry weighing between about 0.5 to 1.5g. However, no consistent pattern was obvious across other species of fish, chemicals, and life stages.

Caution is needed when using SMAVs or GMAVs as summary statistics for ranking species sensitivity or setting criteria. Reviews of the protectiveness of chemical concentrations or criteria that rely in large part upon published mean acute values for species of special concern such threatened species, or their surrogates, may be subject to considerable error if the underlying data points are not examined. This may include analyses such as SSD, interspecies correlation estimates (ICE, Asfaw *et al*. (2004), or any other relative sensitivity comparisons that uses mean acute values at the family, genus, or species level.

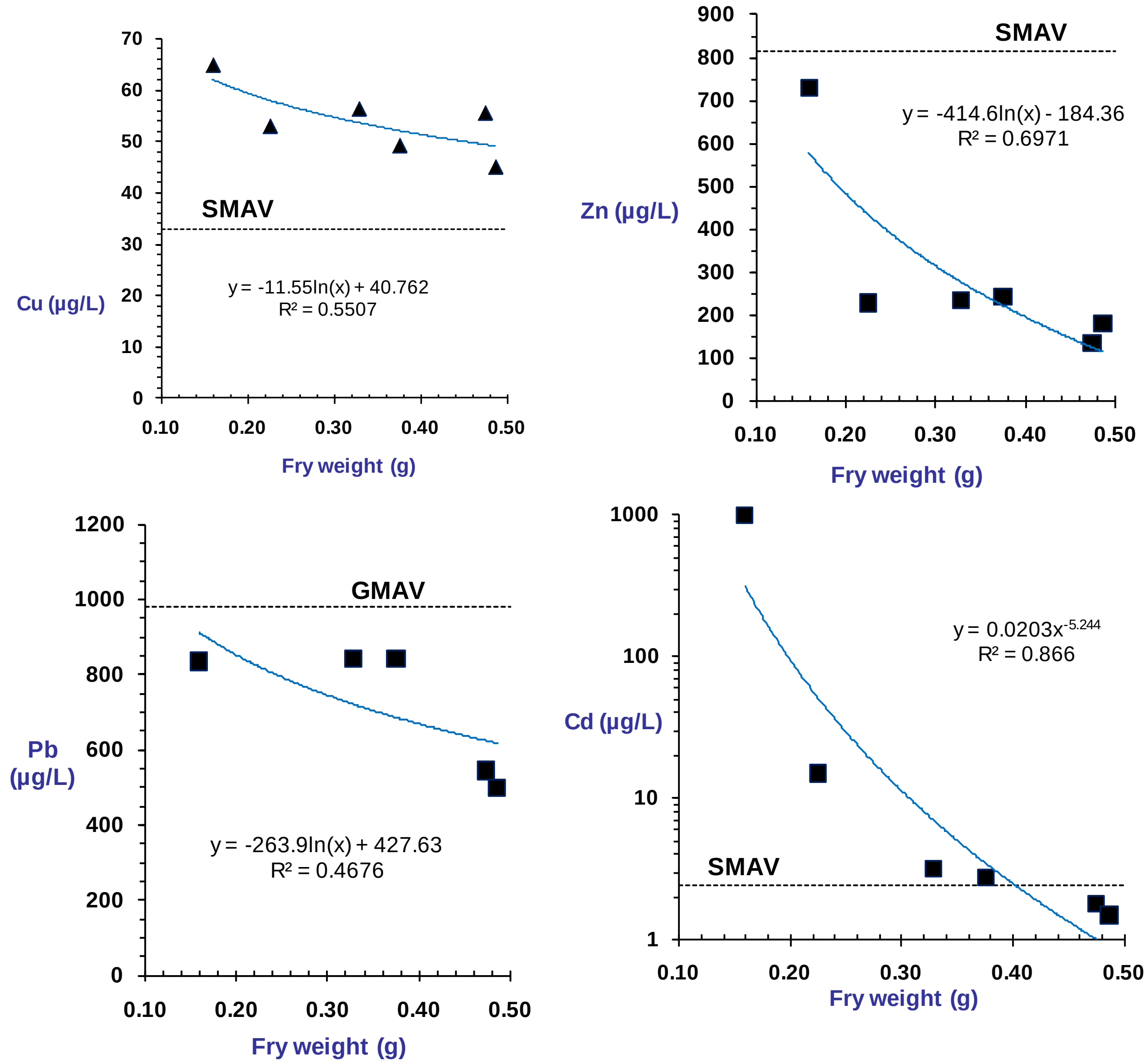



**Figure 2.4.1.1. Size-developmental stage patterns with coho salmon from 2 to 7 weeks post hatch, data from Chapman (1975). Species and genus mean acute values (SMAVs and GMAV) are from the respective criteria documents (EPA 1984b, 1984a, 1985, 1987b), adjusted to test water hardness. All tests used Willamette River water, TOC 3.4 mg/L, hardness 22 mg/L.**

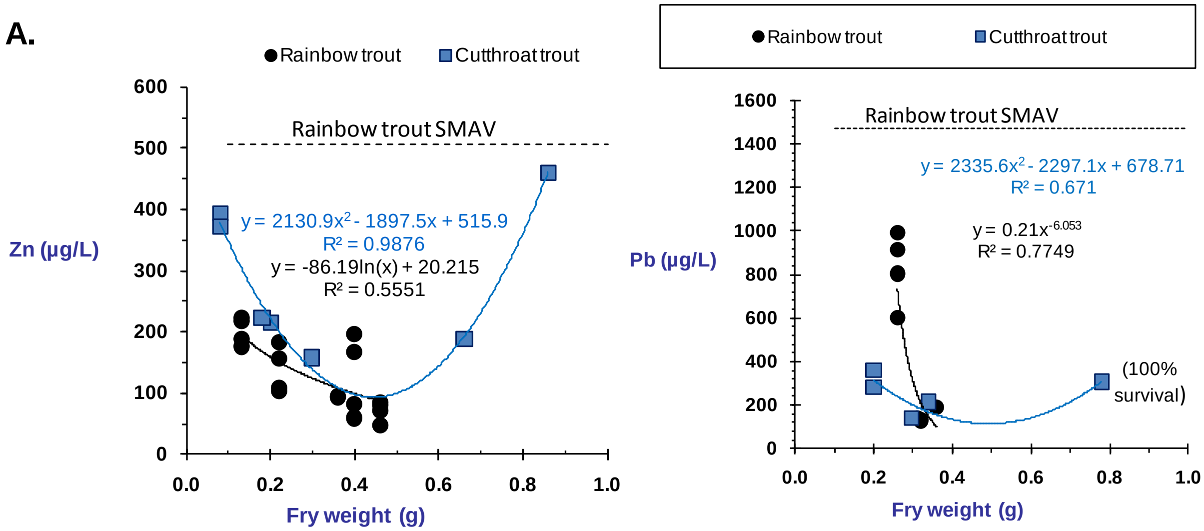



**Figure 2.4.1.2. Relations between size of swim-up rainbow and cutthroat trout and toxicity to zinc and lead sensitivity in renewal tests conducted in water from the South Fork Coeur d'Alene River, Idaho. Data from (Mebane *et al*. 2012). All test values adjusted to a median test hardness of 35 mg/L $CaCO_3$ using hardness-toxicity regressions from (Mebane *et al*. 2012). SMAVs were adjusted using the hardness-criteria equations from the respective criteria documents.**

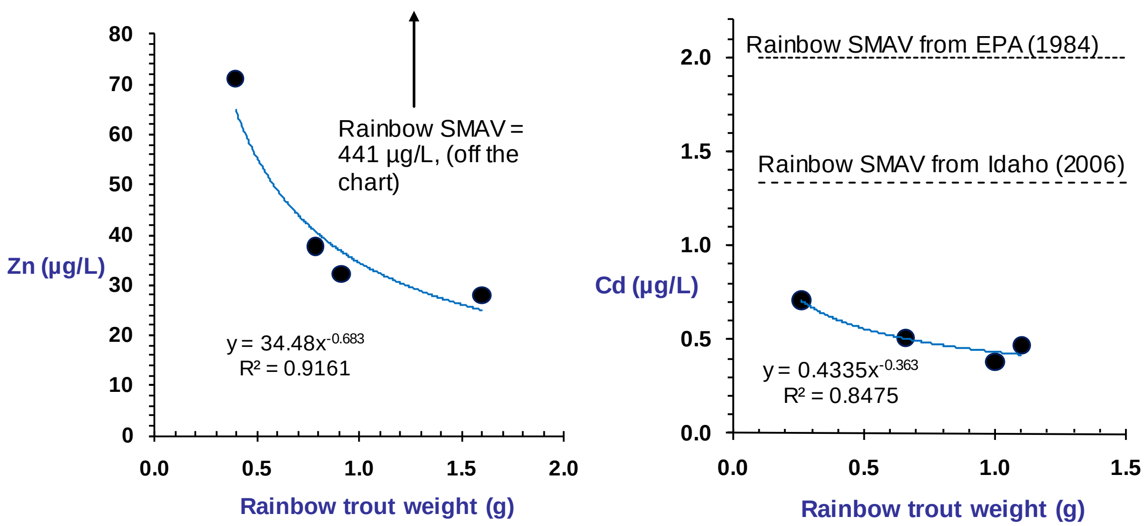



**Figure 2.4.1.3. Resistance to cadmium and zinc toxicity decreased with increasing size over a weight range of 0.2 to 1.6g for swim-up rainbow trout. Data from Hansen (2002a) and Stratus (1999) using 96-h probit $LC_{50}$ values. All tests conducted at a hardness of 30 mg/L and pH of 7.5 SMAV values were adjusted using the hardness-criteria equations from the respective criteria documents.**

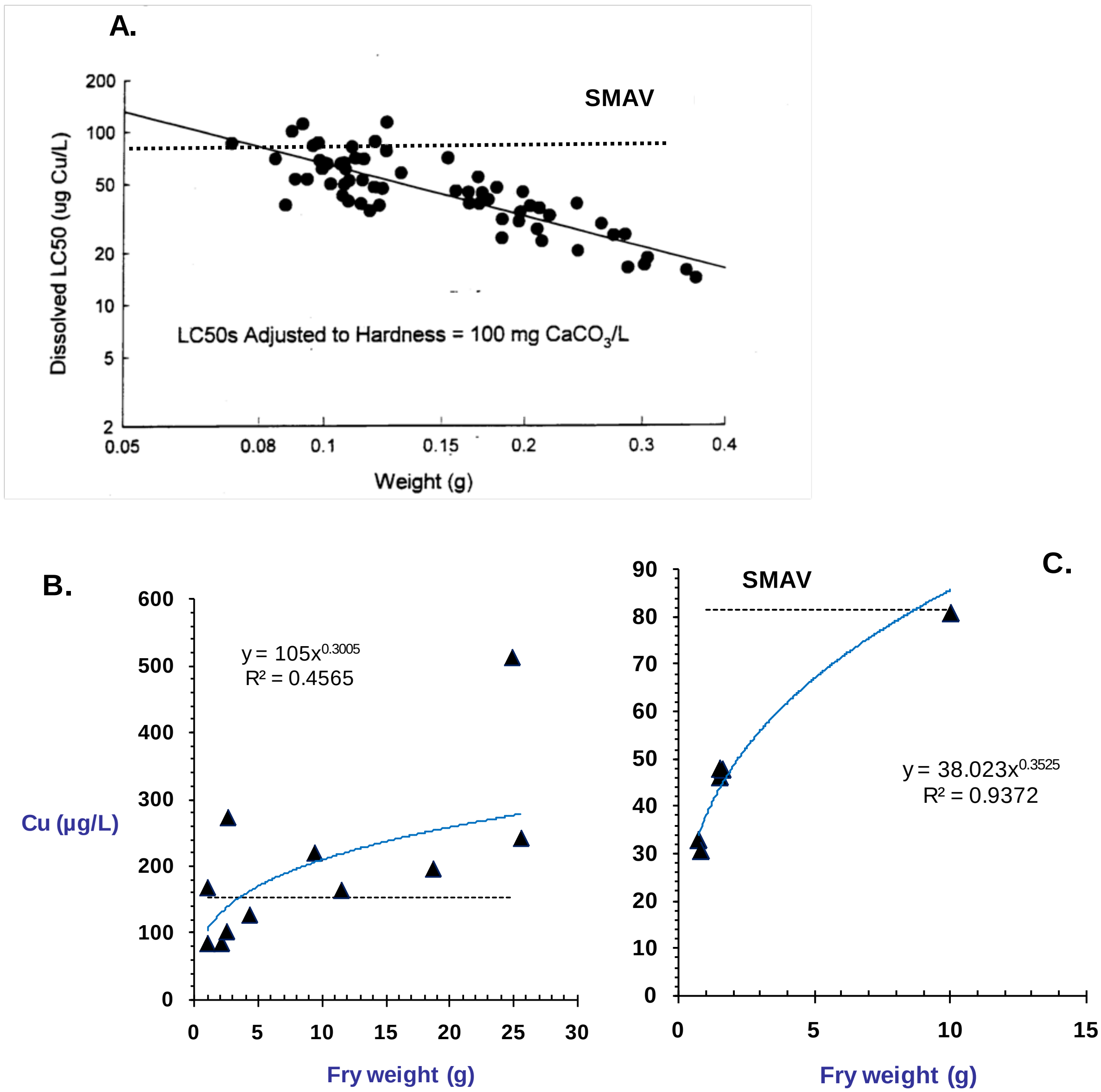



**Figure 2.4.1.4. Resistance to copper toxicity decreased with increasing size over a weight range of 0.06 to 0.4g for swim-up rainbow trout, but above about 1g weight, resistance to copper toxicity increased with increasing size. Dashed lines indicate hardness-adjusted rainbow trout species mean acute value (SMAV) from EPA (1984). A. Relation between copper toxicity and the size of swim-up rainbow trout (<0.5g), from renewal tests conducted in water from the Clark Fork River, MT (Erickson *et al*. 1999); B. Relation between copper toxicity and the size of larger juvenile rainbow trout (>0.7g, older than swim-up fish), data from Chakoumakos *et al*'s (1979) tests under uniform water conditions (hardness 194 mg/L); C. Rainbow trout of difference sizes tested under uniform conditions at hardness 99 to 102 mg/L, data from Howarth and Sprague (1978).**

*2.4.1.4. Effects of Acclimation on Susceptibility to Chemicals*

Exposure to sublethal concentrations of organic chemicals and other metals may result in pronounced increases in resistance to later exposures of the organisms. With metals, the increased resistance may be on the order of two to four times for acute exposures, but may be much higher for some organic contaminants (Chapman 1985). However, the increased resistance can be temporary and can be lost in as little as 7 days after return to unpolluted waters (Bradley *et al*. 1985; Sprague 1985; Hollis *et al*. 1999; Stubblefield *et al*. 1999). For this reason, EPA's Guidelines specify that test results from organisms that were pre-exposed to toxicants should not be used in criteria derivation (Stephan *et al*. 1985).

However, there is a less obvious source of acclimation that is not precluded by the Guidelines and influences chronic values and thus chronic criteria. Several tests have shown that life stages typically sensitive to toxins (e.g., fry stage) become more resistant when toxicity tests were initiated during resistant early life stages (ELS, e.g., embryo stage). This suggests that acclimation to toxin(s) during ELS exposure may lead to greater resistance in later life stages in comparison to the same life stages of naïve fish (fish which had no previous exposure) (Chapman 1978a; Spehar *et al*. 1978; Chapman 1994; Brinkman and Hansen 2004, 2007). The Guidelines could actually be interpreted to exclude chronic exposures that did not pre-expose, and acclimate fish to metals as eggs (Stephan *et al*. 1985), which was probably unintended.

Chapman (1994) exposed different life stages of steelhead (*Oncorhynchus mykiss*) for the same duration (3 months) to the same concentration of copper (13.4 µg/L at a hardness of 24 mg/L as $CaCO_3$). The survival of steelhead which were initially exposed as embryos was no different from that of the unexposed control fish, even though the embryos developed into the usually-sensitive swim-up fry stage during the exposure. In contrast, steelhead which were initially exposed as swim-up fry without the opportunity for acclimation during the embryo state, suffered complete mortality (Figure 2.4.1.4). Brinkman and Hansen (2007) compared the responses of brown trout (*Salmo trutta*) to long-term cadmium exposures that were initiated either at the embryo stage (i.e., ELS tests) or the swim-up fry stage (i.e., chronic growth and survival tests). In three comparative tests, fish that were initially exposed at the swim-up fry stage were consistently two to three times less resistant than were the fish initially exposed at the embryo stage.

These studies support the counterintuitive conclusion that because of acclimation, longer-term tests or tests that expose fish over their full life cycle are not necessarily more sensitive than shorter-term tests which are initiated at the sensitive fry stage. Conceptually, whether this phenomenon is important depends on the assumed exposure scenario. If it were assumed that spawning habitats would be exposed, then the less-sensitive ELS tests would be relevant. However, for migratory fishes such as listed salmon and steelhead, their life histories often involve spawning migrations to headwater reaches of streams, followed downstream movements of fry shortly after emerging from the substrates, and followed by further seasonal movements to larger, downstream waters to overwinter (Willson 1997; Baxter 2002; Quinn 2005). These life history patterns often correspond to human development and metals pollution patterns such that headwater reaches likely have the lowest metals concentrations, and downstream increases could occur due to point source discharges or urbanization.

From the discussion in the Guidelines of the types of chronic data with fish that are acceptable for use in criteria development, it is clear that the intent was to capture information on the most sensitive life stage of a fish species. Unfortunately, the wording of the Guidelines could be interpreted to preclude the use of the more sensitive chronic growth and survival tests that were initiated with salmonid fry stage, and specify the use of the less sensitive ELS tests (Stephan *et al*. 1985, at p. 44).

<u>Summary:</u> In chronic tests with salmonids and metals, the Guidelines inadvertently favor a test method (ELS tests) that may be inherently biased toward insensitivity because acclimation can occur during the insensitive egg stage of exposure. Thus, Species Mean Chronic Values listed in criteria documents may be also be biased high.

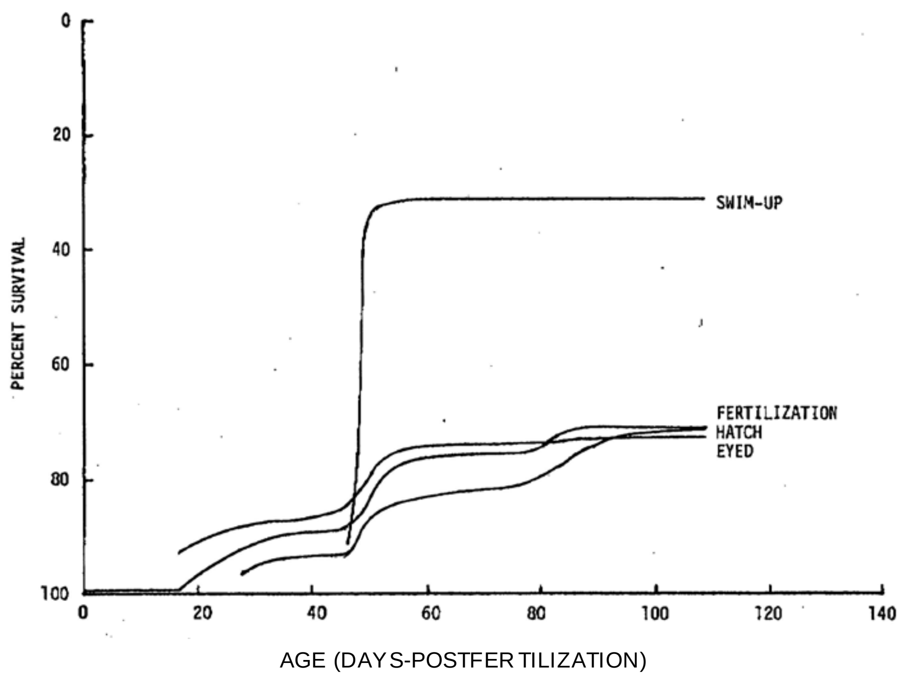



**Figure 2.4.1.5. Effect of developmental stage at the onset of continuous copper exposure (13.4 µg/L) on the survival of juvenile steelhead trout (figure from Chapman 1994).**

*2.4.1.5. Implications of the use of the "chronic value" statistic in setting criteria*

A related issue with the derivation of chronic criteria is the test statistic used to summarize chronic test data for species and genus sensitivity rankings. Literature on chronic effects of chemicals often contains variety of measurement endpoints, different terms, and judgments by the authors of what constitutes an acceptable or negligible effect. While the Guidelines give a great deal of advice on considerations for evaluating chronic or sublethal data (Stephan *et al*. 1985, at p.39), those considerations were not usually reflected in the individual criteria documents reviewed for this consultation. In practice for most of the criteria documents reviewed, "chronic values" were simply calculated as the geometric mean of the lowest tested concentration that had a statistically significant adverse effect at the 95% confidence level (lowest observed effects concentration [LOEC]) and the next lower tested concentration (no observed effects concentration [NOEC]). The "chronic value" as used in individual criteria documents is effectively the same thing as the maximum acceptable toxicant concentration (MATC) used in much environmental toxicology literature, even though the MATC term is never used in the Guidelines. This MATC approach has the potential to seriously underestimate effects because the statistical power in typical toxicity tests is fairly low. A bias in many ecotoxicology papers is to focus on avoiding "false accusations" of a chemical with 95% accuracy (i.e., Type I error or false positive, the risk of declaring an effect was present when in fact the apparent effects only occurred by chance). Often no consideration whatsoever is given to the companion problem, known as Type II error, or false negatives, (i.e., declaring no adverse effects occurred when in fact they did but because of the limited sample size or variability, were not significant with 95% confidence).

The magnitude of effect that can go undetected with 95% confidence in a NOEC statistic can be large, greater than 30% on average for some endpoints, and much higher for individual tests (Crane and Newman 2000). This problem is compounded with the "chronic value" or MATC when calculated in its most common form as the geometric mean of a NOEC and LOEC. For instance, 100% of juvenile brook died after being exposed to 17 µg/L copper for 8 months; this was considered the LOEC for the test. The next lowest concentration tested (9.5 µg/L) had no reduced survival relative to controls (McKim and Benoit 1971). Therefore, the only thing that can be said about the geometric mean of these two effect concentrations, i.e., the chronic value of 12.8 µg/L that was used in the chronic copper criteria (EPA 1985d) is that it represents a concentration that can be expected to kill somewhere between all or no brook trout in the test population. Similarly, Grosell *et al*. (2006a) showed that the NOECs and LOECs for reduced growth in snails exposed to lead corresponded with about a 57% and 90% growth reduction, and over 70% reduced growth for the MATC. Animals suffering such severe stunted growth may not even reproduce, so the MATC would not seem to be a very acceptable maximum toxicant concentration. Suter *et al*. (1987) evaluated published chronic tests with fish for a variety of chemicals and found that on the average the MATC represented about a 20% death rate and a 40% reduction in fecundity. They noted that "*although the MATC is often considered to be the threshold for effects on fish populations, it does not constitute a threshold or even a negligible level of effect in most of the published chronic tests. It corresponds to a highly variable level of effect that can only be said to fall between 0% and 90%.*" Barnthouse *et al*. (1989) further extrapolated MATC-level effects to population-level effects using fisheries sustainability models and found that the MATC systematically undervalued test responses such as fecundity, which are both highly sensitive and highly variable.

One implication of this issue is that because the MATC chronic values typically used in criteria documents under review may represent substantial adverse effects for that test species, the criteria on the whole will be less protective than the intended goal of protecting 95% of the species. How much less protective is unclear and probably varies among the criteria datasets. One dataset from which a hypothetical NOEC-based chronic criterion could readily be recalculated and compared with the usual MATC criteria was a 2006 cadmium criteria update

(Mebane 2006). In this comparison, the MATC-based chronic criteria would protect about 92% of the aquatic species in the dataset at the NOEC level. Because the NOEC statistic also can reflect a fairly sizable effect (Crane and Newman 2000), it may be that at least with Cd, the true level of protection is closer to about 90% than the 95% intended by the Guidelines.

A specific question for interpreting ecotoxicological data to evaluate the protectiveness of species listed under the ESA is, what level of effect is "insignificant?" "Insignificant effects" have been defined in this context to "*relate to the size of the impact and should never reach the scale where take occurs*" and "*based on best judgment, a person would not be able to meaningfully measure, detect, or evaluate insignificant effects*" (USFWS and NMFS 1998). To evaluate what test statistic best approximated a "true" no-effect concentration for evaluating risks to ESA-listed species, we made a limited comparison of NOECs versus regression or distribution-based methods for estimating no- or very low effects concentrations. The alternative statistics evaluated were the lower 95$^{th}$ percentile confidence limit of the concentration affecting 10% of the test population (LCL- EC10), or estimates of the EC1 or EC0 (1% or 0% effects). NMFS concluded that the EC0 was the preferred, best estimate of no-effect value from a toxicity test. However, if data were insufficient to calculate an EC0 or other regression based approaches, the NOEC may be the best available statistic for estimating "insignificant" effects (Appendix B).

<u>Summary:</u> The Chronic Value statistic is calculated by splitting the difference between an adverse effects concentration (the LOEC) and a concentration expected to have low adverse effects (the NOEC). However, in practice the NOEC can have more adverse effects than implied by the term "NOEC", and splitting the difference between two adverse effects concentrations produces another adverse effect concentration. Thus the Chronic Value statistic used to set chronic criteria through ACRs, etc., in practice produces an uncertain level of effect and may result in less protection than intended by the EPA Guidelines. This has been estimated to result in a level of protection was closer to about 90% of the species represented in an SSD than the 95% intended by the Guidelines.

#### *2.4.1.6. The assumption that dividing a concentration that killed 50% of a test population by two will result in a safe concentration*

One challenge for deriving aquatic life criteria for short-term (acute) exposures is that the great majority of available data is for mortality, which is a concentration that kills 50% of a test population. A fundamental assumption of EPA's criteria derivation methodology is that the FAV, the $LC_{50}$ for a hypothetical species with a sensitivity equal to the 5$^{th}$ percentile of the SSD, may be divided by two in order to extrapolate from a concentration that would likely be extremely harmful to sensitive species in short-term exposures (kill 50% of the population) to a concentration expected to kill few, if any, individuals. This assumption, which must be met for acute criteria to be protective of sensitive species, is difficult to evaluate from published literature because so few studies report the data behind an $LC_{50}$ test statistic. While $LC_{50}$s are almost universally used in reporting short-term toxicity testing, they are not something that can be "measured" but are statistical model fits. An acute toxicity test is actually usually a series of four to six tests run in parallel in order to test effects at different chemical concentrations. An

$LC_{50}$ is estimated by a statistical distribution or regression model which generates an $LC_{50}$ estimate, usually a confidence interval, and then all other information is thrown away. Thus, while the original test data included valuable information on what concentrations resulted in no, low, or severe effects, that information is lost to reviewers unless the unpublished raw lab data are available to them.

The assumption that dividing an $LC_{50}$ by two will result in a no- or very low effects concentration rests on further assumptions of the steepness of the concentration-response slope. Several examples of tests with metals which had a range of response slopes are shown in Figure 2.4.1.6. We selected these examples from data sets that were relevant to salmonid species in Idaho and for which the necessary data to evaluate the range of responses could be located (Chapman 1975, 1978b; Marr *et al*. 1995b; Marr *et al*. 1999; Mebane *et al*. 2010; Mebane *et al*. 2012).

The citations are to reports with detailed enough original data to examine the mortality at the $LC_{50}$ concentration divided by two. The vast majority of published data was inadequate for this comparison, because usually only the $LC_{50}$s are reported, not the actual responses by concentration. We examined around 100 tests for this comparison. The examples shown in Figure 2.4.1.6 range from tests with some of the shallowest concentration-response slopes located to very steep response slopes. In the shallowest tests (panels *A and E*), an $LC_{50}/2$ concentration would still result in 15% to 20% mortality. However, a more common pattern with the metals data was that an $LC_{50}/2$ concentration would probably result in about a 5% death rate (panels B and F), and in many instances, no deaths at all would be expected (panels C and D).

In one of the few additional published sources that gave relevant information, Spehar and Fiandt (1986) included effect-by-concentration information on the acute toxicity of chemical mixtures. Rainbow trout and *Ceriodaphnia dubia* were exposed for 96 and 48 hours, respectively, to a mixture of six metals, each at their presumptively "safe" acute CMC. In combination, the CMC concentrations killed 100% of rainbow trout and *Ceriodaphnia*, but 50% of the CMC concentrations killed none (Spehar and Fiandt 1986). This gives support to the assumption that dividing a lethal concentration by two would usually kill few if fish, although it does not bode well for arguments of the overall protectiveness of criteria concentrations in mixtures.

Other reviews include Dwyer *et al*. (2005b) who evaluated the "$LC_{50}/2$" assumption with the results of the acute toxicity testing of 20 species with five chemicals representing a broad range of toxic modes of action. In those data, multiplying the $LC_{50}$ by a factor of 0.56 resulted in a low (10%) or no-acute effect concentration. Testing with cutthroat trout and cadmium, lead, and zinc singly and in mixtures, Dillon and Mebane (2002) found that the $LC_{50}/2$ concentration corresponded with death rates of 0% to 15%.

<u>Summary:</u> The assumption that one-half of an $LC_{50}$ concentration for a sensitive test, i.e., a concentration near the $5^{th}$ percentile of the ranked species sensitivities, will result in little or no deaths was supported by several data sets plus two published articles. While up to 20% mortality was calculated, in most cases the expected morality associated with a $LC_{50}/2$ was less than 10% and often zero.

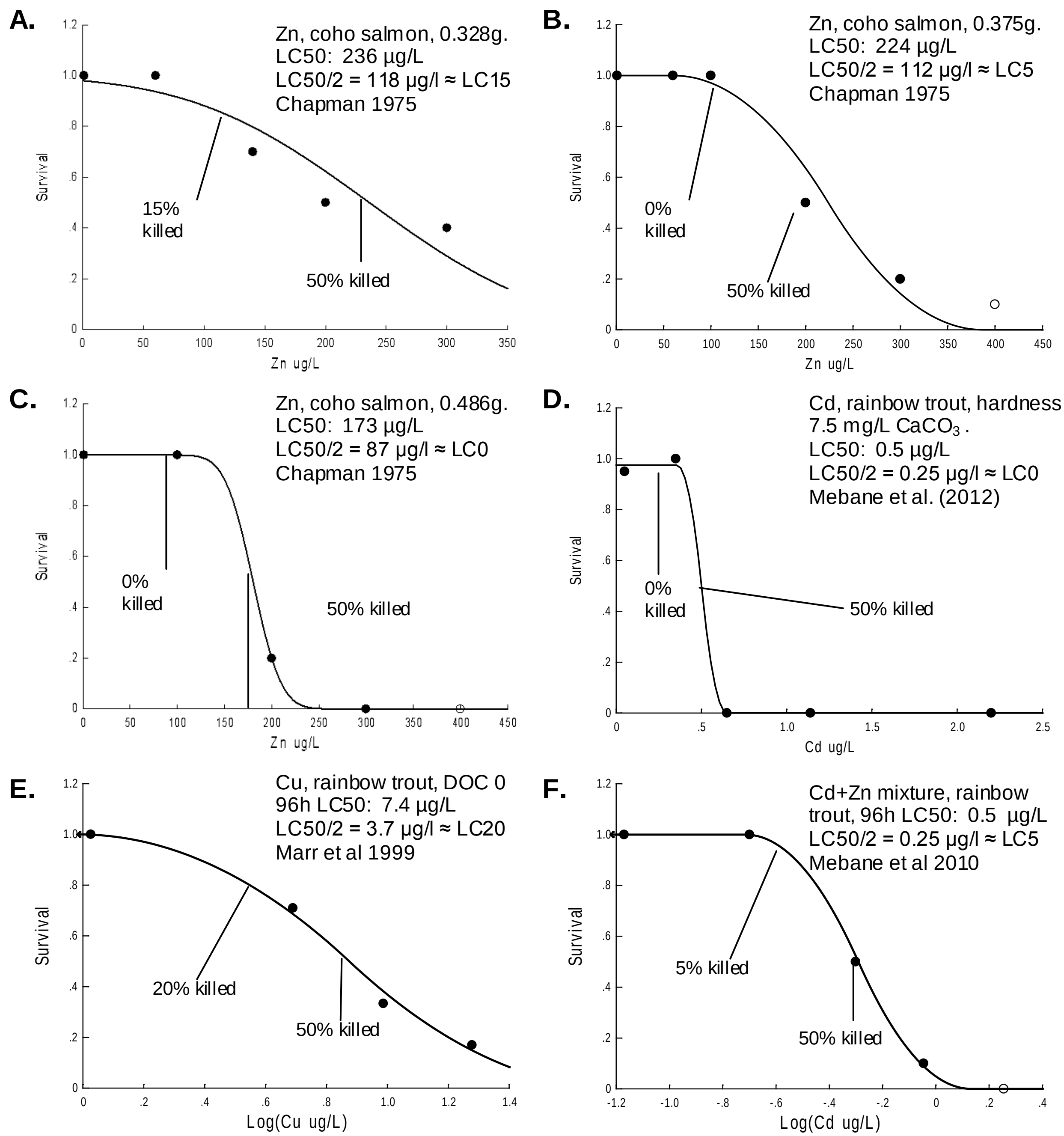



**Figure 2.4.1.6. Examples of percentages of coho salmon or rainbow trout killed at one-half their $LC_{50}$ concentrations with cadmium, copper, and zinc.**

*2.4.1.7. Issue of Using Flow Through, Renewal, or Static Exposure Test Designs*

One area of controversy in evaluating toxicity test data or risk assessments or criteria derived from them has to do with potential bias in how test organisms are exposed to test solutions. Exposures of test organisms to test solutions are usually conducted by variations on three techniques. In "static" exposures test, solutions and organisms are placed in chambers and kept there for the duration of the test. The "renewal" technique is like the static technique except that test organisms are periodically exposed to fresh test solution of the same composition, usually once every 24 or 48 hours, by replacing nearly all the test solution. In the "flow-through" technique, test solution flows through the test chamber on a once-through basis throughout the test, usually with at least five volume replacements/day (ASTM 1997).

The term "flow-through test" is commonly mistaken for a test with flowing water, i.e., to mimic a lotic environment in an artificial stream channel or flume. This is not the case; rather the term refers to the once-through, continuous delivery of test solutions (or frequent delivery in designs using a metering system that cycles every few minutes). Flows on the order of about 5-volume replacements per 24 hours are insufficient to cause discernible flow velocities. In contrast, even very slow moving streams have velocities of around 0.04 ft/sec (a half inch per second) or more. At that rate, a parcel of water would pass the length of a standard test aquarium (~2 ft) in about 48 seconds, resulting in about 3,600 volume replacements per day. At more typical stream velocities of about 0.5 ft/sec would produce over 20,000 volume replacements/day.

Historically, flow-through toxicity tests were believed to provide a better estimate of toxicity than static or renewal toxicity tests because they provide a greater control of toxicant concentrations, minimize changes in water quality, and reduce accumulation of waste products in test exposure waters (Rand *et al*. 1995). Flow-through exposures have been preferred in the development of standard testing protocols and water quality criteria. The EPA Guidelines first advise that for some highly volatile, hydrolysable, or degradable materials, it is probably appropriate to use only results of flow-through tests. However, this advice is followed by specific instructions that if toxicity test results for a species were available from both flow-through and renewal or static methods, then results from renewal or static tests are to be discounted (Stephan *et al*. 1985). Thus, depending upon data availability, toxicity results in the criteria databases may be a mixture of data from flow through, renewal, or static tests, raising the question of whether this could result in bias. In the 1985 Guidelines, the rationale for the general preference for flow-through exposures was not detailed, but it was probably based upon assumptions that static exposures will result in $LC_{50}$s that are biased high (apparently less toxic) than comparable flow-through tests or because flow-through tests are assumed have more stable exposure chemistries and will result in more precise $LC_{50}$ estimates.

With metals, renewal tests have been shown to produce higher $EC_{50}$s (i.e., metals were less toxic), probably because of accretion of dissolved organic carbon (DOC) (Erickson *et al*. 1996; Erickson *et al*. 1998; Welsh *et al*. 2008). However, in contrast to earlier EPA and American Society for Testing and Materials (ASTM) recommendations favoring flow-through testing, Santore and others (2001) suggested that flow-through tests were biased low because copper complexation with organic carbon, which reduces acute toxicity, is not instantaneous and typical flow-through exposure systems allowed insufficient hydraulic residence time for complete copper-organic carbon complexation to occur. Davies and Brinkman (1994) similarly found that cadmium and carbonate complexation was incomplete in typical flow-through designs, although in their study incomplete complexation had the opposite effect of the copper studies, with cadmium in the aged, equilibrium waters being more toxic. A further complication is that it is not at all clear that natural flowing waters should be assumed to be in chemical equilibria because of tributary inputs; hyporheic exchanges; and daily pH, inorganic carbon, and temperature cycles. Predicting or even evaluating risk of toxicity through these cycles is complex and seldom attempted (Meyer *et al*. 2007a), in part because pulse exposures cause latent mortality (i.e., fish die after exposure to the contaminant is removed), a phenomenon that is often overlooked or not even recognized in standard acute toxicity testing.

When comparing data across different tests, it appears that other factors such as testing the most sensitive sized organisms or organism loading may be much more important than if the test was conducted by flow through or renewal techniques. For instance, Pickering and Gast's (1972) study with fathead minnows and cadmium produced flow-through $LC_{50}$s that were lower than comparable static $LC_{50}$s (~ 4,500 to 11,000 µg/L for flow-through tests versus ~30,000 µg/L for static tests). The fish used in the static tests were described as "immature" weighing about 2g (2000 mg). The size of the fish used in the Pickering and Gast (1972) their flow-through acute tests were not given, but is assumed to have been similar. In contrast, 8- to 9-day old fathead minnow fry usually weigh about 1 mg or less (EPA 2002c). Using newly hatched fry weighing about 1/1000$^{th}$ of the fish used by Pickering and Gast (1972) in the 1960s, cadmium $LC_{50}$s for fathead minnows at similar hardnesses tend to be around 50 µg/L with no obvious bias for test exposure. Similar results have been reported with brook trout. One each flow-through and static acute tests with brook trout were located, both conducted in waters of similar hardness (41 to 47 mg/L). The $LC_{50}$ of the static test which used fry was < 1.5 µg/L whereas the $LC_{50}$ of the flow-through test using yearlings was > 5,000 µg/L (Carroll *et al*. 1979; Holcombe *et al*. 1983).

<u>Summary:</u> When all other factors are equal, it appears that renewal tests may indicate chemicals are somewhat less toxic (e.g., higher $LC_{50}$s), but there is no clear consensus whether this indicates that renewal tests are biased toward lower toxicity than is "accurate" or whether conventional flow-through tests are biased toward higher toxicity. Comparisons with data across studies suggest that factors such as the life stage of exposures, can dwarf the influence of flow-through or renewal methods for the acute toxicity of at least metals.

#### *2.4.1.8. The "Water-Effect Ratio" Provision*

The water-quality criteria for metals proposed in this action include a Water Effects Ratio (WER) in their equations. The purpose of WERs is to empirically account for characteristics other than hardness that might affect the bioavailability and thus toxicity of metals on a site-specific basis. Because the WERs are directly incorporated into the criteria equations, no separate action is needed to change the criteria values using a WER. Following EPA's (EPA 1992) precedent, the default WER value for the proposed criteria is 1.0 "*except where the Department assigns a different value*" (Idaho Department of Environmental Quality 2011, at 210.03.c.iii. ).

The concept of adjusting metals criteria to account for differences in their bioavailability in site-waters has long been a precept of water quality criteria (Carlson *et al*. 1984; EPA 1994; Bergman and Dorward-King 1997). The WER approach uses one or more standard-test species (usually *Ceriodaphnia* and/or fathead minnows) which are tested in tandem in dilution waters collected from the site of interest and in a standard reconstituted laboratory water. The results in the laboratory water are presumed to represent the types of waters used in tests used in EPA criteria documents. The WER is the ratio of the test $LC_{50}$ in site water divided by the $LC_{50}$ in laboratory water; the ratio is then multiplied by the aquatic life criteria to obtain a WER-adjusted site-specific criteria. The approach has probably been most used with copper because of the profound effect of DOC to ameliorate toxicity, which is not correlated with hardness.

The main problem with the concept and approach is trying to define a single "typical" laboratory dilution water that reflects that used in criteria documents. Testing laboratories may generate valid results using all sorts of different dilution waters including dechlorinated tap water, natural groundwaters (well waters), natural surface waters such as Lake Superior or Lake Erie, and reconstituted waters made from deionized water with added salts. The widely used "Interim Guidance on Determination and Use of Water-effect Ratios for Metals" (Stephan *et al*. 1994b) specified using recipes from EPA or ASTM for making standardized water that results in a water hardness with unusually low calcium relative to magnesium concentrations compared to that of most natural waters ("hardness" is the sum of equivalent concentrations of calcium (Ca) and magnesium (Mg) and is discussed more in Section 2.4.2, "The Influence of Hardness on Metals Toxicity"). This has the effect of making metals in the reconstituted laboratory waters made by standard recipe more toxic than would be expected in waters with more natural proportions of calcium and magnesium. This is because at least for fish and some invertebrates and copper, calcium reduces toxicity somewhat but magnesium affords little or no protection (Welsh *et al*. 2000a; Naddy *et al*. 2002; Borgmann *et al*. 2005b).

The effect of this issue is that unrepresentative lab waters can generate low $EC_{50}$ values which when used as a denominator with higher $EC_{50}$s from site waters can produce extremely high-biased values. For instance, in WER testing on the Boise River, Idaho, a stream receiving treated municipal wastewater effluent, testing with *Ceriodaphnia* and copper resulted in mean site:lab WER of 18.4, which when multiplied by the copper CMC at a hardness of 40 mg/L would result in a WER adjusted CMC of 132 µg/L. Yet the *Ceriodaphnia* $EC_{50}$s in that same site water ranged from 18.6 to 60 µg/L (CH2M Hill 2002). Thus, the published WER procedure would generate a site-specific acute copper criterion that was three to seven times higher than concentration that killed 50% of a sensitive species in that same site water. Such a grossly unprotective site-specific criteria was argued for on the grounds that it was procedurally in accordance with the Idaho metals criteria under consultation, because it follows from the WER equation and definition in the NTR and derivative Idaho criteria. Because it arguably followed EPA's 1994 Interim Guidelines for developing Water Effect Ratios (Stephan *et al*. 1994b), whatever the outcome was, was therefore procedurally acceptable.

Both EPA and IDEQ have made steps to reduce the bias that could be introduced by low $EC_{50}$ values in laboratory waters compared with site waters. The EPA (2001a) effectively eliminated the issue by setting the WER as the lesser of the site water $EC_{50}$/ lab water $EC_{50}$ ratios or the ratio of site water $EC_{50}$ divided by the SMAV from an updated criteria dataset. When this latter calculation was applied to the Boise River dataset, it produced an average copper WER of 2.6 instead of 18.4 and produced a site-specific acute copper criterion of 18.5 µg/L for a hardness of 40 mg/L (CH2M Hill 2002). Given the *Ceriodaphnia* $EC_{50}$s of 18.6 to 60 µg/L in site water, this approach may not fully protect species as sensitive as *Ceriodaphnia* but it's an improvement. The IDEQ (2007a) regulations at subsection 210.03.c.iii specify that calcium and magnesium ratios should be similar to those in EPA's criteria laboratory waters or the water body for which WERs are to be applied. However, such an approach was used in the Boise River project and exorbitantly high WERs still resulted so it is not clear that the WER approach can be corrected in this way. Further, IDEQ's implementation procedures for NPDES permits call specifically for the use of EPA's 1994 interim procedures (IDEQ 2007a, at subsection 210.04) although IDEQ has the discretion to use "other scientifically defensible methods" as they see fit.

Other approaches by EPA that might be used as an interim, operational substitute include establishing criteria on a more mechanistic basis that can directly account for the factors that affect toxicity. One example is the biotic ligand model (BLM) which is supposed to capture the major interactions between metals concentrations, competition, and complexation that control bioavailability and thus toxicity (Di Toro *et al*. 2001; Niyogi and Wood 2004). For copper, BLM was used as the basis of EPA's (2007a) updated aquatic life criterion, which for copper at least, should negate much of the need for empirical WER testing. The predictiveness of the copper BLM over a wide range of environmental conditions makes the BLM a more versatile and effective tool for deriving site-specific water quality criteria compared to the WER method (EPA 2000c; Di Toro *et al*. 2001).

This provision has rarely been used in Idaho, but NMFS is recommending a term and condition to help reduce future risk if WERs are developed in critical habitat for listed salmon and steelhead.

<u>Summary:</u> While seldom used to date, the WER is a fundamental part of the formula-based water quality criteria for metals. In guidance and practice, the manner in which WERs are developed has a substantial risk of undermining the protectiveness of criteria. Procedures that are consistent with the action evaluated in this opinion could result in criteria concentrations that were higher than concentrations that were acutely toxic to sensitive organisms when tested in the same site water. Two alternate procedures could achieve the intent of the WER provision (to adjust criteria based on site-specific conditions). First, the WER could be calculated by using the lower ratio from either (a) the site water $EC_{50}$/ lab water $EC_{50}$ ratios or (b) the ratio of site water $EC_{50}$ divided by the species mean acute value (SMAV) for that test organism (e.g., Ceriodaphnia dubia, fathead minnow, or rainbow trout) from a criterion dataset as described by EPA (2001a). Second, with copper the EPA (2007) BLM-based criteria is intended to adjust for site-specific water quality differences (EPA 2007a; DiToro *et al.* 2001).

### *2.4.1.9. Issue of Basing Criteria on Dissolved or Total-Recoverable Metals*

One difference between the proposed action and the NTR as first published by EPA (1992) is that the proposed metals criteria are defined on the basis of "dissolved" metals rather than for "total recoverable" metals. "Dissolved" metals are those that pass through a 0.45 µm filter, and “total recoverable” metals are determined from unfiltered samples, and thus consist of both dissolved and particulate or colloidal phases. Metals sorbed to particulates are subject to gravity and will eventually settle from undisturbed water whereas dissolved metals are truly in solution and will not settle from gravity.

This criteria change was based on a 1993 EPA policy statement that “*it is now the policy of the Office of Water that the use of dissolved metal to set and measure compliance with WQSis the recommended approach, because dissolved metal more closely approximates the bioavailable fraction of metal in the water column than does total recoverable metal. This conclusion regarding metals bioavailability is supported by a majority of the scientific community within and outside the Agency. One reason is that a primary mechanism for water column toxicity is adsorption at the gill surface which requires metals to be in the dissolved form.*” (Prothro 1993).

To implement Prothro’s (1993) policy change, metals criteria had to be recalculated on a dissolved basis. Because the tests in the acute and chronic datasets used to derive metals criteria were mostly reported total recoverable rather than dissolved metals, in order express metals criteria on a dissolved metals basis, a conversion was needed. To do so, Stephan (1995) evaluated what data were available on the proportions of dissolved versus total recoverable metals in different laboratories that contributed data used in the EPA metals criteria. The resulting conversion factors ranged from 0.32 with chromium (III) to 0.99 with chronic zinc. With lead, because its solubility usually decreases as hardness increases, the conversion factor for lead varies with hardness, ranging from 1 at hardness 25 mg/L to 0.69 at hardness 200 mg/L. For most metals, the conversion factors were close to 1 indicating that for the laboratory conditions under which the toxicity tests in the datasets were conducted, almost all metals were present in dissolved form (Stephan 1995)

Because no supporting documentation was given by Prothro (1993) in support of their conclusions, they are hard to evaluate. There is theoretical support for the assumption that metals need to be in dissolved form to adsorb to the gill surface (Wood *et al*. 1997), and it does seem logical to assume that metals bound to particulates would be less toxic. However, no compelling evidence was found that particulate bound metals can be assumed to be non-toxic. Only two studies were located that examined the toxicity of particulate metals in controlled experimental studies. Both found toxicity associated with particulate bound copper (Brown *et al*. 1974; Erickson *et al*. 1996).

Erickson *et al*. (1996) estimated that the adsorbed copper has a relative toxicity of almost half that of dissolved copper, and noted that the assumption that toxicity can be simply related to dissolved copper was questionable, and a contribution of adsorbed copper to toxicity cannot be generally dismissed (Erickson *et al*. 1996). One possible reason for the observed toxicity from particulate-bound copper is that adsorbed metals could become desorbed, becoming more bioavailable, as the pH of water moving across fish gills decreases. If the pH of water where a fish is living is 6 or greater, then the pH will be lowered as water crosses the gill (Playle and Wood 1989). Most ambient waters in the Snake River basin action area have pH greater than 6.

A further manner in which particulate bound metals could become biologically active is through sediment or food exposure. For instance, in Panther Creek, a tributary to the Salmon River, Idaho, total copper concentrations were measured at greater than twice that of dissolved concentrations (Maest *et al*. 1995). Copper was also greatly elevated in biofilms (algae and detritus) and sediment, and correlations between copper concentrations in benthic invertebrates and biofilms were stronger than were correlations between invertebrates and water or sediment (Beltman *et al*. 1999). Copper sorbed to sediments was also bioavailable and toxic to benthic invertebrates when exposed to Panther Creek sediments after the sediments were transferred to clean overlying water (Mebane 2002a). In this stream at the time of those studies, dissolved copper consistently exceeded dissolved criteria values, so these studies do not directly help with the question of whether streams with low contamination that largely comply with dissolved criteria could result in sediment contamination at hazardous concentrations. Others have reported toxicity from metals contaminated freshwater sediments even when overlying waters mostly are at dissolved criteria (Canfield *et al*. 1994; Besser *et al*. 2008).

Attempting to define, evaluate, and manage risks associated with contaminated sediments by basing criteria on total recoverable metals would likely be so indirect as to be ineffective. However, in the absence of such efforts the assumption that metals sorbed to particles are in effect biologically inert and can safely be ignored is questionable. The effect of this stance is to give up some conservatism in aquatic life criteria for metals.

<u>Summary:</u> The component of the action to define metals criteria as applying only to the dissolved fraction of metals rests on the rationale that metal particulates are less toxic than dissolved metals. Criteria are adjusted from total to dissolved metals fraction through conversion factors. The total to dissolved conversion factors for metals criteria were set in a generally conservative manner and are close to 1 for most metals. While the conversion factors per se are not a conservation problem, the concept of basing criteria solely on the dissolved fraction may not always be protective. While we concur that for divalent metals (e.g., cadmium, copper, lead, nickel, zinc), the particulate fraction is less toxic, the particulate fraction is not necessarily non-toxic. Conceptually, the particulate fractions of metals and inorganics could contribute to foodweb exposure pathways from sediments or biofilms to macroinvertebrates to fish. This is of particular concern for substances with primarily dietary routes of exposure (e.g., arsenic, mercury, and selenium).

*2.4.1.10. Mixture Toxicity: criteria were developed as if exposures to chemicals occur one at a time, but chemicals always occur as mixtures in effluents and ambient waters*

In point or nonpoint pollution, chemicals occur together in mixtures, but criteria for those chemicals are developed in isolation, without regard to additive toxicity or other chemical or biological interactions (Table 2.4.1.1). Whether the toxicity of chemicals in mixtures is likely greater or less than that expected of the same concentrations of the same chemicals singly is a complex and difficult problem. While long recognized, the “mixture toxicity” problem is far from being resolved. Even the terminology for describing mixture toxicity is dense and has been inconsistently used (e.g., Sprague 1970; Marking 1985; Borgert 2004; Vijver *et al*. 2010). One scheme for describing the toxicity of chemicals in mixtures is whether the substances show additive, less than additive, or more than additive toxicity. The latter terms are roughly similar to the terms “antagonism” and “synergism” that are commonly, but inconsistently used in the technical literature.

For both metals and organic contaminants that have similar mechanisms of toxicity (e.g., different metals, different chlorinated phenols), assuming chemical mixtures to have additive toxicity has been considered a reasonable and usually protective (Norwood *et al*. 2003; Meador 2006). This conclusion is in conflict with the way effluent limits are calculated for discharge of toxic chemicals into receiving water. Each projected effluent chemical concentration occurring during design flow is divided by its respective criterion, along with adjustments for variability and mixing zone allowances (EPA 1991). Thus, each substance would be allowed to reach one “concentration unit” and any given discharge or cleanup scenario would likely have several concentration units allowed, which is sometime referred to as cumulative criterion units.

Experimental approaches in the literature usually report “toxic units” (TUs) based on observed toxicity in single substance tests, rather than criterion units. In this “concentration addition” scheme, toxicity of different chemicals is additive if the concentrations and responses can be summed on the basis of “TUs.” For instance, assume for simplicity that cadmium is more toxic than copper to a species, with the an $EC_{50}$ of 4 µg/L for cadmium, and an $EC_{50}$ of 8 µg/L for copper. We will also call each single metal $EC_{50}$ a TU. The toxicity of mixtures could be estimated as follows:

4 µg/L Cd + 0 µg/L Cu = $\frac{4\ \mu\text{g/L}}{4\mu\text{g/L/TU}}+\frac{0\ \mu\text{g/L}}{8\mu\text{g/L/TU}}$ =1 TU, (obviously, for a single substance), *or*

2 µg/L Cd + 4 µg/L Cu = $\frac{2\mu\text{g/L}}{4\mu\text{g/L/TU}}+\frac{4\mu\text{g/L}}{8\mu\text{g/L/TU}}$ =0.5+0.5=1 TU.

Using this approach, some studies have shown significant additive toxicity. For instance, Spehar and Fiandt (1986) exposed rainbow trout and *Ceriodaphnia dubia* simultaneously to a mixture of five metals and arsenic, each at their acute CMC, which by definition were intended to be protective. There were no survivors. In chronic tests, adverse effects were observed at mixture concentrations of one-half to one-third the approximate chronic toxicity threshold of fathead minnows and daphnids, respectively, suggesting that components of mixtures at or below no effect concentrations may contribute significantly to the toxicity of a mixture on a chronic basis (Spehar and Fiandt 1986).

A common outcome in metals mixture testing has been that metals combinations have been less toxic than the sum of their single-metal toxicities, i.e., show less than additive toxicity or are antagonistic (Finlayson and Verrue 1982; Hansen *et al*. 2002c; Norwood *et al*. 2003; Vijver *et al*. 2011; Mebane *et al*. 2012). The other possibility, more than additive toxicity (also called synergistic effects) are rare with metals although it has been shown with pesticides (Norwood *et al*. 2003; Laetz *et al*. 2009).

<u>Summary:</u> The water criteria evaluated in this opinion were all developed as if only one chemical was present at a time. However, in the real world chemicals always occur in mixtures. As result, criteria and discharge permits based upon them may afford less protection than intended. Measures to address this potential underprotection need to be included in discharge permits.

***The efficacy of whole-effluent toxicity tests to evaluate mixture toxicity.*** The EPA's approach to the mixture toxicity problem in effluents, including effects of substances without numeric criteria or unmeasured substances, has been to recommend an integrated approach to toxics control (EPA 1991, 1994). The EPA has long recognized that numerical water quality criteria are an incomplete approach to protecting or restoring the integrity of water. A major part of EPA's strategy for measuring and controlling such potential issues has been through the concept of an integrated approach to toxics control, where meeting numerical criteria is but one of three elements. The other two elements are: (1) The concept of regulating whole effluents through whole- effluent toxicity (WET) testing; and (2) through biological monitoring of ambient waters that receive point or nonpoint discharges (EPA 1991, 1994). Because of assumptions that: (1) Chemicals will inevitably occur in ambient waters in mixtures rather than occurring chemical by chemical in the fashion that criteria are developed; and (2) it's not possible to know all the potential contaminants of concern in effluents and receiving waters, let alone measure them, it is not feasible to predict effects by chemical concentrations alone. Thus, the EPA developed procedures for testing the whole-toxicity of effluents and receiving waters, including procedures for identifying and reducing toxicity (e.g., Mount and Norberg-King 1983; Norberg-King 1989; Mount and Hockett 2000). In practice, some consideration of the potential for aggregate toxicity through WET testing is made by EPA for major permits that they administer in Idaho.

Test procedures for WET testing are intended to be practical for permitted dischargers or test laboratories to carry out as a routine monitoring tool. Thus, to simplify testing, improve test repeatability, and to facilitate interpretation of test results by dischargers and permit compliance staff, the EPA has limited WET testing requirements to select standard test species and test conditions (EPA 2002a, 2002c). Most commonly, EPA has required monitoring for chronic WET through testing of two species, fathead minnows and the cladoceran ("water flea") *Ceriodaphnia dubia*. Both tests are administered as 7-day tests. *Ceriodaphnia* have a short life-cycle, so even though the test is only 7 days, it spans three broods, and so can be considered a "true" chronic test that includes all or most of an organism's life cycle. In contrast, the 7-day fathead minnow "chronic" test only spans about 1% of the 2-year or so life span of a fathead minnow and is more properly called a short-term method for predicting chronic toxicity.

The rationale and performance of WET testing for predicting or protecting against impairment have been complicated and controversial and have been debated in conferences and articles, among them a special issue of the journal *Environmental Toxicology and Chemistry* (v19, 1, January 2000) and an entire book (Grothe *et al*. 1996). Issues with WET testing include whether the tests are sensitive, and whether any single species toxicity test can meaningfully predict in stream effects or lack thereof. For instance, Clements and Kiffney (1996) noted that *Ceriodaphnia* effluent tests were correlated with effects detected from stream microcosms or field surveys, but the latter two tended to be more sensitive than the Ceriodaphnia effluent tests. Conversely, Diamond and Daley (2000) and de Vlaming *et al*. (2000) found that the chronic WET methods were useful for predicting ambient impairment.

The best comparison of the sensitivity of WET tests in relation to listed salmon, steelhead and their prey is probably a series of tests conducted at the same laboratory with the same dilution water with copper and different species (Table 2.4.1.2). Neither the *Ceriodaphnia* or 7-day fathead minnow test were as sensitive as 30- or 6-day chronic tests with rainbow trout; the

*Ceriodaphnia* were about twice as resistant as the rainbow trout, and the 7-day fathead minnow test was almost five times as resistant as the longer rainbow trout test. Dwyer *et al*. (2005a) also found that the *Ceriodaphnia* test was considerably more sensitive than the 7-day fathead test to a complex "effluent" comprised of a mixture of pesticides, chlorinated organic compounds, ammonia, and metals. The low sensitivity of the 7-day fathead minnow test might be because the species is inherently less sensitive to some substances than salmonids or because a 7-day exposure is too short to be an accurate "short-term" chronic measurement (Suter 1990; Lazorchak and Smith 2007).

Comparisons with other metals were less reliable because they required comparing tests across studies and regression-based hardness normalizations (Table 2.4.1.3). Focusing on the more sensitive *Ceriodaphnia* test, sensitivity comparisons were made for four metals with rainbow trout (treating rainbow trout as a surrogate for listed salmon and steelhead). The comparisons used the most convenient, readily available statistics that were comparable across tests, even though those statistics do not reflect protective concentrations in of themselves (e.g. EC20, MATC, see "*Implications of the use of the "chronic value" statistic*"). A sensitivity ratio of 1.0 or less suggests that *Ceriodaphnia* are at least as sensitive as the salmonid surrogate and that the WET testing should be protective for aggregate, direct toxicity of waste mixtures in effluents (Table 2.4.1.2). The comparisons suggest that for cadmium and zinc the *Ceriodaphnia* test would be almost as sensitive or more sensitive as the average rainbow trout test; however, for copper and lead. Chinook salmon or rainbow trout could be much more sensitive than the *Ceriodaphnia*.

A further consideration beyond these simple comparisons of whether reduced survival or reproduction in Ceriodaphnia test results occurred at higher or lower concentrations than mortality to listed salmonids, is whether WET tests such as Ceriodaphnia can be used as a proxy indicator of sublethal effects of chemicals to salmonids, such as olfactory impairment. The limited information available suggests that they can be used in this way, at least for copper. Toxicity of copper to aquatic organisms can often be predicted using a "biotic ligand model" or BLM. The BLM uses geochemical speciation modeling to model bioaccumulation of copper on the organisms' gills or their other biological tissues in contact with water (i.e., their "biotic ligands"), and then uses an empirical species-specific toxicity adjustment to predict effects (Appendix C). This empirical species-specific toxicity adjustment was initially done to predict killing organisms with different sensitivities following short-term exposures (EPA 2007a). However, it has been successfully expanded to predict olfactory impairment (or lack thereof) in coho salmon or behavioral avoidance in rainbow trout or Chinook salmon (Appendix C; Meyer and Adams 2010). These analyses suggest that on the average, adverse effects predicted for Ceriodaphnia dubia would occur at lower copper concentrations than would olfactory impairment or avoidance behavior in rainbow trout, based upon lower modeled critical accumulation values for Ceriodaphnia dubia (0.06 vs. 0.19 nmol/g wet weight (Appendix C; Meyer and Adams 2010).

In contrast, the Ceriodaphnia WET test has been shown to be able to predict adverse effects in benthic macroinvertebrate communities in streams, but that the Ceriodaphnia WET test appeared less sensitive than the more complex stream communities (Clements and Kiffney 1996). This suggests that with a sensitivity adjustment, the Ceriodaphnia WET test could be used to predict whether effluents were likely to adversely modify critical habitats by reducing the benthic macroinvertebrate forage base for rearing salmonids.

**Table 2.4.1.2. Relative sensitivity of standard 7-day WET tests with *Ceriodaphnia* and fathead minnows to rainbow trout with copper under directly comparable test conditions (ASTM moderately-hard water, hardness 170 mg/L).**

| Organism | Test duration | EC25 for the most sensitive endpoints (µg/L) | Source |
|---|---|---|---|
| Rainbow trout | 30-days (starting with fry) | 21 | (Besser *et al*. 2005b) |
| Rainbow trout | 60-days (starting with eggs) | 25 | (Besser *et al*. 2005b) |
| Fathead minnow | 30-days | 12-24 (range of 3 replicate tests) | (Besser *et al*. 2005b) |
| Fathead minnow | 7-days | 103 | (Dwyer *et al*. 2005a) |
| Ceriodaphnia dubia | 7-days | 51 | (Dwyer *et al*. 2005a) |

**Table 2.4.1.3. Relative sensitivity of the standard WET *Ceriodaphnia dubia* 7-day test in relation to a surrogate salmonid for listed salmon and steelhead (rainbow trout except where noted), pooled from data compilations**

| Metal | *Ceriodaphnia dubia* SMCV (µg/L) | Surrogate salmonid SMCV (µg/L) | Sensitivity Ratio (*C.dubia* ÷ Salmonid) | Notes (source) |
|---|---|---|---|---|
| Cd | 2.04 | 1.7 | 1.2 | MATC, (Mebane 2006) |
| Cu | 19 | 23.8 | 0.8 | EC20s, (EPA 2007a); |
| Cu | 19 | 5.9 | 3.2 | Chinook salmon biomass EC20 (EPA 2007a); |
| Pb | 46 | 28 | 1.6 | Rainbow trout, geometric mean of 5 tests, normalized to hardness 50; (Mebane *et al*. 2008); C. dubia is from a single test at hardness 52 mg/L, pH 7.56 (Mager *et al*. 2011a) (note) |
| Zn | 33 | 113 | 0.3 | NOECs; (Van Sprang *et al*. 2004) |

Note: Much new data with C. dubia and chronic toxicity of Pb has been recently generated (Parametrix 2010; Mager *et al*. 2011a). While this was too much to synthesize and estimate whether C. dubia are usually more or less than salmonids, recent toxicity values with C. dubia indicate the sensitivities overlap those of rainbow trout and the species may be much more sensitive than previously indicated (Jop *et al.* 1995; Mebane *et al*. 2008)

Summary: Our review generally supports EPA's concept of assessing mixture toxicity of criteria substances under consultation through WET testing and instream bioassessment. However, the more sensitive of the two commonly used chronic WET tests, the three-brood *Ceriodaphnia dubia* test was sometimes less sensitive than chronic tests with salmonids. The 7-day fathead minnow test was consistently less sensitive than chronic salmonid tests in the data reviewed. This suggests that to be protective of listed salmonids, the assessment triggers for the

*Ceriodaphnia* test might have to be scaled to account for sensitivity and or differences in tolerable risk for a threatened species versus a zooplankton.

In much of EPA's (2000a) biological evaluation of the action, and elsewhere in the present opinion, the effects of criteria provisions or substances are evaluated linearly, one-by-one. Despite this simplification, in the environment chemicals in water never occur in isolation, but rather always occur as mixtures. The toxicity of mixtures is probably dependent upon many factors, such as which chemicals are most abundant, their concentration ratios, differing factors affecting bioavailability, and organism differences. Because of this complexity, accurate predictions of the combined effects of chemicals in mixtures appear to be beyond the present state of the ecotoxicology practice.

Here, despite the complexities and many exceptions, we make a general assumption that, at their criteria concentrations, the effects of chemicals in mixtures would likely be more severe than would be the same concentration of the mixture components singly.

Addressing mixture toxicity through the use of WET testing and instream bioassessment are practical and reasonable approaches for addressing the expected increased toxicity of a given concentration of a chemical in the presence of other chemicals. However, the assessment triggers on WET tests may not be sensitive enough to protect listed salmonids with reasonable certainty, and biomonitoring has not always been well defined. Measures for implementing biomonitoring are provided in Section 2.9 and Appendix E

*2.4.1.11. Frequency, Duration and Magnitude of Allowable Criteria Concentration Exposure Exceedences.*

For simplicity, much of the discussion of the water quality criteria that are the subject of this consultation treats the criteria as though they were defined solely as a concentration in water. However, the action actually defines aquatic life criteria in three parts: a concentration(s), a duration of exposure, and an allowable exceedence frequency. All of EPA's criteria recommendations define criteria using a statement similar to the following:

> "The procedures described in the 'Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and their uses' indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the 4-day average concentrations of [the chemical] do not exceed [the 'chronic' criterion continuous concentration] more than once every 3 years on the average and if the 1-hour average concentration does not exceed [the 'acute' criterion maximum concentration] more than once every 3 years on the average."

The 4-day and 1-hour duration and averaging periods for criteria were based upon judgments by EPA authors that included considerations of the relative toxicity of chemicals in fluctuating or constant exposures. The EPA's (1985) Guidelines considered an averaging period of 1 hour most appropriate to use with the criterion maximum concentration or (CMC or "acute" criterion) because high concentrations of some materials could cause death in 1 to 3 hours. Also, even when organisms do not die within the first few hours, few toxicity tests attempt to monitor for latent mortality by transferring the test organism into clean water for observation after the chemical exposure period is over. Thus, it was not considered appropriate to allow concentrations above the CMC for more than 1 hour (Stephan *et al*. 1985). Recent criteria documents (e.g., EPA 2007a) have used an averaging period of 24 hours for their CMC, although no explanation could be found for the deviation from the 1985 Guidelines and thus, the issue of latent toxicity might not have been considered.

A review of more recent information supported EPA's judgments from the 1980s that if an averaging period is used with acute criteria for metals, it should be short. Some of the more relevant research relates the rapid accumulation of metals on the gill surfaces of fish to their later dying. When fish are exposed to metals such as cadmium, copper, or zinc, a relatively rapid increase in the amount of metal bound to the gill occurs above background levels. This rapid increase occurs during exposures on the order of minutes to hours, and these brief exposures have been sufficient to predict toxicity at 96 to 120 hours. The half saturation times for cadmium and copper to bind to the gills of rainbow trout may be on the order of 150 to 200 seconds (Reid and McDonald 1991). Several other studies have shown that exposures well under 24 hours are sufficient for accumulation to develop that is sufficient to cause later toxicity (Playle *et al*. 1992; Playle *et al*. 1993; Zia and McDonald 1994; Playle 1998; MacRae *et al*. 1999; Di Toro *et al*. 2001). Acute exposures of 24 hours might not result in immediate toxicity, but deaths could result over the next few days. Simple examination of the time-to-death in 48- or 96-hour exposures would not detect latent toxicity from early in the exposures. The few known studies that tested for latent toxicity following short-term exposures have demonstrated delayed mortality following exposures on the order of 3 to 6 hours (Marr *et al*. 1995a; Zhao and Newman 2004, 2005; Diamond *et al*. 2006; Meyer *et al*. 2007a). Observations or predictions of appreciable mortality resulting from metals exposures on the order of only 3 to 6 hours supports the earlier recommendations by Stephan and others (1985) that the appropriate averaging periods for the CMC is on the order of 1 hour.

The 4-day averaging period for chronic criteria was selected for use by EPA with the CCC for two reasons (Stephan *et al*. 1985). First, "chronic" responses with some substances and species may not really be due to long-term stress or accumulation, but rather the test was simply long enough that a briefly occurring sensitive stage of development was included in the exposure (e.g., Chapman 1978a; Barata and Baird 2000; De Schamphelaere and Janssen 2004; Grosell *et al*. 2006b; Mebane *et al*. 2008). Second, a much longer averaging period, such as 1 month would allow for substantial fluctuations above the CCC. Whether fluctuating concentrations would result in increased or decreased adverse effects from those expected in constant exposures seems to defy generalization. A comparison of the effects of the same average concentrations of copper on developing steelhead, *Oncorhynchus mykiss*, that were exposed either through constant or fluctuating concentrations found that steelhead were about twice as resistant to the constant exposures as they were to the fluctuating exposures (Seim *et al*. 1984). Similarly, *Daphnia magna* exposed to daily pulses of copper for 6 hours at close to their 48-hour $LC_{50}$ concentrations had more severe effects after 70 days than did comparisons that were exposed to constant copper concentrations that were similar to the average of the daily fluctuations (Ingersoll and Winner 1982). In contrast, cutthroat trout exposed instream to naturally fluctuating zinc concentrations survived better than fish tested under the same average, but constant zinc concentrations (Nimick *et al*. 2007; Balistrieri *et al*. 2012). Thus, literature reviewed either supports or at least do not contradict EPA's position on averaging periods.

The third component of criteria, EPA's once-per-3-years allowable exceedence policy was based on a review of case studies of recovery times of aquatic populations and communities from locally severe disturbances such as spills, fish eradication attempts, or habitat disturbances (Yount and Niemi 1990; Detenbeck *et al*. 1992). In most cases, once the cause of the disturbance was lifted, recovery of populations and communities occurred on a time frame of less than 3 years. The EPA has subsequently further evaluated the issue of allowable frequency of exceedences through extensive mathematical simulations of chemical exposures and population recovery. Unlike the case studies, these simulations addressed mostly less severe disturbances that were considered more likely to occur without violating criteria (Delos 2008). Unless the magnitude of disturbance was extreme or persistent, this 3-year period seemed reasonably supported or at least was not contradicted by the information we reviewed.

A more difficult evaluation is the exceedence magnitude, which is undefined and thus not limited by the letter of the criteria. Thus, by the definition, a once-per-3-year exceedence that has no defined limits to its magnitude, could be very large, and have large adverse effects on listed species. However, within the 4-day and 1-hour duration constraints of the criteria definitions, some estimates of the potential magnitude of exceedences that could occur without "tripping" the duration constraints can be calculated. This is because environmental data such as chemical concentrations in water are not unpredictable but can be described with statistical distributions, and statements of exceedence probabilities can be made. Commonly with water chemical data and other environmental data, the statistical distributions do not follow the common bell-curve or normal distribution, but have a skewed distribution with more low than high values. This pattern may be approximated with a log-normal statistical distribution (Blackwood 1992; Limpert *et al*. 2001; Helsel and Hirsch 2002; Delos 2008).

The following three hypothetical scenarios are intended to illustrate contaminant concentrations that could occur without violating the exceedence frequency and duration limitations of the proposed criteria (Figure 2.4.1.7). The scenarios use randomly generated values from a log-normal distribution with different variabilities and serial correlations. Serial correlation refers to the pattern in environmental data where values at time one are often highly correlated with values at time two and so on. For example, a hot day in summer is much more likely to be followed by another hot day than a bitterly cold day, a low chemical concentration during stable low flows on a day in September will most likely be followed by low chemical concentration the next day, a high chemical concentration in a stream during runoff on a day in April will more likely to be repeated by another high concentration, and so on (Helsel and Hirsch 2002; Delos 2008). Under Scenario 1, effects could be appreciable since the mean concentrations are close to the criteria, and organisms would have little relaxation of exposure for recovery. Under Scenario 2, effects to a population of sensitive organisms would presumably be slight, since the mean concentrations were well below the criterion, and the exceedence magnitude was slight followed by a recovery opportunity. Scenario 3 might be more likely in runoff of nonpoint pollutants from snowmelt or stormwater. In these scenarios, sensitive populations could experience effects ranging from appreciable reductions if the contaminant pulse hit during a sensitive part of their life history, to no effect if it hit during a resistant phase or if the listed species was less sensitive than the species that drove the criteria calculations.

An actual event that was very similar to Scenario 3 occurred when an upset at a large, industrial mining operation caused elevated cadmium concentrations in Thompson Creek, a tributary to the upper Salmon River in Idaho. In April 1999, a pulse of cadmium about 30X higher than background, 2.6 times higher the chronic criterion, and equal to the acute criterion was detected. The duration of exceedence was probably greater than a day and less than a week. By August 1999, when a biological survey was conducted, few if any adverse effects could be detected in the benthic community structure. Whether subtle differences between unaffected upstream survey sites were lingering effects of the disturbance or just differences in naturally patchy stream invertebrate communities was unclear. However, it does suggest that benthic communities in similar mountain streams would be either resilient to, or recover quickly from criteria exceedences of this magnitude (Mebane 2006, pp. 47,62).

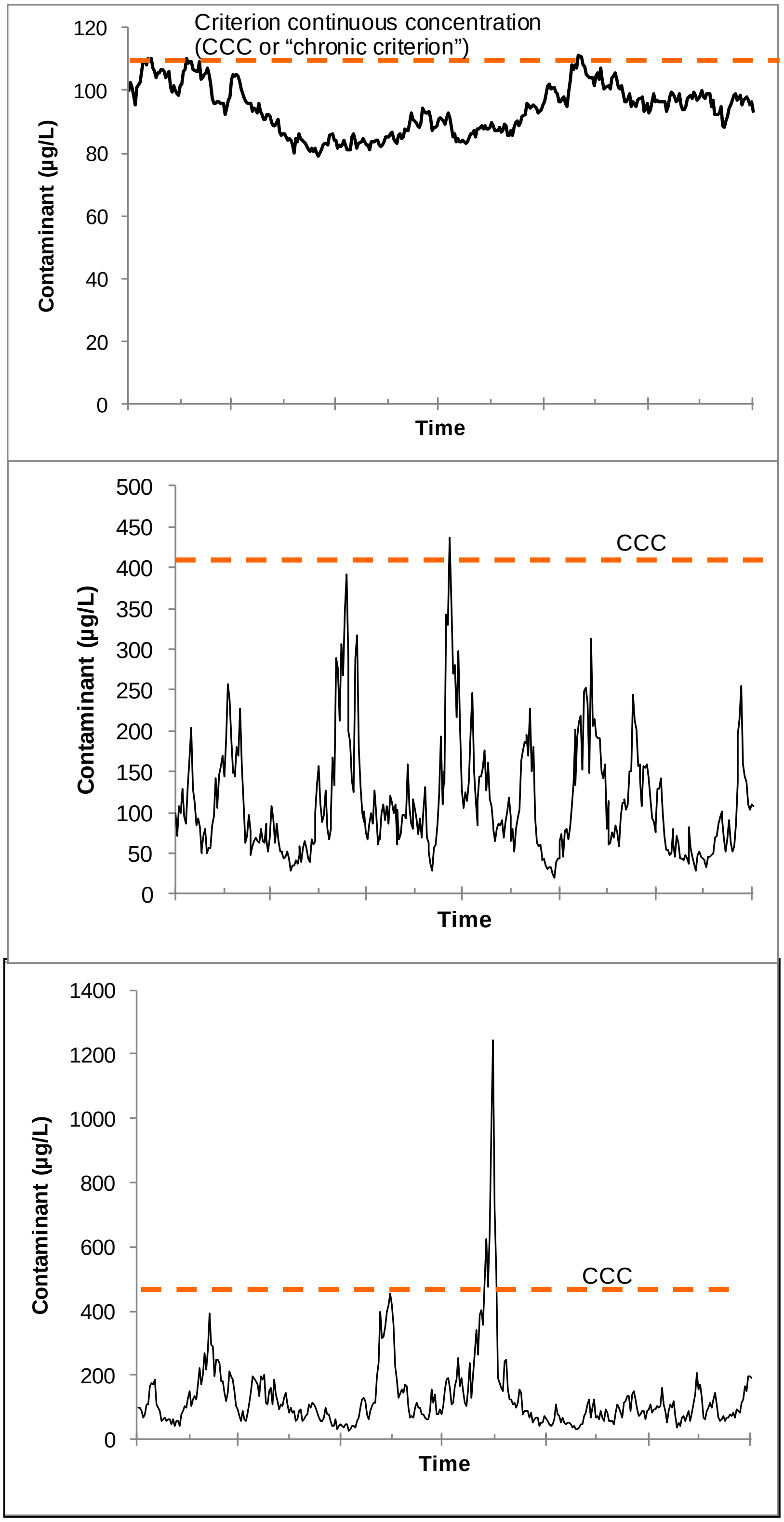
These hypothetical scenarios used a simplified, fixed criterion, whereas in actuality, some of EPA's criteria vary and may be positively correlated with the concentrations of metals in water. If the criteria accurately reflect risks from varying environmental conditions, and if ambient conditions co-vary with and are positively correlated with criteria, this will tend to lessen risks resulting from ambient increases in concentration. In cases where the criteria were positively correlated with the contaminants, such as in the following Section 2.4.4 example for Pine Creek with cadmium or the BLM-copper example for Panther Creek, the frequency and magnitude of exceedences is expected to be less than if the criteria and contaminant concentrations did not rise and fall together. This is because the contaminant and another water quality parameter that mitigates toxicity have common sources and rise and fall together, such as cadmium and calcium in Pine Creek where the source for both is probably weathering of gangue rock and spring snowmelt and runoff appears to dilute both.

In the Panther Creek example, copper and DOC tended to rise and fall together with snowmelt and runoff, similarly mitigating exceedence frequency and magnitude. This was the case in all examples examined. In the Panther Creek example, the hardness-based criterion is negatively correlated with copper concentrations, which gives the impression of risks of copper being exacerbated due to lower hardness corresponding with higher copper. However, this impression is probably misleading because copper risks indicated from the hardness-based criteria are often the opposite from risks indicated by BLM-based criteria, which is considered to more accurately represent the copper risks (Section 2.4.4; Appendix C).

While NMFS did not locate any plausible examples of negative correlations between contaminants and important factors modifying toxicity, it is likely that such scenarios do occur somewhere because if the event that releases the contaminant, such as a runoff pulse from a storm or snowmelt, caused a contaminant spike from washing accumulations into a stream and at the same time lowered the pH and hardness, then the magnitude of exceedences could be more severe. Such a circumstance could be plausible for metals such as cadmium, lead, or zinc in which hardness is a major modifier of toxicity.

Further, the actual possibility that an extreme exceedence would occur and be "allowed" under the exceedence policy seems unlikely. This is because in natural waters seasonal and hydrologic factors tend to cause concentrations to be serially correlated, that is low concentrations follow low concentrations and high concentrations follow high concentrations (Helsel and Hirsch 2002;

Delos 2008). Thus for an extreme exceedence to be allowable under the chronic criteria 4-day average concentration definition, it would also have to not exceed the 1-hour acute criteria definition. A very large exceedence of the sort illustrated in Figure 2.4.1.7, Scenario 3, would likely span across more than one, 1-hour averaging period for acute criteria and "violate" the one exceedence per 3-year recurrence interval term. While there are no regulatory limits on the upper concentration of an exceedence of the 1-hour acute criteria, the idea that a chemical concentration in a natural water could rapidly rise to acutely toxic concentrations and then drop back down to below criteria seems like a remote possibility. In urban watersheds with high proportions of impervious surface, runoff is flashier than in forested watersheds, and short-term pulse exposures could occur in those settings Booth *et al.* (2002). In the predominately forested areas of the action areas, such scenarios seem less likely.




Scenario 1: Contaminant concentrations have low variability, and while the CCC is only briefly exceeded, the average exposure concentration is only slightly lower than the criterion. Such a scenario might result from a stable effluent discharged into a flow regulated receiving water .

Scenario 2: Contaminant concentrations are more variable, and while the frequency and magnitude of criterion exceedences are similar to scenario 1, average concentrations are well below the CCC in this scenario. Such a scenario might result from nonpoint pollutants resulting from snowmelt or precipitation into an unregulated stream, such as stormwater from a mining operation.

Scenario 3: Contaminant concentrations have the same variability as scenario 2, but by chance a high magnitude criterion exceedence of 12X above the average concentrations occurred. Unless the acute criterion for this substance was at least 12X higher than the CCC, such an exceedence would not be allowable because the 1-hour acute criterion averaging period would also be exceeded.

**Figure 2.4.1.7. Three example allowable scenarios for criteria exceedence magnitudes**

Summary: The 1-hour and 4-day exceedence durations for acute and chronic criteria respectively are supported by the science as reasonable and adequately protective. Whether the allowable 1 in 3 years exceedence frequency is sufficiently protective was difficult to evaluate, in part because the magnitude of allowable exceedences is undefined. However, the likelihood that a runoff pulse could both rise and fall so high within an hour that it could cause acute effects without exceeding the acute criteria seems unlikely. This does remain an aspect of uncertainty regarding the protectiveness of criteria.

#### *2.4.1.12. Special Consideration for Evaluating the Effects of the Action on Critical Habitat*

Fundamentally, the analyses of water quality criteria for toxic substances included in this Opinion are most directly analyses of the "water quality" features of the PCE's of critical habitat. The WQS directly characterize and define the conditions and quality of surface waters that listed salmon and steelhead experience, either as incubating embryos in the interstices of spawning gravels, or as juveniles and adults in the water column. Analyzing whether the action would represent an "adverse modification" of water quality is at least conceptually more straightforward than whether these modifications would jeopardize the continued existence of listed species. This is because quantitative causal predictions relating habitat change to species population changes and long-term viability are uncertain. Many simplifying assumptions are required, including things like specifics of species life histories, other interacting physical and biological factors, the nature and magnitude of assumed exposures such as whether the exposures are joint or separate, continuous or intermittent, magnitude of exceedences, and so on. Quantitative models relating water quality changes to extinction risks may provide value in a relative sense for evaluating relative risks of different "what if" scenarios (e.g., McCarthy *et al*. 2004; Baldwin *et al*. 2009; Mebane and Arthaud 2010). However, except for cases of extreme-risk with very high extinction probabilities (perhaps for example, Spromberg and Scholz 2011), the absolute projections from quantitative models of habitat and population changes may be thought of as mathematical speculation. Further, all mathematical population models will project some extinction risk, and policy definitions or scientific consensus are elusive on how much habitat modification or extinction risk is too much under narrative Endangered Species Act definitions (DeMaster *et al*. 2004; McGowan and Ryan 2009; McGowan and Ryan 2010; Owen 2012).

The types of adverse effects reported in the scientific literature that we consider to directly or indirectly reduce survival or reproduction included such things as reductions in survival, growth, swimming performance, ability to detect or evade predators (e.g., chemoreception), ability to detect or capture prey, ability to detect and avoid harmful concentrations of chemicals, homing ability, disease resistance, certain fish health indicators that have been related to survival or growth such as gill or liver tissue damage, spawning success, or fecundity. For evaluating what severity of effects to invertebrates would be considered an appreciable enough reduction in forage to reduce the conservation value of habitats for freshwater rearing, if a general reduction in diversity or abundance of invertebrates was expected at criteria conditions, we would consider that to be "appreciable." Because salmonids are opportunistic feeders, effects to a single invertebrate species for example, might not be important. This assumption must be tempered by the availability of data. Often data were available for very few invertebrate species, so if few data were available, but they indicated adverse effects, that could be considered a diminishment in water quality and habitat value.

Examples of types of effects that we do not consider to be sufficiently severe to represent an "appreciable diminishment" of water quality and thus the value of critical habitat include simple bioaccumulation of chemical in tissues, enzyme changes, gene expression or transcription, molecular changes, or other markers of exposure that may be considered sub-organismal, without known correlation to other changes such as reduced growth or survival. A human-health analogy of the latter types of effects would be those considered asymptomatic or sub-clinical, that is, not rising to the level that caused negative symptoms.

Because multiple criteria (acute and chronic aquatic life criteria, human health based water quality criteria) for the same substances would apply to any given area of critical habitat, we compared adverse effects indicated from short-term experiments of 4 days or less duration to the acute criteria that are intended to protect against short-term effects, and compared adverse effects shown in longer-term studies to the proposed chronic criteria. Human health-based criteria were only evaluated if they were both more stringent than chronic criteria and if the chronic criteria failed to be fully protective. In Idaho, water quality criteria for the protection of "fishable" beneficial uses based on avoiding health risks from consuming tainted fish, were clearly intended to be some sort of backstop to the aquatic life criteria because the human-health based criteria explicitly apply to waters designated for "cold water biota" and "salmonid spawning" aquatic life uses (Table 1.3.1).

For most of the substances, there were at least some conflicts in the scientific literature where for the same species and similar types of experiments, one study might find no ill effects from a given concentration and another might find severe effects. Thus, we considered the overall strength of the evidence for or against the protectiveness of criteria.

***Sediments.*** If sufficiently elevated, toxic pollutants in ambient water may adversely modify critical habitat through contamination of stream and lake bed sediments. In general, sediment contamination by toxic pollutants adversely modifies critical habitat because the particulate forms of toxicants are either immediately bioavailable through re-suspension, or are a delayed source of toxicity through bioaccumulation or when water quality conditions favor dissolution at a later date. Specifically, contaminated sediments are expected to influence: (1) The intra-gravel life stages of listed salmon and steelhead; (2) the food source of listed salmonids; and (3) the fish through direct ingestion or deposition on the gill surfaces of particulate forms of toxicants. However, other than for mercury, it is not clear whether moderately-elevated concentrations in water (i.e, up to criteria concentrations), would be likely to result in concentrations in bed sediments that are elevated to a degree that would pose appreciable risks to listed salmonids or their prey.

The proposed criteria do not explicitly account for exposure to contaminants via sediments. NMFS recognizes that considerable technical and practical problems exist in defining water quality criteria on a sediment basis, and that this is presently the subject of considerable research and debate. Nevertheless, most organic and metal contaminants adsorb to organic particulates and settle out in sediments. Thus, at sites where there have been past discharges, or where there are continuing discharges of contaminants into the water column, sediments form a long-term repository and a continuing source of exposure that must be addressed if the water quality component of critical habitat is to be protected. Further, although these substances may not readily be transferred into the water column, they may still be available to salmonids through food chain transfer from their benthic prey, or through ingestion of sediment while feeding, as has been described in preceding sections. Not having water quality criteria that consider uptake through direct ingestion or food chain transfer leaves potential routes for harm to listed species that the proposed criteria do not directly address.

***Salmonid Prey Items.*** An important type of indirect adverse effect of toxic substances to listed salmon and steelhead is the potential reduction of their invertebrate prey base. This is because for many substances, invertebrates tend to be among the most sensitive taxonomic groups and because juvenile salmonids depend on aquatic invertebrates during freshwater rearing. Known effects of specific substances to invertebrates are discussed specifically in those sections; however, some general considerations and assumptions applicable to all substances follow.

First, in instances of a pulse of chemical disturbance such as insecticide spraying of forests or crops, effects to aquatic invertebrate communities ranging from increased drift to catastrophic reductions can result (Ide 1957; Gibson and Chapman 1972; Wallace and Hynes 1975; Wallace *et al*. 1986). In such cases, even if the fish are not directly harmed by the chemical, the temporary reduction in food from the reduction in invertebrate prey can lead to reduced growth, and reduced growth in juvenile salmonids can in turn be extrapolated to reduced survival and increased risk of population extinction (Kingsbury and Kreutzweiser 1987; Davies and Cooke 1993; Baldwin *et al*. 2009; Mebane and Arthaud 2010). However, such severe effects would not be expected in waters with chemical concentrations similar to the maximum allowed by aquatic life criteria. The criteria are intended to only allow adverse effects to a small minority of the species in aquatic communities, and for most substances, the analyses of individual criteria that follow in Sections 2.4 are consistent with this expectation (*although copper has exceptions*).

This begs the question, whether the loss of a minority of invertebrate prey species could lead to a reduction in forage for juvenile salmonids that in turn could affect growth and survival? To address that question, NMFS reviewed a large number of studies on food habits of salmonids in streams, lakes, and reservoirs. [5] The body of evidence indicates that juvenile salmonids are opportunistic predators on invertebrates, and so long as suitable, invertebrate prey items are abundant and diverse, the loss of a few "menu items" probably would not result in obvious, adverse effects. Suitable invertebrate prey items for juvenile salmonids are those that are small enough to be readily captured and swallowed, and vulnerable to capture (i.e., not taxa that are burrowers or are armored (Keeley and Grant 2001; Suttle *et al*. 2004; Quinn 2005)). Some otherwise apparently suitable taxa such as water mites (Hydracarina) appear to taste bad to salmonids and others, like copepods, are too small to provide much energy for the effort it takes to eat them (Keeley and Grant 1997). Freshwater aquatic invertebrates have such great diversity (over 1200 species in Idaho alone, Mebane 2006), that they have some ecological overlap and redundancy, so that the loss of a few species would be unlikely to disrupt the stream or lake ecology greatly (Covich *et al*. 1999). However, this apparent ecological redundancy is compromised in streams that have already lost substantial diversity to pollution. For instance, in

[5] Over 90 were reviewed, although only a handful are listed here.

copper-polluted Panther Creek, Idaho, during springtime in the early 1990s, the total count of invertebrates was just as abundant as in reference sites, although the abundance was composed of fewer species. Yet in October, the abundance in the polluted reaches was less than 10% of reference (Mebane 1994). With reduced diversity, after a single species hatches and leaves the streams, a large drop in remaining abundance can occur. Because all species don't hatch at the same time, with greater diversity, the swings in abundance would be less severe. Further, in copper-polluted tributaries to Panther Creek, the usually abundant mayflies were scarce and had been replaced by unpalatable mites and low-calorie copepods (Todd 2008).

One consistent theme in the literature on the feeding of salmonids in streams is the persistent importance of mayflies and chironomid midges (Chapman and Quistorff 1938; Chapman and Bjornn 1969; Sagar and Glova 1987, 1988; Mullan *et al*. 1992; Clements and Rees 1997; Rader 1997; White and Harvey 2007; Iwasaki *et al*. 2009; Syrjänen *et al*. 2011). In lakes zooplankton are disproportionally important, and as stream size increases and gradients drop, amphipods become popular food items with migrating and rearing juvenile salmon and steelhead (Tippets and Moyle 1978; Rondorf *et al*. 1990; Muir and Coley 1996; Budy *et al*. 1998; Karchesky and Bennett 1999; Steinhart and Wurtsbaugh 2003; Teuscher 2004). However, salmonids are opportunistic and will shift their feeding to whatever is abundant, accessible, and palatable, and have sometimes have been reported with their stomachs full of unexpected prey such as snails or hornets (Jenkins *et al*. 1970; NCASI 1989; Mullan *et al*. 1992).

In general, the body of the evidence suggests that there is some ecological redundancy among aquatic stream and lake invertebrates, and if a small minority of invertebrate taxa were eliminated by chemicals at criteria concentrations, but overall remain diverse and abundant, then aquatic invertebrate overall community structure and functions, and forage value of critical habitats would likely persist. However, case-by-case consideration of the data is required because the previous assumption is tempered by the fact that aquatic insects are typically underrepresented in criteria datasets and toxicity testing in general (Mebane 2010; Brix *et al*. 2011).

Some of the anticipated effects will be to food items for juvenile salmonids, a vital component of juvenile rearing and migration habitat. Reductions in food quantity would result in limited resources to rearing and migrating fish, which can be expected to reduce population viability through increased mortality. Under-nourishment can alter juvenile salmon ability to avoid predators and select habitat within rearing drainages. Mortality can also be expected during migration, as under-nourished juveniles will not be able to withstand the rigors of migration.

Changes in species composition could have the same results. Biomass quantity is not necessarily a substitute for prey suitability, as differing prey behavior patterns and micro-habitat needs can reduce the foraging efficiency of juvenile salmonids. However, juvenile salmonids are opportunistic predators, and the loss of a minority of taxa might not be a severe indirect effect if other prey were still diverse and abundant as described above.

***Effects to Other Elements of Critical Habitat.*** Approval of the proposed criteria may also indirectly affect safe passage conditions and access. Safe passage conditions and access to other habitats may be prevented or modified if a passage barrier exists in a section of stream because of insufficient mixing at an effluent outfall, or dilution capacity is insufficient to provide a passage corridor. To avoid these forms of adverse modification of critical habitat, the application of criteria must be protective of listed species. To determine this we evaluated if the action as proposed would provide safe passage in the manners described in Appendix F Salmonid Zone of Passage Considerations.

There appears to be little to no relation between adverse changes in water quality caused by adoption of the proposed criteria and effects to the remaining essential features of critical habitat, including: (1) Water quantity; (2) riparian vegetation; (3) instream cover/shelter; (4) water velocity; (5) floodplain connectivity; (6) water temperature; and (7) space.

### 2.4.2. The Effects of Expressing Metals Criteria as a function of Water Hardness

Some of the metals criteria under review in this consultation are hardness-dependent, meaning that rather than establishing a criterion as a concentration value, the criteria are defined as a mathematical equation using the hardness of the water as the independent variable. Thus, in order to evaluate the protectiveness of the hardness-dependent criteria, it was first necessary to evaluate the hardness-toxicity relations. The criteria that vary based on site-specific hardness are Cd, Cu, Cr (III), Pb, Ni, Ag, and Zn. Hardness measurements for calculating these criteria are expressed in terms of the concentration of $CaCO_3$, expressed in mg/L, required to contribute that amount of calcium plus magnesium. In the criteria equations, hardness and toxicity values and expressed as natural logarithms to simplify the math. In a general sense, these are referred to by the shorthand “ln(hardness) vs. ln(toxicity)” relations.

In the 1980s, hardness was considered a reasonable surrogate for the factors that affected toxicities of several metals. It was generally recognized that pH, alkalinity and hardness were involved in moderating the acute toxicity of metals. While it wasn’t clear which of these factors was more important, because pH, alkalinity, and hardness were usually correlated in ambient waters, it seemed reasonable to use hardness as a surrogate for other factors that might influence toxicity (Stephan *et al*. 1985). In the case of copper, dissolved organic matter or carbon (DOM or DOC) was also recognized as being important. It was assumed that DOC would be low in laboratory waters and might be high or low in ambient waters, and that hardness-based copper criteria would be sufficiently protective in waters with low DOC and conservative in waters with high DOC (EPA 1985). Most of these relations were established in acute testing, and they were assumed to hold for long-term exposures (chronic criteria). Whether that assumption is reliable was and continues to be unclear. For instance, in at least two major sets of chronic studies with metals conducted in waters with low and uniform DOC concentrations, water hardness did not appear to have a significant effect on the observed toxicity in most cases (Sauter *et al*. 1976; Chapman *et al*. 1980).

In the two decades since the NTR metals criteria were developed, a much better understanding has been developed of the mechanisms of acute toxicity in fish and factors affecting bioavailability and toxicity of metals in water. Generally, acute toxicity of metals is thought to be moderated by complexation of metals, competition for binding sites on the surface of the fish's gill, and binding capacity of the gill before a lethal accumulation ($LA_{50}$) results (Wood *et al*. 1997; Playle 1998). The interplay of these factors has been modeled through biogeochemical "gill surface models" or "biotic ligand models" (BLMs) (Di Toro *et al*. 2001; Niyogi and Wood 2004). For brevity, "BLM" as used here refers to both.

While BLMs are conceptually applicable for developing water quality guidelines for many metals, the BLM approach is most advanced for copper. The EPA's (2007a) recommended national criteria for copper are based on the BLM. Santore *et al*. (2001) validated acute toxicity predictions of the copper BLM by demonstrating that it could predict the acute toxicity of copper to fathead minnows and *Daphnia* within a factor of 2 under a wide variety of water quality conditions. The predictive capability of the BLM with taxonomically distinct organisms is evaluated in detail in Appendix C. With fathead minnows, rainbow trout, Chinook salmon, planktonic invertebrates (various daphnids), benthic invertebrates (freshwater mussels and the amphipod *Hyalella* sp.) tested in a variety of natural and synthetic waters, predictions were always strongly correlated with measured acute toxicity. In several field studies, adverse effects to macroinvertebrate communities appear likely to have occurred at concentrations lower those allowed by EPA's (2007) chronic copper criterion. Still, the 2007 BLM-based copper criterion was a least as or more protective for macroinvertebrate communities than were EPA's 1985 and 1995 hardness-based criteria for copper.

For copper, the research leading to development of the BLM generally refutes the general relevance of the hardness-toxicity relation in ambient waters (e.g., Meador 1991; Welsh *et al*. 1993; Erickson *et al*. 1996; Markich *et al*. 2005). This is because the important factors that influence copper bioavailability are, in rough order of importance, DOC $>\approx$ pH $>>$ $>$Ca $>$ Na $\approx$ alkalinity $\approx$ Mg. Hardness is likely correlated with pH, calcium, Na, and alkalinity in natural waters, but DOC and hardness are not expected to rise and fall together.

For lead, the situation is probably similar with hardness being less important than DOC in many waters where DOC is abundant, although the BLM for lead is less advanced. With lead, calcium hardness was an important modifier of toxicity in laboratory waters with low DOC concentrations. However, at DOC concentrations reflective of many ambient waters ($>\approx$ 2.5 mg/L DOC), DOC was more important (Grosell *et al*. 2006b; Meyer *et al*. 2007b; Mager *et al*. 2011b).

In contrast, for cadmium, nickel, and zinc, the BLM and experimental data generally support the hardness-toxicity assumption in that acute toxicity to fish is influenced by water chemistry variables that are usually correlated with hardness (e.g., calcium, pH, Na, alkalinity, magnesium, in rough order of importance). The DOC is less important (Niyogi and Wood 2004). For silver, the protective effects of hardness are modest for acute or chronic silver toxicity in early life stages, juvenile, and adult rainbow trout and similar to the protection afforded to acute silver toxicity in juvenile and adult rainbow trout (Morgan *et al*. 2005).

For cadmium and zinc, or copper under conditions of low organic carbon, the ratios of calcium to magnesium influences the protective influence of hardness. Under the NTR and Idaho criteria, hardness is determined for a site, expressed as mg/L of $CaCO_3$, and input to the criteria equations for each metal. In natural waters considerable variation can occur in the calcium: magnesium ratio contributing to site-specific water hardness. Studies show significant differences in toxicity for some metals depending on this ratio. In general, calcium provides greater reductions in toxicity than magnesuim. For example, in the case of cadmium and zinc, the presence of calcium is protective against toxicity whereas magnesium, sodium, sulfate ions and the carbonate system appear to give little to no protection (Carroll *et al*. 1979; Davies *et al*. 1993; Alsop *et al*. 1999). Welsh *et al*. (2000b) and Naddy *et al*. (2002) determined that calcium also afforded significantly greater protection to fish against copper toxicity than magnesium.

The calcium:magnesium ratio in natural waters of Idaho vary by about two orders of magnitude (Appendix A). Median molar ratios of calcium:magnesium across a USGS/IDEQ network of 56 sites across Idaho monitored from 1989 to 2002 range from 0.56 to 9.73, and median ratios at all sites except one exceeded 1.3 (Hardy *et al*. 2005). In several important salmon and steelhead streams, calcium to magnesium ratio ranges are on the order of 8:1 in Valley Creek, between 4:1 and 7:1 in the upper Salmon River basin above the Pahsimeroi River, between 0.8:1 and 4:1 in Pahsimeroi River tributaries, 2:1 in the Pahsimeroi River, 1.5:1 in the Lemhi River, and 3:1 in the Salmon River at Salmon (Clark and Dutton 1996). In the review included as Appendix A, some of the lowest ratios were found outside the action area in the Coeur d'Alene region and in south-central and southeastern Idaho. Generally, these analyses indicate that the issue of hardness-toxicity relations failing and not being protective because of low calcium:magnesium ratios is not a big concern within the range of anadromous fish in Idaho.

*2.4.2.1. The Use of a "Hardness Floor" in Calculating Metals Limits.*

The Idaho hardness-dependent criteria, like the NTR criteria restrict the hardness values used in calculating the criteria to the range of 25 mg/L to 400 mg/L (EPA 1992). For high hardness values this is probably generally protective because the usual pattern of decreasing toxicity with increasing hardness breaks down at high hardness values. Heijerick and others (2002) found that at hardness values greater than 325 mg/L as calcium carbonate, no linearity, and even a decrease in 48-hour $EC_{50}$s, was observed with *Daphnia magna* and zinc. With copper and fathead minnows, above hardnesses of 150 mg/L, $LC_{50}$s apparently approached an asymptote (Erickson *et al*. 1996), and with copper and *Daphnia* at hardness of 400 mg/L and above, no relation was observed between hardness and toxicity (Gensemer *et al*. 2002). Thus, while an upper hardness ceiling of 400 mg/L might be too high, the concept of an upper ceiling is logical.

In contrast, at low hardness values this hardness floor is logically underprotective. What follows is a review of the history relating to the hardness floor issue, scientific investigations relevant to the hardness-toxicity relationship at low-hardnesses, and ambient hardness in Idaho.

***History of the Hardness Floor***. The EPA's 1992 NTR low-end hardness floor appears to have been an administrative invention associated with the promulgation of the NTR (EPA 1992); we found no support for it in any of EPA's scientific literature policy analyses that was available to date. The EPA's Guidelines (Stephan *et al*. 1985) defines a general scheme for developing criteria with increased conservatism (more protective) when data are sparse and uncertainties high. Their Guidelines specifically describes adjusting criteria based on factors that affecttoxicity, including the general ln(hardness) vs. ln(toxicity) relationship. NMFS did not find the suggestion of imposing a low-end floor on hardness-toxicity relations in the Guidelines or any of the individual criteria documents from the 1980's was any suggestion of imposing a low end floor on hardness-toxicity relations found (e.g., EPA 1984b; 1985c; 1987b). Thus the notion of making unprotective assumptions about water quality criteria in the absence of supporting data or theory is generally counter to the EPA's science approach in the criteria process. Further, the low-end hardness floor notion is contrary to results of EPA research that specifically investigated metals toxicity at very low hardness. For example, Cusimano *et al*. (1986) tested the toxicities of cadmium, copper, and zinc to rainbow trout at low hardness (9 mg/L).

It appears that EPA tacitly recognized the error of the 1992 low-end hardness floor shortly thereafter. No hardness floor appeared with the metals criteria contained in the 1995 Great Lakes Initiative (40 CFR 132.6) nor in EPA's 1997 California Toxic Rule (40 CFR 131.37), and EPA's 1999 national recommended water quality compilation was silent on hardness floor (EPA 1999b). In 2002, EPA directly repudiated the 1992 hardness floor policy, asserting that while data below hardness of about 20 mg/L are limited, "*capping hardness at 25 mg/L without additional data or justification may result in criteria that provide less protection than that intended by EPA's Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses (EPA 822/R-85-100) or 'the Guidelines.' Therefore, EPA now recommends that hardness not be capped at 25 mg/L, or any other hardness on the low end*" (EPA 2002b). The EPA further recommended that "*if there is a state or tribal regulatory requirement that hardness be capped at 25 mg/L, or if there are any situation-specific questions about the applicability of the hardness-toxicity relationship, a Water Effect Ratio (WER) procedure should be used to provide the level of protection intended by the Guidelines*" (2002b).

Beyond the preceding quoted sentence, NMFS located no further details on how to use the WER procedure to remedy the hardness floor issue.

***Hardness-toxicity patterns in soft water.*** Fish maintain their internal mineral balance through osmoregulation, and the greater the difference between their internal plasma mineral balance, and the mineral content of the water they live in, the greater the energy required to maintain homeostasis. In waters with very dilute mineral content (soft water), the energy requirements to maintain their mineral balance, or ionic balance, can be high. Compared to hard water, costs of these energy requirements to maintain ionic balance in soft water include reduced growth, reduced swimming ability, and reduced ability to recover from severe exercise (McFadden and Cooper 1962; Wood *et al*. 1983; Wood 1991; Kieffer *et al*. 2002; Dussault *et al*. 2008; Wendelaar Bonga and Lock 2008). In very soft water, fish may be on the verge of ionoregulatory problems, and because metals also disrupt ionic balance, any increase in metals may result in plasma ion loss (Playle *et al*. 1992; Van Genderen *et al*. 2008). The similarity in responses of fish to soft water acclimation and metals exposure suggest that simple extrapolation of hardness-toxicity relations that were developed at high hardnesses to soft waters may underestimate the additive responses and thus underestimate metals toxicity in very soft waters.

Empirical evidence and theoretical considerations both argue against the assumption that the general pattern of increasing toxicity of metals with decreasing hardness stops at 25 mg/L. However, the slope might be expected to be different than that at higher hardnesses and there are both rationales and data suggesting that the slope would be shallower or steeper at low hardnesses. Based on calculations of cation competition and aqueous complexation, Meyer (1999) predicted that for divalent transition metals such as cadmium, copper, and zinc, the slope of hardness-toxicity relations, as ln(hardness) vs. ln($LC_{50}$) was likely to start shallowing below a hardness of about 20 mg/L and would reach a slope of zero at a hardness of about 3 mg/L. The 3 mg/L hardness floor theorized by Meyer (1999) was a data-free prediction because tests of hardness-toxicity relations at low hardnesses seem limited to minimum hardnesses of about 5 to 10 mg/L for cadmium, copper, lead, nickel, and zinc (Miller and Mackay 1980; Cusimano *et al*. 1986; Long *et al*. 2004; Sciera *et al*. 2004; Mebane 2006; Deleebeeck *et al*. 2007; Mebane *et al*. 2012). Morgan *et al*. (2005) did test the comparative effects of silver to rainbow trout at hardnesses of 2, 150, and 400 mg/L, but because the soft water exposure caused adverse effects without any metals addition, and the wide range of hardnesses tested, the data were insufficient to directly evaluate Meyer's theoretical 3 mg/L hardness-toxicity floor. However, because exposure to 2 mg/L hardness water by itself caused a doubling in mortality rates and increased time to hatch for rainbow trout embryos, the notion of 3 mg/L hardness-toxicity floor may be moot. One of the more comprehensive studies of metals toxicity was by Van Genderen *et al*. (2005). They found that over a hardness range of 6 to 40 mg/L in laboratory waters with low organic matter, there was a linear trend between copper toxicity to fathead minnows and hardness. They observed a species-specific slope between ln (hardness) and ln ($LC_{50}$) of (0.795 for hardness ranging from 6 to 40 mg/L as $CaCO_3$) was less than the pooled value for all species developed for EPA's (1985) copper dataset (0.9422 for hardness ranging from 13 to 400 mg/L as CaCO3). Van Genderen *et al*. (2005) suggested that the lower slope indicated that the influence of cation competition changes in low-hardness waters. The implications of these differing slopes are that Van Genderen *et al*.'s (2005) results showed that as hardness declined, copper becomes more toxic but because of the shallower slope, the increases in toxicity were not as great as predicted by EPA's (1985) steeper slope.

However, a safer interpretation of the general relationship between water hardness and metals toxicity is that aquatic organisms are likely more sensitive to metal exposure than would be expected by hardness-toxicity relations determined at higher ambient hardnesses. This is because fish have higher energy requirements to maintain homeostasis in soft water, and may be more sensitive to metals that inhibit ionoregulation (Greco *et al*. 1995; Taylor *et al*. 2000; Taylor *et al*. 2003; Van Genderen *et al*. 2005; Van Genderen *et al*. 2008; Wendelaar Bonga and Lock 2008). The increased sensitivity of fish to metals in very soft water may persist after fish that were acclimated or incubated in very soft water move into higher hardness water. Mebane *et al*. (2010) incubated rainbow trout in waters above the confluences of two streams, one with very-soft water (average hardness around 11 mg/L) and one with harder water with an average hardness of about 21 mg/L. Then the fish were exposed to cadmium and zinc in the harder of the two waters. The fish that had reared in the stream with softer water were about twice as sensitive as were trout that had been incubated in the higher hardness water (Mebane *et al*. 2010). This has implications for salmonid life histories and habitats. Water hardness tends to be lowest near the headwaters of streams and increase downstream, and some salmonids tend to ascend streams to spawn in the upper reaches of watersheds and after emerging, their fry move downstream into higher hardness waters.

In Section 2.4.2 of this analysis, NMFS show plots of metals toxicity vs. hardness for various salmonid species at various life stages for cadmium, copper, lead, nickel, and zinc. For at least cadmium, copper, and zinc, those plots show a general relationship of decreasing resistance by the fish with decreasing hardness, a pattern that did not stop at a hardness limit of 25 mg/L $CaCO_3$. However, meta-analyses in this manner have limitations for analyzing specific relations between variables such as hardness-toxicity relations. This is because toxicity to salmonids and other fishes can vary by other factors which can obscure the patterns of interest. The influence of different sizes or developmental state is well known to be important, but other factors could influence the results. These include the strain or stock of fish; incubation or acclimation history conditions; water characteristics other than hardness such as pH, ionic composition, organic matter or particulates; and water renewal rates and frequencies. Data pooling such as was done for the summaries of effects for individual metals later in Section 2.4 is sometimes a beneficial and necessary means of generalizing study findings because this broader view may sometimes reveal patterns that may not be apparent in smaller, individual studies. However, important patterns can be lost.

The following data sets illustrate how pooling data that are only influenced by a few such factors can greatly confound hardness-toxicity relations. In an effort to develop site-specific water quality criteria for a soft-water river, the South Fork Coeur d'Alene River, Idaho, toxicity tests were conducted with cutthroat trout and rainbow across a range of water hardnesses (Mebane *et al*. 2012). Rainbow trout were used to develop hardness-toxicity relations. All the rainbow trout were obtained as eggs from a single supplier (Mt Lassen Trout Farms, Red Bluff, California) and incubated on site; all tests were done in the same test facility, and were directed by the same people. However, because it is seldom feasible to always test fish, at say, 30-days post hatch, some tests were run with fish of slightly different ages. In contrast, some tests were run side-by-side to specifically examine hardness variability using the same batch of fish at the same time, using waters collected from different waters with different hardnesses (Mebane *et al*. 2012).

With zinc, Figures 2.4.2.1 and 2.4.2.2 illustrate how hardness-toxicity patterns were always stronger when hardness was varied within a test series using the same batch of fish at the same time, than were patterns from meta-analyses that pooled data from across tests. The most complete data are with zinc. A simple comparison of hardness-toxicity relations with zinc from cutthroat trout fry over a hardness range of 11 to 63 mg/L shows that hardness can explain nearly 100% of the variability in toxicity. In contrast, when Mount Lassen rainbow trout are pooled across different years and batches, hardness explains less than half of the variability. Yet when the Mount Lassen rainbow trout results are grouped by concurrent test groups, the subgroup hardness toxicity relations explain from around 85% to 98% of the variation in toxicity compared to about 38% when pooled across groups. The reasons for the differences between groups are unclear, although differences in the sizes of fish might be a factor since the largest fry (average 0.46g wet weight) were most sensitive. Other testing has found that in the range of 0.2 to 1.0g, smaller fry tended to be more resistant to zinc toxicity (Hansen *et al*. 2002c).

With zinc, at a hardness of 10 mg/L, the Idaho acute and chronic criteria would both be about 17 µg/L (Table 2.4.2.1), which is similar to an estimated $EC_{50}$ of about 21 µg/L for rainbow trout in waters with hardness of about 7 mg/L (Mebane *et al*. 2012), which was the lowest hardness test found. A concentration killing 50% of the test organisms can hardly be considered protective. If instead, the criteria were calculated with the ambient hardness of 7 mg/L, the criteria would be 12 µg/L, and if calculated with the proposed hardness floor of 25 mg/L the criteria would be 36 µg/L. At 36 µg/L, the lowest concentration actually tested, 80% of the rainbow trout were killed in this test.

With nickel, the most sensitive organisms appear to be zooplankton with approximate thresholds of adverse effects ($EC_{10}$s) of about 3 to 7 µg/L in very-soft water with hardness of 6 mg/L (Deleebeeck *et al*. 2007) compared to threshold of adverse effects for rainbow trout of <35 µg/L at hardness 27 to 39 mg/L (Nebeker *et al*. 1985). The NTR chronic nickel criterion is well above these values at 49 µg/L. However, Idaho's revised criteria, proposed for approval by EPA (Table 2.4.2.1) are 16, 7, and 5 µg/L at hardnesses of 25, 10, and 6 mg/L.

For lead, a different shortcoming of these types of hardness-toxicity comparisons becomes apparent in Figure 2.4.2.3. As with zinc, cutthroat trout sensitivity to lead is strongly influenced by hardness, with a reasonable spread of hardnesses of a range of 11 to 56 mg/L explaining about 80% of the variability in cutthroat $EC_{50}$s for lead. For rainbow trout, the range of hardnesses for six tests was only 20 to 32 mg/L, and when all rainbow trout tests were pooled and regressed against hardness, the results had no explanatory value ($r^2 = 0.05$). The only tests conducted as a series (the three points with the highest $EC_{50}$s) only varied from 23 to 32 mg/L, still only resulted in a regression explaining 48% of the variability (not shown).

Cadmium from the South Fork Coeur d'Alene testing shows a similar pattern with an inadequate spread of the hardness data *(*Figure 2.4.2.4*)*. If all tests were pooled, the resulting relation is weak with a best fit regression only explaining only about 36% of the variability; when the regression is limited to the four concurrent tests, hardness can explain about 68% of the variability.

This problem of an inadequate spread in the hardness as the independent variable in regressions or pooling disparate data is a common limitation in hardness-toxicity meta-analyses of found data. For example, Meyer *et al*. (2007b) includes a comprehensive review of metals toxicity versus hardness. Their plots often show clumps of poorly distributed hardness values. Two unpublished reviews focusing on soft-water metals toxicity hardness relations showed similar patterns (CEC 2004a; Lipton *et al*. 2004). Mebane (2006, p.20) pooled hardness-toxicity data for rainbow trout and cadmium from across a variety of studies for a total of 37 studies. The plot shows a fair amount of scatter and hardness explained about half the variability in the cadmium acute toxicity data with rainbow trout ($r^2 = 0.56$). In contrast, hardness-toxicity data for brown trout where most data were from a single study that explicitly tested cadmium toxicity across a wide range of hardness showed a much tighter relation between hardness and acute toxicity ($r^2 = 0.97$).

These comparisons show that pooling datasets may also wash out patterns that are only apparent in the smaller, synoptic datasets.

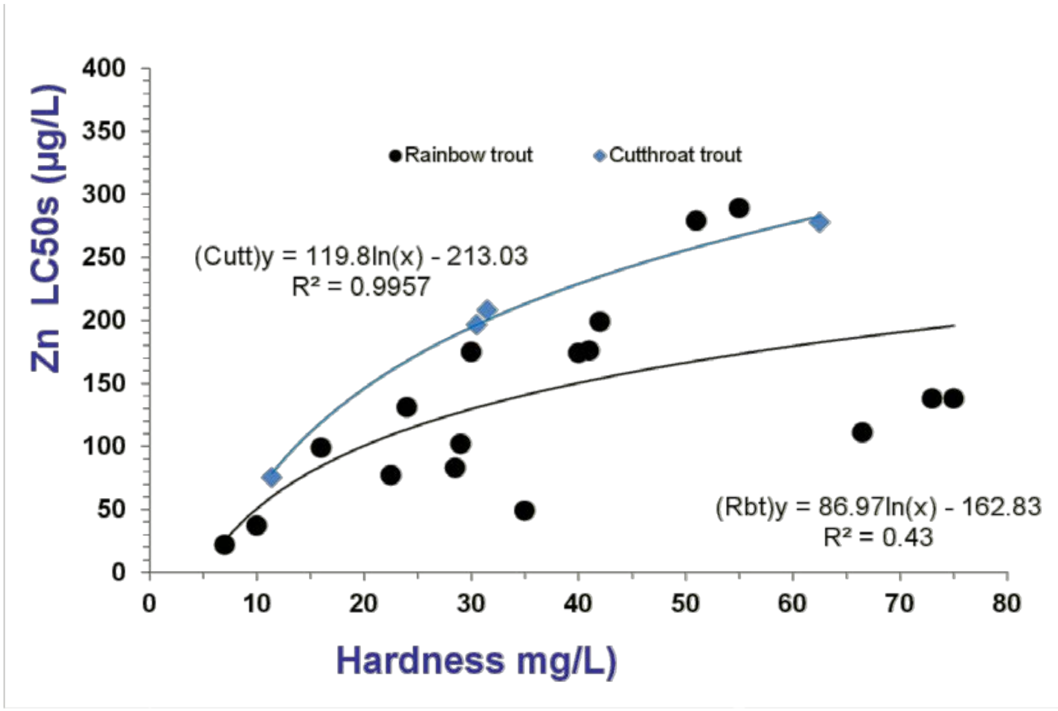



**Figure 2.4.2.1. Zinc toxicity versus water hardnesses for swim-up stage rainbow trout pooled across test groups and westslope cutthroat trout (data from Mebane *et al*. (2012).**

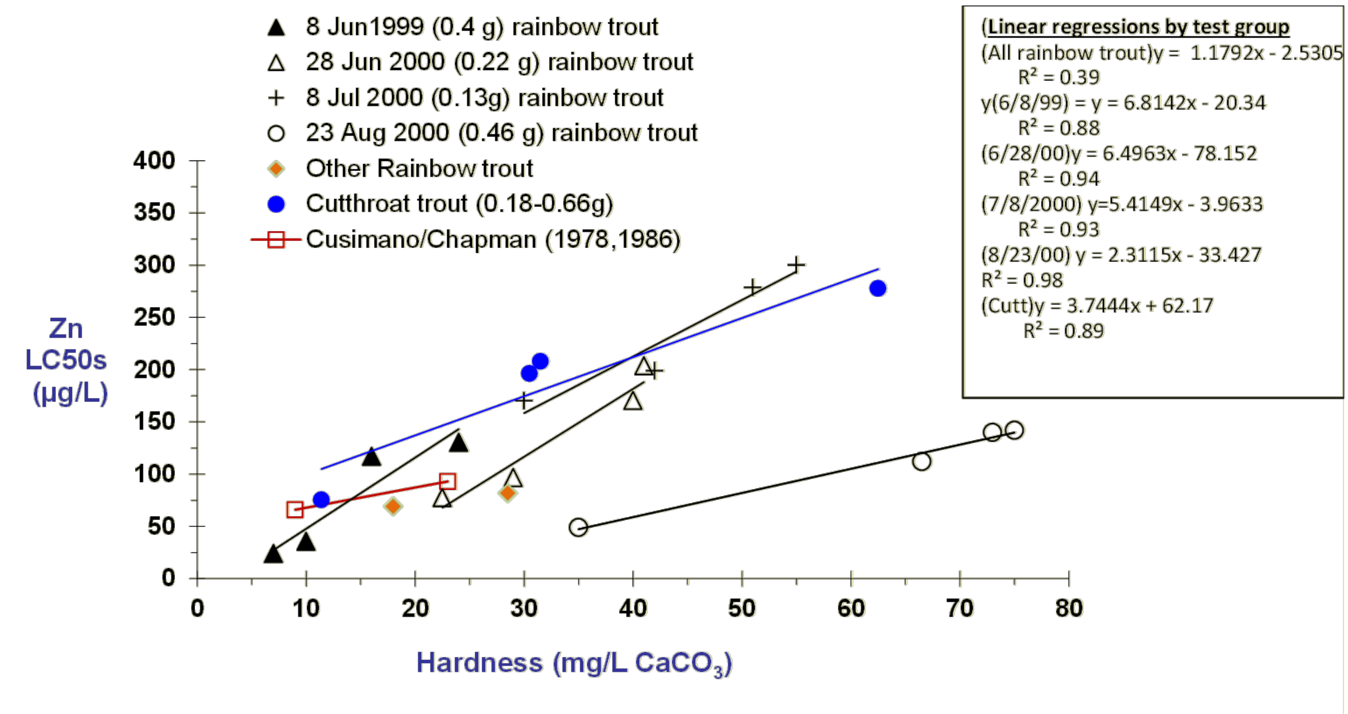



**Figure 2.4.2.2. Zinc (Zn) toxicity versus water hardnesses for swim-up stage rainbow trout by concurrent test groups, cutthroat trout, and steelhead tested under similar conditions by the same people (average fry weights are in parentheses) Data from (data from Mebane *et al*. (2012) except for steelhead data which are from Cusimano, Brakke and Chapman (1986)and Chapman (1978b).**

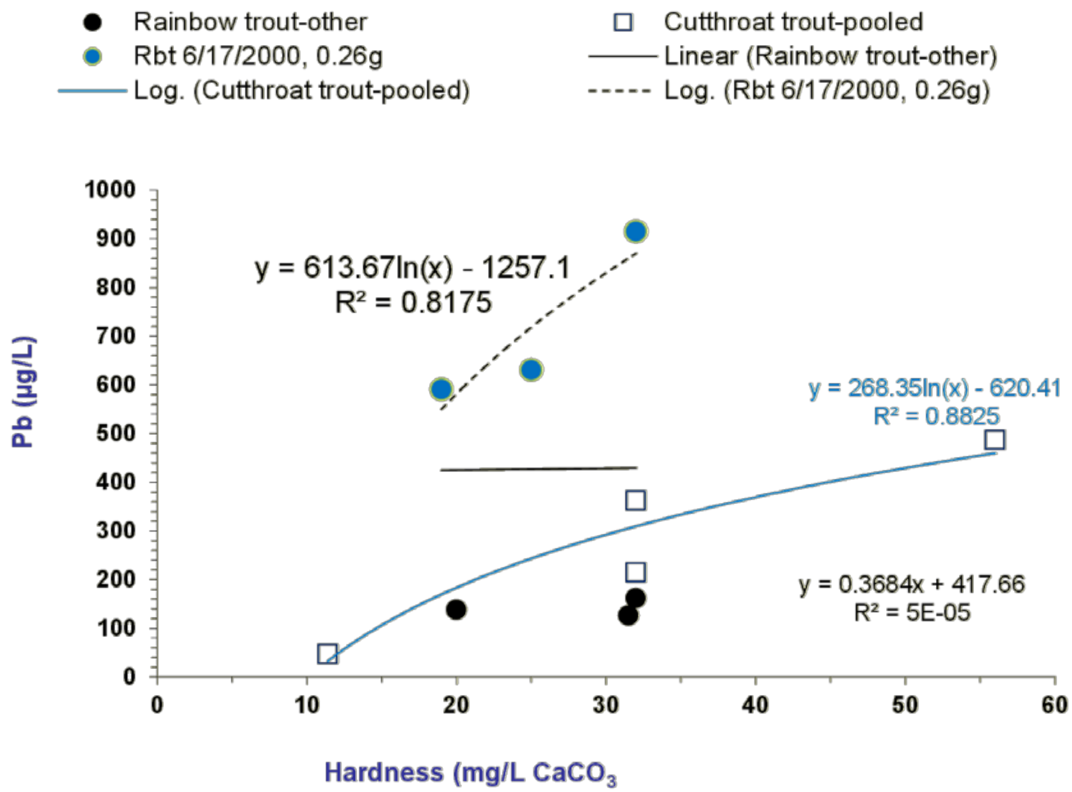



**Figure 2.4.2.3. Lead (Pb) toxicity versus water hardnesses for swim-up stage rainbow trout either pooled across test groups, or separated into synoptic and other tests and pooled westslope cutthroat trout (data from Mebane *et al*. (2012).**

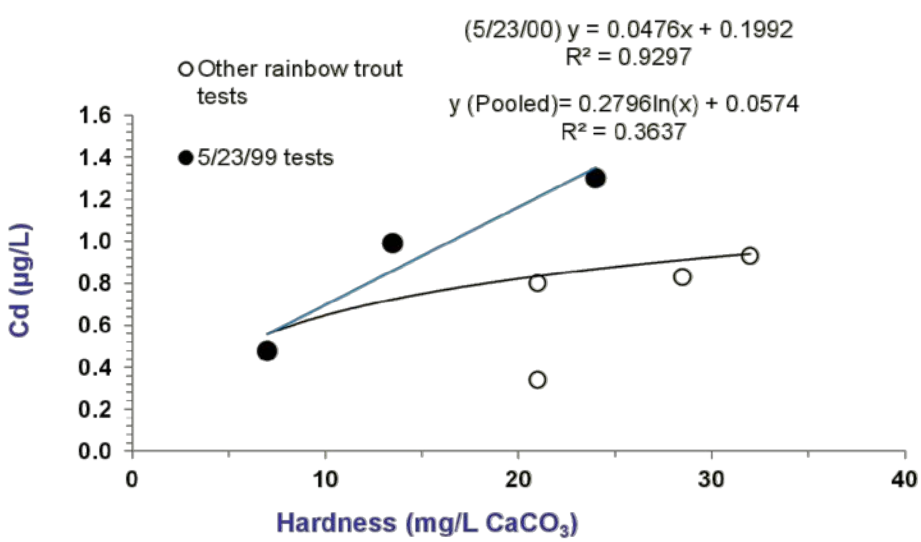



**Figure 2.4.2.4. Cadmium (Cd) toxicity versus water hardnesses for rainbow trout tested under the same conditions on 5/23/99 versus "other" rainbow trout tested by the same people in the same facility, using the same source of fish eggs, same water sources, but using fish that were a few weeks apart in age (data from Mebane *et al*. (2012).**

***Relevance of the hardness floor issue in the action area***. Nationally, about 20% of the freshwaters can be considered "softwater" (Figure 2.4.2.5). Within the range of listed salmon or steelhead "salmon country" in Idaho, water hardness tends to decrease from south to north (Figure 2.4.2.6). In the Salmon River drainage in the southernmost portion of the range of anadromous fish in Idaho ("salmon country"), water hardnesses are highly variable, apparently depending on the bedrock geology. Hardnesses are relatively high in drainages with carbonate rock (e.g., Lemhi and Pahsimeroi river drainages), intermediate in watersheds with volcanic rock, and very low in the granitic drainages of the Idaho Batholith. The Idaho Batholith is the dominant geologic feature of much of central Idaho (Appendix A, Thomas *et al.* 2003; Hardy *et al.* 2005). Hardnesses as low as 4 mg/L have been measured in softwater areas of Idaho (Figure 2.4.2.6); however, the true minimum hardnesses in streams in granitic watersheds are probably close to that of snowmelt, which is in the range of 0.5 to 1 mg/L total hardness (Clayton 1998).


**Figure 2.4.2.5. Soft-water ecoregions of the USA where most water hardness values are <50 mg/L CaCO3 (Whittier and Aitkin 2008).**

The magnitude of likely effects of the hardness floor on criteria values is probably substantial in waters with the lowest hardnesses within the range of anadromous salmonids in Idaho. The best data sets are from monitoring of waters into which effluents from hard rock mines are discharged. Several major active and inactive mining operations are present in the Salmon River drainage. The inactive operations still discharge effluents and some are regulated by EPA under the NPDES program.

Historically, mining also occurred in the Clearwater River and drainages in the Hells Canyon reach of the Snake River such as around the old mining towns of Cuprum and Florence. However, these mining districts played out and there has been no large scale mining activity in these areas in at least the last 50 years or so. The hardness floor issue in Idaho's salmon country is only relevant to industrial mining. Within salmon country, NPDES effluent limits have been imposed by EPA on one major urban wastewater treatment plant (city of Lewiston), many minor wastewater discharges from small towns and consolidated sewage treatment districts, and two major forest products facilities. NMFS reviewed the fact sheets detailing known or suspected pollutants and calculations of the reasonable potential to exceed metals criteria for these current discharges. Other than the mines, none of the facilities had measured or projected metals concentrations that approached having reasonable potential to exceed any metals criteria. In the case of the city of Lewiston, the maximum concentrations measured in the undiluted effluent exceeded criteria by nine times for copper and about three times for cadmium and zinc. However, the EPA "reasonable potential to exceed" determination assumes that dilution with river water will be allowed using 25% of the receiving water flows, and it is only necessary for facilities to comply with WQS after mixing and dilution. The city of Lewiston discharges into a large river (the Clearwater River) with a minimum dilution ratio of 37 to 1, which would dilute these metals to well below criteria. (*http://yosemite.epa.gov/r10/water.nsf* accessed February 2008). See also Appendix D on issues with mixing zones and dilution assumptions.

Within the Salmon River drainage, the mining operations tend to be located high in watersheds where the waters may have quite low hardness values. In EPA Region 10's effluent limits calculations, EPA tends to use the $5^{th}$ percentile of measured hardness values, which is a conservative approach. Estimated ranges of water hardnesses for major mining discharges within the ranges of listed salmonids are summarized in Table 2.4.2.1. The hardness floor is a substantive concern in about 75% of the receiving waters.

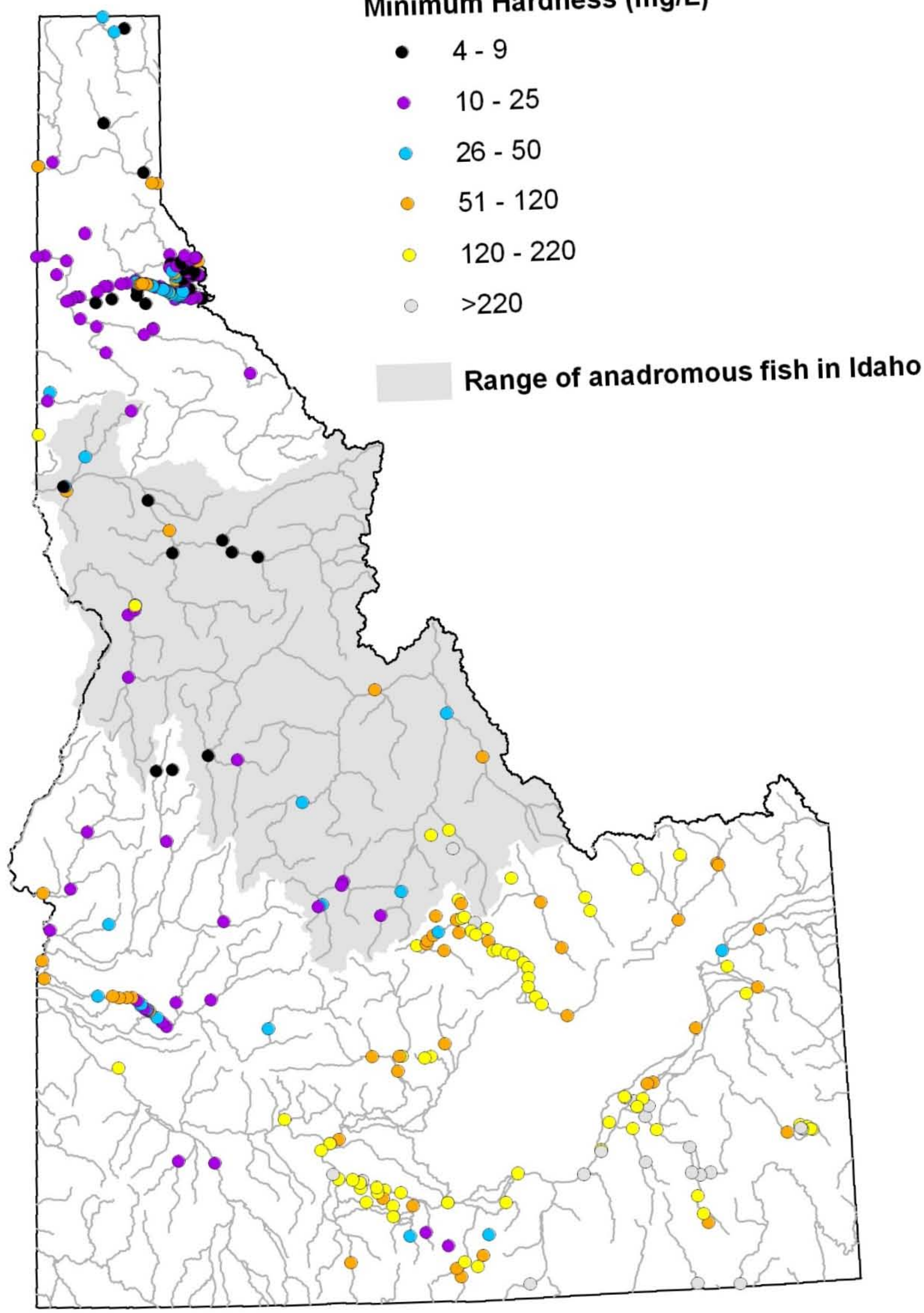



**Figure 2.4.2.6. Minimum hardness values measured at 323 sites in Idaho between 1979-2004 (data from Appendix A)**

**Table 2.4.2.1. Ranges of low hardnesses observed in Salmon River basin receiving waters of industrial mine effluents or nonpoint source mine runoff (limited to major facilities discharging to waters either designated as critical habitats for listed salmonids or at least some portions are accessible and presumably used by listed salmonids.**

| Stream | Location | Hardness "range" (mg/L as $CaCO_3$) | Statistics for "range" | Source |
|---|---|---|---|---|
| Napias Creek | Downstream of mine effluent | 4 – 6 | 5th percentiles during high and low-flow tiers respectively | EPA |
| Big Deer Creek | Downstream of mining-affected tributary (Bucktail Cr) | 9 – 36 | Range of 6 observations | (Maest *et al*. 1994) |
| Salmon River | Upstream of permitted but inactive outfall from Thompson Cr. Mine | 15 – 54 | Range of 14 observations | USGS |
| Jordan Creek | Downstream of mine effluent | 16 – 39 | 5th percentiles during high and low-flow tiers respectively | EPA |
| Panther Creek | Downstream of mining-affected tributary (Blackbird Cr) | 17 – 48 | Range of 68 observations | (Maest *et al*. 1994) |
| Yankee Fork | Upstream of mine effluent | 19 – 54 | Range of 47 observations | Note |
| Yankee Fork | Downstream of mine effluent | 24 – 149 | Range of 47 observations | Note |
| Squaw Creek | Downstream of mine effluent | 45 – 110 | 5th percentiles during high and low-flow tiers respectively | EPA |
| Thompson Creek | Downstream of mine effluent | 55 – 85 | 5th percentiles during high and low-flow tiers respectively | EPA |

Notes: "EPA" data from factsheets accessed January 2008 from *http://yosemite.epa.gov/r10/water.nsf*. USGS data from site 13296500, Salmon River below the Yankee Fork, *http://waterdata.usgs.gov/id/nwis/qw*; Yankee Fork data courtesy of B. Tridle, Hecla Mining Co.

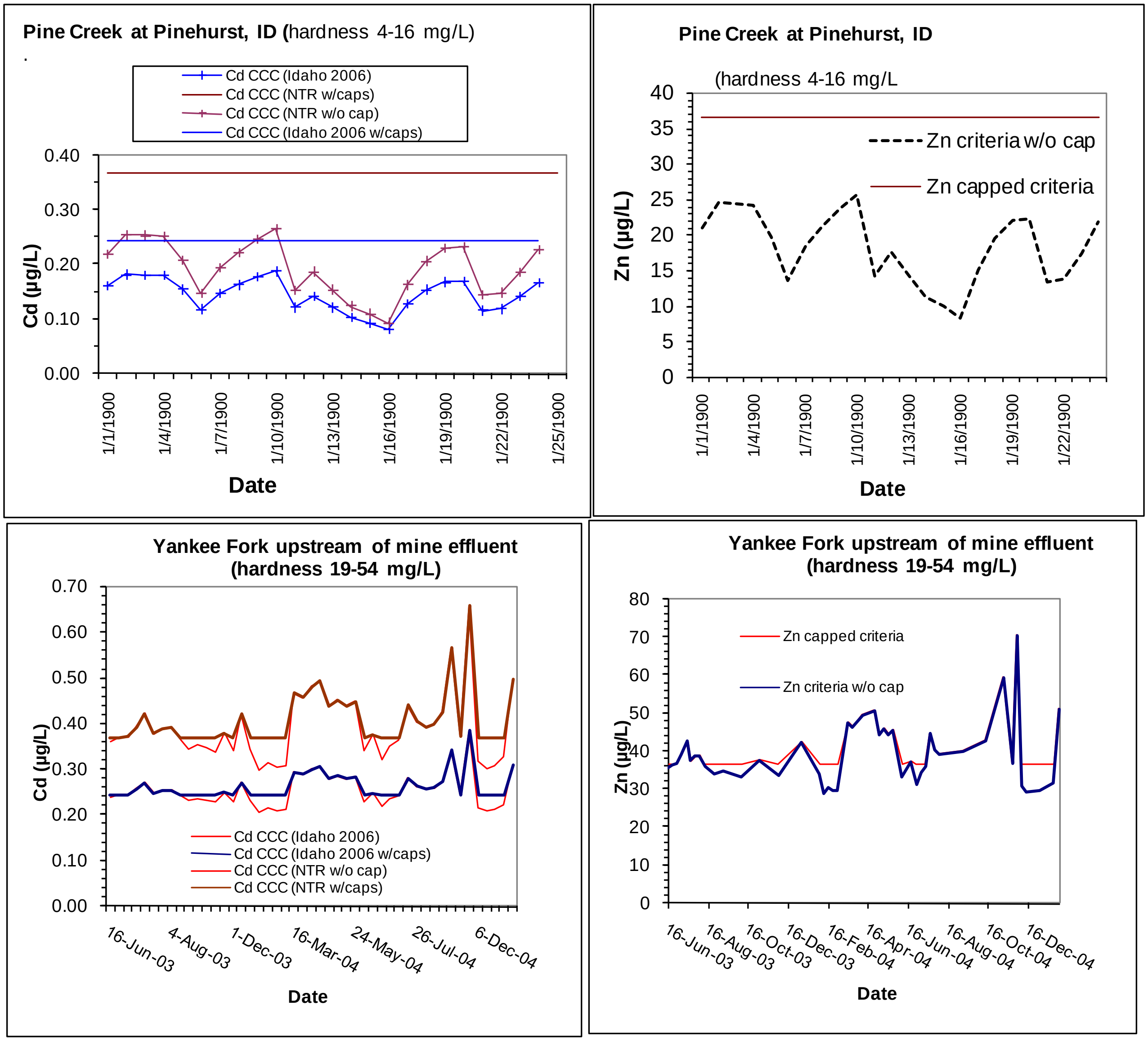



**Figure 2.4.2.7. Examples of the effects of the "hardness floor" on cadmium and zinc criteria in very-soft and soft water settings. In the Pine Creek example (top), all hardness observations were less than the 25 mg/L floor. In this very-soft water example, applying a hardness floor would result in the criteria being considerably less protective than intended by EPA Guidelines at all times, with the floor-limited criteria as much as 3X higher. The Yankee Fork example (bottom) is probably more typical of soft water streams in the Salmon River drainage. There the floor has little or no effect during much of the record, at the worst the floor-limited criteria were about 1.25X higher than the hardness-dependent criteria.**

Thus, there are many streams in the Salmon River and Clearwater River drainages in Idaho where hardness concentrations average less than 25 mg/L, for which concentrations of contaminants with hardness ameliorated toxicity should be calculated on actual site conditions, and which have active metals discharges.

The magnitude of likely effects of the hardness floor on criteria values is compared graphically in Figure 2.4.2.6. The first illustration, from Pine Creek, a tributary to the South Fork Coeur d'Alene River, Idaho, is located outside the salmon country area of interest, but is shown because it is probably similar to the streams with very low hardnesses and because it had a robust data set. In this example, the "floor-limited" criteria values are up to three times higher than criteria calculated on relevant site hardness values. In this stream, because the hardness never rises above 25 mg/L, the hardness-floor-limited criteria plot as horizontal lines. While the hardness of Pine Creek is very low, ranging from only 4 to 16 mg/L in Figure 2.4.2.1, it is not uniquely low. In the North Fork Payette River at McCall, Idaho, measured hardnesses only ranged from 6 to 7 mg/L, n=9 (Hardy *et al*. 2005). Since the North Fork Payette River upstream of McCall shares similar geology as much of the adjacent South Fork Salmon River drainage, similarly low hardness values are presumed to occur in the South Fork Salmon River drainage.

In the more intermediate example of the Yankee Fork upstream of mine effluent, the floor-limited criteria are only biased high (unprotective) compared to the uncapped criteria by 1.2 times or less.

#### *2.4.2.2. Summary of Effects of the Hardness Floor for Calculating Metals Criteria*

Exposure of listed Snake River salmon and steelhead to levels of metals in discharges at proposed criteria levels will result in adverse effects. Many of the streams in the Salmon River and Clearwater River drainages of Idaho also have hardness concentrations that average less than 25 mg/L which is the current floor in the hardness equation. For copper and lead, hardness is less important than DOC, but if DOC is low, toxicity does increase below the hardness floor. For nickel, and zinc, acute toxicity to fish rises as hardness declines below the 25 mg/L. For silver, acute toxicity increases modestly in early life stages, below the hardness floor.

The use of a hardness floor of 25 mg/l in calculating metals discharge limits will allow for increased exposures of listed fish to levels of metals that result in adverse effects. These effects range from a direct increase in mortality to decreases in growth and survival of juvenile Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River Sockeye salmon and Snake River Basin steelhead.

### 2.4.3. The Effects of EPA Approval of the Arsenic Criteria

Arsenic is been well known for its high dietary toxicity to humans for hundreds of years, and arsenic poisoning was a popular method of political assassination and murder starting at least in the Middle Ages. To mammals, arsenic is carcinogenic, mutagenic, and teratogenic, and at high enough dietary exposures can be directly lethal. Compared to mammalian toxicology, relatively little work has been done with fish at environmentally relevant exposures (Sorenson 1991).

At environmentally relevant concentrations, adverse effects in fish from arsenic are most likely from dietary rather than waterborne exposures as discussed below. Arsenic and selenium interact with each other in various metabolic functions and each element can substitute for the other to some extent, which could partly explain the reported protective effect of selenium against some arsenic-linked diseases (Plant *et al*. 2007).

The water quality criteria concentrations that are evaluated as part of this action are: acute criterion, not to exceed 360 µg/L; and chronic criterion, not to exceed 190 µg/L. Also applicable to all waters in the action area is a recreational use criterion of 10 µg/L. Whereas all of Idaho's aquatic life criteria are expressed as dissolved metals, the IWQS are ambiguous whether the human health based 10 µg/L is expressed as dissolved or total recoverable arsenic. The rules only state that the criteria addresses "inorganic arsenic only" (IDEQ 2007a). The latter provision is unexplained and is curious because organic arsenic species probably have different bioavailablity and toxicity than inorganic species. Plant *et al*. (2007) stated that organic arsenic forms are likely more bioavailable and toxic than inorganic forms, although as discussed later in the section, organic arsenic may be less toxic than inorganic arsenic in the diet of fish. Presumably the human-health recreational use standard was intended as total arsenic since those "fishable and swimmable" criteria address exposures from incidental consumption of water while swimming or eating fish. Neither swimmers nor fish can be expected to filter their water prior to ingestion.

The human health based criteria apply to all waters in Idaho unless there are specific exclusions. The IWQS have one such exclusion, Bucktail Creek, a small stream contaminated by mine waste. Bucktail Creek is a tributary to Big Deer Creek, which is a tributary to Panther Creek, which in turn a tributary to the Salmon River, in the Middle Salmon-Panther hydrologic unit (Figure 1.4.3.1). The Middle Salmon-Panther hydrologic unit is designated as critical habitat for Snake River spring/summer Chinook salmon. This critical habitat designation is defined to include river reaches presently or historically accessible (except reaches above impassable natural falls (NMFS 2004). Most of the Big Deer Creek watershed, including Bucktail Creek is located above an impassable natural fall. Within a mile upstream from the mouth of Big Deer Creek with Panther Creek, a series of natural cascades and waterfalls block upstream passage by anadromous fish (Reiser 1986). Therefore Bucktail Creek is not considered to be within the critical habitat designations for either Snake River spring/summer Chinook salmon or steelhead. Designated critical habitat for Snake River Basin steelhead is defined specifically by water body; only the lowest reach of Big Deer Creek, not including Bucktail Creek, is designated critical habitat for Snake River basin steelhead.

#### *2.4.3.1. Species Effects of Arsenic Criteria*

Arsenic toxicity does not vary significantly with hardness (Borgmann *et al*. 2005a). Because IDEQ has inclusive rules for designated aquatic life and recreational uses, the human-health related criteria also apply in all designated critical habitats and waters inhabited by listed salmon and steelhead in Idaho (IDEQ 2007a).

***Acute Arsenic Criterion.*** No studies were found that reported acute toxicity to juvenile or adult salmonids at arsenic concentrations close to the acute criterion. All studies NMFS reviewed indicate that acute toxicity, including to alevins, occurs at concentrations that are significantly higher than the acute criterion (e.g., Buhl and Hamilton 1990). Ambient arsenic concentrations in surface water are never known to approach the acute criterion.

***Chronic Arsenic Criterion.*** A conclusion that can be drawn from a recent comprehensive review of arsenic toxicology in fishes by McIntyre and Linton (2011) is that arsenic is not very toxic in classic toxicity tests with exposures through water. The results of Birge *et al*. (1978, 1981) suggests that chronic arsenic toxicity from waterborne exposures occurs to developing embryos of listed salmonids at concentrations below the chronic criterion. Rainbow trout embryos were exposed to arsenic for 28 days (4-days post-hatching) at 12°C to13°C and a hardness of 93 mg/L to 105 mg/L $CaCO_3$ in static tests. Concentrations of 42 to 134 µg/L were estimated to be associated with the onset of mortality, as LC1 and LC10 respectively (Birge *et al*. 1980). No detail of the results of this test were reported beyond these statistical effects estimates, making these results impossible to critically review. Acclimation appears to enhance resistance to chronic arsenic toxicity (Dixon and Sprague 1981; EPA 1985a). Studies reviewed in Eisler (1988a) and EPA (1985a) indicate that chronic effects do not occur in other lifestages until concentrations are at least about an order of magnitude higher than the levels determined by Birge *et al*. (1978, 1981) to be detrimental to developing embryos. The reported concentrations associated with chronic embryo and fry mortality were much lower than the chronic criterion.

***Dietary toxicity of arsenic.*** Cockell *et al*. (1991) fed rainbow trout arsenic contaminated food under standard laboratory conditions for 12 to 24 weeks and correlated signs of toxicity with diet and tissue arsenic concentrations. They found that the threshold for the onset of organ damage (gall bladder inflammation and lesions) was between 13 and 33 mg/kg arsenic in food. Woodward *et al.* (1994, 1995) fed rainbow trout a diet made from invertebrates collected from the metals contaminated Clark Fork River, Montana, which resulted in lower growth and survival of the fish fed the metals contaminated wild diet. However, because these wild metals-contaminated invertebrates were contaminated with several metals including arsenic, and the effects were equally correlated both with arsenic and copper, effects could not be attributed to either. Subsequently Hansen *et al*. (2004) collected metals-contaminated sediments from the Clark Fork River, reared aquatic earthworms (*Lumbriculus*) in them, and fed the *Lumbriculus* to rainbow trout. Fish fed the *Lumbriculus* diet had reduced growth and physiological effects, and the presence of effects was strongly correlated with arsenic but not to other elevated metals. Bull trout collected from mining-influenced Gold Creek in northern Idaho, showed similar liver damage with inflammation, necrosis and cellular damage. Arsenic was elevated in the sediments, periphyton, and macroinvertebrates, and fish tissues, and was correlated with the liver damage (Kiser *et al*. 2010). Erickson *et al*. (2010) further implicated arsenic as the causative agent by experimentally mixing arsenic into clean sediments, rearing *Lumbriculus* in them, and feeding the *Lumbriculus* to rainbow trout. The rainbow trout fed the worms that had been raised in arsenic dosed sediments again had reduced growth and disrupted digestion. Erickson *et al*. (2010) is difficult to directly compare to feeding studies with field collected invertebrates because Erickson *et al.* (2010) did not report what tissue concentrations bioaccumulated in fish following 30 days on a diet of arsenic enriched invertebrates. Still, the Erickson *et al.* (2010) study produced similar effects to those from field-collected diets with controlled exposures to contaminated field sediments and strongly implicated arsenic as an important stressor.

Together these studies have shown that inorganic arsenic in the diet of rainbow trout are associated with reduced growth, organ damage and other physiological effects at concentrations in the diet of about 20 mg/kg dry weight (dw) and above (Cockell 1991; Hansen *et al*. 2004; Erickson *et al*. 2010). Ranges of reported effects in other species are wider. Damage to livers and gall bladders occurred in lake whitefish (*Coregonus clupeaformis*) fed arsenic contaminated diets as low as 1 mg/kg food dw (Pedlar *et al*. 2002). Adverse effects of dietary arsenic to salmonids are summarized in Table 2.4.3.1. Bioaccumulation of arsenic in prey organisms to concentrations higher than 30 mg/kg dw has been documented from the Clark Fork River, Montana; Boulder River, Montana; the Coeur d'Alene River, Idaho; and Panther Creek, Idaho. Concentrations of arsenic in these streams have been measured at higher than background (<~ 5µg/L) but were never documented at concentrations even approaching the chronic water quality criterion of 190 µg/L dissolved arsenic (Table 2.4.3.2). Review of waterborne arsenic concentrations collected from the same waters suggests that bioaccumulation of arsenic in invertebrate prey organisms to concentrations harmful to salmonids appears to be able to occur in streams with dissolved arsenic concentrations on the order of 10 µg/L or less. These studies focused mostly on the effects of arsenic on organs and growth; however at least one study has shown that arsenic in fish diets can affect reproduction, although the single dietary exposure tested was higher (135 mg/kg dw) than in the studies mentioned with salmonids (Boyle *et al*. 2008).

Field studies of resident trout populations in streams influenced by natural geothermal drainage in Yellowstone National Park give indirect evidence of tolerance to elevated arsenic or perhaps density-dependent compensation to low-level toxicity. Goldstein *et al*. (2001) found that naturalized rainbow and brown trout were at least present in some streams with arsenic concentrations in water that were greatly above typical background concentrations. Arsenic was elevated both in water and invertebrates collected from the Snake River at the southern boundary of Yellowstone National Park (Table 2.4.3.2). Trout and sculpin densities at that location appeared robust in comparison to surveys at other least-disturbed rivers in Idaho and the Pacific Northwest (Maret *et al*. 1997; Mebane *et al*. 2003), so arsenic concentrations on the order of 30 µg/L in water and 11 mg/kg in insect tissues were causing no obvious harm to resident fish populations.

Most of the fish feeding and field studies reported total arsenic, without speciation analyses of whether the arsenic was in inorganic or organic forms. Recent evidence suggests that organic arsenic in the diet of salmonids is less toxic than inorganic arsenic (Table 2.4.3.1). Whether the arsenic that occurs in salmonid prey items in streams occurs predominately in inorganic or organic forms is unknown, but is assumed here to be primarily inorganic. Whether dissolved or particulate arsenic contributes more to arsenic risk is also debatable, but the present evidence suggests particulate arsenic may be more of a concern. The Idaho water quality criteria are based on dissolved arsenic, the rationale for which is unstated in EPA's criteria documents. Arsenic is a metalloid rather than a metal, but apparently for regulatory purposes, arsenic was simply considered another metal like cadmium or zinc without any known analysis. While the information is sparse, field data suggests that dissolved arsenic may be far less important as a source to aquatic food webs than particulate and sediment sorbed arsenic. This suggests that the dissolved arsenic criterion may be less relevant than a sediment, dietary, or tissue residue based criterion.

**Table 2.4.3.1. Relevant concentrations of arsenic in the diet of juvenile fish that were associated with adverse effects**

| Fish Species | Diet source | Effect | Arsenic in diet (mg/kg dw) | Reference |
|---|---|---|---|---|
| Cutthroat trout | Metals-contaminated invertebrates collected from the Coeur d'Alene R, ID | Reduced growth, liver damage | 14-51 | (Farag *et al*. 1999) |
| Cutthroat trout | " " " | None apparent | 2.6-3.5 | Farag *et al*. (1999) |
| Rainbow trout | Metals-contaminated invertebrates collected from the Clark Fork River, MT | Reduced growth, impaired digestion | 19 – 42 | Woodward *et al*. (1994,1995) |
| Rainbow trout | " " " | None apparent | 2.8-6.5 | Woodward *et al*. (1994,1995) |
| Rainbow trout | *Lumbriculus* (aquatic earthworms) contaminated using Clark Fork River sediments | Reduced growth, impaired digestion, liver and gall bladder degeneration | 21 | (Hansen *et al*. 2004) |
| Rainbow trout | Diet of *Lumbriculus* exposed to arsenic | Reduced growth | 34 | (Erickson *et al*. 2010) |
| Rainbow trout | Diet (pellets) amended with arsenate | Reduced growth, impaired digestion, gall bladder inflammation | 33 | (Cockell *et al*. 1991) |
| Rainbow trout, subadult | Diet (pellets) amended with arsenite | Reduced growth | ≥ 51 | (Hoff *et al*. 2011) |
| Rainbow trout | Diet (live or pellets) amended with inorganic arsenic (arsenite or arsenate) | Reduced growth | >≈ 20 mg/kg | (Erickson *et al*. 2011) |
| Rainbow trout | Diet (live or pellets) amended with organic arsenic | Reduced growth | >≈ 100 mg/kg | (Erickson *et al*. 2011) |
| Rainbow trout | " " " | None apparent | 13 | Cockell *et al*. (1991) |
| Lake Whitefish | Diet (pellets) amended with As | Liver and gall bladder damage, no effects on growth | ≥1 | (Pedlar *et al*. 2002) |

**Table 2.4.3.2. Relevant concentrations of arsenic in stream water, sediment, and in the tissues of aquatic invertebrates collected from the same streams. Selected undiluted mine effluent concentrations from within the action area are included for comparison. Unless otherwise noted, concentrations are averages, values in parentheses are ranges**

| Location and notes | Arsenic in water (filtered, µg/L) | Arsenic in water (unfiltered, µg/L) | Arsenic in sediment (mg/kg dw) | Arsenic in invertebrate tissues, average (mg/kg dw) |
|---|---|---|---|---|
| Effects thresholds (j) | | | 7-33 | ~ 20 |
| "Typical" USA river waters, not in enriched areas | | 0.1 – 2 (l) | | |
| Idaho rivers– statewide assessment (h) | | 2.3 (0.06 – 17) | | |
| Stream sediments, USGS national median | | | 6.3 (l) | |
| Panther Cr, ID, 1992-93, mining influenced reaches (a, f, i) | <1 | 102 (max) | 27-888 | 76 (f) |
| Blackbird Creek, ID (a) | 1.1 | 158 (max) | 939 | |
| South Fork Coeur d'Alene (b, c) | 0.4 – 4 | 13 (max) | 180 | 42 (d) |
| Clark Fork River at Galen, MT (b,d) | 15 (3-53) | 20 (4-80) | 170 (3) | 21(e) |
| Snake River leaving Yellowstone NP, WY (b,e) | 34 (8-55) | Nm | 38 | 11 (f) |
| Snake River at King Hill, ID (b,e) | 3 (0.5 – 7) | 4 (2-9) | 5 (4-7) | 1 (0.5 – 2) (f) |
| Hecla Grouse Creek gold mine, near Custer, Idaho (k) | 2.4 (<1-5) | 7 (<5 – 55) | | |
| Thompson Creek molybdenum mine, nr Clayton, Idaho (l) | 2 – 4 (projected max for new discharge was 30) | | | |

nm- not measured. (a) (Beltman *et al*. 1994; Maest *et al*. 1994; Beltman *et al*. 1999); (b) USGS Water-Quality Data for the Nation, *http://nwis.waterdata.usgs.gov/nwis/qw* ;(c) (Farag *et al*. 1998); (d) (Hansen *et al*. 2004); (e) (Ott 1997); (f) Community sample, (g) caddisfly *Hydropsyche* sp. (h) (Essig 2010) (i) (Mebane 2002a); (j) Effects thresholds for invertebrate residues are from this review; values for sediment are MacDonald *et al*.'s (2000a) threshold and probable effect concentrations. (k) R. Tridle, Hecla Mining Company, unpublished data, Jan 2008 (l) Thompson Creek mine "NPDES" wastewater permit factsheets, accessed January 2008 from *http://yosemite.epa.gov/r10/water.nsf* . (l) (Plant *et al*. 2007)

***Tissue concentrations of arsenic associated with chronic responses in fish.*** McIntyre and Linton (2011) report that regardless of exposure route or form, bioaccumulated fish tissue concentrations associated with chronic effects were remarkably similar among fish. Adverse effects appear likely to occur when whole-body tissue concentrations reach about 2 to 5 mg/kg wet weight (ww). The critical tissue residue concentrations in liver associated with reduced growth may be somewhat lower, around 0.7 to 1.0 mg/kg ww. This range of critical liver concentrations was supported by recent research reported by Hoff *et al*. (2011) who showed a change point in growth of rainbow trout when arsenic in liver reached about 6 mg/kg dw, which would be equivalent to about 1 to 1.5 mg/kg ww.

In a similar study in the Coeur d'Alene River basin, Idaho, Farag *et al*. (1999) fed fish invertebrates collected from mining influenced reaches and reported reduced growth, liver degeneration, and fish tissue concentrations ranging from about 0.5 to 1.2 mg/kg ww. In contrast, arsenic in fish fed a reference diet collected from a minimally polluted reach of the North Fork Coeur d'Alene River ranged from about 0.2 to 0.3 mg/kg ww (Farag *et al*. 1999). Other metals were also elevated in the fish, particularly lead, although results from Erickson *et al*. (2010) and Hansen *et al*. (2004) argue that most of the toxicity in Farag's study was probably attributable to arsenic.

Whole-body arsenic residues associated with reduced growth in fish following feeding studies (>≈ 0.6 mg/kg ww) are difficult to compare to surveys that only sampled edible fillets (muscle). In a probabilistic study of fish captured from 55 randomly selected river sites throughout Idaho, Essig (2010) obtained a median arsenic concentration of 0.06 mg/kg ww, ranging from <0.13 to 0.31 mg/kg ww in muscle fillets. The highest value in Essig's (2010) report was from a brown trout collected from a geothermally influenced reach of the Portneuf River. In targeted collections of trout in the Stibnite Mine area, arsenic concentration in fillets were up to 0.96 mg/kg, fresh weight), considerably higher than the maximum value from Essig's (2010) randomized survey. In the Stibnite study, arsenic in muscle fillets was considerably lower than in the remaining trout carcasses (e.g., organs, bone, viscera, skin) after the fillets had been removed. Arsenic in fillets ranged from <0.25 to 0.96 mg/kg fresh weight versus 0.32 to 6.3 mg/kg fresh weight in the remainders (Woodward-Clyde 2000).

***Behavioral and neurotoxic effects.*** Despite profound neurotoxic effects of arsenic in mammals, there appears to have been minimal research with behavioral and neurotoxic effects of arsenic in fish. However, the available information reviewed suggests that behavioral effects could be important at very low exposure concentrations. Arsenic impaired long-term memory in zebrafish exposed for 96 hours to arsenic concentrations as low as 1 µg/L before avoidance trials. Measurement of elevated levels of oxidized proteins in brain tissue of fish exposed to 10 µg/L arsenic suggested that the observed effects may have been related to oxidative stress in brain tissue (McIntyre and Linton 2011).

The information reviewed indicates that at environmentally relevant concentrations, arsenic in the diets of salmonids poses significant risks for reduced growth. Reduced growth in turn, may lead to reduced survival or reproduction.

#### *2.4.3.2. Habitat Effects of Arsenic Criteria*

***Toxicity to Food Organisms.*** The limited data available suggests that the risk of toxicity to salmonid food organisms is lower than the risk of toxicity to salmonids from eating arsenic exposed organisms. However, we did not locate any studies that tested invertebrates using environmentally relevant exposures through arsenic enriched periphyton or sediments, and conducted through full life exposures or obviously sensitive life stages.

Norwood *et al*. (2007) related bioaccumulation of arsenic in *Hyalella azteca*, a benthic invertebrate common in slow moving rivers and lakes, to mortality in 4-week exposures. Lethal body concentrations associated with 25% and 50% mortality were about 9 and 10 mg/kg dw respectively. *Hyalella* exposed to Panther Creek, Idaho sediments for 10 days had a trend of decreasing growth and survival with increasing arsenic concentrations (Mebane 1994, 2002a). However, arsenic in Panther Creek sediments was also correlated with cobalt and copper, and correlations between decreased *Hyalella* survival and cobalt and copper concentrations in sediments were stronger than for arsenic, and thus adverse effects were attributed to copper and or cobalt (Mebane 1994, 2002a). However, arsenic bioaccumulation in *Hyalella* probably takes more than 10 days to reach saturation (Norwood *et al*. 2006) and in general, 10-day Hyalella tests can be considerably less sensitive than 4 to 7 week tests (Ingersoll *et al*. 1998). Thus, the Panther Creek study may not have had the necessary duration for detecting effects of arsenic-contaminated sediments.

Irving *et al*. (2008) exposed mayfly nymphs to tri- and pentavalent arsenic in water-only exposures for 12 days. For trivalent arsenic, the threshold of growth effects was about 100 µg/L. However, arsenic levels accumulated by the mayfly nymphs in their study (1.2 to 4.6 µg/g dw) were far lower than those reported from stream locations with far lower water concentrations of arsenic but that had elevated arsenic in diet or sediments, suggesting that the water-only exposures may have underrepresented likely environmental exposures. Crayfish collected from Australian streams disturbed by mining activities had up to 100 mg/kg dw arsenic in their tissues. Levels of arsenic in the tissues of the crayfish were similar to those found in the sediment, thus it is highly likely that the primary exposure to arsenic for the crayfish came from the sediment (Williams *et al*. 2008).

Other data we reviewed on arsenic toxicity to aquatic macroinvertebrates were from water only exposures that are unlikely to have much relevance to toxicity under environmental conditions (EPA 1985a; Eisler 1988a; Canivet *et al*. 2001). Results reported in Eisler (1988a) suggest that gammarid amphipods may experience acute toxicity at concentrations of trivalent arsenic that are below the chronic criterion. Canivet *et al*. (2001) similarly found increased mortality of gammarid amphipods and heptagennid mayflyies at about 100 µg/L which is lower than the chronic criterion of 190 µg/L.

#### *2.4.3.3. Summary of Effects for Arsenic*

If only direct water exposures were considered, arsenic would be of minimal concern to listed salmonids at typical ambient concentrations or at the criteria concentrations under review. The risk of harm from short-term water-only exposures to arsenic concentrations at the acute criterion is unlikely enough to be considered a minor risk for short-term exposures.

The chronic criterion appears to avoid chronic adverse effects to the adult and juvenile salmonid life stages from water-only exposures; however, arsenic concentrations below the chronic criterion have been reported to cause mortality in salmonid embryos. The chronic arsenic criterion is far higher than concentrations of arsenic sufficient to bioaccumulate in invertebrates to concentrations that cause harm to the salmonids that feed on them. Bioaccumulation of arsenic in prey organisms to concentrations that could be harmful to salmonids has occurred in streams at exposures less than 10 µg/L. As such, adverse effects can occur at the chronic criterion, through reduced growth of juveniles via food web transfer.

### 2.4.4. The Effects of EPA Approval of the Copper Criteria

Copper toxicity is influenced by chemical speciation, hardness, pH, alkalinity, total and dissolved organic content in the water, previous exposure and acclimation, fish species and life stage, water temperature, and presence of other metals and organic compounds that may interfere with or increase copper toxicity. Adverse effects of copper to salmonids that have been documented at environmentally relevant concentrations include reduced growth and reproductive impairment. A host of initially sublethal physiological and behavioral effects to salmonids have been documented following copper exposures including interference with immune response and reduced disease resistance, reduced swimming stamina, damage to olfactory cellular tissue, impaired olfactory function, which in turn impairs ability of fish to avoid predators, find prey, and migrate from and to their natal streams. Benthic macroinvertebrate communities that form the food base of salmonids in freshwater streams appear particularly sensitive to copper, compared to other metals. The Idaho copper criteria under review in this Opinion are hardness dependent. At a hardness of 100 mg/L the acute criteria for copper is 17 µg/L and the chronic criteria for copper is 11 µg/L .

#### *2.4.4.1. Species Effects of Copper Criteria*

***Acute toxicity.*** Available toxicity test data indicate that, under certain conditions, juvenile salmonids can be killed by copper concentrations equal to the final acute value (FAV) used to define the acute criterion. Because acute toxicity data are commonly reported only as the concentrations lethal to 50% of the test population ($LC_{50}$s), and because 50% test population is a severe effect, the protectiveness of acute criterion is not evaluated by comparing it directly to $LC_{50}$ data. Rather, $LC_{50}$ data are compared to the FAV, which is equal to 2X the acute criterion. The assumption in the criteria derivation and in this opinion is that dividing an $LC_{50}$ value by 2 will result in a concentration that kills few if any organisms. This assumption was critically reviewed in Section 2.4.1.6 and in Appendix B. In this manner, the acute criterion, which is intended to protect against short-term exposures in the environment is compared to short-term $LC_{50}$ toxicity data. Because the chronic criterion only comes into play for exposure scenarios longer than 96-hours, the acute criterion regulates allowable concentrations from >1-to-96 hours.

The studies reviewed indicate that $LC_{50}$s for adult listed salmon and steelhead are slightly higher than the proposed criterion that is the FAV divided by two. This is consistent with older summaries that found $LC_{50}$ values for adult salmon and trout were well above the proposed acute criterion (EPA 1985d; Eisler 1998a). Figure 2.4.4.1 shows all acute data NMFS reviewed, for tests in waters with hardness less than 200 mg/L, irrespective of lifestage. (We consider waters with hardness of less than 200 mg/L more representative of waters in the action area.) Although most of the $LC_{50}$ values are higher than the FAV, a substantial minority are lower. Many of the tests for which the FAV would not be protective fall in two general categories: test waters with low hardness; and waters in which magnesium contributes much of the measured hardness values, that is Ca:Mg ratios are lower than in most of the tests used to develop criteria (Welsh *et al*. 2000a; Naddy *et al*. 2002). However, others appear to capture sensitive life stages or stock. For instance, Chinook salmon exposed to copper in pH 7.7 at hardness 35 mg/L resulted in an $LC_{50}$ of 7.4, which is lower than the hardness adjusted FAV of 13 µg/L. Rainbow trout tested in hardness 25 mg/L at pH 6 yielded a $LC_{50}$ of 2.4 µg/L which is less than the FAV of 9.2 µg/L at hardness 25 mg/L (Fig. 2.4.4.1, data from Stratus (1996;1998).

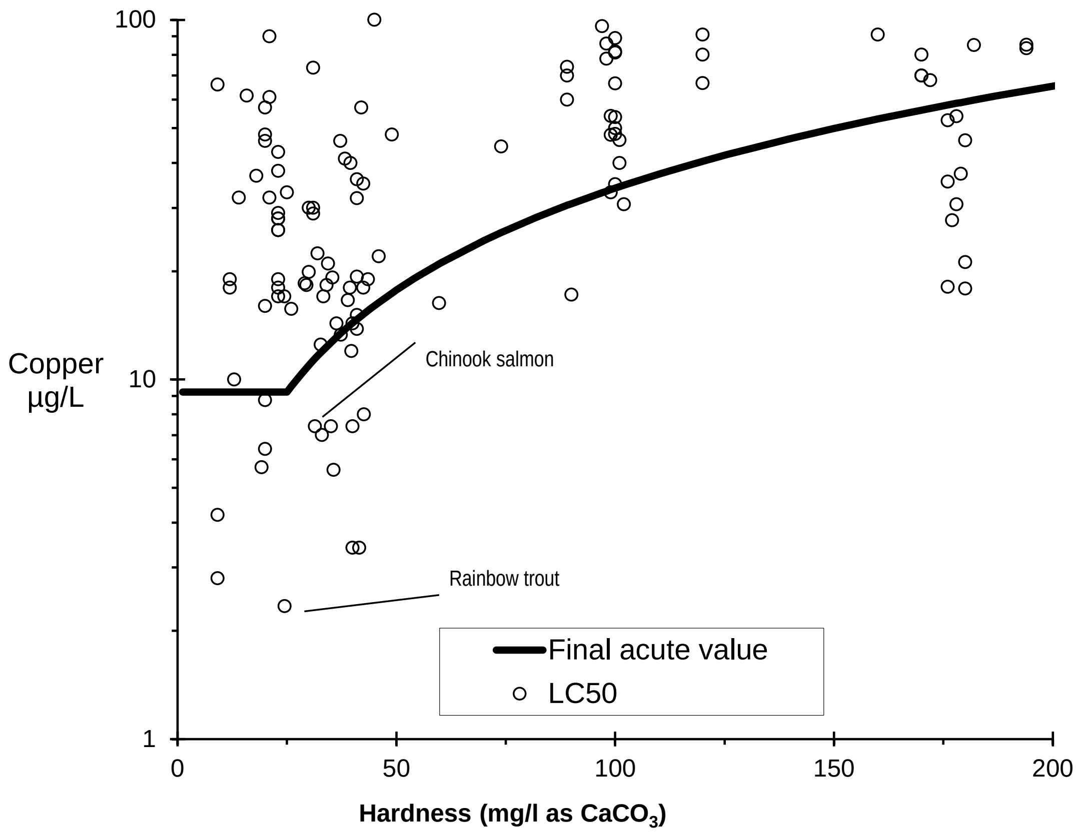



**Figure 2.4.4.1. Comparison of 96-hour $LC_{50}$s for salmonids with copper and the Idaho criterion final acute values, calculated for hardnesses up to 200 mg/L as CaC03. $LC_{50}$s limited to species within the genera Oncorhynchus, Salvelinus, and Salmo. If all $LC_{50}$ values fell above the line, that would suggest that for the most part, few mortalities would be likely at criterion concentrations.**

***Chronic Toxicity***. Numerous adverse effects have been reported that were attributable to long-term exposures of salmonids and other fish to copper. "Chronic effects" as used here refer to effects resulting from long-term exposures, and effects from such long-term exposures can include mortality or sublethal effects.

The most sensitive endpoint in some chronic tests with copper and fish was reproductive impairment, as reduced fecundity (Mount 1968; Mount and Stephan 1969; McKim and Benoit 1971; Suter *et al*. 1987). However, with anadromous steelhead and salmon, presumably long-term exposure of adults to copper in freshwater would be unlikely, since adults are either only passing through migratory areas or are exposed on their spawning grounds for a few weeks or less. Thus, the risk of chronic effects from copper is higher for juvenile fish.

Reduced immune response and disease resistance is an effect of copper that appears to be understudied, considering its potential implications. Stevens (1977) reported that pre-exposure to sublethal levels of copper interfered with the immune response and reduced the disease resistance in yearling coho salmon.

Other chronic effects include damage to olfactory tissues, reduced swimming speed, and reduced growth (Table 2.4.4.1).

***Growth effects and population-level risks***. Comparisons of available chronic copper effects data with salmonids and the Idaho chronic criteria were unfavorable to the criteria. In contrast to the acute $LC_{50}$ data for salmonids with copper where at least most values were higher than the Idaho final acute value, with the Idaho chronic criterion about as many adverse effects were documented to occur at or below the criterion concentrations as above (Figure 2.4.4.2). Relevant studies are described in more detail in Table 2.4.4.1.

A common chronic effect observed with copper exposure has been reduced growth in laboratory toxicity tests with salmonids. In tests in soft water, copper concentrations CCC caused about a 4% to 7.5% reduction in the lengths of Chinook salmon and rainbow trout, depending on the statistical model used to analyze the toxicity data (Table 2.4.4.1). However, the relevance of subtle and sometimes transitory growth reductions under laboratory conditions to natural-origin populations may not be obvious. One study used population modeling to estimate the relevance of subtle and sometimes transitory growth reductions under laboratory conditions to natural-origin populations (Mebane and Arthaud 2010). Demographic data from Marsh Creek, Idaho, was used as a “model” headwaters population of Snake River spring/summer Chinook salmon to develop the population model (Mebane and Arthaud).

The size of juvenile salmon as they first migrate from Marsh Creek is a strong predictor of their survival during the initial part of their seaward migration. Growth reductions in laboratory tests were extrapolated to reduced survival in the wild through the size-survival correelations of migrating juvenile fish. Reductions in growth predict disproportionate reductions in survival of migrating juveniles. For average sized migrants, a 4% to 7.5% length reduction predicts about a 14% to 26% reduction in survival from Marsh Creek to the LGD, the next downstream census point, 640 km downstream. The study used these changes in juvenile survival rates to adjust the life stage survival rates in the population model, to estimate the population-level consequences of low-level copper stress on juvenile Chinook salmon.

The study projected population-level risks for up to six generations (30 years). Risks of severe decline or quasi-extinction were slightly higher under the copper-influenced scenarios, compared to baseline risks with no copper. Severe declines or quasi-extinction were defined as a 90% reduction of adult spawners or five-consecutive runs with less than 25 spawners each year respectively. Risks of "quasi-extinction" rather than absolute extinction were projected because of biological and mathematical difficulties reaching true zero in the population model. Risks of severe decline occurring in a single spawning run over a 30-year projection were about 75% for the baseline scenario, and 76% to 79% for the copper CCC scenarios. Quasi-extinction risk projections for the same time period averaged 23% for the baseline scenario and 26% to 31% for the copper CCC scenarios (Mebane and Arthaud 2010).

Projections of population recovery times differed more between the scenarios than did the risks of decline. The baseline scenario was projected to meet a relative recovery threshold of 500 adults in about 11 years, and the 4% to 7.5% copper growth reduction scenarios were projected to meet the recovery threshold in the 18 to 28 years (Mebane and Arthaud 2010). The model results mentioned here all assumed density dependence, that is, the population cannot increase above an assumed carrying capacity). While the modeling used a real population to increase realism, all of these risks and population projections should be interpreted in a relative sense in comparison between the scenarios, not as absolute predictions.

***Chemosensory and Behavioral Effects.*** Sensory system effects are generally among the more sensitive fish responses and underlie important behaviors involved in growth, reproduction, and (ultimately) survival (i.e., predator avoidance). Recent experiments on the sensory systems and corresponding behavior of juvenile salmonids contribute to more than 4 decades of research and show that dissolved copper is a neurotoxicant that directly damages the sensory capabilities of salmonids at low concentrations. (Hecht *et al.* 2007). These effects can manifest over a period of minutes to hours and can persist for weeks. To estimate toxicological effect thresholds for dissolved copper in surface waters, Hecht *et al.* (2007) calculated benchmark concentrations (BMCs) for juvenile salmonid olfactory function based on recent data. The BMCs ranged from increases of 0.18 to 2.1 µg/L above background copper concentrations, corresponding to reductions in predator avoidance behavior of approximately 8% to 57%. The BMC examples represent the increases in dissolved copper concentration above background copper concentrations, which were up to 3 µg/L in the tests used to derive the BMCs. These levels are expected to affect the ability of juvenile salmonids to avoid predators in freshwater. These BMCs are much less than the corresponding acute Idaho criteria of 20 µg/L, and even the chronic criteria of 13 µg/L (for a hardness of 120 mg/L for the conditions of a test that was used in the derivation of the BMC, Table 2.4.4.1). These BMCs thresholds for juvenile salmonid sensory and behavioral responses fall within the range of other low sublethal endpoints affected by dissolved copper such as behavior, growth, and primary production, which is around 0.75–2.5 µg/L (Hecht *et al*. 2007).

Studies showing diminished predator avoidance behaviors of juvenile salmon in the presence of elevated copper have subsequently been expanded through predation experiments (McIntyre 2012). Short-term (30 min) copper exposure made prey easier for predators to detect and capture. The primary impact of copper on predator-prey dynamics in her study was faster prey detection, manifested as faster time to attack and time to capture. Cutthroat trout were more effective predators on copper-exposed coho during predation trials, as measured by attack latency, survival time, and capture success rate. The shift in predator-prey dynamics was similar when predators and prey were co-exposed to copper. The onset of these effects occurred at concentrations less than the acute criterion for copper: predatory cutthroat trout captured and ate juvenile coho salmon that had been exposed to 4.5 µg/L copper in only about 1/3 of the time needed to capture and eat coho that had not been exposed to copper (McIntyre 2012). For the water hardness of the test chambers, 56 mg/L, the acute criterion was 10 µg/L.

***Hardness and Other Parameters as Predictors of Copper Toxicity.*** A number of water quality characteristics influence the toxicity of copper. A conclusion that generally seems to hold across most data and studies we reviewed is that in laboratory waters that have low and uniform DOC present, increasing hardness will usually result in alkalinity and pH naturally increasing as well. In this case, decreasing acute copper toxicity will be expected. However, this pattern may not be consistent for chronic copper toxicity in similar laboratory waters, and it most certainly does not hold for natural waters that have variable DOC and pH.

Chakoumakos *et al*. (1979) determined that hardness and alkalinity influenced the $LC_{50}$ of copper to cutthroat trout, whereas pH had greater influence on the speciation of copper involved in toxicity. They recommended that water quality criteria for copper include all three parameters: hardness, alkalinity, and pH. Miller and Mackay (1980) determined that the incipient lethal concentration of copper varied more rapidly with changes in alkalinity in moderately hard (98 mg/L) water than in soft (12 mg/L) water. Conversely, Lauren and McDonald (1986) varied pH, alkalinity, and hardness independently and determined that alkalinity was an important factor reducing copper toxicity to juvenile rainbow trout with no significant influence of increasing hardness. Lauren and MacDonald (1986) argued that the degree of acclimation to ambient hardness levels could explain the difference in results. Meador (1991) found that both pH and DOC were important in controlling copper toxicity to *Daphnia magna*. Welsh *et al*. (1993) evaluated the importance of DOC in affecting the toxicity of copper to fathead minnows and suggested that water quality criteria be reviewed to consider the toxicity of copper in waters of low alkalinity, moderately acidic pH, and low DOC concentrations. Applications of gill models to copper binding also consider complexation by DOC, speciation and competitive effects of pH, and competition by calcium ions. Welsh *et al*. (1993) varied several test water qualities independently and found that pH, hardness, sodium, DOC, and suspended solids have important roles in determining copper toxicity. They also suggested that it may be difficult to sort out the effects of hardness based on simple toxicity experiments.

The data NMFS reviewed also suggested that increasing hardness affords more protection for acute copper exposures than for chronic. Hansen *et al*. (2002b) found a clear relationship between ACRs and water hardness, with lower ACRs at higher hardness levels. Similarly with acute and chronic exposures of copper to *Daphnia magna*, Chapman *et al*. (1980) found that increasing hardness from about 50 to 200 mg/L consistently increased the acute resistance of

*Daphnia* to copper, but with chronic exposures, resistance only increased with increasing hardness from 50 to 100 mg/L; increasing hardness from 100 to 200 mg/L provided no additional resistance to copper. These results have disturbing implications for a chronic copper criterion because they contradict a fundamental assumption in the criteria derivation (EPA 1985d) that that chronic toxicity is similarly modified by water hardness as acute criteria, and the chronic criterion varies with hardness as a fixed proportion of the acute criteria.

Tests that used natural waters or approximated natural waters by varying DOC along with hardness and other parameters have repeatedly found that hardness is a minor influence on the toxicity of copper to aquatic invertebrates and fish (Appendix C; Hyne *et al*. 2005; Markich *et al*. 2005; Wang *et al*. 2009). The results of these studies indicate that the use of site calcium plus magnesium hardness only as input to an equation to derive a criterion for copper may not be sufficiently protective of listed salmon and steelhead, and that the criteria need to also consider the influences of DOC and pH as key water quality variables that are more important for modulating toxicity. This issue is described in more detail in the Section 2.4.2, "*The Influence of Hardness on Metals Toxicity*" and Appendix C.

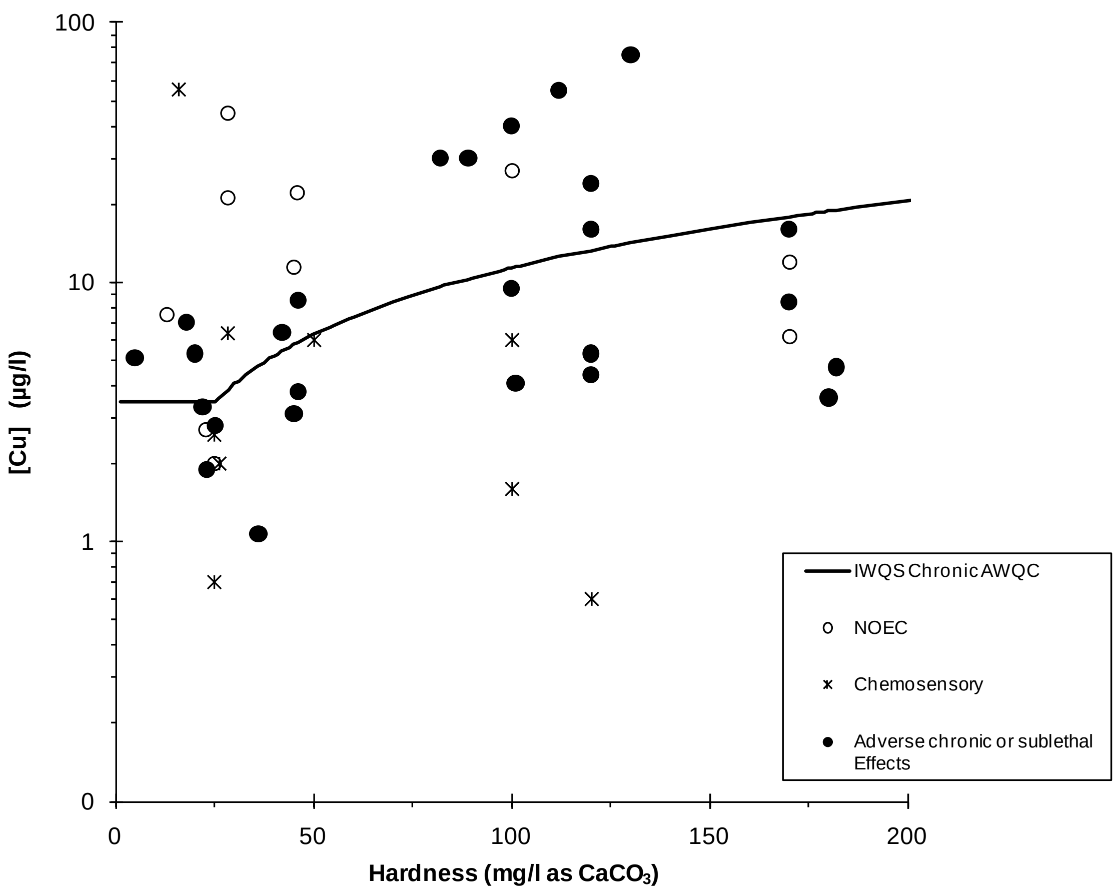



**Figure 2.4.4.2. Comparison of the copper Idahochronic criterion and adverse chronic or sublethal effects and estimates of no-effect concentrations to salmonids.**

**Table 2.4.4.1. Relevant effects and risk ratios of copper to salmonids or other ecosystem components, emphasizing effects that occurred at lower concentrations than the relevant Idaho criteria. Long-term effects (> 4 days to occur) are compared to the chronic criterion, short-term sublethal effects to the Idaho acute criteria, or for acute $LC_{50}$s, the Idaho final acute value. Risk ratios greater than 1.0 are considered harmful.**

| Species | Effect | Exposure duration | Hardness (mg/L) | Effect statistic | Effect concen-tration (µg/L) | Criterion (µg/L) | Risk ratio (r=NTR/ effect) concen-tration | Source/ Notes |
|---|---|---|---|---|---|---|---|---|
| | **Sublethal effects** | | | | | | | |
| Coho salmon (juvenile) | Reduced olfaction and compromised alarm response | 3 hours | 120 | EC10 - EC50 | 0.18 to 2.1 | 20.2 | 112 to 9.6 | 1 |
| Coho salmon (juvenile) | Reduced olfaction and compromised alarm response | 3 hours | 120 | ~EC25 | 0.6 | 20.2 | 34 | 1 |
| Coho salmon (juvenile) | Shorter time to get captured and eaten | 3 hours | 56 | ~EC50 | 5 | 10 | 2 | (McIntyre 2012) |
| Chinook salmon (juvenile) | Avoidance in laboratory exposures | 20 minutes | 25 | LOEC | 0.75 | 4.6 | 6.1 | 2 |
| Rainbow trout (juvenile) | Avoidance in laboratory exposures | 20 minutes | 25 | LOEC | 1.6 | 4.6 | 2.9 | 2 |
| Chinook salmon (juvenile) | Loss of avoidance ability | 21 days | 25 | LOEC | 2 | 3.5 | 1.7 | 2 |
| Atlantic salmon (juvenile) | Avoidance in laboratory exposures | 10 minutes | 20 | LOEC | 2 | 4.6 | 2.3 | 3 |
| Coho salmon | Delays and reduced downstream migration of copper exposed juveniles | 6 day | 95 | LOEC | 5 | 10.9 | 2.2 | 4 |
| Chinook salmon | Reduced growth (as weight) | 120 days | 25 | EC10 | 1.9 | 3.5 | 1.8 | 5 |
| Rainbow trout | Reduced growth (as weight) | 60 days | 25 | EC10 | 2.8 | 3.5 | 1.2 | 6 |
| Rainbow trout | Reduced growth (as weight gain) | 56 days | 102 | EC10 | 4.1 | 11.5 | 2.8 | (Hansen *et al*. 2002b) |
| Rainbow trout | Reduced critical swimming speed, pH 6 | 30 days | 30 | EC10 | 5 | 4.1 | 0.8 | (Waiwood and Beamish 1978) |
| Rainbow trout | Reduced growth rate, pH 7.5 | 30 days | 30 | EC25 | 6 | 4.1 | 0.8 | (Waiwood and Beamish 1978) |

| Species | Effect | Exposur e duration | Hardness (mg/L) | Effect statistic | Effect concen-tration (µg/L) | Criterion (µg/L) | Risk ratio (r=NTR/ effect) concen-tration | Source/ Notes |
|---|---|---|---|---|---|---|---|---|
| Rainbow trout | Reduced growth rate, pH 6 | 30 days | 30 | EC25 | 2 | 4.1 | 2 | (Waiwood and Beamish 1978) |
| Brook trout | Delayed growth (as weight) | 23 weeks | 45 | EC10 | 3.1 | 5.7 | 1.9 | 7 |
| Brook trout | Reduced growth (as weight) | 3 months | 45 | EC10 | 8.5 | 5.7 | 0.7 | 8 |
| Brook trout | Slight mortality | 3 months | 45 | EC10 | 17 | 5.7 | 0.3 | 7 |
| Brook trout | Complete mortality | 22-months | 45 | EC100 | 17 | 5.7 | 0.3 | 7 |
| Brook trout | Reduced growth (as weight) | 60 days | 37 | EC10 | 1.1 | 4.9 | 4.4 | 9 |
| Brook trout | Reduced growth (as weight) | 60 days | 187 | MATC | 6.3 | 19.3 | 3.1 | 9 |
| Brook trout | Reduced growth (as weight) | 60 days | 181 | EC10 | 4.8 | 18.1 | 4 | (Besser *et al*. 2001a) |
| | **Habitat effects: Adverse effects to ecosystem components** | | | | | | | |
| Ecosystem function | Reduced photosynthesis | ~ 1 year | 49 | LOEC | 2.5 | 6.2 | 2.5 | 10 |
| Ecosystem structure | Loss of invertebrate taxa richness in a mountain stream | ~ 1 year | 49 | LOEC | 5 | 6.2 | 1.2 | 11 |
| Macroin-vertebrate community | abundance (total individuals) | 10-d | 60 | EC50 | 6 | 7.3 | 1.2 | 12 |
| Snail, *Leptoxis praerosa* | 80% mortality in *in situ* river exposures | 114-d | 136 | LOEC | 6.3 | 14.8 | 2.3 | 13 |
| Idaho springsnail | 25% mortality | 28-d | 170 | EC25 | 11 | 17.9 | 1.6 | 15 |
| Bliss Rapids snail | 25% mortality | 28-d | 170 | EC25 | 14 | 17.9 | 1.3 | 15 |
| Snake River pebblesnail | 25% mortality | 28-d | 170 | EC25 | 10 | 17.9 | 1.8 | 15 |

| Species | Effect | Exposure duration | Hardness (mg/L) | Effect statistic | Effect concen-tration (µg/L) | Criterion (µg/L) | Risk ratio (r=NTR/ effect) concen-tration | Source/ Notes |
|---|---|---|---|---|---|---|---|---|
| Sculpin, *Cottus bairdi* (MO) | 97% mortality | 28-d | 100 | LOEC | 7.8 | 11.4 | 1.5 | 14 |
| Sculpin, *Cottus bairdi* (MO) | No mortality or growth effects | 28-d | 100 | NOEC | 30 | 11.4 | 0.4 | 14 |
| | **Acute Lethality** | | | | | | | |
| Steelhead/ Rainbow trout (fry) | Death (pH 7) | 96 h | 9.2 | $LC_{50}$ | 2.8 | 4.6 | 3.3 | 16 |
| Steelhead/ Rainbow trout (fry) | Death (pH 5.7) | 96 h | 9.2 | $LC_{50}$ | 4.2 | 4.6 | 2.2 | 16 |

**Table notes (data sources):** 1. (Hecht *et al*. 2007; Sandahl *et al*. 2007); 2. (Hansen *et al*. 1999); 3. (Sprague *et al*. 1965); 4. (Lorz and McPherson 1976, 1977); 5. (Chapman 1982); 6. (Marr *et al*. 1996) ; 7. (McKim and Benoit 1971); 8. (McKim and Benoit 1974); 9. (Sauter *et al*. 1976); 10. (Leland and Carter 1985), 11. (Leland *et al*. 1989); 12. (Clements *et al*. 1989); 13. (Reed-Judkins *et al*. 1997); 14. (Besser *et al*. 2009); 15. (Besser *et al*. 2007); 16. (Cusimano *et al*. 1986)

### *2.4.4.2. Habitat Effects of Copper Criteria*

***Toxicity to Food Organisms***. Copper is highly toxic to many freshwater invertebrates (Kiffney and Clements 2002; Mebane 2002a). Aquatic macroinvertebrates are sensitive to both dissolved and particulate copper, and some taxa can be more sensitive than salmonids (e.g., Kemble *et al*. 1994). Data in EPA (1985d) list relatively high $LC_{50}$s, which would apparently indicate that the proposed criteria are usually protective of invertebrates that juvenile salmon and steelhead feed on. However, compilations of short-term $LC_{50}$s tend to do a poor job of reflecting the sensitivities of metals to invertebrates in field conditions. The compilations indicate that stream invertebrates are not very sensitive to metals, but effects observed in field surveys tend to indicate that stream invertebrates are very sensitive to copper stress (Buchwalter *et al*. 2007). For these reasons, we consider field surveys more relevant indicators of metals effects than acute toxicity testing.

At concentrations less than or near the Idaho chronic criterion, elevated copper in water can adversely affect invertebrate communities that salmonids rely on for food (Table 2.4.4.1; Figure 2.4.4.3). Invertebrate communities in rivers also may be sensitive to elevated copper levels in the sediments. Most commonly, the reported effects to the invertebrate community are changed composition to pollution-tolerant taxa, rather than by reducing overall abundance (Canfield *et al*. 1994; Clements and Kiffney 1994; Beltman *et al*. 1999; Mebane 2002a). However, this might reflect sampling bias, because most invertebrate surveys reviewed were made in the summer. When invertebrates were collected in spring and autumn 1992 in Panther Creek, Idaho, a salmon stream contaminated by copper well in excess of the Idaho chronic criteria, total biomass was much lower in the copper-influenced areas. A possible explanation for seasonally low biomass is that when the diversity was lower, and then a dominant, pollution tolerant insect taxa hatched and left the stream, the remaining biomass was lower than in unaffected areas with more diverse communities (Mebane 1994). Seasonal differences in copper effects have also been observed in invertebrates in pond communities, where effects of copper were more severe in cold, springtime conditions (6°C to 9.5°C) than in warmer summer (23°C to 28°C) or fall (15°C to 9.5°C) conditions (Winner *et al*. 1990).

Panther Creek, Idaho, has been the subject of detailed analyses of benthic macroinvertebrate communities and copper (among many other analyses). It is emphasized because prior to becoming polluted by copper in the 1960s, Panther Creek supported major runs of Chinook salmon and steelhead. The loss of habitat in Panther Creek resulting from water quality degradation from the Blackbird Mine was specifically cited as a contributing factor leading to the decline of the Snake River spring/summer Chinook salmon species (NMFS 1991). Prior to the mid-1990s, measured copper concentrations in Panther Creek were always well in excess of proposed criteria, so associated biological effects are not directly relevant to the question of whether adverse effects would be expected at criteria concentrations. Since then, restoration efforts have led to pronounced reductions in copper contamination to the point that the Idaho chronic criterion is mostly met. Thus, recent conditions in Panther Creek field surveys are very relevant to the present review because it offers a real-world view of biological conditions in a stream with copper present at close to the criteria concentrations under review.

Metrics calculated for benthic macroinvertebrates from Panther Creek in September 2005 and 2006 are shown in relation to the mean Idaho copper chronic criterion exceedence factors (Figure 2.4.4.3). An exceedence factor is the measured copper concentration at a location divided by the criterion for that sampling effect. The exceedence factors were calculated from chemical sampling from March to September of the year shown. Three measures of the macroinvertebrate community that seemed particularly relevant to their role in the food web of listed salmonids were examined: (1) Stream macroinvertebrate index (SMI) scores; (2) mayfly abundance; and (3) the abundance of organisms that were considered vulnerable to predation by salmonids. The SMI is an additive index comprised of nine measures of community diversity, dominance, or presence of pollution sensitive or intolerant species. It was derived as a measure of similarity or dissimilarity of macroinvertebrates to minimally disturbed reference conditions in the different ecological regions of Idaho (Jessup and Gerritsen 2002). The SMI and its component metrics relates to overall biological condition of stream ecosystems.

Abundance of mayflies was considered separately because mayflies have repeatedly been found to be important in the diets of juvenile salmonids in streams (Sagar and Glova 1987, 1988; Mullan *et al*. 1992; Clements and Rees 1997; Rader 1997; White and Harvey 2007; Syrjänen *et al*. 2011). Because mayflies are often also sensitive to copper, their loss in a stream food web could require shifting to other food items that are less preferred by salmonids. The third metric, abundance of taxa that are vulnerable to predation by juvenile salmonids, is broader than just mayflies. This metric was derived by assigning all organisms collected in the stream samples to one of three broad functional groups (i.e., burrowing, armored, and vulnerable to predation) based on life history traits influencing availability to steelhead fry (Suttle *et al*. 2004).

The comparisons of these metrics with copper exceedence factors in Panther Creek shows that even when copper concentrations were generally lower than the Idaho chronic criteria, the concentration gradient was still correlated with effects on the macroinvertebrate community (Figure 2.4.4.3). If copper only adversely affected macroinvertebrate communities at concentrations above the criteria, no correlation would be expected between copper and the macroinvertebrate metrics across a gradient of sub-criterion levels. The macroinvertebrate-copper exceedence patterns varied between years. In 2005, increasing copper concentrations were correlated with declining SMI scores (Figure 2.4.4.3). In 2005, relations between copper exceedence factors and mayfly abundance or vulnerable prey abundance were weak or nonexistent. In 2006, the pattern was reversed (Figure 2.4.4.3).

Together these comparisons show that relatively low levels of copper apparently affect macroinvertebrate communities, but that relations are more complex than can fully be explained in these simple correlations. For example, the copper gradient in Panther Creek tended to increase upstream to downstream along with temperatures that increased as the elevation dropped. The temperature gradient did not explain the macroinvertebrate patterns as well as the copper gradient; still it is an example of why patterns in field studies may be "noiser" than field or laboratory experiments. The changes in the stream macroinvertebrate communities did not obviously extend to adverse effects to the salmonid fishes, which are of most interest in this evaluation. There were no obvious decreases in various field measures of the salmonid populations at the sites with low-copper influence compared with upstream reference sites (e.g., overall abundances, age-class strength, condition factors of salmonids) (EcoMetrix 2006, 2007).

Sediments with elevated copper that were collected from Chinook salmon and steelhead habitat in Panther Creek, Idaho and tested in a laboratory setting with clean overlying water caused high mortality to *Hyalella azteca*, a freshwater benthic crustacean (Mebane 2002a). The resident benthic invertebrates collected from the same locations as the copper-contaminated sediments had reduced diversity compared to reference collections. Unlike the sediment toxicity tests, adverse effects to the instream invertebrates could not be attributed solely to either copper in the sediments or in water, because copper was elevated in both (Mebane 2002a). Elevated copper in sediments is also associated with elevated copper in benthic invertebrate tissues in field studies conducted in metals-contaminated streams (e.g., Ingersoll *et al*. 1994; Woodward *et al*. 1994; Beltman *et al*. 1999; Besser *et al*. 2001b). Uptake and toxicity of copper by invertebrates is strongly influenced by the amount of acid-volatile sulfide in the sediments or by the amount of organic carbon in the sediments (Besser *et al*. 1995; Mebane 2002a).

In summary, habitat effects of elevated copper levels to listed salmon and steelhead include reductions in preferred invertebrate taxa that have been shown to influence the seasonal availability of food for juvenile salmonids. These reductions have been observed even with relatively low concentrations near the Idaho chronic criteria. Logically, reductions or changes in prey availability could translate to adverse effects on juvenile salmonid populations. However, in the Panther Creek field studies that we reviewed in some detail, no obvious extensions of macroinvertebrate effects to the salmonid fishes were observed. This suggests either or both that juvenile salmonids are able to switch prey when preferred prey are diminished, or that the food web effects were too subtle to tease out of the natural variability inherent in field monitoring studies without going to extraordinary means.

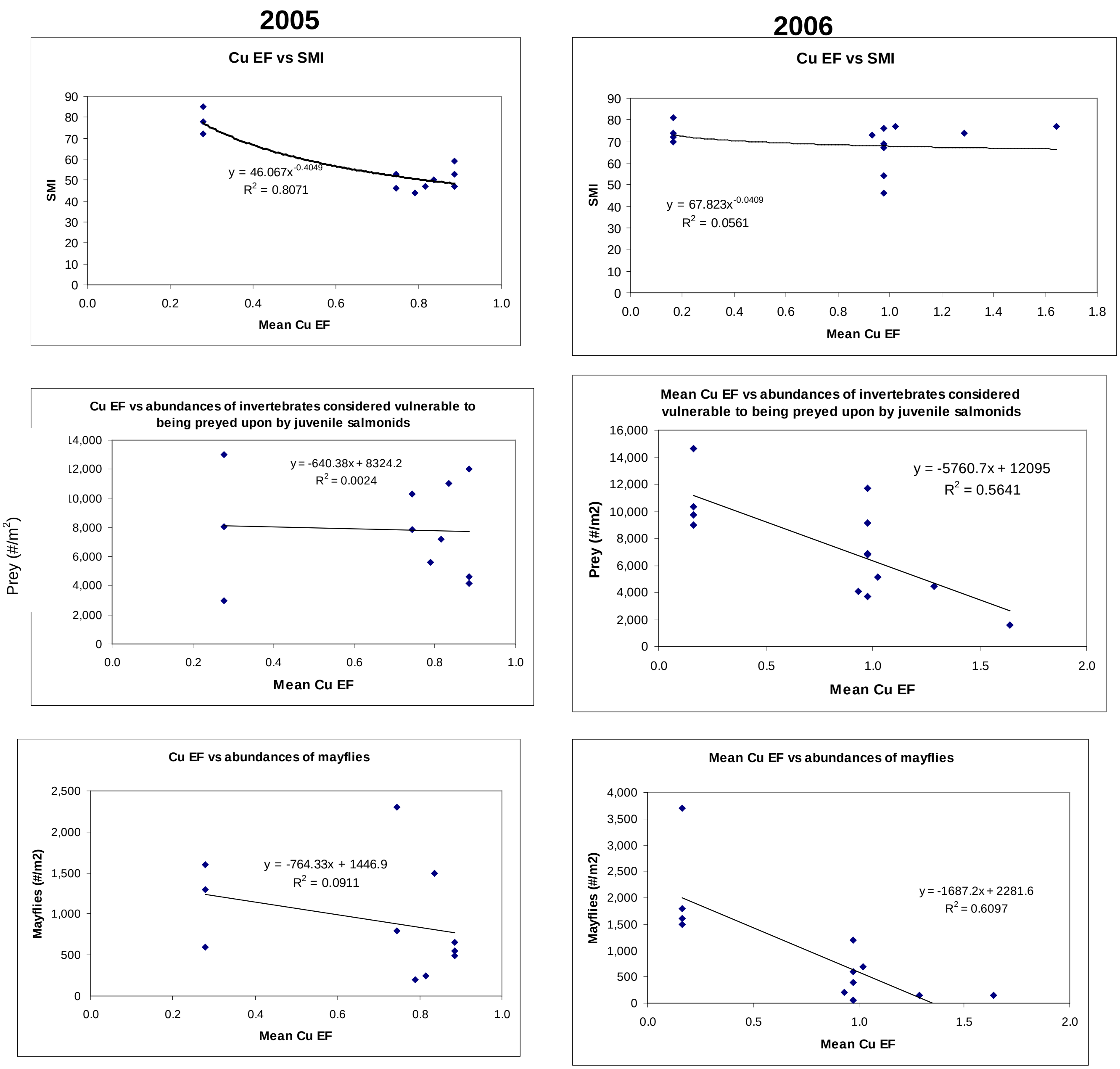



**Figure 2.4.4.3. Correlations of relevant macroinvertebrate metrics with mean exceedence factors of the chronic criterion for stations monitored in Panther Creek, Idaho, September 2005 and 2006 (EcoMetrix 2006, 2007).**

***Bioaccumulation and dietary effects of copper***. There is tremendous variation between fish species in the amount of copper that is accumulated for a given exposure. Copper is more strongly bioconcentrated in invertebrates than in fish, and is more commonly found in tissues of herbivorous fish than in carnivorous fish from the same location (Sorensen 1991). In salmonids, copper has been determined to accumulate in liver, gill, muscle, kidney, pyloric caecae, and spleen tissues, and the concentrations of copper in fish tissues reflect the amount of bioavailable copper in the environment (Farag *et al*. 1994; Camusso and Balestrini 1995; Saiki *et al*. 1995; Sorensen 1991; Marr *et al.* 1996). The kidneys and gills are not thought to play a significant role in copper detoxification (Sorensen 1991). Both waterborne and dietary pathways have been associated with bioaccumulation in salmonids.

A series of dietary toxicity studies was conducted that involved feeding young rainbow trout diets prepared from invertebrates collected from the metals-contaminated Clark Fork River in Montana (Woodward *et al*. 1994; 1995; Farag *et al*. 1994). Results of these studies showed that fish fed a diet of pellets prepared from metal enriched invertebrates had reduced growth and physiological abnormalities relative to fish fed similar diets prepared from invertebrates from reference areas or less contaminated portions of the Clark Fork River. The Clark Fork watershed is enriched with several metals, though copper was generally considered to be the metal of greatest concern, and the adverse effects described in these articles were attributed to copper. However, a subsequent feeding study with invertebrates exposed to Clark Fork sediments in a controlled setting again produced adverse effects in rainbow trout but found that the effects were correlated with arsenic but not with copper (Hansen *et al*. 2004). Similar testing with experimentally exposed invertebrates under controlled conditions to single-metal sediment formulations, rather than field-contaminated sediments, also found no adverse effects of dietary copper exposure, but did find reduced growth and survival with the fish exposed to dietary arsenic, at comparable concentrations that had been measured in invertebrate diets from the previous studies with field-collected invertebrates (Erickson *et al*. 2010).

In a substantive review of the issue, Schlekat and others (2005, p. 141) observed that “We found no studies that demonstrate adverse effects resulting from diet-borne metals in systems in which water quality criteria were apparently being met. However, this could be a reflection of poorly designed approaches or a lack of appropriate data rather than an indication that such effects are not possible.” [Note: “metals” in this quotation refers to cadmium, copper, lead and zinc; mercury, metalloids such as arsenic, and non-metal inorganics such as selenium were not addressed]. Other studies have reached similar conclusions (Mount *et al*. 1994; Dethloff and Bailey 1998; Taylor *et al*. 2000).

Thus while bioaccumulation of copper could result from dietary exposure near the Idaho chronic criterion concentration, the available information indicates that no appreciable adverse effects from dietary exposure to copper will occur at close to criteria concentrations.

### *2.4.4.3. Summary for Copper*

The results of this analysis suggest that concentrations below the proposed acute and chronic criteria for copper can cause acute and chronic toxicity to salmon and steelhead. At the lower range of hardness values encountered in Idaho streams and lakes the acute standard could result in injury and death.

Listed salmon and steelhead can experience a variety of adverse effects at or below the chronic Idaho copper criterion. These include:

- Deprivation of chemosensory function which in turn causes maladaptive behaviors including the loss of ability to avoid copper, and the loss of ability to detect chemical alarm signals. Appreciable adverse effects can be expected with increases as small as 0.6 µg/L above background concentrations.

- Reduced growth in juvenile Chinook salmon and rainbow trout under conditions of low hardness and low organic carbon.

- Because survival of juvenile salmon and steelhead in their migration to sea is strongly size-dependent, small reductions in size will result in disproportionately larger reductions in survival during migration to sea. Using population modeling, growth reductions at the chronic copper criterion were projected to result in slight increases in extinction risk and pronounced delays in recovery time in a model Chinook salmon population.

- The diversity and abundance of the macroinvertebrate food base for rearing juvenile salmon and steelhead could be reduced at copper concentrations near or below the Idaho chronic criterion.

While a variety of adverse effects relevant to listed salmonids have been demonstrated at copper concentrations less than the copper criteria under consultation, the most important issue is that the hardness-toxicity equation embedded into the criteria commonly results in fundamentally inaccurate and misleading indications of risk in critical habitats. This is because the best available science indicates that organic carbon is a more important mediator of copper risks than water hardness. During late summer or fall base flow conditions, copper would be expected to be most toxic because organic carbon tends to be low. Yet this is the time of year that hardness tends to be highest, and the hardness-based copper criteria wrongly indicate that copper would be of least risk at this time of year (Appendix C).

### 2.4.5 The Effects of EPA Approval of the Cyanide Criteria

The cyanide group (CN) includes free cyanide (HCN and $CN^-$), simple cyanide salts, (e.g. KCN, NaCN), metal-cyanide complexes, and in some organic compounds. The most bioavailable and toxic forms are free cyanide (Gensemer *et al*. 2007). The EPA's (1985e) criteria considered cyanide toxicity to mostly result from HCN but because the cyanide ion $CN^-$ readily converts to HCN at pH values that commonly exist in surface waters, cyanide criteria were stated in terms of free cyanide expressed as CN. Free cyanide is extremely toxic and fast acting, and its fast action was one reason for EPA's (1992) expression of acute criteria based on 1-hour average concentrations. The EPA recommends measuring free cyanide at the lowest occurring pH and also measuring total cyanide during the monitoring of freshwater systems. In cases where total cyanide concentrations are significantly greater than free cyanide concentrations, EPA recommends evaluating the potential for dissociation of metallocyanide compounds (EPA 1985e).

The criteria being analyzed for cyanide are 22 µg/L for acute exposure and 5.2 µg/L for chronic exposure. A difference between Idaho's cyanide criteria, which is being evaluated in this Opinion, and the cyanide criteria as originally developed by EPA and initially promulgated for Idaho by EPA (1992) is that Idaho's cyanide criteria are defined as Weak Acid Dissociable (WAD) cyanide (EPA 2000a). While not explicitly explained, this definition is probably used because direct measurement of free cyanide was not routinely offered by many environmental test laboratories until fairly recently, and as result a criteria based on free cyanide would be difficult to analytically measure and implement. Interpreting the criteria as total cyanide would include iron-cyanide and other metal-cyanide complexes that are considerably less reactive and toxic than free cyanide. Weak acid dissociable cyanide analyses were a compromise between free and total cyanide measurements and WAD cyanide includes metal-cyanide complexes such as zinc-, nickel-, copper-, and cadmium-cyanide easily dissociate under weakly acidic conditions (pH 5-6).

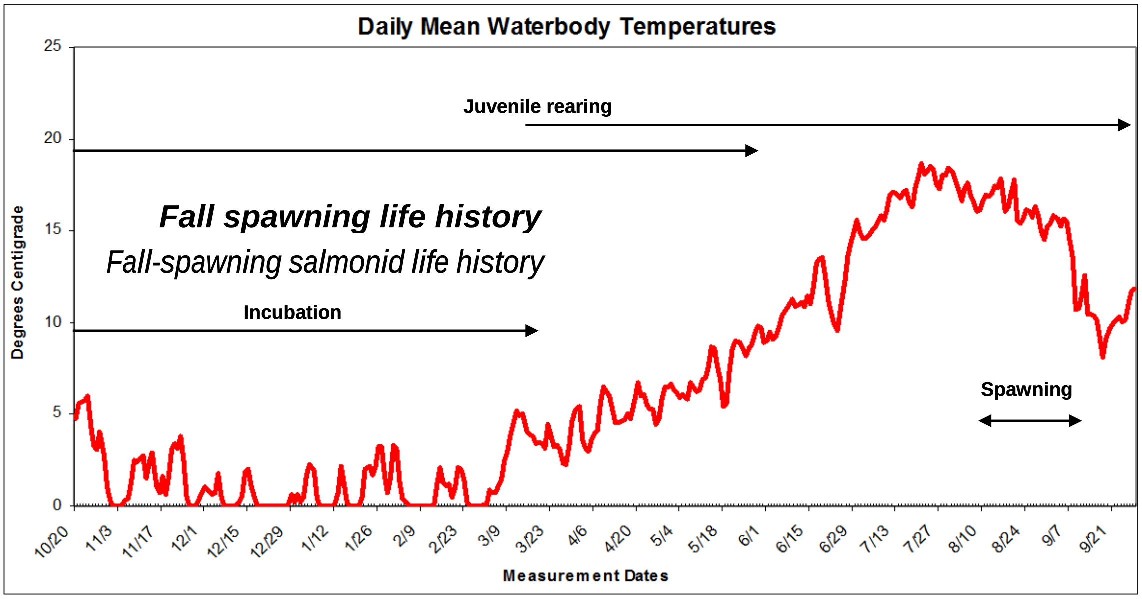
The relevance of these cyanide definition and analytical testing issues for the present Opinion is that for a given environmental sample collected from an effluent or stream that contains cyanides, analyzing the sample for WAD cyanide would produce a higher value than if it could be analyzed for free cyanide. Likewise, using free cyanide concentrations from a toxicity test cited in this Opinion is more protective than using a WAD cyanide concentration. This adds a degree of conservatism to the present analyses, although the magnitude of this it cannot be quantified because the degree of difference between WAD and free cyanide would depend on the sample.

***Temperature and cyanide toxicity.*** Whereas with metals, water hardness or DOC are often important modifiers of toxicity, with cyanide, temperature has a strong influence on toxicity. A number of tests with different species indicated a marked positive correlation between resistance to HCN and temperature rather than the negative one that might be expected from general stress models. This increased toxicity at lower temperatures has been observed with rainbow trout, brook trout, yellow perch, fathead minnows, and bluegills (Smith *et al*. 1978; Kovacs and Leduc 1982b, 1982a). The most robust dataset was probably from Kovacs and Leduc (1982a) from which a temperature-toxicity relationship for rainbow trout can be estimated as: $LC_{50}$ = (T°C)*3.167+6, $r^2 = 0.97$

When a water quality characteristic such as temperature is apparently related to the toxicity of a substance, the EPA Guidelines (Stephan *et al*. 1985) for developing aquatic life criteria provide two approaches: (1) Direct incorporation of the characteristic into the criteria; or (2) data acceptability. In Approach 1, "*if the acute toxicity of the material to aquatic animals apparently has been shown to be related to a water quality characteristic such as hardness or particulate matter for freshwater animals or salinity or particulate matter for saltwater animals, a Final Acute Equation should be derived based on that water quality characteristic*." (Stephan *et al*. 1985). Examples of this include criteria for ammonia which are based on temperature and pH (EPA 1999a), and most metals criteria that are based on hardness, or EPA's 2007 copper criteria, based upon multiple water quality characteristics.  In Approach 2, "*results of acute tests conducted in unusual dilution water. e.g., dilution water in which total organic carbon [TOC] or particulate matter exceeded 5 mg/L, should not be used [in a criterion dataset], unless a relationship is developed between acute toxicity and organic carbon or particulate matter or unless data show that organic carbon, particulate matter, etc., do not affect toxicity*." (Stephan *et al*. 1985).

While test waters warmer than 6°C could hardly be considered "unusual" (or waters with particulates or >5 mg/L TOC, for that matter), temperature clearly affects the toxicity of cyanide, and the Guidelines are clear that such characteristics should be incorporated in criteria.  Why that was not done in the case of cyanide is unexplained in the criteria document.

In cold-temperate climates such as the Idaho action area, it follows that if the cyanide criteria were not adjusted for temperature, only the coldest test results (6°C) should be used.  For example, fall-spawning Chinook salmon progeny in the Snake River usually emerge from gravels at water temperatures of about 5.5 to 9°C (Connor *et al*. 2002).  If data were available on the effects of cyanide at temperatures of 6°C, 12°C, and 15°C, on the incubation and hatching of eggs from a salmonid with a fall-spawning life history only data from the 6°C exposure would be relied upon.  Similarly, since juvenile salmonids from either fall- or spring-spawning species can be expected to be exposed to near-freezing temperatures for long periods (Figure 2.4.5.1), only the $LC_{50}$s obtained from the coldest tests would be used in a final assessment.  For the cyanide data set, these would be the tests conducted around 6°C or below.



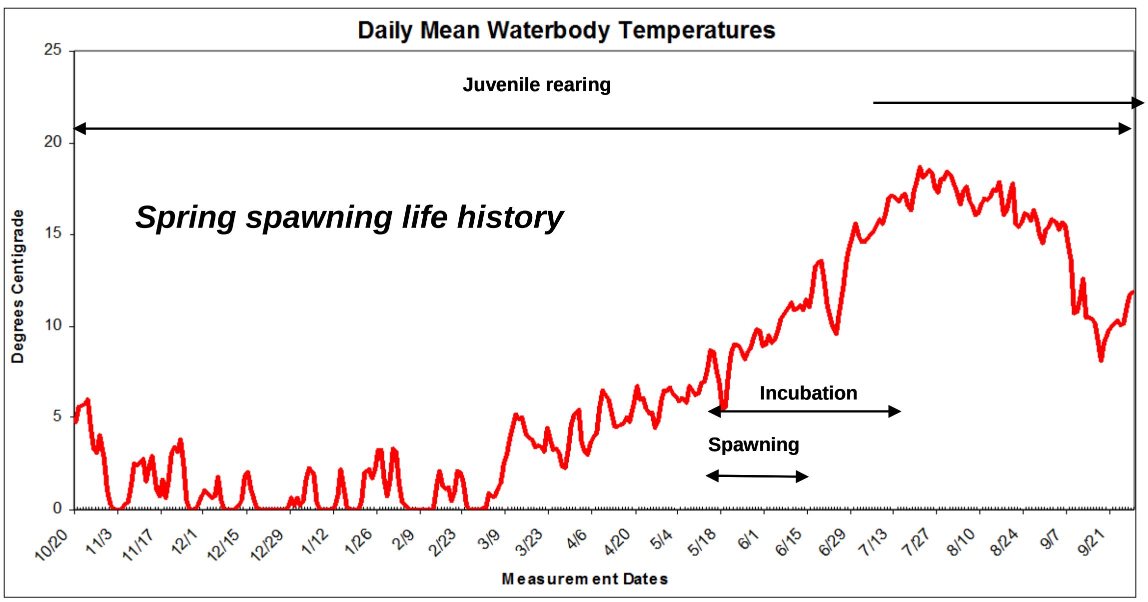




**Figure 2.4.5.1. Examples of the occurrence of different salmonid life stages and annual temperature patterns for a coldwater stream: salmonid species with (top) fall-spawning life histories (e.g., Chinook salmon, coho salmon, Atlantic salmon, brown trout, bull trout, brook trout), and (bottom) spring-spawning life histories (e.g., steelhead, rainbow trout, cutthroat trout, most non-salmonid fishes). Temperature data from the Salmon River, Idaho near Sunbeam, Idaho; data from Idaho Department of Environmental Quality, 2002 water year.**

**Table 2.4.5.1. Contrasting effects of cyanide on salmonids at different temperatures. For lethal effects data, if $LC_{50}$s are greater than the Final Acute Value of 44 µg/L that is assumed to indicate lack of harm at acute criteria concentrations; for sublethal effects, lowest effects concentrations should be greater than 5.2 µg/L.**

| Species | Effect | Exposure duration | T (°C) | Effect statistic | Effect concen-tration (µg/L) | Source/ Notes |
|---|---|---|---|---|---|---|
| | **Lethal effects** | | | | | |
| Rainbow trout | Killed | 4 d | 6 | $LC_{50}$ | 27 | (Kovacs and Leduc 1982a) |
| " | Killed | 4 d | 12 | $LC_{50}$ | 40 | (Kovacs and Leduc 1982a) |
| " | Killed | 4 d | 18 | $LC_{50}$ | 65 | (Kovacs and Leduc 1982a) |
| Rainbow trout | Killed | 4-d | 10 | $LC_{50}$ | 57 | (Smith *et al*. 1978) |
| | **Sublethal effects** | | | | | |
| Rainbow trout | Reduced swimming performance | 20 d | 6 | No effect threshold | <4.8 | (Kovacs and Leduc 1982b) |
| " | Reduced swimming performance | 20 d | 12 | No effect threshold | <9.6 | (Kovacs and Leduc 1982b) |
| " | Reduced swimming performance | 20 d | 18 | No effect threshold | 43 | (Kovacs and Leduc 1982b) |
| " | Reduced swimming performance | | | No effect threshold | <10 | (a) |
| " | Reduced growth | 20 d | 6 | No effect threshold | <4.8 | (Kovacs and Leduc 1982b) |
| " | Reduced growth | 20 d | 12 | No effect threshold | <9.6 | (Kovacs and Leduc 1982b) |
| " | Reduced growth | 20 d | 18 | No effect threshold | 24 | (Kovacs and Leduc 1982b) |
| " | Reduced growth in fish forced to exercise | 20d | 10 | LOEC | 9.6 | (b) |
| Brook trout | Reduced egg production | | | 18% reduction in spawned eggs/female | 5.6 | (Koenst *et al*. 1977) |
| Atlantic salmon | Abnormal embryo and larval development | | | LOEC | 9.6 | (Leduc 1978) |

(a) EPA 1985e, citing Broderius 1970; (b) EPA 1985e, citing McCracken and Leduc 1980

*2.4.5.1. Species Effects of Cyanide Criteria*

***Acute Cyanide Criterion***. The acute criterion under review is 22 µg/L, which is the FAV divided by two. Because the FAV was derived from $LC_{50}$ data, and available acute data for cyanide are $LC_{50}$s, and a concentration killing 50% of the test population obviously cannot be used directly to judge the protectiveness of the acute criteria. Thus the $LC_{50}$s are compared to the FAV rather than the acute criterion. Following the assumption that dividing a $LC_{50}$ by two will likely kill few if any fish (Section 2.4.1.6), it also follows that $LC_{50}$s need to be higher than the FAV (44 µg/L) in order to assume that little mortality would result at the acute criterion. This turns out not to be the case when temperatures were 12°C or less (Table 2.4.5.1), indicating that the acute criterion cannot be considered fully protective under these conditions.

Billard and Roubaud (1985) determined that sperm of rainbow trout had lower fertilization success when they were exposed for 15 minutes directly to 1 µg/L cyanide in a sodium and potassium chloride buffered diluent that kept the sperm immobile. This concentration is below the chronic criterion. However, spermatozoa become motile when released into unbuffered, natural waters and only survive for a few minutes (Billard and Roubaud 1985; Farag *et al*. 2006). Thus effects demonstrated by Billard and Roubaud (1985) may not relate to natural waters.

***Chronic Cyanide Criterion.*** The chronic cyanide criterion is 5.2 µg/L. Kovacs and Leduc (1982) observed chronic toxicity effects on growth in terms of average fat gain and dry weight when juvenile rainbow trout were exposed to 5 µg/L at 6°C. At 12°C, toxicity effects were determined at concentrations greater than or equal to 10 µg/L. As with acute toxicity, chronic effects were inversely related to water temperature in the study. All measures of growth were affected significantly at an exposure concentration of 15 µg/L at all temperatures tested (6°C to 18°C). The results of Kovacs and Leduc (1982) suggest there is potential for reduced growth at the proposed chronic criterion when temperatures are 6°C or lower.

Kovacs and Leduc (1982b) also found that after a 20-day exposure to sublethal cyanide the swimming ability of rainbow trout was reduced at all cyanide concentrations tested in the range of 5 µg/L to 45 µg/L, with the effect increasing at lower temperatures. Although cyanide-exposed fish had returned to normal or near normal growth rates, their swimming impairment suggests biochemical disturbance and perhaps tissue damage as observed by Dixon and Leduc (1981).

Kovacs and Leduc (1982b) noted that at low water temperatures (4°C to 5°C), under conditions where metabolism is depressed, fish are under some stress to maintain their life processes. This is evidenced by a greater water content of fish, less food availability in nature, greater specific dynamic action, assimilation, and food conversion efficiency. Under such conditions, another stressor such as cyanide would have a serious effect on fish production and even on long-term survival. Their study indicated that at 6°C, a concentration as low as 5 µg/L HCN can cause marked reduction in fat synthesis and swimming performance. In Idaho waters, low water temperatures prevail for much of the year. Therefore, for a more realistic appraisal of our water pollution problems, toxicity to fish at low temperatures needs to be evaluated.

We did not locate any tests for reproductive impairments with exposures of listed species or very close surrogates (e.g., other genus *Oncorhynchus* tests) for this analysis. Tests with bluegill and brook trout suggest that fish reproduction can be severely inhibited at concentrations close to the chronic criterion. Kimball *et al*. (1977) tested the effects of long-term cyanide exposure on bluegills and found severe adverse effects at the lowest concentration tested, which was the same as the chronic criterion concentration of 5.2 µg/L. They noted at p. 345 that "*Spawning is completely inhibited at 5.2 µg/L HCN and presumably, is inhibited to some extent at lower level*."

*2.4.5.2. Habitat Effects of Cyanide Criteria*

***Toxicity to Food Organisms.*** Although cyanide toxicity varies extensively among invertebrate taxa, available data for the types of aquatic insects and crustaceans that juvenile salmonids feed on indicate that they are similarly or less sensitive to cyanide compared with listed salmon and steelhead (EPA 1980e, 1985e; Eisler 1991). Aquatic invertebrates do not appear to be adversely affected by concentrations that are protective of fish. As documented below, cyanide does not appear to bioaccumulate because of its short-lived nature and the ability of aquatic organisms to depurate the compound. The proposed criteria are likely to be protective of the food sources of listed salmon and steelhead.

***Bioaccumulation.*** There is no evidence of significant bioaccumulation of cyanide in fish at levels below the proposed chronic criterion because the compound is easily metabolized (EPA 1985e). Lanno and Dixon (1996) determined that bioconcentration occurred in juvenile rainbow trout exposed to a cyanide level (8 µg/L) which is close to the chronic criterion, but did not observe any significant toxic effects. Other evidence exists that cyanide levels are elevated in fish tissues when subjected to long-term chronic exposure, but cyanide depuration occurs relatively quickly when fish move to clean water (Eisler 1991; Lanno and Dixon 1996). Therefore, potentially adverse effects related to cyanide bioaccumulation are unlikely to be observed in listed salmon and steelhead.

***Water Chemistry.*** Cyanide in the water column at the proposed acute and chronic criteria concentrations during the colder seasons will result in the water quality being unsuitable for listed salmonids as described above in the temperature and cyanide toxicity section.

*2.4.5.3. Summary for Cyanide*

The proposed acute and chronic criteria can expose listed salmonids to harmful cyanide concentrations under specific situations. The acute criterion cannot be considered to be reliably protective when water temperatures drop to about 6°C or lower. Further, Leduc (1984) found that cyanide concentrations at the chronic criterion in water colder than 6°C may be associated with chronic toxicity effects. Temperatures in streams within the action area routinely drop below 6°C.

2.4.6. The Effects of EPA Approval of the Mercury Criteria

Mercury is hazardous to fish because of its strong tendency to bioaccumulate in muscle tissue and because it is a potent neurotoxin that causes neurological damage which in turn leads to behavioral effects which in turn lead to reduced growth and reproductive effects (Wiener *et al*. 2003; Weis 2009; Sandheinrich and Wiener 2010; Kidd and Batchelar 2011). Methylmercury is a highly neurotoxic form that readily crosses biological membranes, can be rapidly bioaccumulated through the water, and is taken up primarily through the diet (which accounts for more than 90% of the total amount of methylmercury accumulated by fish). Both organic and inorganic mercury bioaccumulate, but methylmercury accumulates at greater rates than inorganic mercury. Methylmercury is more efficiently absorbed, and preferentially retained than inorganic mercury (Scheuhammer 1987, Wiener 1995). Methylmercury is biomagnified between trophic levels in aquatic systems and in general proportion to its supply in water (Wattras and Bloom 1992). In fish tissue accumulated mercury consists almost entirely of methylmercury (Bloom 1992; Hammerschmidt *et al*. 1999; Harris *et al*. 2003). Toxicity of methylmercury is therefore particularly important with respect to effects to higher trophic level fish and other organisms (Sorensen 1991; Nichols *et al*. 1999).

Inorganic mercury is absorbed less readily and is eliminated more rapidly than methylmercury. In fact, intestinal absorption of inorganic mercury is limited to a few percent of methylmercury, for which absorption is nearly complete (Scheuhammer 1987; Wiener *et al*. 2003). Inorganic mercury appears to have the greatest effect upon the kidneys, while methylmercury is a potent embryo and nervous system toxicant. Methylmercury readily penetrates the blood brain barrier, produces brain lesions, spinal cord degeneration, and central nervous system dysfunctions. Long-term dietary exposure to mercury has been shown to cause instability, inability to feed, and diminished responsiveness. The central nervous system is the site of the most extensive damage due to mercury exposure.

### *2.4.6.1. Species Effects of Mercury Criteria*

The acute and chronic criteria for dissolved mercury under consultation are 2.1 µg/L and 0.012 µg/L (12 ng/L), respectively (EPA 1985g). The EPA has also developed a human health criterion, in which fish tissue concentrations are not to exceed 0.3 mg/kg ww (66 FR 1344; EPA 2001). This standard was adopted in Idaho in 2005 and is applicable to all designated critical habitats and waters inhabited by listed salmon or steelhead (IDEQ 2005).

***Acute Mercury Criterion.*** The acute mercury criterion is about 175 times higher than the chronic criterion and about 1,000 times higher than typical ambient concentrations (Table 2.4.6.2). All criteria applications contemplated under the Idaho standards (cleanup actions and discharge limits) would also involve application of the chronic criterion. As a practical matter the acute criterion would never be relevant for determining discharge limits to any receiving water since it is hydrologically inconceivable that the critical flows used by EPA with the acute criteria for calculating short-term maximum discharge limits (lowest 1-day average flows in a 10-year period, abbreviated as a 1Q10) would be anywhere close to 175 times lower than the critical flows used for calculating long-term average discharge limits (lowest 7-day average flows occurring in a 10-year period 7Q10). An example is given later in this Opinion in Appendix D, where the question of implementing criteria through limiting effluent volumes is treated in more detail. For Thompson Creek, the 7Q10 is 2.1 cfs which very close to the 1Q10 of 2.05 cfs. Thus the 1Q10 is 1.02 times lower than the 7Q10. The possibility that the 1Q10 and the 7Q10 could differ by 175 is discountable. Nevertheless, even though the acute mercury criterion is unlikely to be applied as a practical matter, the following analysis summarizes the available acute toxicological information for mercury.

Most available data suggest that listed salmon and steelhead are not susceptible to acute toxicity from direct exposure to mercury in water water at concentrations approaching the 2.1 µg/L acute criterion (Kidd and Batchelar 2011). Many "acute" type of studies NMFS reviewed exposed fish to mercury in water for much longer than the 4 days typical of "acute" exposures. The EPA (1985g) reported $LC_{50}$ values for salmonids exposed to inorganic mercury that ranged between 155 µg/L and 420 µg/L. For organic mercury, reported $LC_{50}$s ranged from 5 µg/L to 84 µg/L, depending on the chemical form, with a phenylmercuric compound ($LC_{50}$ = 5 µg/L) being the most toxic. Buhl and Hamilton (1991) exposed coho salmon and rainbow trout alevins and parr to mercuric chloride, and determined average $LC_{50}$s that ranged between 193 µg/L and 282 µg/L. Devlin and Mottet (1992) determined a methylmercury $LC_{50}$ equal to 54 µg/L for coho salmon embryos exposed for 48 days. Niimi and Kissoon (1994) exposed rainbow trout sub-adults to 64 µg/L of mercuric chloride until the fish died. The average time to death was 58 days. In another exposure to 4 µg/L of methylmercury chloride, they determined that the fish lived more than 100 days. The lowest effect level noted from an "acute" type study was an LC10 of 0.9 µg/L following a 28-day exposures of rainbow trout embryo's to mercury, with a no-effect (LC1) estimated of 0.2 µg/L (Birge *et al.* 1980)

Available information on sublethal effects from direct acute exposure is sparse. Rainbow trout were attracted to 0.2 µg/L mercuric chloride in 80 minute exposures, which is about a factor of 10 lower than the acute criterion (Black and Birge 1980).

The reported $LC_{50}$s for life stages beyond the embryo are well above the acute criterion. The results of these studies suggest collectively that the proposed acute mercury criterion is unlikely to cause mortality. Behavioral alterations at a concentration 10 times lower than the acute criterion were reported, but even that concentration is ~20 times higher than the chronic criterion.

***Chronic Mercury Criterion.*** The EPA's 1984 chronic aquatic life criterion for mercury is something of a misnomer, since its establishment had nothing to do with the chronic effects of mercury on aquatic life. Rather, the criterion was intended to protect the "fishable" uses of aquatic life which in this case is to avoid allowing bioaccumulation in fish at mercury levels that would impair marketability of fish. The chronic criterion was established with the objective of avoiding fish from bioaccumulating mercury to concentrations that were predicted to exceed the Food and Drug Administration's (FDA) (1984) action level of 1 mg/kg fresh weight for the sale of commercially caught fish. "Fresh weight" is synonymous with wet weight, ww, which is more commonly used in the ecotoxicology literature. All tissue residue values for mercury are given as ww unless otherwise indicated.

The marketability approach of setting chronic criterion for mercury replaced EPA's (1980j) approach which was similar to that used for other substances. The EPA (1980j) followed an extrapolated species-sensitivity distribution to obtain a Final Acute Value of 0.0017 µg/L (1.7 ng/L), which was divided by an ACR of 3.0 to obtain a freshwater final chronic value of 0.00057 µg/L (0.57 ng/L).

The physiological effects of direct exposure to mercury at ambient concentrations near the chronic criterion are the result of dietary bioaccumulation. This is due to the strong tendency of mercury to bioaccumulate, discussed further in the next section. In the environment virtually all mercury exposure to fish is from dietary sources, so concentrations in water are not meaningful for direct water-only exposures (Wiener and Spry 1996; Wiener *et al*. 2003). Literature from water borne exposures may be useful; however, in instances where waterborne exposures were used as a means to achieve tissue burdens. However, in these instances the relevant media to evaluate is the tissue burden, not the water concentrations.

Wiener and Spry (1996) noted that water-borne concentrations in natural streams are unlikely to be high enough to result in direct toxicity effects. In a broad survey of mercury in freshwater systems in California and other areas including the lower Columbia River, Gill and Bruland (1990) failed to locate any water bodies containing levels of mercury above or approaching the dissolved criterion although many of these same water bodies were mercury impaired due to elevated concentrations in fish. Similar findings have been reported from other areas (Becker and Bigham 1995; Watras *et al*. 1998; Castro *et al*. 2002; Hope and Rubin 2005; Wiener *et al*. 2006; IDEQ 2007b; Chasar *et al*. 2009; Essig 2010).

Sublethal effects of the proposed chronic criterion may occur from long-term exposure in the natural environment effects, since ambient water mercury concentrations that are near or below the proposed chronic criterion have been associated with bioaccumulation (see below). For example, Davis Creek Reservoir in California is highly contaminated by mercury and has dissolved organo-mercury concentrations around 2.4 ng/L and total dissolved mercury concentrations around 12 ng/L These concentrations of mercury in water are similar in magnitude to the proposed chronic criterion, and were associated with fish tissue concentrations of 2.5 mg/kg ww (Gill and Bruland 1990) that were almost 10 times higher than apparently safe the tissue concentrations of 0.2 to 0.3 mg.kg ww that appear to be safe for fish (later in this section).

Hence, available information suggests that listed salmon and steelhead are unlikely to be killed outright by direct exposure to water concentrations equal to the proposed chronic criterion. However, in all reports from field situations reviewed, effects of direct exposure are likely to be overshadowed by effects from bioaccumulation.

*2.4.6.2. Habitat Effects of Mercury Criteria*

***Toxicity to Food Organisms.*** Little information was located indicating appreciable risk of adverse effects to invertebrates prey items themselves. Rather, the most significant concern from the perspective of listed salmon and steelhead is bioaccumulation from eating aquatic invertebrates that themselves have elevated mercury levels, not changes in aquatic invertebrate production due to mercury toxicity.

***Bioaccumulation.*** Food chain transfer is by far the most important exposure pathway in aquatic ecosystems (Hall *et al*. 1997; Wiener *et al*. 2003). Aquatic systems have complex food webs including several trophic levels, and primary producers in aquatic systems may themselves accumulate more mercury from water and sediment than their soil-based counterparts in terrestrial systems. Rates of bacterial methylmercury production in water and sediment ultimately determines the potential of an aquatic system to develop a mercury bioaccumulation problem (EPA 1997b). Aquatic predators including salmonids are most susceptible to bioaccumulating mercury, and thus their tissue concentrations may best reflect the amount of mercury available to aquatic organisms in the environment. For example, in comparisons of bottom feeding fish with fish that feed on plankton, invertebrates, and vertebrates, Wren and MacCrimmon (1986) determined that the greatest mercury concentrations were found in piscivorous fish species and that mercury content increased with higher trophic levels.

Fish store most mercury as methylmercury in their muscle, even when they are exposed to inorganic mercury. Methylmercury both bioconcentrates and biomagnifies across trophic levels, and corresponding, field-measured bioaccumulation factor (BAFs) can be in the millions for top trophic level fish (Nichols *et al*. 1999). Methylmercury accumulates at greater rates than inorganic mercury because it is more efficiently absorbed and is preferentially retained (Scheuhammer 1987; Wiener 1995).

Rates of bioaccumulation are thought to be affected by numerous factors such as the number of trophic levels present, food web structure of the aquatic ecosystem, abundance of sulfur reducing bacteria and concentration of sulfates, amount of dissolved oxygen, water temperature, organic carbon availability, pH, the nature of the mercury source, and other parameters (Porcella *et al*. 1995). The uptake of mercury and methylmercury in fish increases with ambient water concentration, water temperature, size and age of the fish, breeding status, and food ingestion rate. Decreases in pH have also been correlated with increasing methylmercury uptake (Wren and MacCrimmon 1986; Ponce and Bloom 1991).

Diet is the primary route of methylmercury uptake by fish in natural waters, and contributes more than 90% of the amount accumulated. The assimilation efficiency for uptake of dietary methylmercury in fish is probably 65% to 80% or greater. To a lesser extent, fish obtain mercury from water passed over the gills, and fish also methylate inorganic mercury in the gut (Wiener and Spry 1996).

Sediments are an important reservoir for mercury in freshwater systems. Mercury in sediments can become available for food chain transfer, and instances of elevated mercury in sediment corresponding with elevated mercury in fish have been documented (Maret 1995; Clark and Maret 1998; Suchanek *et al*. 2008; Scudder *et al*. 2009; EPA 2011). Mercury may accumulate in bed sediments to levels that greatly exceed levels associated with probable adverse effects to benthic communities even when mercury in surface water was far lower than the chronic criterion of 12 ng/L. One well documented instance was from Onondaga Lake, New York, where dissolved mercury in the epilimnion was about 1 ng/L and mercury in the hypolimnium was up to 10 ng/L (Bloom and Effler 1990). Mercury in sediments were always above 1 mg/kg dw, often above 5 mg/kg dw, and exceeded 25 mg/kg dw in some samples. Mercury in sediments was strongly correlated with mercury in invertebrate tissues (Becker and Bigham 1995). In addition to the role of mercury bound in sediment as an entry point to trophic pathways, direct adverse alterations to benthic communities are probable when mercury in sediment exceeds 1 mg/kg dw (MacDonald *et al*. 2000a).

***Toxicity of dietary mercury to fish.*** Concentrations of mercury that would be expected to elicit ecologically significant adverse effects in fish when ingested as prey were estimated by DePew *et al*. (2012). They concluded that chronic dietary exposure to low concentrations of MeHg may have significant adverse effects on natural-origin fish populations. Adverse effects on behavior resulting from dietary concentrations of mercury usually occurred above 0.5 mg/kg ww. However, adverse effects on reproduction occurred with dietary concentrations of mercury at 0.2 mg/kg ww or lower. DePew *et al*. (2012) noted that although their thresholds were intentionally conservative, they still may underestimate the magnitude of effects experienced by natural-origin fish because their thresholds were derived from laboratory tests conducted under favorable conditions. In the wild, additional environmental stressors related to foraging, predation, temperature fluctuation, and other potentially toxic contaminants are present (Depew *et al*. 2012).

***Mercury tissue residues in fish associated with the presence or absence of adverse effects.*** While the risks of mercury neurotoxicity to humans from eating fish has been the subject of much concern and research, and effects on fish-eating wildlife have been reasonably well documented, there is yet considerable uncertainty regarding effects of mercury on fish themselves. Scheuhammer *et al*. (2007) summarized the state of the knowledge succinctly: "*Compared with humans and mammalian and avian wildlife, relatively little is known of the toxicological significance to fish of environmentally realistic exposures to methylmercury*." Ranges of estimates of "safe" tissue residues of mercury in various fish tissues are greater than an order of magnitude. Fish can survive in laboratory environments with mercury concentrations in tissues elevated far above those encountered in the wild. In the brain, concentrations of 7 mg/kg ww or greater probably eventually kill fish in laboratory environments, and for mercury sensitive species, brain-tissue concentrations of 3 mg/kg ww or greater probably indicate significant toxic effects. For axial muscle tissue, concentrations of 6 to 20 mg/kg ww have been associated with toxicity in laboratory studies (Wiener and Spry 1996). However, subsequent to the review of Wiener and Spry (1996), more ecologically relevant studies have been devised that have detected effects associated with endocrine disruption, neurotoxicity, and reproductive impairment (Table 2.4.6.1*)*

Given the high neurotoxicity of methylmercury, the exposure levels causing adverse behavioral effects are probably much lower than exposure levels causing overt toxicity. Many fish behaviors are sensitive and ecologically relevant indicators of contaminant toxicity, affected at lower levels than those causing direct mortality. The neurotoxic effects of exposure to sublethal concentrations of methylmercury can impair the ability of fish to locate, capture, and ingest prey and to avoid predators (Wiener *et al*. 2003). For example, Fjeld *et al*. (1998) showed that the feeding efficiency and competitive ability of grayling (*Thymallus thymallus*) exposed as eggs to waterborne methylmercury chloride for 10 days and having yolk-fry with mercury concentrations of 0.27 mg/kg ww or greater, were impaired when fish were tested 3 years later.

The NOEC from Fjeld *et al*'s (1998) study (0.09 mg/kg ww in embryos) would translate to a mean concentration in maternal muscle tissue of about 0.7 (range 0.15 to 1) mg/kg ww based on various ratios of mercury concentrations in eggs or maternal fillets in brook trout reported by McKim *et al*. (1976). Similar calculations by USFWS (2003) and IDEQ using relative mercury concentrations in different tissues of yellow perch or other data resulted in somewhat higher extrapolations of the 0.09 embryo NOEC concentration to muscle tissue concentrations ranging from 0.45 to 1.8 mg/kg ww. Similarly, estimates of maternal muscle tissue concentrations that would produce an LOEC embryo residue of 0.27 mg/kg ww range from 1.35 to 5.4 mg/kg ww (Fjeld *et al*. 1998; USFWS 2003; IDEQ 2005). These tissue residue effect estimates are roughly similar (within a factor of two) to tissue effects in Atlantic salmon parr. In Atlantic salmon parr, methylmercury concentrations of 0.69 mg/kg ww were associated with brain lesions and behavioral alterations (Berntssen *et al*. 2003).

Mercury tissue residues associated with the presence or absence of adverse effects are summarized in Table 2.4.6.1. Generally, the most sensitive effects of long-term exposures of a variety of fish species to methylmercury have been reproductive or behavioral effects, with concentrations greater than about 0.3 mg/kg ww in whole bodies or axial muscle tissues likely to be harmful to fish (Table 2.4.6.1). However, adverse effects at concentrations lower than this range are possible. Cutthroat trout with whole-body mean mercury burdens of only about 0.05 mg/kg ww collected from a mountain lake had significant changes in metabolic, endocrine, and immune-related genes, compared to fish from lakes with lower mercury concentrations in trout (Moran *et al*. 2007). Possible steroidogenesis effects in white sturgeon collected from the lower Columbia River, as reduced androgen levels in the sperm, were suggested to correspond with a mean muscle mercury concentration of 0.2 mg/kg ww (Webb *et al*. 2006). These reports indicate that a true threshold for the absence of effects from mercury accumulation could be considerably lower than 0.3 mg/kg ww. However, to borrow a phrase from human health care, changes in gene expression or steroid concentrations may be considered “sub-clinical,” that is, not of health significance unless further work relates sub-organismless effects such as these to some other organism-level effect such as altered behaviors, reduced growth, or impaired reproduction.

Another recent review reached fairly similar conclusions on what tissue burdens of mercury were unsafe. Sandheinrich and Wiener (2010) concluded that effects on biochemical processes, damage to cells and tissues, and reduced reproduction in fish have been documented at methylmercury concentrations of about 0.3 to 0.7 mg Hg/kg ww in the whole body and about 0.5 to 1.2 mg Hg/kg ww in axial muscle.

**Table 2.4.6.1. Examples of mercury tissue residues co-occurring with the presence or absence of adverse effects.**

| Organism | Residue concentration (mg/kg wet weight, ww) | Tissue | Effect | Source |
|---|---|---|---|---|
| Edible fish | 0.3 | Muscle | Acceptable risk for most human consumption of recreationally caught fish (including subsistence), except for vulnerable populations | (EPA 2001b; IDEQ 2005) |
| Edible fish | 1.0 | Muscle | Acceptable risk for human consumption of commercially caught fish. | (FDA 1984) |
| Multiple fish species | 0.2 | Whole body | Tissue threshold-effect level based largely on sublethal endpoints (growth, reproduction, development, behavior), calculated to be protective of juvenile and adult fish. | (Beckvar *et al*. 2005) |
| Multiple fish species | 0.2 | Whole body | 5.5% or 33% injury to juvenile or early life stage fish respectively, where injury is limited to endpoints such as survival, reproductive success, and lethal developmental abnormalities. | (Dillon *et al*. 2010) |
| Grayling, *Thymallus thymallus* | 0.27 | Yolk-fry | Impaired feeding efficiency and competitive ability 3 years after exposure as yolk-fry | (Fjeld *et al*. 1998) |
| Grayling, *Thymallus thymallus* | 0.09 | Yolk-fry | No-observed effects on feeding efficiency and competitive ability 3 years after exposure as yolk-fry | (Fjeld *et al*. 1998) |
| Grayling, *Thymallus thymallus* | 2.2 (0.45 – 3) | Muscle | Estimated range of maternal muscle tissue concentrations that would result in yolk-fry concentrations of 0.27 mg/kg, using organ tissue ratios derived from McKim *et al*. (1976). | This review |
| Grayling, *Thymallus thymallus* | 0.7 (0.15 to 1) | Muscle | Estimated range of maternal muscle tissue concentrations that would result in NOEC yolk-fry concentrations of 0.09mg/kg, using organ tissue ratios derived from McKim *et al*. (1976). | This review |
| Atlantic salmon | 0.69 | Brain | Brainstem lesions and behavioral alterations | (Berntssen *et al*. 2003) |

| Organism | Residue concentration (mg/kg wet weight, ww) | Tissue | Effect | Source |
|---|---|---|---|---|
| Atlantic salmon | 0.61 (0.4 – 0.8) | Muscle | Estimated mean and range of muscle (fillet) concentrations resulting in brain residues of 0.69 mg/kg that caused lesions and behavioral alterations, using organ tissue ratios derived from McKim *et al*. (1976). | This review |
| Brook trout | 0.6 – 0.8 | muscle | No-effect (EC0) for 2nd and 3rd generation reduced egg production per female, using regression analysis | (this review, using data from McKim *et al*. 1976) |
| Brook trout | 1.6 | muscle | No-effects apparent through 3 generations, based on statistical hypothesis testing | (McKim *et al*. 1976) |
| Brook trout | 4 | muscle | No-effects apparent until the progeny attempted to reproduce nearly 2 years after exposure began. Females showed abnormal behavior before spawning and ultimately all died. | (McKim *et al*. 1976) |
| Cutthroat trout | ~0.055 | Whole body | Increased expression of genes related to stress response, immune responses, metabolism, and contaminant detoxification. | (Moran *et al*. 2007) |
| White sturgeon | 0.19 | muscle | Possible steroidogenesis threshold | (Webb *et al*. 2006) |
| Mummichogs | 0.2 to 0.47 | Whole body | Reduced male survival, reduced the ability of offspring to successfully reproduce, and altered sex ratios in offspring | (Matta *et al*. 2001) |
| Golden shiners | 0.5 | Whole body | Abnormal behavior in response to a model predatory bird, slower to regroup afterwards | (Webber and Haines 2003) |
| Fathead minnows | 0.86 | Whole body | Reduced spawning success | (Drevnick and Sandheinrich 2003) |
| Fathead minnows | 1.5 | Whole body | Reduced spawning success | (Hammerschmidt *et al*. 2002) |
| Fathead minnows | 0.7 | Whole body | Reduced reproductive behavior and spawning success | (Sandheinrich and Miller 2006) |
| Walleye | 0.25 | whole body | Reduced growth and gonadal development | (Friedmann *et al*. 1996) |

Coincidentally, this low risk threshold of 0.2 to 0.3 mg/kg ww is almost the same as the 0.3 mg/kg ww water quality standard the IDEQ has adopted subsequent to the present action to protect people eating edible portions of recreationally caught fish. As implemented, because of uncertainty in sampling and analysis of fish, IDEQ (2005) applies a 20% uncertainty factor to fish data, and would consider effluent limits and reductions necessary if average concentrations in the highest trophic level present exceeded 0.24 mg/kg ww. In waters inhabited by threatened or endangered species, the criteria would be applied to the highest trophic level of fish present; elsewhere the criteria would be applied to average trophic level of fish present in the water body (IDEQ 2005). Because the mercury aquatic life criteria are expressed as concentrations in water, but the adverse effects of mercury to fish are related to tissue concentrations, the relations between water and tissue residue concentrations need to be considered.

***Factors influencing mercury tissue concentrations in fish.*** So far, our analysis has shown that concentrations of mercury in fish tissue residues are more meaningful for evaluating risk to fish, and that the lowest thresholds for adverse effects of mercury to fish reported in the literature were around 0.2 to 0.3 mg/kg ww, as measured in muscle fillets, or for small fish, whole carcasses. This leads to two additional and related questions:

1. The Idaho chronic criterion under review is a concentration in water (12 ng/L in filtered samples). If listed salmon and steelhead had long-term exposure to the 12 ng/L chronic criterion, what concentrations would be predicted in the fish tissues?

2. What concentrations in water would likely result in bioaccumulation to the low-risk tissue residue thresholds in fish of about 0.2 to 0.3 mg/kg ww?

Attempting to answer these questions first requires a consideration of the factors that influence mercury concentrations in fish. The bioaccumulation of methylmercury in fish is influenced by an array of abiotic, biotic, and ecological variables. However, because the elimination of methylmercury from the tissues of fish is very slow, as a rule, within a species, older and bigger fish tend to have the highest mercury tissue burdens, and because mercury biomagnifies within food chains within a community, predatory fish (i.e., higher trophic levels) will generally accumulate more mercury than non-predatory fish. Because juvenile salmonids and other fish tend to strictly feed on small invertebrates but may switch to preying on smaller fish as they grow larger, these trophic levels are not rigid within species.

The relationship between mercury bioaccumulation and trophic level may put listed steelhead and salmon at lower risk of mercury toxicity than strictly freshwater fish. Most salmonids only start becoming predominantly piscivorous when they reach about 30 cm in length, although in lakes habitats salmonids tend to start preying on fish at about 15 cm. Most listed steelhead and salmon smolts are less than 20cm in length when they leave their freshwater habitats (Quinn 2005; Mebane and Arthaud 2010). Many studies have examined the interrelationship between trophic level and size or age of fish, to the point that broadscale, predictive models have been developed. The "Environmental Mercury Mapping, Modeling, and Analysis' (EMMMA) project is a statistical model and national data set (31,813 samples) *http://emmma.usgs.gov/* that allows prediction of mercury levels in different fish species by fish length and various sampled locations (Wente 2004). Model results for the Snake River at Lewiston, Idaho, predict that piscivorous fish such as bass or pikeminnows would exceed 0.3 mg Hg/kg by the time they reach about 20 cm (8 in.). Similar-sized Chinook salmon or rainbow trout would only be expected to have about 0.1 to 0.06 mg Hg/kg respectively (Figure 2.4.6.1). This suggests that in the larger migratory rivers in which the top predators (the highest trophic level) are pikeminnows or centrarchids such as bass or perch, if the 0.3 mg Hg/kg ww water quality standard were met, mercury tissues expected in anadromous steelhead or salmon would be less than the 0.2 mg/kg adverse effect threshold.

In waters where salmonids are the top predators, most evidence suggests that the larger and older non-anadromous fish would be more at risk of mercury toxicity, and that in waters where these fish met the 0.3 mg Hg/kg standard, all the smolt-sized salmonids would be at considerably lower risk. The length-concentration curves the Snake River show this pattern, as do empirical patterns of mercury vs. length in rainbow and brown trout collected from streams and reservoirs in southern Idaho and northern Nevada (Figures 2.4.6.1 and 2.4.6.2). For example, for fish modeled at the Snake River at Lewiston, adult, 15 inch smallmouth bass tend to have at least four times greater mercury tissue concentrations than do 8-inch Chinook salmon or rainbow trout. Thus, if conditions were such that mercury in smallmouth bass was no higher than 0.3 mg/kg, concentrations in smolt-sized fish would be considerably less than 0.3 mg/kg, and would likely be on the order of 0.08 mg/kg or lower (Figure 2.4.6.1). These patterns of higher mercury residues in older fish and fish at higher trophic levels have been repeatedly reported in the literature (Becker and Bigham 1995; Watras *et al*. 1998; Hope and Rubin 2005; McIntyre and Beauchamp 2005; Wiener *et al*. 2006; IDEQ 2007b; Peterson *et al*. 2007; Chasar *et al*. 2009; Scudder *et al*. 2009). Exceptions were noted. In cases where sediments have elevated mercury concentrations, benthic insects can be a greater source of dietary mercury than forage fish, and fish that preyed on benthic insects had higher mercury burdens than exclusively piscivorous fish (MacRury *et al*. 2002). Within the action area, mercury in sediment can be highly elevated in localized areas associated with historic gold mining (Frost and Box 2009), and thus the sediment-insect route of exposure could be locally important. Similarly, in surveys in western streams, juvenile Chinook salmon had unexpectedly high mercury burdens with a mean tissue mercury concentration of 0.30 (0.212 to 0.411) mg/kg ww and a mean length of 330 (range, 260 to 400) mm. The mercury burdens in these Chinook salmon were fifth highest of 75 species sampled across the western United States (Peterson *et al*. 2007). These Chinook salmon were collected from the Klamath River, California, (S. Peterson, personal communication) and these fish were larger than Snake River Chinook salmon smolts, so mercury levels might be expected to be lower in Snake River salmon smolts.

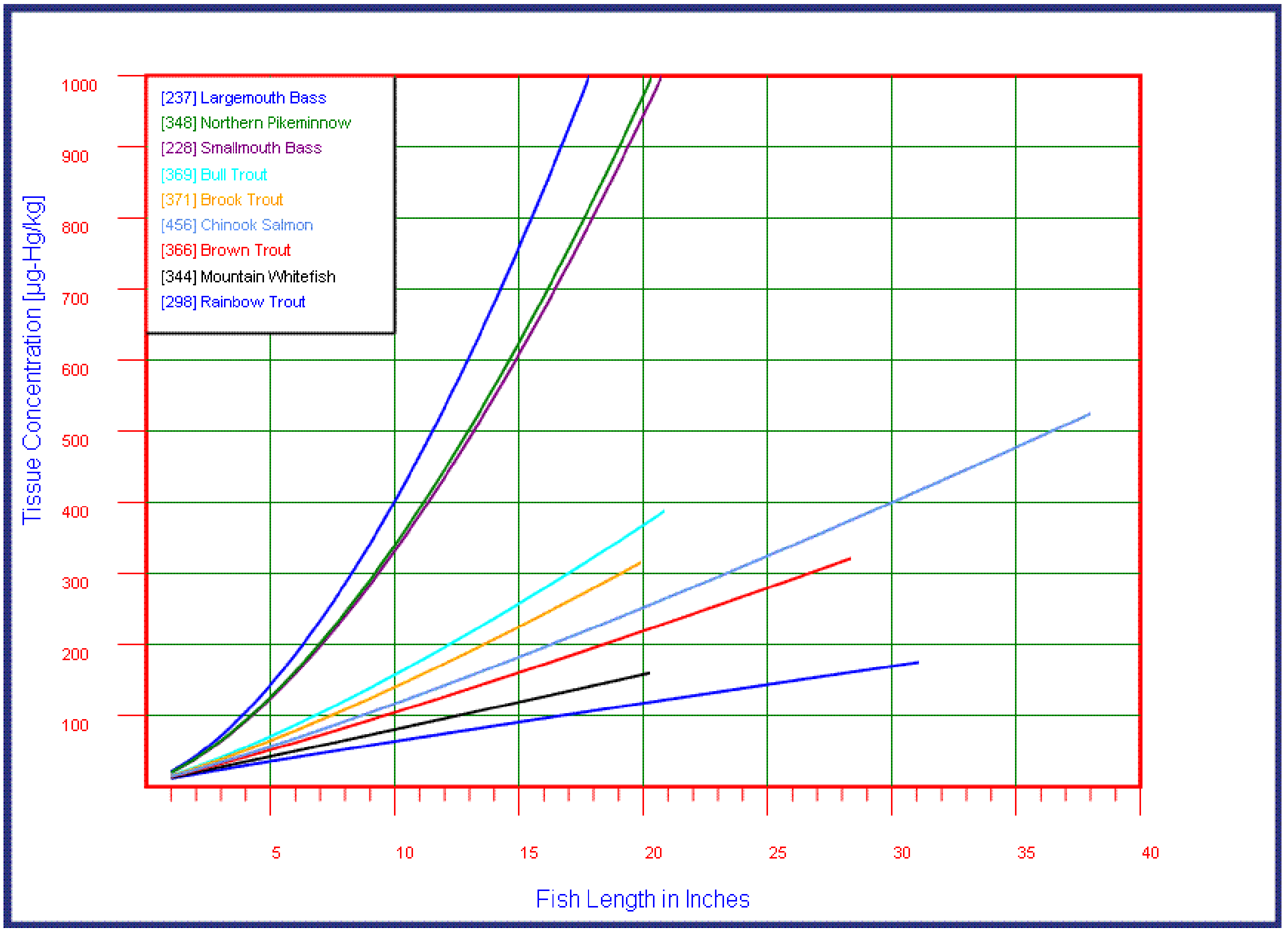



**Figure 2.4.6.1. Modeled concentrations of mercury in fish-tissue for the Snake River at Lewiston, standardized by length and species (modeled and figure generated using the Environmental Mercury Mapping, Modeling, and Analysis (EMMMA) website *http://emmma.usgs.gov/*)**

A further complexity in understanding risks of mercury toxicity associated with tissue burdens in fish is interactions with selenium. Selenium in fish tissue tends to reduce mercury toxicity when present at greater than a 1:1 molar ratio with mercury. Under this scenario, both the bioaccumulation rates of mercury may be lessened and mercury burdens that do accumulate tend to be less toxic (Chen *et al*. 2001; Belzile *et al*. 2006; Ralston *et al*. 2007). The mechanisms behind these patterns are unclear, and it has both been hypothesized that selenium reduces the activity and toxicity of mercury or that low-levels of mercury make fish and some mammals more susceptible to selenium deficiency (Khan and Wang 2009). Nevertheless, in Idaho and the western United States, the patterns are that in the great majority of instances, selenium is present at a 1:1 or greater molar ratio with mercury (Peterson *et al*. 2009; Essig 2010). This suggests that the risks of adverse effects of mercury in the action area are lower than those observed in conducted at lower selenium to mercury ratios. This observation, in conjunction with: (1) The age and trophic level patterns where juvenile anadromous salmonids tend to have lower exposure to mercury through feeding; and (2) the IDEQ policy of triggering actions for their mercury fish tissue-based water quality standard for existing discharges at 0.24 mg/kg ww (IDEQ 2005) , indicate that the 0.3 mg/kg ww fish tissue based water quality criteria for mercury would likely be sufficiently protective against risks of adverse effects to listed salmon and steelhead, and their habitats.

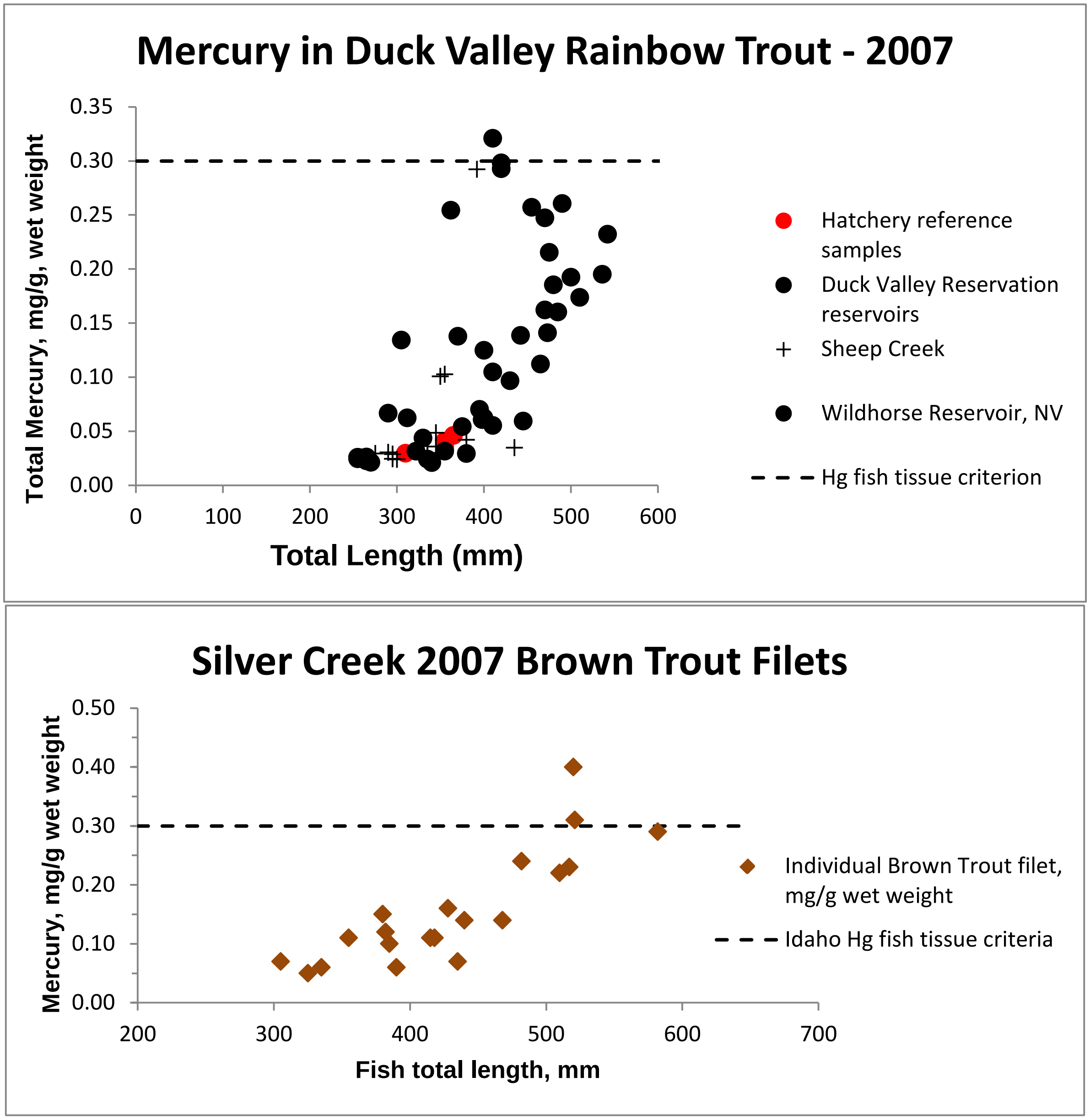



**Figure 2.4.6.2. Concentrations of mercury in salmonid tissues from waters in southern Idaho and northern Nevada versus length. Data from Maret and MacCoy (2008)**

**Table 2.4.6.2. Examples of mercury concentrations in water and or mercury tissue burdens in fish muscle tissue (unless noted otherwise), in relation to the minimal effects threshold of 0.2 mg/kg in tissue and the 12 ng/L criteria in water under consultation.**

| Location or situation | Hg in unfiltered water (ng/L) | Hg in fish tissue (mg/kg, ww) | Fish species | Source |
|---|---|---|---|---|
| Idaho rivers, statewide average (range) in rivers | 0.94 (<0.15 to 6.8) | 0.16 (<0.04 to 1.1) | Various species | (Essig 2010) |
| Idaho rivers, 90th percentile | 1.6 | 0.34 | Various species | (Essig 2010) |
| Chinook salmon returning to Idaho hatcheries | – | 0.149 (0.131 to 0.191) | Chinook fillets | (Essig 2010) |
| Chinook salmon returning to Idaho hatcheries | – | 0.06 (estimated from 0.24 dw) | Chinook, whole carcass | (Felicetti *et al*. 2004) |
| Yankee Fork Salmon R | 3 – 4.6 | 0.08 – 0.19 | Mtn. whitefish, wb | (Rhea *et al*. 2013) |
| Yankee Fork Salmon R | 3 – 4.6 | 0.08 – 0.17 | Shorthead sculpin, wb | Rhea *et al*. 2013) |
| Yankee Fork Salmon R | 3 – 4.6 | <0.05 | Cutthroat trout | (Mebane 2000, citing unpub. USFWS data) |
| Lemhi R. | 0.7 – 0.92 | 0.13 | Mtn. whitefish | (Essig 2010) |
| Pahsimeroi R. | 0.35 – 0.51 | 0.10 | Mtn. whitefish | (Essig 2010) |
| Johnson Creek at Yellow Pine, ID (tributary to SF Salmon R) | 0.70 | – | | (Essig 2010) |
| Camas Creek, ID (tributary to MF Salmon R) | 0.68 | 0.06 | Mtn. whitefish | (Essig 2010) |
| Lochsa R. | 0.54 | 0.05 | Cutthroat trout | (Essig 2010) |
| Selway R. | 0.4 | 0.049 - 0.057 | Cutthroat trout | (Essig 2010) |
| Selway R. | 0.4 | 0.83 | Mtn. whitefish | (Essig 2010) |
| Selway R. | 0.4 | 0.15 | Brook trout | (Essig 2010) |
| Salmon R. ds NF Salmon R | – | 1.1 | N. pikeminnow | D. Essig, p. comm |
| Salmon R. ds NF Salmon R | – | 0.25 | Mtn. whitefish | D. Essig, p. comm |
| Salmon R. ds SF Salmon R | 0.98 – 1.1 | 0.68 | N. pikeminnow | (Essig 2010) |
| Salmon R. ds SF Salmon R | 0.98 – 1.1 | 0.58 | Smallmouth bass | (Essig 2010) |

| Location or situation | Hg in unfiltered water (ng/L) | Hg in fish tissue (mg/kg, ww) | Fish species | Source |
|---|---|---|---|---|
| SF Salmon R. | 1.4 | – | | (Essig 2010) |
| Sugar Creek, tributary to EFSF Salmon River (average and range, 2011 through 2013, n=12) | 2520 (12 to 26,300) | Not measured | | *http://nwis.waterdata.usgs.gov*; site 13311450 |
| NF Clearwater R | 0.23 | 0.11 | Kokanee salmon | (Essig 2010) |
| Stanley Basin lakes | ~ 0.3 (a) | 0.11 – 0.16 | Bull trout | (Essig and Kosterman 2008) |
| Snake R. (Brownlee outlet) | 1.23 | 0.47 – 0.77 | Smallmouth bass | (Essig and Kosterman 2008) |
| Portneuf R., downstream of Lava Hot Springs | 1.89 – 6.8 | 0.4 – 1.1<br>0.25 | Brown trout<br>Rainbow trout | (Essig and Kosterman 2008; Essig 2010) |
| Portneuf R., upstream of Lava Hot Springs | 0.21 | 0.32 – 0.68 | Cutthroat/ rainbow hybrid | (Essig 2010) |
| Salmon Falls Reservoir, ID | 2.3 (0.76 - 4.25) | 0.7 – 1.4 | Walleye | (IDEQ 2007b) |
| Salmon Falls Reservoir, ID | 1.4 – 2.3 | 0.35 | Rainbow trout | (IDEQ 2007b) |
| Silver Creek, ID | 0.15 – 1.45 | 0.5 –and 0.67 (mean of 10 fish each | Brown trout | (Essig 2010) |
| Wilderness lakes in the Seven Devils Mtns, ID and the Wallowa Mtns., OR (geometric means, assuming 80% moisture) | Not measured | 0.03<br>0.05<br>0.05<br>0.43 | Rainbow trout (n=85)<br>Cutthroat trout (n=11)<br>Brook trout (n=230)<br>Lake trout (n=1) | (Eagles-Smith *et al*. 2013) |
| Lake contaminated by chlor-alkali plant wastes | 6 (2 – 12) (filtered) | 1.1<br>0.68<br>0.48 | Piscivores<br>Planktivores<br>Benthivores | (Bloom and Effler 1990; Becker and Bigham 1995) |
| TMDL target for the Willamette R., OR | 0.92 | 0.3 | median for higher trophic level fish | (Hope *et al*. 2007) |
| Streams with a variety of land uses in OR, WI, and FL | 0.6 – 5.7 | 0.02 – 1.2 | Predatory fish | (Chasar *et al*. 2009) |
| Median for streams across the USA | 2.09 | 0.085 | Salmonids | (Scudder *et al*. 2009) |

wb – whole-body, converted from dry weight using 27% moisture; Note a – estimated value from NF Payette River near Grandjean (0.28 ng/L) and Big Wood River, near Galena (0.26 ng/L). Both proxy sites drain watersheds with some similarities in geology and land uses as the Stanley Basin lakes.

***Concentrations of mercury in water associated with mercury tissue residues of concern.*** NMFS examined a variety of matched samples of mercury in water and fish tissue to help evaluate concentrations in water that might produce mercury in fish at concentrations that could be adverse (Table 2.4.6.2). The data presented were selected from datasets that would be directly applicable or at least relevant to salmonids in stream or large river habitats that make up most of the action area. Repeatedly, these matched samples show that mercury concentrations in fish commonly approach or exceed the lowest adverse effect threshold of 0.2 ~ 0.3 mg/kg, even though the mercury concentrations in water were commonly an order of magnitude lower than the Idaho chronic mercury criterion.

This observation leads to the question that if the 12 ng/L water criterion for mercury would likely permit too high mercury concentrations in fish, what concentrations in water likely would likely result in low risk to fish? NMFS took two approaches to answering this question, back calculating from tissue to water using BAFs and by using a regression between matched water and tissue concentrations from Essig's (2010) large study of mercury in fish and water in Idaho rivers.

Using data reported by Essig (2010), a linear relationship between total mercury in river water and fish tissues was calculated for this Opinion: (Tissue residue (µg/kg) = 66 (L/ng))•(total Hg (ng/L) +98.9 µg/kg, $r^2 = 0.22$, $p<0.00001$). The regression equation suggests that a water concentration of 0.9 ng/L total dissolved mercury would, on the average, result in a fish tissue concentration of about 300 µg/kg. This estimated water concentration is effectively the same as the 0.92 ng/L concentration of dissolved mercury in water selected as a TMDL target for the Willamette River, Oregon (Hope *et al*. 2007). Both values are a full order of magnitude lower than the Idaho chronic criterion of 12 ng/L under consultation. Estimating the fish tissue concentration that might result if waters actually were at their allowed 12 ng/L dissolved concentration requires extrapolation, since few rivers or lakes approach this water concentration in their surface waters, even among waters with substantial mercury pollution in their food webs, such as Lake Onodaga, New York, Salmon Falls Reservoir, Idaho, or Brownlee Reservoir, Idaho/Oregon (Table 2.4.6.2). If the linear relationship from the Idaho river data held from the maximum measured value (5.5 ng/L) to 12 ng/L, the predicted fish tissue concentrations would be around 0.9 mg/kg (0.67 to 1.9, 95$^{th}$ confidence intervals).

An alternative approach is to use BAFs to estimate potential mercury tissue residues from mercury concentrations in water measured in the field. We examined two sources of BAF estimates; those compiled by DeForest *et al*. (2007) using field data compiled from peer-reviewed literature and technical reports (e.g., USGS Water-Resources Investigations reports); and the Idaho probabilistic survey of mercury species in water and mercury in muscle tissue of edible fish (Essig 2010). DeForest *et al*'s (2007) analysis included BAFs from water concentrations in excess of 12 ng/L, Essig's (2010) BAFs were all developed from a more limited range of mercury in water concentrations, 0.2 to 5.5 ng/L. The BAFs we estimated from these independent data sets were remarkably similar (Table 2.4.6.3). From the BAFs derived from both studies, on the average, a water body that was at the NTR (Idaho) chronic criterion of 12 ng/L would be expected to eventually bioaccumulate to produce fish mercury residues of around 3 mg/kg ww, with a range of 0.5 to 20 mg/kg or more. Thus, even the lowest BAF estimates used with the 12 ng/L water criterion would predict a muscle tissue residue greater than the 0.3 mg/kg ww threshold selected here. A muscle tissue residue of 3 mg/kg is 10 times higher than the 0.3 mg/kg threshold for risks of adverse effects selected from this review. Adverse effects in fish that have been linked with muscle tissue residues on the order of 3 mg/kg ww include complete reproductive failure, brain damage, and severe behavioral abnormalities (Table 2.4.6.1). If the BAF estimates were used to backcalculate potential total mercury concentrations in water from the 0.3 mg/kg fish tissue concentration, the total mercury in water values would range from about 0.2 to 7 ng/L, with a geometric mean value of about 1.7 ng/L. If rounded to the nearest integer to avoid implying greater precision than one significant digit, this implies on the average if rivers had total mercury concentrations less than 2 ng/L, predicted concentrations of mercury in fish tissue would be expected to be less than 0.3 mg/kg wet weight.

**Table 2.4.6.3. Ranges of potential tissue concentrations that would result from (A) applying field-based BAFs to the chronic mercury water quality criterion of 12 ng/l, and (B) ranges of water concentrations that would result from applying BAFs to low-risk tissue concentrations. Calculations showing the laboratory water-only bioconcentration factor (BCF) used in EPA (1985g) to derive the 12 ng/L criterion are also included for comparison.**

**A. Estimated mercury concentrations resulting in fish if mercury in water were 12 ng/L:**

| Scenario | Water total Hg (ng/L) | Total Hg BAF | Predicted fish tissue (ng/kg ww) | Predicted fish tissue (mg/kg ww) |
|---|---|---|---|---|
| DeForest *et al*. (2007) BAFs | | | | |
| Geometric mean | 12 | 263,362 | 3,160,344 | 3.2 |
| Minimum | 12 | 40,857 | 490,284 | 0.5 |
| Maximum | 12 | 4,110,638 | 49,327,656 | 49.3 |
| Essig (2010) BAFs | | | | |
| Average | 12 | 249,480 | 2,993,756 | 3.0 |
| Geometric mean | 12 | 178,968 | 2 147 620 | 2.1 |
| Minimum | 12 | 42,632 | 511,579 | 0.5 |
| Maximum | 12 | 1,635,294 | 19,623,529 | 19.6 |
| EPA (1985) BCF | | | | |
| Fathead minnow | 12 | 81,700 | 1,000,000 | 1.0 |

**B. Estimated mercury concentrations in water resulting in fish tissue concentrations of 0.3 mg/kg ww:**

| Scenario | Fish tissue (mg/kg ww) | Fish tissue (ng/kg ww) | Total Hg BAF | Predicted water total Hg (ng/L) |
|---|---|---|---|---|
| DeForest *et al*. (2007) BAFs | | | | |
| Geometric mean | 0.3 | 300,000 | 263,362 | 1.1 |
| Minimum | 0.3 | 300,000 | 40,857 | 7.3 |
| Maximum | 0.3 | 300,000 | 4,110,638 | 0.1 |
| Essig (2010) BAF values | | | | |
| Average | 0.3 | 300,000 | 249,480 | 1.2 |
| Geometric mean | 0.3 | 300,000 | 178,968 | 1.7 |
| Harmonic mean | 0.3 | 300,000 | 138,215 | 2.2 |
| Minimum | 0.3 | 300,000 | 42,632 | 7.0 |
| Maximum | 0.3 | 300,000 | 1,635,294 | 0.2 |
| EPA (1985) BCF | | | | |
| Fathead minnow | 0.3 | 300,000 | 81,700 | 3.6 |

*2.4.6.3. Summary for Mercury*

The 1984 chronic mercury criterion was back calculated from the FDA limit for allowable mercury content in commercially marketed seafood (1.0 mg/kg ww), using a bioconcentration factor derived from a laboratory water-only (aquaria) methylmercury exposures with fathead minnow (USEPA 1985g). Thus, the criterion derivation had no consideration of ecological effects of mercury or effects of mercury to sensitive species. In the 25 plus years since this fish marketability-based criterion was developed, much new information on the effects of mercury on the fish themselves, not just their marketability, has been developed. The newer information both reflects that: (1) The older bioconcentration values considered in the 1984 chronic criterion were about four times lower than the average bioaccumulation factors obtained in field settings; and (2) that adverse developmental effects in fish occur at <1 mg/kg.

Severe adverse effects have been observed in fish that accumulated mercury in their muscle tissue, including brain damage, behavioral abnormalities, and reproductive failure. However, effects of methylmercury on fish are not limited to neurotoxicity, but also include histological changes in the spleen, kidney, liver and gonads. These effects have been observed in multiple species of freshwater fish at tissue concentrations of methylmercury well below 1.0 mg/kg ww (Sandheinrich and Wiener 2010).

## 2.4.7. The Effects of EPA Approval of the Nickel Criteria

The acute and chronic nickel criteria being consulted on are 470 µg/L and 52 µg/L respectively (Table 1.3.1).

*2.4.7.1. Species Effects of Nickel Criteria*

Nickel poisoning in fish can cause respiratory stress, convulsions, and loss of equilibrium prior to death. Adverse respiratory effects occur through destruction of gill tissues by ionic nickel and subsequent blood hypoxia. Other effects include decreased concentrations of glycogen in muscle and liver tissues, simultaneous increases in lactic acid and glucose in the blood, and interference with metabolic oxidation-reduction processes (Eisler 1998b). In general, the egg and embryo stages of salmonids are the most, and older stages the least, sensitive to nickel toxicity (Nebeker *et al.* 1985). Nickel is thought to have lower inherent toxicity to fish than other criteria metals for which aquatic criteria have been developed (Niyogi and Wood 2004).

Available toxicity test data indicate that juvenile and adult salmon and steelhead are protected from acute effects of nickel at the acute criterion (Figure 2.4.7.1). However, several studies have determined that mortality of salmonid embryos occurs over longer-term exposures to concentrations that are below the Idaho chronic criterion:

> Birge *et al*. (1978) determined a 30 day $LC_{50}$ for rainbow trout embryos of 50 µg/L at a water hardness between 93 mg/L and 105 mg/L.

> Eisler (1998b) cite an $LC_{10}$ of 11 µg/L, no hardness given, for rainbow trout embryos exposed from fertilization through hatching.
>
> Birge *et al*. (1981) concluded that nickel concentrations of 10 µg/L would not impair reproduction of most aquatic species although adverse effects at concentrations were not substantially greater than this.
>
> In Eisler's (1998b) review, $LC_{50}$s were reported of 60 µg/L and 90 µg/L at water hardness of 125 and 174 mg/L, respectively, for rainbow trout embryos that were exposed from fertilization through hatching.
>
> Nebeker *et al*. (1985) found that the sensitivity of rainbow trout to long-term nickel exposures varied depending upon the developmental stage the test was started. Unlike tests with some other metals (cadmium, zinc, and maybe copper and Pb), tests initiated with swim-up fry were much less sensitive than tests started with either eyed or newly fertilized eggs. Newly fertilized eggs were most sensitive with reduced growth observed at the lowest concentration tested, 35 µg/L at a hardness of 27-39 mg/L. The Idaho chronic criterion at this range of hardnesses is 52 to 71 µg/L, which is higher than the lowest concentration causing adverse effects. However the chronic criterion over this hardness range from EPA's 2013 updated action is 17 to 23 µg/L, which is lower than the adverse effects concentration. This is illustrated in Figure 2.4.7.2, where the "Idaho chronic values" shown are the same as EPA's 2013 updated action.
>
> Brix *et al*. (2004) tested newly fertilized rainbow trout eggs using a similar test design to Nebeker *et al*'s (1985) tests, using a higher hardness dilution water of about 91 mg/L. No adverse effects were reported from exposures up to 466 µg/L.

These results suggest that adverse effects could occur to embryos exposed to nickel concentrations that are lower than the Idaho chronic criterion for nickel which was evaluated by EPA (2000a). However, the contrasting responses of the Nebeker *et al*. (1985) and Brix *et al*. (2004) indicate there is yet considerable uncertainty in risks of nickel to aquatic life. Idaho's current criteria for nickel include a chronic criterion for nickel that is lower than Nebeker's adverse effect level; at hardnesses of 27 to 39 mg/L, the 2002 chronic nickel criterion is 17 to 23 µg/L (Table 1.3.1).

***Behavioral Effects.*** One study was located that suggested behavioral avoidance could potentially occur at concentrations that are below the proposed chronic criterion:

> Giattina *et al*. (1982) determined that rainbow trout fry avoided a nickel concentration equal to 24 µg/L at a mean water hardness of 28 mg/L. This effect concentration is greater than the updated IWQS chronic criterion (18 µg/L for hardness 28 mg/L), which is reflected in EPA's 2013 updated action.

***Hardness as a Predictor of Nickel Toxicity.*** In meta-analyses of acute toxicity data for nickel with Daphnia magna and rainbow trout Meyer *et al*. (2007b) found that, toxicity tended to decrease with increases in alkalinity, pH, and hardness. However, the relations were fairly weak, and a similar analysis of data for fathead minnows showed no relationship between hardness and toxicity (Meyer *et al*. 2007b). Deleebeeck *et al*. (2007) investigated: (1) Whether cladocerans living in soft water (< 10 mg $CaCO_3$/L) are intrinsically more sensitive to nickel than cladocerans living in "hard water" (hardness > 25 mg $CaCO_3$/L) in chronic exposures; and (2) whether a single bioavailability model can be used to predict the protective effect of water hardness on the toxicity of nickel to cladocerans in both soft and hard water. Their results found that water hardness significantly reduced nickel toxicity to both the soft and the hard water organisms tested.

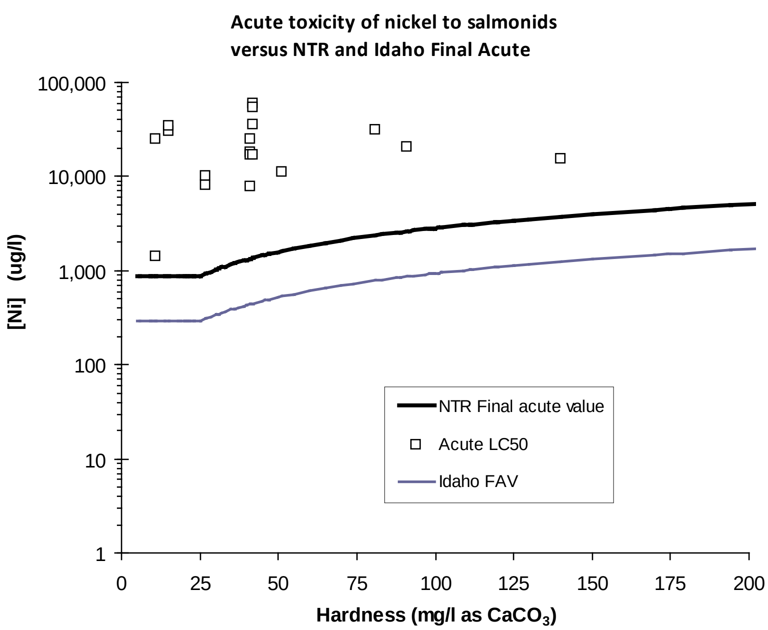



**Figure 2.4.7.1 Acute $LC_{50}$s for nickel with rainbow trout, any life stage (no data on other salmonids) vs. the Idaho and Idaho final acute values (FAVs).**

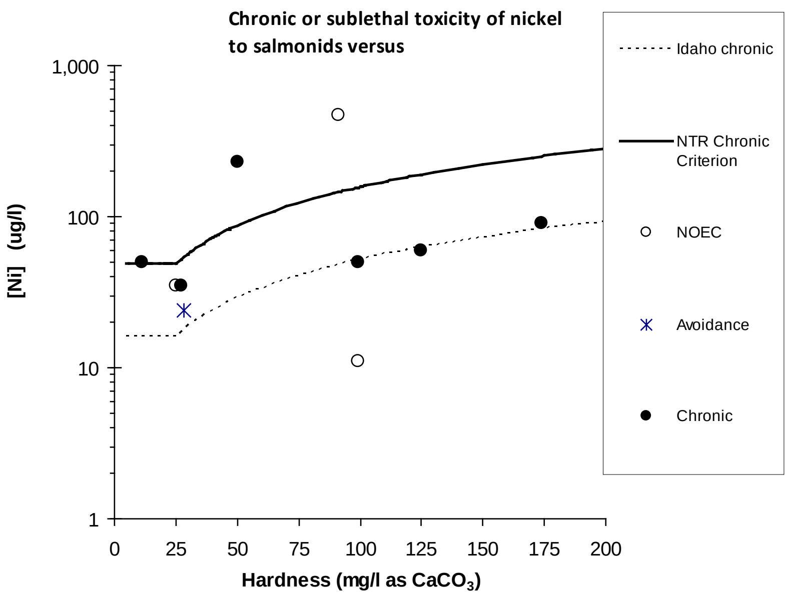



**Figure 2.4.7.2. Chronic effects, no-observed effect concentrations, and avoidance concentrations with rainbow trout vs. the NTR and Idaho chronic values for nickel.**

*2.4.7.2. Habitat Effects of Proposed Nickel Criteria*

***Toxicity to Food Organisms.*** In some instances, nickel can be quite toxic to invertebrates such as zooplankton and amphipods. In soft waters, thresholds of effects ($EC_{10}$ values) for zooplankton range from only about 2 to 40 µg/L in waters with hardnesses ranging from about 6 to 43 mg/L (Deleebeeck *et al*. 2007). Lethal concentrations ($LC_{50}$s) to the freshwater amphipod *Hyalella azteca* ranged from 77 to 147 µg/L in soft and hardwater (18 and 130 mg/L) (Borgmann *et al*. 2005a). This suggests that concentrations causing no or few effects would be in the 20 to 70 µg/L range in hard or soft water, assuming common concentration-response patterns (Section 2.4.1.6). In life cycle tests caddisflies were affected at concentrations greater than 66 µg/L in waters with hardness of about 25 to 30 mg/L (Nebeker *et al*. 1984). Criteria in softwaters comparable to those used in these softwater tests (25 µg/L) would be about 16 µg/L for the state of Idaho's updated criteria (Table 1.3.1). Thus, at least above the "*hardness floor of 25 mg/L*", the state of Idaho's updated chronic nickel criterion would likely be protective of sensitive invertebrates.

***Bioaccumulation.*** Nickel is known to bioaccumulate in salmonids, which can accumulate through both dietary and water-borne exposure routes (EIFAC 1984; Eisler 1998b). Bioconcentration factors vary substantially both within and between species, age of organism, and with exposure concentration. Bioconcentration has been noted to occur in kidney, liver, and muscle tissues of rainbow trout exposed to ambient water concentrations of nickel equal to 1000 µg/L for 6 months, but the test fish were able to depurate much of the accumulated nickel within 3 months after exposure was terminated and were not visibly affected during the experiment (Calamari *et al*. 1982). Studies of saltwater and freshwater fish species have determined that piscivorous fish bioaccumulate greater levels of nickel in muscle tissues than other fish, indicating the potential for biomagnification to occur (albeit to a limited extent according to most studies; EIFAC 1984; Eisler 1998b). There is evidently a risk of bioaccumulation from chronic nickel exposure, but it remains unknown to what extent this is a significant hazard for listed salmon and steelhead.

#### *2.4.7.3. Summary for Nickel*

A striking feature of the information reviewed for nickel toxicity is the tremendous range of effects concentrations. Much work, particularly short-term exposures, has shown adverse effects from nickel at concentrations in the milligrams per liter range, which are hundreds or even thousands of times higher than environmentally relevant concentrations. Yet other work has shown nickel to be about as toxic or more toxic, in long-term exposues than metals more commonly considered to pose a risk to sensitive organisms, such as copper or cadmium. No reports were located of adverse effects from short-term (96-hr) toxicity tests using salmonids at concentrations below the final acute value (two times the acute criterion) for nickel.

During this consultation, EPA revised the proposed chronic criterion for nickel resulted in a level that is considerably more protective of listed salmon and steelhead. Potential adverse effects from exposure to nickel at concentrations at or below the criterion in the revised action are expected to be primarily to sensitive invertebrates which may be a food source for listed species. This affect is expected to be very small.

### 2.4.8. The Effects of EPA Approval of the Selenium Criteria

Selenium in water is a particularly challenging substance to evaluate risks to listed salmon and steelhead because of many contradictions in the available science and controversies of interpretation. Selenium is an essential micronutrient for all animals that have a nervous system, yet it is toxic at not much higher concentrations. At optimal concentrations, selenium is an antioxidant nutrient with positive effects on the immune system in mammals and birds, yet oxidative stress seems to be the principal mechanism of toxicity in animals and may compromise immune function at higher concentrations (Burk 2002; Palace *et al*. 2004; Miller *et al*. 2007; Janz *et al*. 2010). The cell damage caused by oxidative stress in turn can lead to a cascade of symptoms that include edema in developing embryos; teratogenic deformities in offspring; spinal deformities; anemia; cataracts; popeye; pathological alterations in liver, kidney, heart, and ovary; reduced egg viability; and reduced growth of juveniles. Oviparous (egg laying) vertebrates appear to be the most sensitive taxa to selenium toxicity (Lemly 2002; Janz *et al*. 2010). Selenium has been called an insidious threat in waters where it is elevated, because adult fish may appear perfectly healthy, whereas severe effects may be occurring to early life stage fish but not be noticed in routine surveys until a large percentage of the year classes are affected (Lemly 2002). Because of concerns over effects of selenium, a large amount of research has been focused on effects of selenium to wildlife and aquatic life. Over 120 references pertinent to effects of selenium on the freshwater life stages of salmonids were located and reviewed for our analysis.

Idaho's chronic aquatic life criterion for selenium of 5 µg/L is unique in that it is based on "other data" rather than the usual approach that uses the 5$^{th}$ percentile of the SSD in conjunction with an ACR. The "Other Data" provision in EPA's Guidelines for developing aquatic life criteria serves to allow the use of pertinent information that could not be used directly in the usual species ranking, etc. approach. Data from any type of adverse effect that has been shown to be biologically important could be used, such as data from behavioral, biochemical, physiological, microcosm, and field studies. If the "other data" show that a lower criteria value should be used instead of the usual final chronic value, then the CCC would be based on this "other data" (Stephan *et al*. 1985, section X.) To NMFS' knowledge, selenium is the only substance for which the "other data" were sufficiently compelling to adjust a chronic water quality criterion.

The adverse effects attributable to selenium from a well documented field study were both severe (decimation of fish populations in a reservoir with elevated selenium) and occurred at lower selenium concentrations than were calculated from the laboratory studies on toxicity available at the time. In Belews Lake, a reservoir in north-central North Carolina that received fly ash from a coal power plant, selenium concentrations in water reached about 10 µg/L in the main body of the lake. Populations of several fish species suffered recruitment failure and then collapsed. In an arm of the reservoir that had limited circulation with the main body of the lake and selenium in water was below or near the detection limit of 5 µg/L, the fish assemblage was mostly intact. Therefore, EPA set the recommended chronic criterion at the detection limit available during the studies, 5 µg/L. This concentration was EPA's best estimate of a concentration that was intended to be protective, but also generally attainable based upon the information available to them at the time. Subsequently; however, pronounced adverse effects have been discovered in low selenium areas of the reservoir and other locations, at water selenium concentrations down to less than 1 µg/L (EPA 1998). In another twist from the usual approach, the acute criterion was back calculated from the field-based chronic criteria, using a laboratory water based acute:chronic ratio of 8 (EPA 1987a).

The combined notoriety of the Belews Lake and similar cases and the occurrences of severely deformed aquatic bird embryos in western reservoirs and wetlands that received elevated selenium in irrigation return water (e.g., Presser 1994) led to much research on selenium bioaccumulation and toxicity in aquatic organisms. Thus, a large body of knowledge has become available subsequent to EPA's 1987 selenium criteria document.

Recently, several key areas of consensus in the scientific community have formed regarding selenium risks to aquatic life and criteria to protect them:

- Diet is the primary pathway of selenium exposure for both invertebrates and vertebrates.

- Traditional methods for predicting toxicity on the basis of exposure to dissolved concentrations do not work for selenium because the behavior and toxicity of selenium in aquatic systems are highly dependent upon situation-specific factors, including food web structure and hydrology.

- Selenium toxicity is primarily manifested as reproductive impairment due to maternal transfer, resulting in embryotoxicity and teratogenicity in egg laying vertebrates (Janz *et al.* 2010).

Because adverse effects in fish could be better related to selenium residue concentrations in tissues than concentrations in water, recent efforts to evaluate, refine, or develop site-specific aquatic life criteria or thresholds have focused on selenium residues in fish (EPA 1998; DeForest *et al*. 1999; Hamilton 2002, 2003; EPA 2004; deBruyn *et al*. 2008; DeForest 2008; Janz *et al*. 2010). However, while consensus seems to have been reached that an aquatic life criteria for selenium could be based on tissue concentrations, just what number that should be has been the subject of considerable dispute, with proposed threshold values ranging from about 4 to 11 mg/kg, as whole-body dw, part per million (DeForest *et al*. 1999; Hamilton 2003; Skorupa *et al*. 2004; EPA 2004). The EPA (2004) proposed a fish tissue criterion of 7.9 mg/kg dw in fish, with an summer or fall monitoring trigger of 5.8 mg/kg dw, which was primarily based on a dietary toxicity study with bluegill, *Lepomis macrochirus*, under a simulated winter temperature and photoperiod regime. Subsequently, the EPA has conducted additional testing of bluegill under different temperatures and published a call for "scientific information, data, or views on the draft selenium aquatic life criterion" (*http://www.epa.gov/waterscience/criteria/selenium/* and *Regulations.gov docket* (EPA-HQ-OW-2004-0019)). As of April 2009, about 268 responses had been posted to this site, ranging from statements of opinion to complex research reports or original research and interpretative analyses.

This plethora of competing information presented a challenge to resolving the questions of the present review. These include three questions:

1. What concentration of selenium in which fish tissues is a sufficiently low threshold to protect listed salmon and steelhead?

2. Would the effective chronic aquatic life criterion of 5 µg/L selenium in water likely result in bioaccumulation of selenium to levels in tissues that are less than the fish-tissue threshold identified in Question 1 (i.e., is likely protective), or that are greater than the fish-tissue thresholds (i.e., is likely under protective)?

3. If the answer to Question 2 is the latter (underprotective), what concentration in water likely would be protective, i.e., would not result in bioaccumulation to threshold levels in fish?

NMFS reviewed over 120 scientific articles and technical reports in attempts to best answer these questions. These questions are addressed in the "chronic effects" subsection below.

#### *2.4.8.1. Species Effects of Selenium Criteria*

The aquatic life criteria for selenium under consultation are an acute criterion of 20 µg/L and a chronic criterion of 5 µg/L, both expressed as "total recoverable" unlike the dissolved criteria for most metals.

***Acute Selenium Criterion.*** Because risks of selenium to aquatic life are via the food chain, the traditional acute toxicity testing database provides no information of value to understanding selenium toxicity in nature. Since a water-based criterion to protect against short-term exposures is environmentally meaningless (Chapman *et al*. 2009; Janz *et al*. 2010; Janz 2011), it is not reviewed in detail here. For example, 96-hour $LC_{50}$ values for juvenile rainbow trout range from 4,200 µg/L to 47,000 µg/L, which are at least 200 times greater than the acute criterion of 20 µg/L (Janz 2011). Unlike all other EPA criteria documents, the acute selenium criterion was not developed from acute toxicity test data, but was back-calculated from the chronic, field-based criterion.

***Chronic Selenium Criterion.*** Because the chronic criterion was derived from a field study where selenium exposure and effects occurred via reproductive failure linked to bioaccumulation from the food web, not water exposures (EPA 1987a; Sorensen 1991; Lemly 1997), and because adverse effects of selenium are associated with selenium in tissues rather than concentrations in water-only exposures of selenium, only toxicity associated with the bioaccumulation of selenium in tissue residues are considered.

#### *2.4.8.2. Habitat Effects of Selenium Criteria*

***Toxicity to Food Organisms.*** Macroinvertebrates have typically only been considered dietary sources of selenium to higher trophic levels, in part based on Lemly's (1993a) conclusion that the most important aspect of selenium residues in aquatic food chains is not direct toxicity to the organisms themselves, but rather the dietary source of selenium they provide to fish and wildlife species that feed on them. Lemly (1993a) based his conclusion on review of field and laboratory studies in which he found the lowest threshold adverse effects was reduced growth of adult *Daphnia magna* at tissue residues of 20 mg/kg dw, and reduced reproduction occurred at 30 mg/kg dw. Selenium in the diets of fish can cause adverse effects at less than half this concentration. deBruyn and Chapman (2007) challenged that assumption in a commentary which argued selenium may cause toxic effects to some freshwater invertebrate species at concentrations considered "safe" for their predators. Preliminary results presented by Conley *et al*. (2009) further suggested that if mayflies were exposed to selenium through a more natural feeding regime, maternal transfer to eggs and adverse effects to progeny could occur at dietary concentrations as low as about 11 mg/kg dw. Studies of long-term experimental selenium dosing of experimental streams also noted that elevated selenium concentrations affected the structure of macroinvertebrate communities and were more important to ecosystem structure and function than simply through their role as food for fish and birds. For instance, isopods were depressed following long-term exposures to about 10 and 30 µg/L selenium, which may have resulted in a competitive release that directly supported higher densities of amphipods, and indirectly supported an extremely high population density of baetid mayfly and damselfly nymphs (Swift 2002). Thus, assuming that macroinvertebrates simply act as a conduit of selenium to higher trophic levels may not be accurate. However, the literature NMFS reviewed does not indicate that elevated selenium would lead to profound community-level impacts to macroinvertebrates that would limit food resources. For instance, baetid mayflies and amphipods are probably at least as nutritious food sources for juvenile salmonids as isopods. Thus the primary concern with selenium in stream food webs does not appear to be one of food limitation but rather as trophic transfer.

Selenium is an essential trace element for fish at dietary concentrations of 0.1 to 0.5 mg Se/kg dw. In fish, selenium toxicity has been reported to occur at dietary concentrations only seven to 30 times greater than those considered essential for proper nutrition (i.e., > 3 mg Se/kg dw) (Janz *et al*. 2010). There have been efforts to use selenium residue concentrations in salmonid fish prey organisms in effects monitoring and assessment targets. For instance, a food web monitoring plan in Thompson Creek, Idaho, a stream that receives mine wastewater effluents with elevated selenium concentration, set a maximum residue guideline of 4 mg/kg dw in aquatic insects or forage fish (Mebane 2000). However, despite the consensus that diet is the sole important route of selenium exposure to fish, the approach of setting dietary guidelines in monitoring and assessment seems to have received little subsequent attention. Instead, most recent research has been aimed at developing protective guidelines for avoiding selenium toxicity in fish and has focused on effects attributable to the residues in the fish themselves, rather than on defining dietary adverse effect thresholds for selenium in diet.

One study we reviewed in detail tested the effects of organic selenium in the diets of juvenile Chinook salmon (Hamilton *et al*. 1990). They fed the salmon using two diets, one contained meal made from low-selenium mosquitofish (collected from a reference site) fortified with SeMe, and a second diet that contained fish meal made from high-selenium mosquitofish collected from a selenium-laden drain (SLD) located in an intensely irrigated agricultural watershed in California. These diets are likely much more biologically relevant than various studies that studied effects of dietary selenium administered through commercial trout chows or other feeds fortified with sodium selenite or other inorganic selenium species. This is because selenium that has been incorporated into living tissues of plants or animals is likely present as organic selenium (Besser *et al*. 1993).

By analyzing growth reductions occurring after 60 days of exposure to a range of SeMe concentrations in the mosquitofish meal diet, a threshold for the onset of effects was estimated at about 7.6 mg/kg selenium in the diet as dw (Figure 2.4.8.1(A)).

The Hamilton *et al*. (1990) feeding studies were conducted for 90 days, but survival in the control groups dropped from 99% at 60 days to 67% at 90 days. Lower survival occurred in all selenium treatments at 90 days compared to 60 days, and effect concentration percentile (ECp) estimates based on growth or survival were lower as well (i.e., were more sensitive). Because other studies have had high survival rates in similar aged Chinook salmon controls (e.g., Chapman 1982), it seems possible that the lower survival with the 90-day results were influenced by some undetected factor such as disease or parasitism. Thus, only the 60-day results are relied upon here.

***Tissue concentrations of selenium associated with chronic responses in salmonids.*** In natural waters and food chains, selenium most commonly occurs as inorganic selenium in two forms. Selenate, $SeO_4^{2-}$, is an anion that tends to predominate in oxic conditions, and selenite, $SeO_3^{2-}$, is is an anion that tends to predominate in reducing conditions. Both forms are readily taken up by floating or attached algae (phytoplankton or periphyton) or bacteria and then by aquatic invertebrates, fish, and birds. Selenium typically biomagnifies strongly from water to algae, with biomagnification factors ranging from the hundreds to >10,000. Selenium is also converted from the inorganic to organic forms by algae or bacteria. Organic selenium is readily bioavailable to higher trophic levels and bioaccumulates from algae to invertebrates to fish or birds. However, further biomagnification is much lower than from the water to primary producers (Besser *et al*. 1993).

The harmful effects of bioaccumulated selenium on fish have generally been detected through two distinct types of studies and effects. The first, maternal transfer of selenium to developing embryos, may follow from the exposure of adult female fish to selenium with resulting embryo/fry teratogenesis, edema, and mortality. Experimentally, these effects are usually detected by either dietary exposure of broodstock to selenium, or by capturing fish exposed in the wild in areas with elevated selenium, stripping eggs and milt, and evaluating the larval development in the laboratory. The second type of effects and studies are growth reductions or mortality resulting from direct exposure of juveniles to selenium (Janz *et al*. 2010).

Of these two study types, only the latter, the direct exposure of juveniles, is considered relevant to the potential exposures of listed anadromous salmon and steelhead within the action area. Because of the nearly total cessation of feeding by anadromous salmon and steelhead as they re-enter freshwater to start their spawning migration, dietary exposure of adult females within freshwater would be very small. A large body of science on the effects of selenium via maternal transfer or reduced fecundity in non-anadromous salmonids or other fish species such as cutthroat trout, rainbow trout, bull trout, brook trout, northern pike, bluegill and minnow did not seem to be very relevant for estimating effects on listed anadromous species and thus is not further considered (e.g., Gillespie and Baumann 1986; Schultz and Hermanutz 1990; Hermanutz 1992; Hermanutz *et al*. 1992; Hermanutz *et al*. 1996; Kennedy *et al*. 2000; Palace *et al*. 2004; de Rosemond *et al*. 2005; Holm *et al*. 2005; Muscatello *et al*. 2006; Van Kirk and Hill 2007; Rudolph *et al*. 2008; Hardy *et al*. 2010).

NMFS evaluated several studies on direct exposure of juvenile salmonids to selenium were evaluated in attempts to estimate thresholds for “safe” or very low, inconsequential, effects of selenium tissue residue on growth or survival. Summaries of the evaluations are presented in Table 2.4.8.1. Analyses of three of the four studies reviewed resulted in low-effect estimates ranging only from about 5 to 7 mg/kg dw, with one considerably higher estimate at 11 mg/kg dw. The latter estimate is considered the least reliable of the values because it required an extrapolation from liver to whole-body residue, using a relationship established with a different species of fish (Table 2.4.8.1.)

Additionally, results from the feeding study with rainbow trout and organic selenium by Vidal *et al*. (2005) were particularly challenging to interpret, because of lack of monotonic response with increasing dietary exposures. Growth was reduced in all dietary exposures, but the lowest growth reductions occurred at the highest dietary exposure. The lowest dietary exposure resulted in a whole body concentration of 0.58 mg/kg ww, about 2.9 mg/kg dw at 90 days, which was similar to the control concentrations (1.6 to 6.2 mg/kg dw). To further complicate matters, whole body selenium residues in all dietary treatments including the controls peaked at 60 days and then declined by 90 days, making it unclear what residue concentration was most associated with effects.  The threshold of adverse effects concentration selected by Vidal *et al*. (2005) is listed in Table 2.4.8.1, although that seems to have been supported by their informal judgments rather than any statistical analyses.

The remaining two tests considered were with rainbow trout and Chinook salmon by Hunn *et al*. (1987) and Hamilton *et al*. (1990) respectively.  Of these, we place greater reliance for estimating thresholds of effect on the latter study in which reduced survival and growth occurred in juvenile Chinook salmon fed organic selenium (Hamilton *et al*. 1990).  This is because the dietary exposures by Hamilton *et al*. (1990) seemed more relevant to the type of exposures that juvenile salmonids would receive in the wild, the selenium exposures and resulting residues bracketed an environmentally relevant range of concentrations, and the results showed clearly non-adverse and adverse effects, such that robust statistical analyses could be made.

Hamilton *et al*'s (1990) tests included two series in which fish were either fed Oregon moist pellets that had been fortified with organic selenium, as SeMe or a meal made with natural-origin caught forage fish captured from the San Luis Drain, an irrigation wasteway with elevated selenium as well as other contaminants.  Using the SeMe treatment effects after 60 days exposure, by logistic regression, we estimated an essentially no-effect concentration for weight reductions at about 3 mg/kg selenium as whole-body dry weight residues.  Low-effect thresholds as a 10% reduction in weight and a 4% reduction in length ($EC_{10}$ and $EC_{04}$) were both similarly estimated at about 7.6 mg/kg (Figure 2.4.8.1 (B and C) 2.4.8.3*)*.  The $EC_{04}$ statistic for length reduction was used as an estimate of a threshold for low effects that could be biologically important because in population modeling with Chinook salmon, a 4% length reduction was projected to have low risk for increased extinction risk, although it could result in a delay in population recovery (Mebane and Arthaud 2010).

The effects concentrations estimated from the fish fed the San Luis Drain diet were lower than the SeMe fortified feed (i.e., apparently more sensitive to Se).  In the San Luis Drain series, reduced growth occurred in all selenium exposures relative to controls *(*Figure 2.4.8.1(D)). However, inspection of the whole-body selenium and growth curve show that after the first treatment, the slope of the curve is flat with few further reductions in growth until the highest treatment.  If the control treatment were excluded from the regression, then the nonlinear curve fit is almost perfect (Figure 2.4.8.1(E)).  This suggests that something besides selenium may have been in the wild San Luis Drain diets that contributed to the reduced growth and survival, such as unmeasured pesticides or other farm chemicals that presumably would be present in a drain in an intensely cultivated farm region.  Thus we relied on the SeMe test series for selecting no- and low-effects thresholds in preference to the San Luis Drain series.

**Table 2.4.8.1. Effects concentration (EC) estimates for selenium whole-body tissue residues on growth or survival of juvenile salmonids, assuming no maternal pre-exposure. Underlined value indicates EC selected for primary effects analysis.**

| Species | Se, whole-body residues (µg/g dw) | Effects | Exposure | Notes and data sources |
|---|---|---|---|---|
| Chinook salmon | 12 | Reduced survival (EC10) | Seleno-methionine (SeMe) in diet, 60 days | Calculated from data reported in Hamilton *et al*. (1990), using threshold sigmoid regression. |
| Chinook salmon | 6.5 | No effect on survival (EC0) | SeMe in diet, 60 days | Calculated from data reported in Hamilton *et al*. (1990), using threshold sigmoid regression. |
| Chinook salmon | **<u>7.6</u>** | Reduced growth (EC10 for weight ≈ EC04 for length | SeMe in diet, 60 days | Calculated from data reported in Hamilton *et al*. (1990), using threshold sigmoid regression |
| Rainbow trout | 7.2 | Reduced growth, EC04 for length | Sodium selenite in water, 60 days | Calculated from data reported in Hunn *et al*. (1987), using piecewise regression |
| Rainbow trout | 3.5<br>5.3 | EC0 and, EC10 for survival, respectively | Sodium selenite in water, 60 days | Calculated from data reported in Hunn *et al*. (1987), using threshold sigmoid regression |
| Rainbow trout | 6.0 | Reduced growth, LOEC | SeMe in diet, 90 days | LOEC selected by Vidal *et al*.(2005), assuming 20% solids (Jarvinen and Ankley 1999; Essig 2010) |
| Rainbow trout | 11 | NOEC for growth or biochemical parameters | Sodium selenite in low carbohydrate diet, 112 days | Estimated from a Se concentration in liver (38 µg/g dw) reported by Hilton and Hodson (1983), using a liver:whole-body regression developed with bluegill (deBruyn *et al*. 2008) |

Hunn *et al*. (1987) achieved a gradient of tissue residues in juvenile rainbow trout by exposing them to waterborne selenite. Estimates of effect concentrations linked to a given tissue residue after 60 days results in very similar effects concentration estimates as were obtained from Hamilton *et al*'s (1990) Chinook salmon and SeMe test. At 60 days, an $EC_{10}$ for survival of about 5.3 mg/kg, and an EC01 of 3.5 mg/kg were estimated, and for length reductions, an $EC_{04}$ of around 7.2 mg/kg dw was estimated. These effects concentration estimates are similar to those for reduced growth from the juvenile Chinook. The choice to give these results lesser importance in this analysis than those of Hamilton *et al*. (1990) is admittedly debatable, since even though the selenium was derived from different sources (water vs. spiked diet), it may be that once metabolized into fish tissues, selenium may have the same toxic effects on a gram per gram basis (EPA 1998). However, because the effect values are similar, this issue does not have to be resolved for the present Opinion.

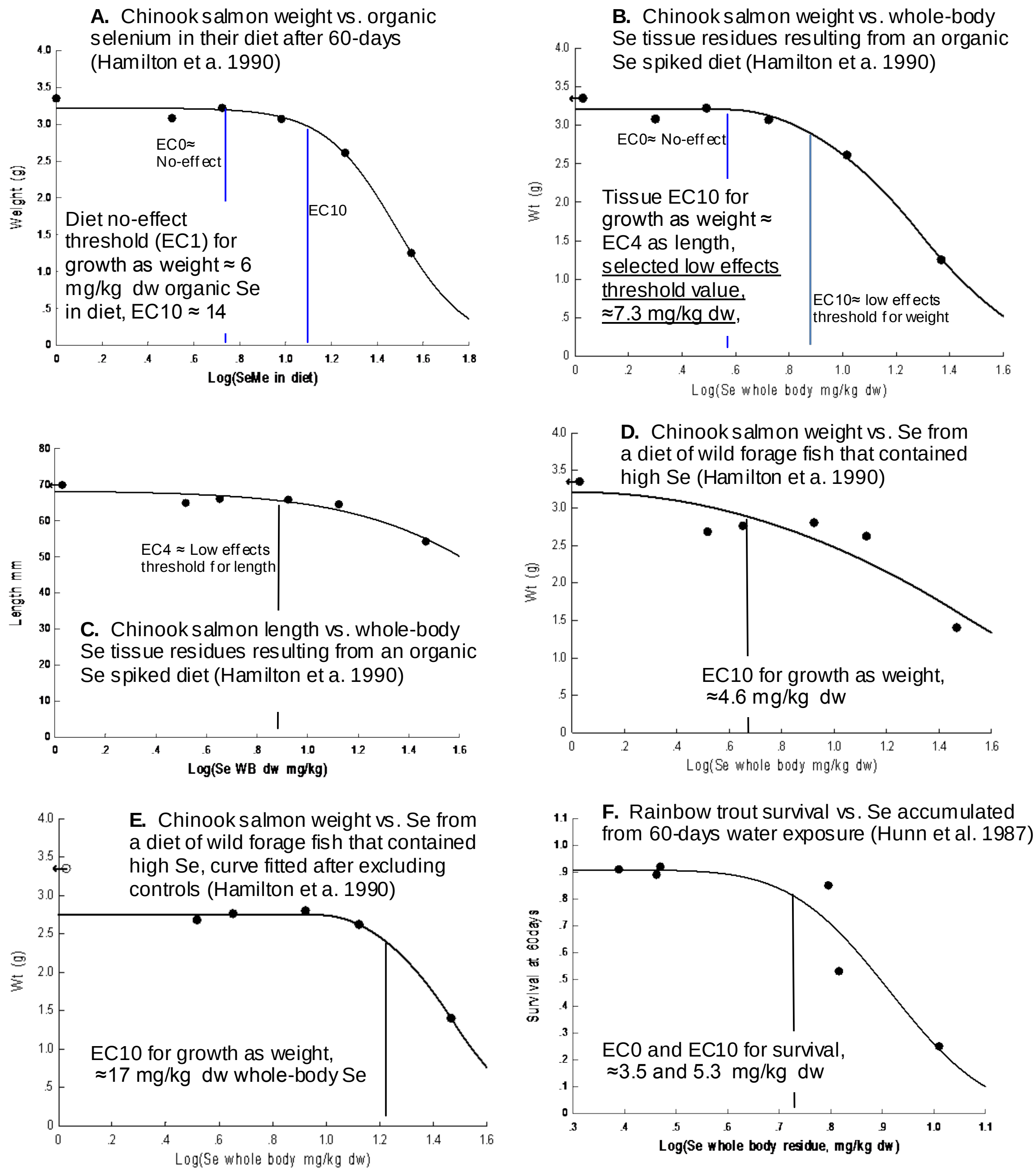



**Figure 2.4.8.1. Estimates of thresholds for no- and low-effect concentrations for selenium in diet and whole-body tissues of juvenile Chinook salmon or rainbow trout. Curve fitting and curve fitting and effects concentration percentiles (ECp) were estimated using threshold sigmoid regression (Erickson 2008). EC10=concentration causing a 10% reduction in growth or survival.**

***Bioaccumulation of selenium through stream food web trophic transfer.*** Now that thresholds have been estimated for no- and low-effects of selenium in the tissues of rearing anadromous salmonids, the next step of the analysis of risks of selenium in water at the chronic criterion concentration is to estimate what tissue concentrations would be expected from ambient concentrations of selenium in streams. These estimates may be made through food web studies in streams and ecosystem models. Traditionally, BAFs have been used to relate water concentrations of a substance to tissue residues, where the BAF is the ww concentration in tissues divided by the concentration in water. However, BAFs are a crude measure and can be unreliable for a variety of reasons including the following: different aquatic ecosystems have shorter or longer food webs; the source of major biomagnification of selenium, water to algae, varies greatly by water body type; and organisms will regulate internal concentrations of micronutrients such as selenium resulting in higher BAFs at low water concentrations and lower BAFs at higher concentrations (Luoma and Presser 2009; Stewart *et al*. 2010).

In contrast, ecosystem food web models have advantages over the traditional BAFs because food web models can account for some of the interrelated factors that contribute to the widely variable BAFs, namely food web type and length, speciation, and water body type. In particular, selenium uptake may differ greatly by water body type such as estuary, lentic freshwater, or lotic (flowing) freshwaters (Orr *et al*. 2006; Luoma and Presser 2009). Ecosystem models of selenium provide a means for relating selenium concentrations in the water column and concentrations in other food chain components, including selenium residues in fish or fish tissue guidelines to prevent adverse effects. The models can then be run either forwards or backwards to predict concentrations of selenium in fish from a given concentration in water, or for a given concentration in fish such as a tissue-based criterion or guideline, then an associated concentration in water can be estimated (Luoma and Presser 2009; Presser and Luoma 2010). Because in the preceding discussion, a selenium tissue concentration of about 7.6 mg/kg dw was estimated, this concentration in fish "$C_{fish}$" can be treated as a given, and the model can be used "backwards" to solve for a corresponding estimated selenium concentration in water "$C_{water}$" (Equation 1):

$$C_{water} = \frac{C_{fish}}{(TTF_{fish}) \bullet (TTF_{invertebrate}) \bullet (K_d)} \qquad \text{(Equation 1)}$$

where:

$C_{water}$ is the allowable water-column concentration of selenium, given a selenium fish tissue guideline;

$C_{fish}$ is the selenium fish tissue residue guideline, in whole-body or muscle, as dry weight;

$K_d$ is the partitioning coefficient between particulates such as benthic biofilms (i.e., algae and associated living and non-living material, sometimes called aufwuchs) and the water-column selenium concentration, Kd is calculated as $K_d = C_{particulate} \div C_{water\text{-}column}$

$TTF_{invertebrate}$ is the trophic transfer factor from biofilm to aquatic insects that graze on biofilm

$TTF_{fish}$ is the trophic transfer factor from aquatic insects to fish that prey on the aquatic insects.

This form of the model is appropriate for a short coldwater food web where the fish prey exclusively on aquatic insects. In a longer food web with fish preying on other fish, such as bull trout in large rivers, then an additional factor would be added for trophic transfer between the forage fish and top predator. Because trophic transfer factors (TTFs) for fish-to-fish are close to 1, if some portion of the fish's diet included other fish, this would not change the model predictions much.

In order to estimate TTFs and $K_d$ values for streams, reliable data on selenium in food web compartments are needed. For listed Snake River salmon and steelhead, the vast majority of their potential exposure within the action area is in flowing streams and rivers; therefore, NMFS did not consider lentic reservoir scenarios. We evaluated extensive data on selenium concentrations in stream foodwebs from two adjacent watersheds located in the upper Salmon River drainage, Idaho. Additionally, a state-wide probabilistic assessment of selenium concentrations in water and fish tissues was recently completed in Idaho (Essig and Kosterman 2008; Essig 2010). These data are summarized in Table 2.4.8.2. The Idaho statewide assessment used a probabilistic approach where a random draw of all stream segments above a certain size in a global information system dataset was used to select sample sites. This approach allowed more robust statistical analyses of median and ranges of selenium concentrations than could be made if known water bodies of concern were targeted.

We examined two adjacent watersheds in the upper Salmon River, Idaho, watershed, the Yankee Fork and Thompson Creek, because of concerns of suspected elevated selenium, mercury, and other metals as result of mining activities. The Yankee Fork has generally elevated selenium contents in stream sediments and alluvium that reflect the generally high selenium contents in the volcanic rocks that underlay the Yankee Fork and the presence of gold and silver selenides in some of the veins that were exploited in the early phases of mining (Frost and Box 2009). In samples from more than 70 locations throughout the watershed, the highest selenium concentrations were obtained from two samples of undisturbed alluvium, reflecting natural sources. A major open-pit bulk-vat leach gold mining operation, the Grouse Creek Mine, operated on Jordan Creek, a tributary to the Yankee Fork, in the mid-1990s. Selenium concentrations in stream sediments showed no pattern attributable to the Grouse Creek Mine (Frost and Box 2009).

Thompson Creek, Idaho is located just east of the Yankee Fork, sharing a watershed divide. A large-open pit molybdenum mine, the TCM, is partially located within the drainage. Overburden from the pit is dumped into two valley-fill waste rock piles. Two small streams that have elevated selenium concentrations drain from the waste-rock piles into Thompson Creek. These discharges are jointly permitted by EPA and the state of Idaho through the NPDES and Idaho state certification programs. Among many other constituents, selenium in water, biofilm, invertebrates, and fish is systematically monitored by personnel from the mine and consultants pursuant to the NPDES permit and certification (Mebane 2000).

**Table 2.4.8.2. Median selenium concentrations in coldwater stream webs relevant to the area of interest, data collected between 2001 and 2008 for all study locations.**

| Location | Water (µg/L) | Biofilm (Periphyton and detritus, mg/kg dw) | Aquatic insects, mg/kg dw | Sculpin, whole-body, mg/kg dw | Salmonids (WB, mg/kg dw) | Sources & notes |
|---|---|---|---|---|---|---|
| Idaho, statewide median from probabilistic sampling | 0.14 | | | | 1.3 | 1 |
| Thompson Creek, Idaho - upstream of mine effluents | 0.9 | 2.6 | 2.7 | 6.2 | 6.7 | 2 |
| Thompson Creek, Idaho - downstream of mine effluents | 3.3 | 5.6 | 7.2 | 9.6 | 7.8 | 2 |
| Yankee Fork, Idaho - upstream of mine effluents | <0.10 | 0.4 | 4.2 | 6.9 | 6.2 | 3 |
| Yankee Fork, Idaho - downstream of mine effluents | 0.21 | 0.4 | 3.8 | 5.4 | 5.6 | 3 |
| McLeod and Smoky River systems, Alberta - reference streams | 0.2 | 1.0 | 4.5 | – | 1.8 | 4 |
| McLeod and Smoky River systems, Alberta - mining influenced stream | 10.7 | 3.2 | 10.0 | – | 8.9 | 4 |

**Table Notes:**

1 – Essig 2010, mean of all collected fish species, muscle tissue, range 0.2 to 14 mg/kg dw, converted from fresh weight using 21.2% solids. Range in water, <0.09 µg/L to 1.75 µg/L.

2 – Water selenium data from Thompson Creek Mining Company, biological data are from August 2003, 2004, and 2007 CEC (CEC 2004b, 2005; GEI 2008). Detection limit for Se in water was 1 µg/L and close to 50% of the samples collected upstream of the mine discharges were <1 µg/L. Maximum likelihood estimates (MLDE) based on the distribution of detected values were used to estimate non-detect concentrations to estimate median concentration (Helsel 2005); however, the resulting estimated median selenium concentration upstream of mine effluents were similar whether the MLE approach or the "data fabrication" approach of using ½ the detection limit for non-detect values was used (0.9 vs. 0.8 µg/L respectively.

2 – Water data from Hecla Mining Company, 2006-2008, detection limit was 0.1 µg/L. Biological data were collected in 2001 and 2002(Rhea *et al*. 2013)

4 – Casey (2007)

Background concentrations of selenium in Thompson Creek are considerably higher than median background concentrations in Idaho, estimated at 0.9 and 0.13 µg/L, respectively. Because no major mining or other human disturbances are known of in the Thompson Creek drainage upstream of the TCM discharges, the elevated instream background selenium concentrations are presumed to be of mostly natural-origin. Concentrations downstream of the discharges from the waste rock dumps are substantially higher, averaging about 3 µg/L and ranging from about 1 to 8 µg/L Se. About 80% of the streams in Idaho are estimated to have baseflow selenium concentrations lower than those in Thompson Creek upstream of mining discharges, and >99% are estimated to have concentrations lower than in Thompson Creek downstream of mine discharge (Table 2.4.8.2.; Essig 2010).

In addition to the three Idaho studies reviewed (statewide, Thompon Creek and Yankee Fork, Table 2.4.8.2), a fourth study relevant to this analysis was from a series of selenium enriched and reference streams in Alberta that had similar invertebrate and resident salmonid species as did the Idaho streams (Casey 2007). With the exception of the Yankee Fork data, all of the data that we used to estimate TTFs between for different food web positions summarized in Table 2.4.8.2 were from water and biota sampling sites that were matched in space and time. The Yankee Fork water data with low enough detection limits to be useful were not matched in time, but because the water selenium concentrations were low and fairly uniform they are assumed to be representative.

**Table 2.4.8.3 Median trophic transfer factors (TTF) and water-biofilm partitioning coefficients of selenium within coldwater stream food webs.**

| | Trophic transfer factors (TTFs) | | | Partitioning coefficients |
|---|---|---|---|---|
| | Aquatic insects | Fish (shorthead sculpin) | Fish (salmonid) | $K_{d\ (water\text{-}biofilm)}$ |
| Thompson Creek, Idaho: upstream of mine effluents (background for this watershed) | 2.7 | 2.2 | 2.4 | 3133 |
| Thompson Creek, Idaho: downstream of mine effluents | 2.0 | 1.3 | 1.0 | 2188 |
| Yankee Fork, Idaho: upstream of mine effluents | 11.0 | 1.6 | 1.5 | 4250 |
| Yankee Fork, Idaho: downstream of mine effluents | 10.5 | 1.4 | 1.5 | 1738 |
| McLeod and Smoky River systems, Alberta: reference streams (Deerlick Cr., Cold Cr.) | 4.5 | – | 1.8 | 5000 |
| McLeod and Smoky River systems, Alberta: mining impacted stream (Luscar Cr.) | 2.6 | – | 2.4 | 299 |

The median selenium concentrations in water, biofilm, insects, and fish given in Table 2.4.8.2 can be used to estimate TTFs and water-particulate partition coefficients ($K_d$) per Equation 1. However, even a casual inspection of the $K_d$ values shows a great range of estimates, with the higher values derived from data collected from reference sites with low selenium, and the low $K_d$ values derived from sites with enriched selenium. This pattern is illustrated in Figure 2.4.8.2 and is reasonably consistent across a gradient of selenium concentrations. This pattern is biologically plausible for a micronutrient for which organisms attempt to regulate internal concentrations and maintain homeostasis by increasing retention when the micronutrient is scarce and increasing uptake when the micronutrient is in excess. This further suggests that risk estimates of selenium in stream food webs could be mistaken if a low $K_d$ from a selenium enriched stream were used to estimate the assimilative capacity for a low-selenium stream using Equation 1, (over predict the amount of selenium that could safely be added). Likewise, if Equation 1 were re-arranged into Equation 2 to predict selenium concentrations in fish resulting from a given water concentration, if the water concentration of interest is much higher than reference conditions such as the selenium CCC of 5 µg/L which is ~5 to 50X higher than reference selenium concentrations (Table 2.4.8.2), but it used with a $K_d$ value that was derived from reference conditions, then the tissue concentrations in fish would be over-predicted, leading to an overestimation of selenium risks associated with the CCC concentration.

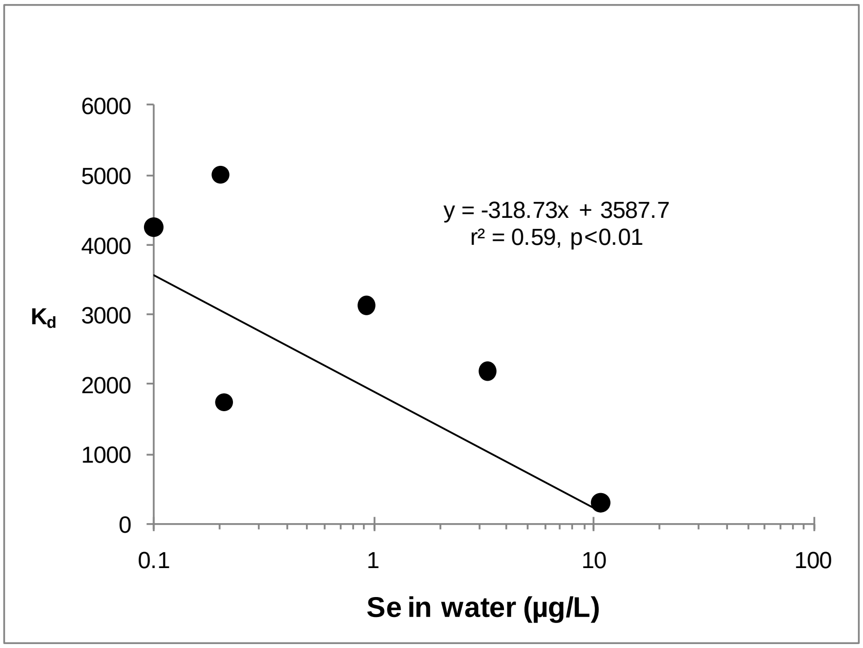



**Figure 2.4.8.2. Apparent concentration dependence of selenium water-particulate partitioning coefficients ($K_d$) from coldwater, salmonid streams.**

Based on this information NMFS estimates a concentration in water that would be expected to be transferred through the food web to a given tissue concentration. Using the juvenile Chinook salmon $EC_{10}$ of 7.6 mg/kg dw selenium in whole bodies from Table 2.4.8.1 as the concentration of concern, with the median TTFs and partition coefficient $K_d$ from Thompson Creek, predicts that this concentration in fish could be reached with a selenium water concentration ($C_{water}$) of about 1.2 µg/L. Median values for Thompson Creek were used in this example because this stream has moderately elevated selenium concentrations that are associated with tissue residues in fish near the Chinook salmon $EC_{10}$ concentration. If the Chinook salmon $EC_{10}$ was instead used with TTFs and $K_d$ from streams with much lower selenium concentrations such as the Yankee Fork or the Alberta reference streams, that would have resulted in a very large denominator and a correspondingly very low projected concentration in water.

$$C_{water} = \frac{7.6\ (C_{fish})}{1.1(TTF_{fish}) \bullet 2.2(TTF_{invertebrate}) \bullet 2690(K_d)} = 0.00121\ mg/L = 1.2\ \mu g$$

However, the primary purpose of this analysis is to estimate whether a given water concentration of selenium, that is the chronic criterion concentration of 5 µg/L, is likely to lead to trophic transfer to levels in fish likely to cause adverse effects. To estimate likely tissue concentrations from a given water concentration of selenium, equation 1 may be rearranged to predict tissue residues from a given water concentration (Equation 2).

$$C_{fish} = \frac{C_{water}}{(TTF_{fish}) \bullet (TTF_{invertebrate}) \bullet (K_d)} \qquad \text{(Equation 2)}$$

Using the CCC of 5 µg/L with the $K_d$ estimate of 1994 from the $K_d$ vs. selenium in water regression (Figure 2.4.8.2), a tissue concentration of about 19.5 mg/kg dw in juvenile salmonids would be projected. Using Hamilton's 60 d growth model, this would relate to about a 50% reduction in weight ($EC_{50}$ was 19.3 mg/kg); a 10% reduction in length, and about a 25% reduction in survival.

Similarly, using Equation 2 iteratively to "titrate" down from a severe effects concentration to a low-effects concentration, a selenium concentration in water of about 2 µg/L with a $K_d$ appropriate for that enriched concentration (average of Thompson Creek segments downstream of mine discharges and Yankee Fork downstream of mine effluent) projects a whole-body tissue concentration for stream resident salmonids of about 7.7 mg/kg dw (Equation 3). Although it was estimated independently, this projected tissue concentration is very close to the low-effects whole-body tissue residue of 7.6 mg/kg dw used to evaluate the protectiveness of the 5 µg/L water chronic criterion.

$$C_{fish} = \frac{0.002\ mg/L(C_{water})}{1.0(TTF_{fish}) \bullet 2.0(TTF_{invertebrate}) \bullet 1963 L/kg(K_d)} = 7.7\ mg/kg \qquad \text{(Equation 3)}$$

The calculated 2 µg/L low risk water concentration corresponds with recommendations of Lemly and Skorupa (2007) for implementing proposed tissue residue-based selenium water quality criteria in a stepped fashion where for waters less than 2 µg/L selenium, dischargers need not be burdened with fish monitoring requirements. The vast majority of streams in the action area have waterborne selenium concentrations <2 µg/L (Table 2.4.8.2). Above 2 µg/L in water, fish tissue monitoring would then be needed to evaluate if selenium was being transferred through the food web to greater than tissue-residue concentrations of concern (7.6 mg/kg dw). If tissue concentrations in fish in a stream influenced by elevated selenium concentrations from point source discharges or non-point sources are greater than this tissue-residue concentration of concern, then actions to reduce anthropogenic selenium loading to the water bodies are presumed necessary. However, it is conceivable that additional site-specific information could indicate that even if tissue residue concentrations in juvenile salmonids exceed 7.6 mg/kg dw, these concentrations are unlikely to be causing adverse effects. This is because in the Yankee Fork and Thompson Creek examples, tissue residue concentrations collected at upstream background sites were elevated to concentrations only slightly lower than this value (Table 2.4.8.2), and as of 2012, salmonid abundances in mining-influenced sections of Thompson Creek showed no obvious declines that could be attributed to elevated selenium concentrations (Janz *et al*. 2010; GEI 2013).

#### *2.4.8.3. Summary for Selenium*

If water concentrations were near the chronic selenium criterion of 5 µg/L indefinitely, selenium would likely be transferred through the food web resulting in selenium concentrations in juvenile salmonids greater than twice as high as a concentration estimated to be low risk for appreciable effects in juvenile salmon or steelhead (~7.6 mg/kg dw in whole bodies). Fish tissue residues resulting from stream food web transfer from a constant water concentration of about 5 µg/L were projected to exceed about 19.5 mg/kg dw in juvenile salmonids. This selenium tissue burden would be projected to result in growth reductions and increased mortality in juvenile anadromous salmonids, on the order of about a 50% reduction in weight, a 10% reduction in length, and about a 25% reduction in survival. Lesser reductions in growth (e.g., a 7.5% reduction) were projected to appreciably increase extinction risks and delay recovery in a modeled Chinook salmon population (Mebane and Arthaud 2010). While their modeling was specific to a Snake River spring/summer Chinook salmon populations from the upper Salmon River, NMFS assumes that the relations between size and survival during downstream migration would also hold for steelhead and sockeye salmon.

### 2.4.9. The Effects of EPA Approval of the Silver Criteria

Silver, in the free ion form, has been noted to be one of the most toxic metals to freshwater organisms and is highly toxic to all life stages of salmonids. Ionic silver is the primary form responsible for causing acute toxicity in freshwater fish (EPA 1980o, 1987b; Eisler 1996; Hogstrand and Wood 1998; Bury *et al*. 1999a). Toxicity varies widely depending on the anion present: Silver nitrate has a much higher toxicity than silver chloride or silver thiosulfate, by approximately four orders of magnitude (Hogstrand *et al*. 1996). Documented effects of silver toxicity in fish include interruption of ionoregulation at the gills, cell damage in the gills, altered blood chemistry, interference with zinc metabolism, premature hatching, and reduced growth rates (Hogstrand and Wood 1998; Webb and Wood 1998).

#### *2.4.9.1. Species Effects of Silver Criteria*

Aquatic life criteria for silver are complicated by the fact that the NTR criteria (EPA 1992) does not follow either of EPA’s two published criteria documents for silver (EPA 1980a, 1987b). No attribution for criteria values were given in the NTR. The Idaho hardness-adjusted acute values for the action (Table 1.3.1) match the hardness-adjusted acute values in EPA (1980o), however the chronic value from EPA (1980o) was not used in the NTR nor in Idaho’s subsequent water quality criteria. The 1987 criteria version concluded that silver toxicity was affected by chloride speciation, but that hardness was a less important modifier of toxicity. The 1987 criteria retained the 1980 fixed chronic concentration of 0.12 µg/L, but replaced the 1980 hardness-adjusted acute criterion with a fixed acute criterion concentration of 0.92 µg/L (EPA 1987b). No explanation was provided in the NTR why EPA went back to their older acute criterion or why the chronic criterion was omitted (EPA 1992).

***Acute Silver Criterion.*** Most studies of acute toxicity have used silver nitrate as the test solution, which is highly soluble and is the most toxic form of silver. Hogstrand and Wood (1998) pointed out that because of the strong modifying influence of naturally occurring ligands in ambient waters on silver toxicity (see below), the likelihood is significantly reduced that dissolved silver concentrations approach levels needed to cause acute toxicity. However, regardless of form, at hardness levels of 200 mg/L and less, considered relevant to the action area, the acute silver criterion is sufficiently low to prevent lethality (Figure 2.4.9.1).

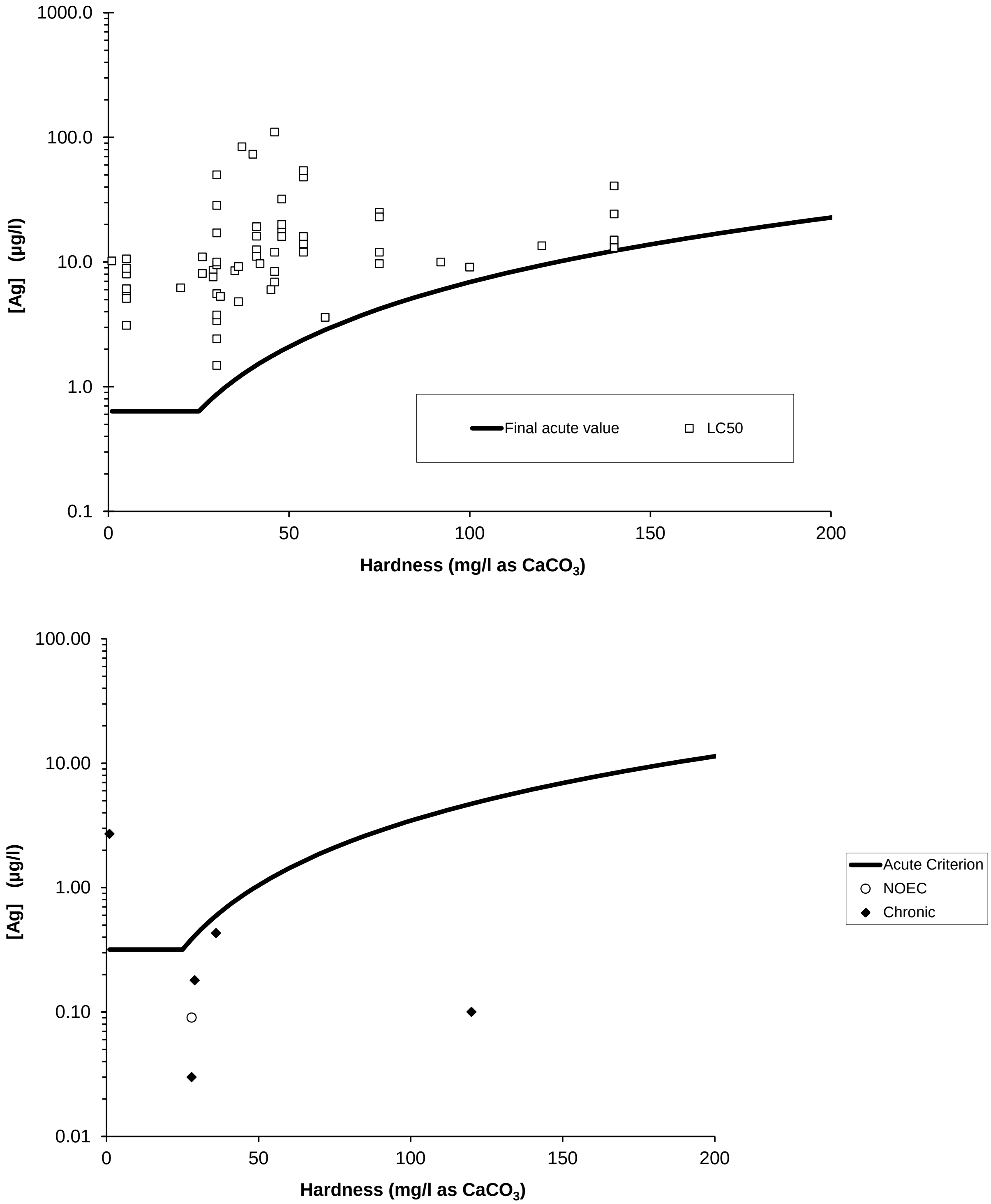



**Figure 2.4.9.1. Acute silver criterion in comparison with acute and chronic silver effects data**

***Chronic Silver Criterion***. Chronic criteria for silver are presented in the AWQC documents (EPA 1987b), and EPA (1980o), and these concluded that the "available data indicate that chronic toxicity to freshwater aquatic life may occur at concentrations as low as 0.12 µg/L." However, no chronic silver criterion was included in the National Toxics Rule (EPA 1992) nor in the proposed action. No explanation for this omission was given in EPA 1992, other than "*with this rule, EPA is promulgating its 1980 criteria for silver, because the Agency believes the criteria is protective and within the acceptable range based on uncertainties associated with deriving water quality criteri*a" (EPA 1992, p. 60883). However, although the word "criteria" is plural, there was only an acute criterion and no chronic criterion proposed for approval in the current action. There are sufficient data available that indicate that the acute criterion, which effectively acts as a chronic criterion, does not avoid chronic toxicity that has been determined to occur at concentrations below the acute criterion:

> The work of Davies *et al*. (1978) suggests that the maximum acceptable silver concentration to prevent chronic mortality in rainbow trout embryos, fry, and juveniles, and avoid premature hatching, is less than 0.17 µg/L for a water hardness equal to 26 mg/L.
>
> Nebeker *et al*. (1983) concluded that the maximum acceptable toxicant concentration to prevent inhibition of growth of steelhead embryos was less than 0.1 µg/L for a water hardness equal to 36 mg/L.

The absence of a chronic silver criterion implies potential mortality at acute criteria concentrations to listed salmonids based on the data and information reviewed here.

***Hardness and Other Parameters as Predictors of Silver Toxicity.*** The acute and chronic toxicities of silver are influenced by hardness, chloride ion, DOC, sulfide, and thiosulfide concentrations, and with pH and alkalinity (Hogstrand and Wood 1998; Erickson *et al*. 1998). For example, Karen *et al*. (1999) determined that increasing hardness from 30 mg/l to 60 mg/l resulted in significantly reducing silver nitrate toxicity. However, it has been shown that hardness is not as important an influence on silver toxicity as was originally thought and is secondary to other water quality constituents. Specifically, chloride ion and DOC concentrations have a significantly greater influence on toxicity (Galvez and Wood 1997; Hogstrand and Wood 1998; Bury *et al*. 1999b; Karen *et al*. 1999; Wood *et al*. 1999). The presence of chloride ion is protective because silver chloride precipitates out of solution readily, although under certain conditions it is possible to observe the formation of the dissolved $AgCl^0$ complex (Erickson *et al*. 1998). Bury *et al*. (1999a, 1999b) determined that chloride and DOC concentrations ameliorated the silver ion inhibition of $Na^+$ influx and gill $Na^+/K^+$-ATPase activity in rainbow trout. Toxicity of silver was found to change very slowly with hardness, where a hundredfold increase in hardness resulted in reducing toxicity only by roughly 50% (Bury *et al*. 1999b) and increased survival time approximately 10 fold (Galvez and Wood 1997). In contrast, only a twofold increase in chloride ion was required to produce toxic effects similar to a hundredfold increase in hardness (Galvez and Wood 1997). Karen *et al*. (1999) observed that DOC was more important than hardness for predicting the toxicity of ionic silver in natural waters to rainbow trout, fathead minnows and *Daphnia magna*. The DOC greatly reduced gill accumulations of silver through complexation. Chloride ion did not reduce gill accumulations of silver because it bound with free silver ($Ag^+$) and accumulated in gills as silver chloride, but reduced toxicity because the silver chloride did not enter cells and disrupt ionoregulation.

A key point from the environmental chemistry and aquatic toxicology literature for silver is overwhelming differences in toxicity between free ionic silver and complexed silver compounds. Most laboratory toxicity tests with silver used silver nitrate because it readily disassociates into ionic silver which tends to remain in solution (Hogstrand and Wood 1998). In contrast, in rivers, streams, lakes, and effluents, ionic silver tends to be vanishingly low, and measureable silver in natural waters and effluents occurs as either silver sulfide, silver chloride, silver thiosulfate, or as complexes with natural DOC (Adams and Kramer 1999; Kramer *et al*. 1999). The differences in effects concentrations obtained between tests using silver nitrate and other forms of silver may be on the orders of magnitude. For instance, Hogstrand *et al*. (1996) obtained a 7-day $LC_{50}$ with rainbow trout and silver nitrate of 9 µg Ag/L, but silver chloride and silver thiosulfate $LC_{50}$s were >100,000 µg Ag/L. Similarly, with fathead minnow, compared to free silver ion resulting from silver nitrate additions, silver chloride complexes were about 300 times less toxic and silver sulfide was at least 15,000 times less toxic (Leblanc *et al*. 1984). When very low and environmentally realistic levels of sulfide were added to a test water (0.0016 mg/L), the $LC_{50}$ of *Daphnia magna* was increased by a factor of 5.5 (Bianchi *et al*. 2002).

### *2.4.9.2. Habitat Effects of Silver Criteria*

***Toxicity to Food Organisms.*** Daphnids appear to be considerably more sensitive to silver than fish, with $LC_{50}$s reported for cladocerans have been below the acute criterion (EPA 1987b). *Daphnia magna* tested in the absence of sulfide in water with a hardness of about 120 mg/L yielded an $LC_{50}$ of 0.22 µg/L (Bianchi *et al*. 2002); which was 20 times lower than the acute criterion value of 4.7 for that hardness. When tested in the presence of environmentally realistic levels of sulfide, the $LC_{50}$ was increased by about 5.5 times (Bianchi *et al.* 2002). Other invertebrate taxa serving as potential food for juvenile salmonids have been determined to experience mortality only at concentrations that are above the acute criterion, Other observed adverse effects include reductions in growth and inhibition of molting (EPA 1987b; Eisler 1996; Call *et al*. 1999). Reduced growth in mayfly larvae occurred at 2.2 µg/L in hardness 49 mg/L water (Diamond *et al.* 1992), which is greater than the acute criterion of 1.1 µg/L for that hardness. Chronic effects appear to be documented only for daphnids when silver concentrations are below the EPA (1987b) acute criterion. Aquatic invertebrates have been reported to accumulate silver more efficiently than fish, in concentrations that are proportional to exposure levels (Eisler 1996; Hogstrand and Wood 1998). Studies involving silver sulfide bioaccumulation through sediment interactions from an amphipod and an oligochaete indicated low potential for salmon and steelhead to accumulate harmful silver concentrations through this exposure pathway (Hirsch 1998a, b).

The proposed silver criteria appear to be protective of salmonid food sources under most circumstances. Adverse effects of the silver criterion to the food organisms of listed salmon and steelhead may be potentially meaningful only when daphnids are a primary food source (e.g., downstream of an impoundment in an otherwise oligotrophic system).

***Bioaccumulation.*** Accumulation of silver is predominantly associated with exposure to its ionic forms rather than complexes. Bioaccumulation occurs primarily in the liver (Hogstrand *et al*. 1996; Galvez and Wood 1997; 1999). Significant food chain biomagnification by fish has been reported to be unlikely because of the low silver concentrations typically encountered in the aquatic environment (Eisler 1996; Hogstrand and Wood 1998; Ratte 1999).

*2.4.9.3. Summary for Silver*

In natural waters silver is likely much less toxic than in most published laboratory experiments because of the strong modifying influence of naturally occurring ligands in ambient waters. Because of this, it appears unlikely that acute toxicity to salmonids at criterion concentrations will occur.

Unlike other criteria considered in this Opinion that all had two part values to protect against short-term and indefinite exposures, for silver only a short-term (acute) criterion is proposed. However, adverse chronic effects, including premature hatching, growth inhibition, and chronic mortality, have been observed at in laboratory settings at concentrations below the proposed single silver criterion. Thus, using a single criterion value that was derived using short-term toxicity data to also protect aquatic life from indefinite exposures may be under-protective. The acute criterion is derived as a function of hardness, which is not supported by more current literature which shows chloride, DOC, and sulfide to be more important factors in mitigating silver toxicity. The potential inadequacies and underprotectiveness of the silver criterion are mitigated by the fact that in the environment, silver occurs in a less toxic form than that used in most of the toxicity tests published in the literature. Significant food chain biomagnification by fish is also possible, but all of these effects appear unlikely to occur because of the low silver concentrations typically encountered in the aquatic environment.

2.4.10. The Effects of EPA Approval of the Zinc Criteria

Zinc is an essential element required for healthy fish, and is present in healthy fish tissues in greater concentrations than other heavy metals. However, increased levels of zinc over natural body concentrations can result in mortality, growth retardation, histopathological alterations, respiratory and cardiac changes, and inhibition of spawning and many other elements critical to fish survival. Exposure to high zinc concentrations can result in damage to the gills, liver, kidney and skeletal muscle and cause a physiological shift to occur, making gas exchange more difficult. Toxicity varies with hardness, pH, alkalinity, dissolved oxygen, water temperature, species and life stage, acclimation, and ambient concentrations of other chemicals in the water (EPA 1987c; Sorensen 1991; Eisler 1993). There is evidence that zinc may be more toxic to fish at cold winter-like temperatures than at warmer, summer-like temperatures (Hodson and Sprague 1975).

*2.4.10.1. Species Effects of Zinc Criteria*

Zinc criteria are hardness dependent. At hardness of 100 mg/L, the acute and chronic criterion are 114 and 105 respectively whereas at hardness 25 mg/L, the criteria are 35 and 32 µg/L zinc. The criteria equations are given a floor at hardness 25 mg/L. In effect, the "floor" at hardness 25 mg/L is an implicit assumption that the general relation of zinc being more toxic at lower hardness only holds to a hardness of 25 mg/L and at hardnesses lower than 25 mg/L, zinc is no more toxic than at 25 mg/L.

***Acute Zinc Criterion.*** Toxicity test data indicate that in most instances, the zinc acute criterion concentrations are unlikely to kill juvenile salmonids (Figure 2.4.10.1). Most studies have found toxic effects at concentrations greater than the proposed criterion for adult and early life stages of salmonids (for example, studies reported in EPA 1987c; Chapman 1978a, 1978b; Chapman and Stevens 1978; Stubblefield *et al*. 1999). Two studies were identified where $LC_{50}$ concentrations were less than the FAV for Zn:

> (Mebane *et al*. 2012) reported $LC_{50}$s for rainbow trout from five of 18 tests that were less than the final acute value for zinc. Two of the tests with effects at zinc concentrations less than the criteria resulted from testing in soft water with hardness less than the Idaho 25 mg/L "hardness floor." Other tests conducted at higher hardnesses had variable results, which were apparently related to some life stages being more sensitive than others.
>
> Hansen *et al*. (2002c) and Stratus (1999) determined 120- and 96-hour $LC_{50}$s for rainbow and bull trout fry that were below both the acute and chronic criteria in low hardness water (shown as several points in a vertical line all at 30 mg/L as $CaCO_3$; Figure 2.4.10.1). (Both references are to the same study, but the 96-hour toxicity values which are used here for comparisons were only reported in the 1999 source; 96- and 120-hour values were similar)

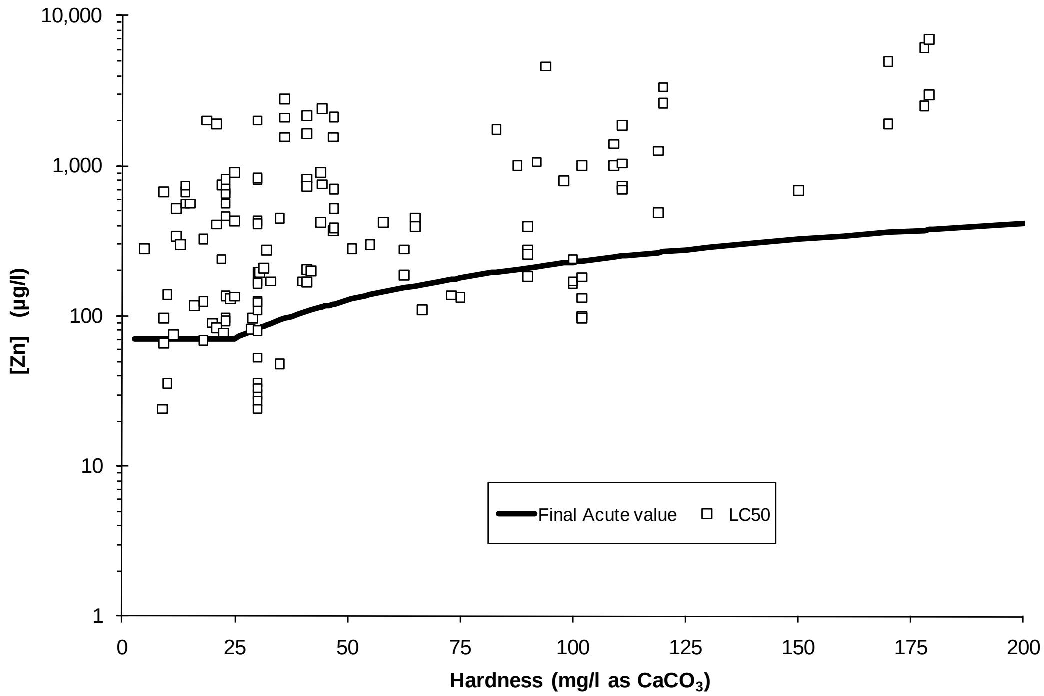



**Figure 2.4.10.1. Comparison of reviewed 96-hour $LC_{50}$s for salmonids with zinc and the Idaho criterion final acute values (FAV), calculated for hardnesses up to 200 mg/L as $CaC0_3$. $LC_{50}$s limited to species within the genera *Oncorhynchus*, *Salvelinus*, and *Salmo.* If $LC_{50}$ values fell above the FAV line, that would suggest few if any mortalities would be likely at criterion concentrations.**

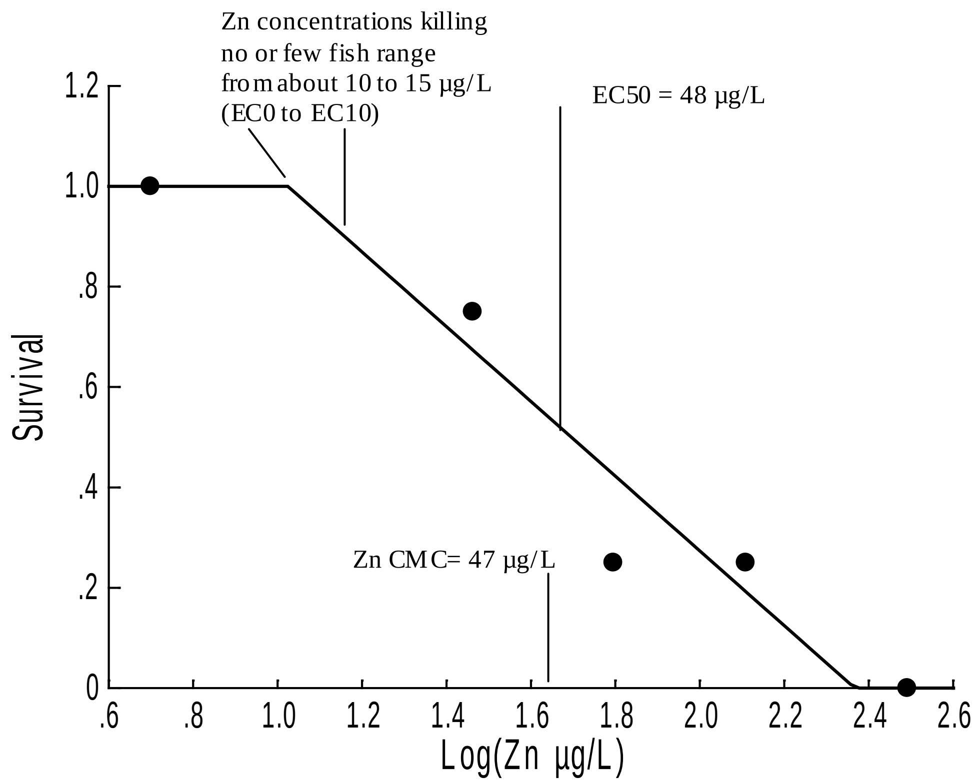



**Figure 2.4.10.2. Example of a 96-hour toxicity test with rainbow trout in which zinc at its acute criterion concentration (CMC) killed about half of the fish tested. At the CMC, few if any fish are supposed to be killed. In this instance, the final acute value that the criterion was based on (i.e., the $LC_{50}$ for a hypothetical organism more sensitive than 95% of organisms) was twice as high as the rainbow trout value.) Rainbow trout data from Mebane *et al*. (2012), test hardness 35 mg/L, 0.5g fish, wet wt.**

***Chronic Zinc Criterion***. The proposed chronic criterion is only approximately 10% less than the acute criterion. The similarity of the chronic and acute criteria may not be of great concern with respect to listed salmon and steelhead because fish have naturally elevated zinc levels in their tissues, are able to regulate tissue zinc concentrations over a range of environmental exposure levels, and exhibit increased resistance to elevated zinc levels with acclimation (Sorensen 1991; Eisler 1993). Acclimation likely explains why some chronic tests values are higher than acute values, because chronic tests conducted as ELS exposures begin the exposures during the metals-resistant egg or alevin life stage (Chapman 1985). If during this resistant stage, the fish acclimate to zinc, when they grow older and enter the more sensitive fry stages they may be more resistant than fish that were initially exposed as fry. For instance, Mebane *et al*. (2008) reported a 68-day zinc $LC_{50}$ of 367 µg/L for rainbow trout compared to a 4-day zinc $LC_{50}$ of 130 µg/L, tested in the same dilution water. In this study, the 68-day test began with the egg life stage where whereas the 4-day test began with fry (Mebane *et al*. 2008). Similarly, Brinkman and Hansen (2004) obtained nearly identical 4-day and 30-day $LC_{50}$s with rainbow trout and zinc when exposures began with fry, but longer ELS exposures that began using eggs yielded much higher (more resistant) values. Chapman (1978b) exposed sockeye salmon to zinc for 21 months beginning with a 3-month adult exposure followed by an 18-mo exposure of embryonic through smolt stages. Zinc concentrations up to 242 µg/L produced no adverse effects on survival, fertility, fecundity, growth, or on the subsequent survival of smolts transferred to seawater. The sockeye salmon became acclimated to zinc, as Chapman found a 2.2 fold increase in tolerance when he tested some of the zinc exposed sockeye salmon along with naïve fish (Chapman 1978b). Overall, most available data indicated that chronic toxic effects were unlikely at concentrations lower than the chronic criterion (Figure 2.4.10.3)

Acclimation is likely related to why salmonids are sometimes present and in apparent good health in some streams that greatly exceed chronic zinc criteria (Chapman 1978a, 1985; Harper *et al*. 2008). However, the information indicate that acquired protection is transient and may be lost in periods as short as 7 days upon return to toxicant-free water (Bradley *et al*. 1985; Stubblefield *et al*. 1999). Thus, even though acclimation can increase the resistance of fish to zinc by factors of around two to three, the protection by acclimation may not be lasting, and neither acclimation nor the fact that salmonids may be self-sustaining in watersheds with elevated zinc refute the demonstrated effects of zinc in fish that have no prior history of zinc exposures previous to testing.

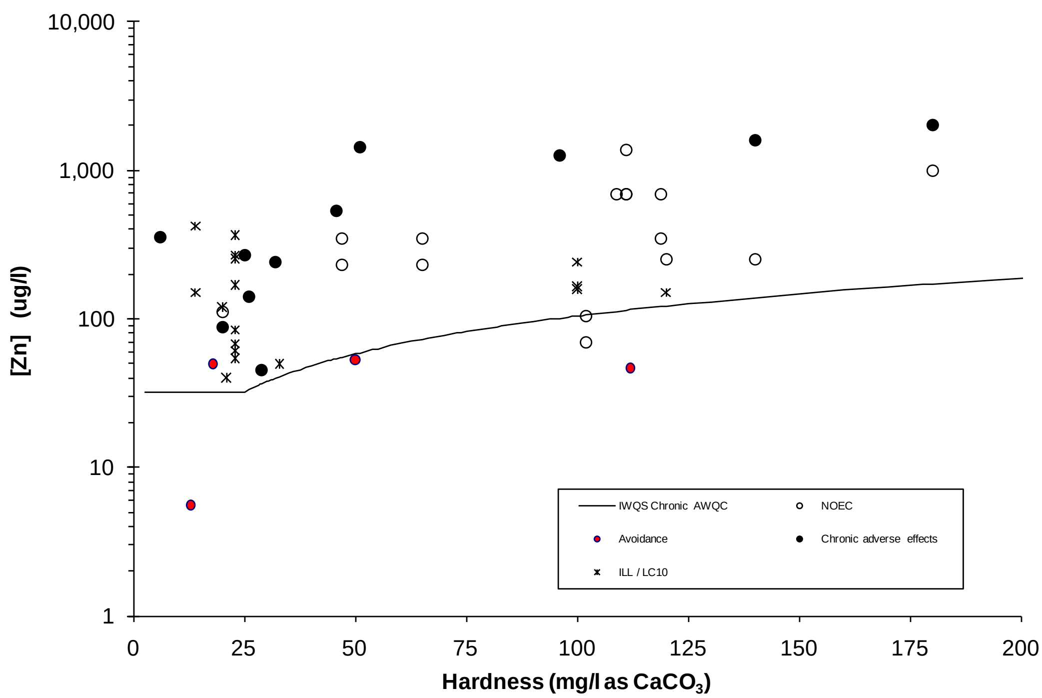



**Figure 2.4.10.3. Comparison of the Idaho chronic criterion and adverse chronic or sublethal effects and estimates of no-effect concentrations to salmonids.**

***Behavioral Effects.*** Behavioral avoidance reactions have been noted to occur in three trout species at zinc concentrations that were below the proposed chronic criterion. Juvenile rainbow trout avoidance was documented at zinc concentrations of 5.6 µg/L at a hardness of 13 mg/L (Sprague 1968) and 47 µg/L at a hardness of 112 mg/L (Black and Birge 1980). Juvenile cutthroat trout avoidance was documented at 53 µg/L at a hardness of 50 mg/L (Woodward *et al*. 1997).

Little study of behavior effects to adult salmonids in relation to zinc has been conducted. There are insufficient and conflicting data available to identify whether these behavioral effects translate into adverse effects in the field. Sprague *et al*. (1965) and Saunders and Sprague (1967) showed that the upstream migrations of Atlantic salmon were disrupted in the Miramichi River, New Brunswick, when zinc concentrations reached about 150 to 180 µg/L and copper reached about 11 to 15 µg/L. From 1990 to1996, water hardness of the Miramichi River measured monthly averaged 10 mg/L, ranging from 4 to 19 mg/L, and DOC averaged 3.7, ranging about 0.5 to 7 mg/L (Komadina-Douthwright *et al*. 1999). Assuming that the major ion and organic carbon content of the Miramichi River do not greatly change year to year, then these values help relate the thresholds for migratory disruption reported by Saunders and Sprague (1967) to criteria values. At a water hardness of 25 mg/L, the acute zinc criterion would be 36 µg/L, which is considerably lower than the apparent migratory disruption threshold of 150 µg/L.

One study asserted that ambient zinc concentrations in an Idaho river disrupted migration of adult Chinook salmon. In an effort to monitor avoidance responses of salmonids to metals in more realistic conditions, Goldstein *et al*. (1999) monitored adult Chinook salmon movements with radio telemetry in the vicinity of the South Fork and North Fork Coeur d'Alene River confluence. Adult male Chinook salmon were captured from Wolf Lodge Creek (a tributary to Lake Coeur d'Alene) and after harvesting their milt, were trucked to the Coeur d'Alene River and released about 2 km downstream of the confluence of the South Fork and North Fork. Half of the released salmon moved upstream; of the half that moved upstream, 70% ascended the North Fork which had a zinc concentration of about 9 µg/L. Thirty percent ascended the South Fork, which had a zinc concentration of about 2200 µg/L. The authors concluded that their study demonstrated that avoidance of metals can disturb critical spawning migrations and may displace or preclude fish from preferred habitats (Goldstein *et al*. 1999). However, because migrating spawning salmon home on their natal stream by chemical imprinting, and Chinook salmon die within a few weeks after spawning, is unclear what the migratory instincts would be of post-spawning male salmon that had been trucked from their natal stream and released in a different watershed shortly before their deaths. Further, since the North and South Forks make up around 70% and 30% of the Coeur d'Alene River flows respectively, and adult salmon movements in rivers in the absence of any homing cues tend to simply follow larger flows (Anderson and Quinn 2007), the conclusion of the authors is debatable. We determined that they proved feasibility of tracking fish movements in a river, and provide guidance for conducting a more ecologically meaningful study, such as using migratory adult cutthroat prior to spawning.

***Hardness as a Predictor of Zinc Toxicity***. Zinc toxicity is known to vary with a number of factors other than hardness, such as pH, alkalinity, temperature, and life stage or size. In many situations, the present criteria appear to avoid harm in circumstances when zinc is the only contaminant present at elevated concentrations and perhaps when jointly elevated with cadmium. An exception exists for at least some juvenile salmonids in water with low hardness values of about 35 mg/L or less, as indicated by the results of Hansen *et al*. (2002c) and Mebane *et al*. (2012). Data reported in EPA (1987c) and the large number of other studies depicted in Figure 2.4.10.1 otherwise indicate that the criteria, expressed as a function of hardness, may often be protective close to the hardness floor of 25 mg/L as $CaCO_3$. Calcium has been determined to reduce zinc uptake directly through both biological acclimation and chemical processes, where protection is additive in nature with increasing calcium concentration (Barron and Albeke 2000). The results of Hansen *et al*. (2002c), however, indicate that some rainbow trout, which are considered a surrogate for listed salmon and steelhead, may be killed at zinc concentrations lower than criteria, particularly as they develop to sensitive sizes during the fry stage.

*2.4.10.2. Habitat Effects of Zinc Criteria*

***Toxicity to Food Organisms.*** Many freshwater insects and crustaceans appear to be tolerant of zinc concentrations that are similar to the acute criterion (Eisler 1993), although some taxa can be more sensitive to chronic effects than salmonids (Kemble *et al*. 1994). Aquatic invertebrates bioaccumulate zinc to a greater degree than salmonids (EPA 1987c; Eisler 1993). Kiffney and Clements (1994) determined that mayflies were sensitive to zinc, and that the response varied with stream size or location in the stream network. Data in EPA (1987c) indicate that the zinc criteria are usually non-lethal to invertebrates that juvenile salmon and steelhead feed on, although in two cases listed in Table 2-1 of EPA (1987c), cladocerans exhibited $LC_{50}$s that were lower than the acute and chronic criteria at a hardness of 45 mg/L. Invertebrate communities in rivers appear to respond to elevated zinc levels in the sediments by changing composition to pollution-tolerant taxa, rather than by reducing overall biomass (Canfield *et al*. 1994; Clements and Kiffney 1994). Working with data from streams in northern Idaho, Dillon and Mebane (2002) found that overall insect taxa richness, mayfly richness, and abundance of metals-sensitive mayflies and other taxa were generally highest at streams with low zinc concentrations. The abundance and diversity of aquatic insects generally declined with increasing zinc concentrations. No threshold of response could be determined, and declines may have begun at zinc concentrations less than the Idaho chronic zinc criteria. While these field data were "noisy," severe alterations on the order of 50% reductions in the abundance or diversity of potential prey items were not obvious at zinc concentrations less than about five times or more greater than the chronic zinc criterion. At lower zinc concentrations close to the criteria, it is not clear if lower reductions in the abundance or diversity of potential prey items would be of a magnitude to adversely affect juvenile salmon foraging ability.

***Bioaccumulation.*** Zinc can clearly bioaccumulate in the environment. Zinc has been found to be elevated in benthic invertebrates in field studies conducted in streams with elevated zinc in sediments or water (Ingersoll *et al*. 1994; Woodward *et al*. 1994; Maret *et al*. 2003; Kiffney and Clements 1996). Farag *et al*. (1994) determined that continuous exposure to zinc at the proposed chronic criterion concentration was associated with bioaccumulation of the metal by juvenile and adult rainbow trout. Mount *et al*. (1994) determined that tissue concentrations increased in rainbow trout fry fed a diet containing enriched levels of zinc.

However, the issue of zinc bioaccumulation in salmonids is confounded by naturally high tissue concentrations and the ability of fish to regulate internal concentrations. Alsop *et al*. (1999) determined that tissue concentrations of zinc in fish exposed to approximately one to two times the acute criterion were not a good indicator of non-lethal, chronic zinc exposure. Recent reviews have concluded that while zinc is bioaccumulated in the environment, there is no evidence for biomagnification in the food chain because zinc concentrations in higher trophic levels are not higher than those in lower trophic levels (Hogstrand 2011; Cardwell *et al.* 2013).

### *2.4.10.3. Summary for Zinc*

Zinc is primarily an acute toxin to salmonids, hence the acute criterion is of greater environmental relevance than the chronic criteria. A confusing aspect of the literature on zinc toxicity to salmonids is the great disparity in reported effects between studies. Across different studies, $EC_{50}$ values for rainbow trout with zinc at similar test hardnesses varied by an order of magnitude. Said differently, zinc at criteria concentrations has been found to be highly toxic and killed most of the fish exposed (Figure 2.4.10.2), but in other tests, concentrations well in excess of the criteria killed no fish. This disparity may be due to differences in the sensitivity of fish at different sizes as they develop. While it is commonly assumed that the smallest organisms will be most sensitive (e.g., ASTM 1997), this is clearly not always the case with zinc. Instead for salmonids, the likely pattern is that the newly hatched, smallest fish appear resistant to zinc, lose resistance as they grow during the first and second months after hatching, and then regain resistance as the fish become older and larger. This suggests that even though most of the studies reviewed that addressed zinc toxicity to listed Snake River salmon and steelhead did not show adverse effects below criteria values (Figure 2.4.10.1 and 2.4.10.3c) the risk from exposure to zinc may have been underestemiated because the studies did not distinguish between sensitive life stages, and not examined effects to listed steelhead and salmonids at their most vulnerable post-hatch stages.

Adverse effects were found at sub-criteria values in tests conducted at hardnesses less than 25 mg/L, a few other tests at moderately low hardness of 35 mg/L with the most sensitive size fish tested (Figure 2.4.10.2), and multiple tests reported by Hansen *et al*. (2002c) with rainbow trout. The preponderance of the information reviewed indicate that in waters with hardness less than about 25 mg/L as $CaCO_3$ the Idaho Zn criteria would not be sufficiently protective of listed Snake River salmon and steelhead if they were exposed at their most sensitive life stages. If alternatively, the current IDEQ zinc criteria were determined using the actual water hardness, instead of the assumed hardness of 25 mg/L, most of those data indicate that the criteria would then be sufficient to avoid harm in most of the studies reviewed.

## 2.4.11. The Effects of EPA Approval of the Chromium III and VI Criteria

Chromium can exist in oxidation states from –II to +VI, but is most frequently found in the oxygenated waters in its hexavalent state (VI). The chromium (III) is oxidized to chromium (VI) and under oxygenated conditions chromium (VI) is the dominant stable species in aquatic systems. The chromium (VI) is highly soluble in water and thus mobile in the aquatic environment.

No single mechanism of impairment has been shown to be responsible for chromium toxicity in fish. The symptoms include changes in tissue histology, temporary reductions in growth, the production of reactive oxygen species, and impaired immune function (Reid 2011).

***Comparison of Chromium Criteria.*** The criteria under review for chromium (III) are 311µg/L acute criterion and 101 µg/L chronic criterion at a hardness of 50 mg/L. Criteria for chromium (VI) are 15µg/L acute, and 10 µg/L chronic and are not hardness dependent. The chromium (III) toxicity is weakly influenced by water hardness. It is unclear if the same if true for chromium (VI), which has been considered more toxic than chromium (III) (EIFAC 1983; Eisler 1986; EPA 1985h). Hexavalent chromium (VI) exists in solution in an anionic rather than cationic form; therefore, calcium competition, one of the main reasons that hardness mitigates toxicity of some metals such as cadmium, nickel, and zinc, does not occur. The acute standards for chromium (III) are unique from analogous standards for the other metals of concern because the total recoverable to dissolved CF (0.316) is substantially smaller.

***Baseline Concentrations of Chromium.*** Although weathering processes result in the natural mobilization of chromium, the amounts added by anthropogenic activities are thought to be far greater. Major sources are the industrial production of metal alloys, atmospheric deposition from urban and industrial centers, and large scale wrecking yards and metals recycling and reprocessing centers (Reid 2011). Because of the rural nature of the action area, transportation costs, and distance to major urban or industrial sources no growth in the business types that discharge chromium is expected in the action area.

The few data on chromium concentrations in Idaho that were located were low relative to aquatic risk concentrations. In the Stibnite Mining District in the EFSFSR basin, total chromium concentrations collected under low flow conditions in September 2011 ranged from <0.2 µg/L to 0.24 µg/L (*http://waterdata.usgs.gov/nwis*, HUC 17060208). In the Blackbird Mining District, concentration of chromium in seeps and adits around the Blackbird Mine were not higher than average background filtered surface water concentrations near the Blackbird Site (<2.9 µg/L) (Beltman and others 1993).

#### *2.4.11.1. Species Effects of Chromium Criteria*

There are more toxicity test data available for chromium (VI) than chromium (III). Toxicity tests on salmonid species indicate that adverse effects do not occur to any life stage of salmonids when exposed to ambient dissolved concentrations at or below the chromium (VI) criteria. This includes the results reported by Birge *et al*. (1978, 1981) and Sauter *et al*. (1976) regarding early lifestage survival. We identified only one study of trivalent chromium toxicity to salmonids, and in this test adverse effects were observed at a concentration that was higher than the chronic criterion (LOEC 48 µg/L vs. the chronic criterion of 24 µg/L). The magnitude of effects at these treatments was fairly slight, with 4% reduction in length of ELS fish after 30 days exposures (Stevens and Chapman 1984).

Patton *et al*. (2007) reported that the survival, development, and growth of early life stage fall Chinook salmon were not adversely affected by extended exposures (i.e., 98 day) to hexavalent chromium ranging from 0.79 to 260 µg/L.

Conflicting results have been obtained from fertilization tests of salmonids under exposures to chromium (VI). Billard and Roubaud (1985) determined that the viability of rainbow trout sperm (but not ova) was adversely affected when exposed directly to a total chromium concentration equal to 5 µg/L, which is below the chronic criterion. Yet Farag *et al*. (2006) found that total chromium concentration ranging from 11 to 266 µg/L or to a chromium (VI) concentration of 130 µg/L did not affect the fertilization process of Chinook salmon or cutthroat trout. Farag *et al*. (2006) suggested that the differences might be because of different species tested, but because cutthroat and rainbow trout are so closely related, the differences seem more likely from the different methodologies used. The time allowed for exposure to chromium during fertilization was 1 minute during Farag *et al*.'s more recent study versus 15 minutes for the study conducted by Billard and Roubard (1985). The shorter time used by Farag *et al*. (2006) more closely mimicked fertilization events that may occur under river conditions where velocities of the water at the substrate are fast and motility of sperm is short-lived. Also, Farag *et al*. (2006) reported that the ova were held in exposure water for 1.5 hours of water hardening after fertilization to more closely mimic natural conditions in which eggs continue to absorb water for approximately 1.5 hours after fertilization. The ova were not exposed to chromium during water hardening in the study performed by Billard and Roubard (1985). Farag *et al*. (2006) concluded that the instantaneous nature of fertilization likely limits the potential effects of chromium on fertilization success. Neither Billard and Roubard (1985) or Farag *et al*. (2006) analyzed chromium speciation for most treatments, but in these oxygenated tests the chromium is expected to be present as chromium (VI) (Reid 2011).

The conflicting results of the Billard and Roubard (1985) and the Farag *et al*. (2006) studies do result in some uncertainty; however, the latter study by Farag *et al*. (2006) seems more persuasive. Thus, the current chronic chromium (VI) criterion of 10 µg/L is likely protective of Chinook salmon fertilization, based on the instantaneous nature of fertilization limiting effects.

***Behavioral Effects***. Anestis and Neufeld (1986) studied avoidance behavior of juvenile rainbow trout exposed to chromium (VI) and determined a threshold concentration for non-acclimated fish that was equal to 28 µg/L, which is above the acute AWQC. Fish that were acclimated to elevated levels of chromium (VI) required higher concentrations to elicit an observable effect. Dauble *et al*. (2001) describe laboratory avoidance/preference tests that showed that juvenile chinook salmon can detect and avoid chromium at concentrations >=54 µg/L under conditions of 80 mg/L hardness. Thus, there is no evidence of altered behavior in salmonids exposed to chromium (VI) concentrations below either the acute or chronic criterion. We did not locate similar data for chromium (III).

#### *2.4.11.2. Habitat Effects of Chromium Criteria*

***Toxicity to Food Organisms.*** The available data suggest that chromium VI may be much more toxic to some aquatic invertebrates than to fish. NMFS did not locate any chronic tests with aquatic insects, but chronic and some acute tests with cladocerans and amphipods were very sensitive, with adverse effects noted at concentrations below the criteria.

The chromium (VI) tested as sodium dichromate chromate, was extremely toxic to the amphipod *Hyalella azteca* in 7-day tests with a $LC_{50}$ of 3.1 µg/L (Borgmann and others 2005a). Hyalella azteca were also found to be highly sensitive to chromium (VI) by Besser *et al*. (2004). They reported the threshold for chromium (VI) toxicity to *H. azteca*, was between 10 and 18 µg/L, with a NOEC of 10 µg/l which is the same as the chronic criterion concentration (Besser and others 2004). Cladocerans have been reported to experience acute and chronic effects at concentrations below the acute and chronic criteria, respectively, for both chromium (III) and (VI). Data in EPA (1985h) indicate reduced survival and reproductive impairment of daphnids at chromium (III) and (VI) concentrations as low as 4 and 10 µg/L, respectively. These concentrations are less than and equal to the chronic criterion for each respective valency. Most studies, however, have determined toxicity to daphnids occurs at higher concentrations than the criterion.

Data summarized in EPA (1985h), EIFAC (1983), and Eisler (1986) suggest that other invertebrate taxa that juvenile salmonids may feed on generally experience mortality at chromium (III) and (VI) concentrations that are well above the acute criterion. More recently, Canivet *et al*. (2001) obtained 240-hour chromium (VI) $LC_{50}$s for larvae of a trichopteran and an ephemeropteran that were well above the acute and chronic criteria.

Salmonid food items appear to be unimpacted by chromium at criteria concentrations under most circumstances. The proposed criteria may only be harmful to food organisms of listed salmon and steelhead if daphnids or amphipods are the primary food source (e.g., downstream of an impoundment in an otherwise oligotrophic system).

***Bioaccumulation***. There is evidence that invertebrates and salmonids bioaccumulate hexavalent chromium when exposed to ambient water concentrations that are above the chronic criterion. Uptake is influenced by water temperature, pH, other contaminant concentrations, fish age and sex, and tissue type (EIFAC 1983; Eisler 1986). Calamari *et al*. (1982) determined that liver, kidney, and muscle tissue concentrations of chromium were elevated in rainbow trout after 30, 90, and 180 days of exposure to 200 µg/L. The fish subsequently were able to depurate some, but not all, of the accumulated chromium within 90 days after exposure ended. At higher concentrations (>2000 µg/L), chromium is known to also accumulate in gill and digestive tract tissues of rainbow trout (Eisler 1986). Gill accumulation appears to continue with exposure, whereas the other tissues may achieve equilibrium in 2 to 4 days. Residues tend to remain high in the liver and kidneys in test fish during post-exposure periods. Eisler (1986) reported that tissue concentrations in excess of 4 mg/kg dw were presumptive evidence of chromium contamination, but the biological significance was not clear. Little is known regarding bioaccumulation at concentrations that are below the chronic criteria.

#### *2.4.11.3. Summary for Chromium*

Data reviewed by NMFS indicate few direct adverse effects to listed salmonids at concentrations less than the chronic trivalent or hexavalent chromium criteria. Studies on the effects of hexavalent chromium to salmon sperm are contradictory with one test indicating it is toxic at concentrations below the chronic criteria, and a more recent study showing no effects at criteria concentrations.  Because the more recent study that showed no effects appeared to use a more relevant exposure duration, NMFS find it to be more relieable and concludes that direct adverse effects of chromium to listed salmonids are unlikely at or below criteria.

The amphipod *Hyalella azteca* suffered adverse effects at a test concentration below the chronic criterion in one study but not in another.  Because so few data on long-term effects of chromium to benthic invertebrates are available, this test is interpreted as suggesting adverse effects to food sources are possible.  Bioaccumulation of chromium clearly occurs when water concentrations are high, but relevant data are absent regarding the effects to salmonids when water-borne concentrations are below the chronic criterion.  Because adverse effects to the species or critical habitat should never reach the scale where take occurs, the effects of the proposed action for chromium are very minor.

### 2.4.12.  The Effects of EPA Approval of the Lead Criteria

The acute lead (Pb) criterion proposed for approval is 65 µg/L, and the proposed chronic criterion is 2.5 µg/L, as dissolved (filtered) metals at a hardness of 100 mg/L.

***Baseline concentrations of lead.*** In natural waters, lead is usually complexed with particulate matter resulting in much lower dissolved than total concentrations (Mager 2011).  For instance, in the lead contaminated Coeur d'Alene River of northern Idaho, dissolved lead concentrations rarely exceed 20 µg/L whereas total concentrations often exceed 100 µg/L.  A maximum dissolved lead concentration of 420 µg/L was reported for this location (Clark 2002; Balistrieri and Blank 2008).  The Coeur d'Alene River is north of occupied habitat, as is the Clark Fork River, Idaho, where up to 60 µg/L dissolved lead has been reported (Hardy and others 2005). Within the action area, reliable lead data are sparse but the measured concentrations are quite low.  The highest lead concentration obtained by the Idaho IDEQ/USGS statewide monitoring program within the action area was from the Hells Canyon reach of Snake River near Anatone, Washington  (7 µg/L).  All other measurements from within the Clearwater and Salmon River basins and the Snake River downstream of Hells Canyon dam were <1 µg/L (Hardy and others 2005).  Mebane (2000) reported lead concentrations in the upper Salmon River near the TCM as high as 2 µg/L, but most values were <0.2 µg/L.

#### *2.4.12.1.  Species Effects of Lead Criteria*

Lead toxicity is influenced by species and life stage, metal speciation including whether in organic or inorganic form, hardness, pH, water temperature, and the presence of other metals that act either synergistically or antagonistically depending on the element.  Elevated lead concentrations are associated with long-term effects to salmonids and other fish including: spinal curvature and other deformities; anemia; caudal chromatophore degeneration (black tail); caudal fin degeneration; destruction of spinal neurons; aminolevulinic acid dehydratase inhibition in blood cells, spleen, liver, and renal tissues; reduced swimming ability; increased mucus formation and coagulation over body and gills and destruction of respiratory epithelium; scale loss; elevated lead in blood, bone, and kidney; muscular atrophy and paralysis; teratogenic effects; inhibition of growth; retardation of maturity; changes in blood chemistry; testicular and ovarian histopathology; and death. Fish embryos appear to be more sensitive to lead than older fry and juvenile stages (Hodson *et al*. 1982; EPA 1985f; Eisler 1988b; Sorensen 1991; Farag *et al*. 1994; Mager 2011). Organic lead compounds are generally more toxic than inorganic. Aquatic organisms are influenced more by dissolved than by total lead, because lead characteristically precipitates out in aqueous environments to bed sediments (Eisler 1988b; Sorensen 1991).

***Acute Lead Criterion.*** Available data suggest that toxic effects of lead on salmonids occur above the proposed acute and chronic criteria concentrations. However, the data exhibit wide variation (Figure 2.4.12.1), and there are limited lead toxicity test data available for salmonids, particularly for sublethal or indirect effects. Results for the early life stage are less conclusive than for adults, and there is conflicting evidence regarding the effects. Fish embryos and fry have been found to be more sensitive to lead in terms of effects to development than older life stages (Sorenson 1991, Mebane *et al*. 2008).

***Chronic Lead Criterion.*** We identified several studies that indicate the chronic criterion is at or below the NOEC level for the early life stage (Figure 2.4.12.2). For example, Sauter *et al*. (1976) determined that the threshold for adverse chronic effects to rainbow trout eggs and fry occurred at a lead concentration between 71 µg/L and 146 µg/L, both of which are well above the chronic criterion. Davies *et al*. (1976) determined that in soft water (hardness ~30 mg/L), adverse developmental effects occurred to eggs and sac-fry when exposure concentrations were between 4.1 µg/L and 7.6 µg/L, which are above the proposed chronic criterion. When the eggs were not exposed, effects to sac-fry were found when exposure concentrations were between 7.2 µg/L and 14 µg/L in soft water, and between 190 µg/L and 380 µg/L in hard water (300 mg/L). Other bioassays involving adult trout and their offspring in soft water indicated that there were no adverse reproductive effects occurring when lead concentrations were around 6 µg/L (Davies *et al*. 1976); this level is also above the proposed chronic criterion. The results of Birge *et al*. (1978; 1981) indicated that rainbow trout embryos exposed for more than 4 days can begin to die when lead concentrations are between 2.5 µg/L and 10.3 µg/L, and hardness is 100 mg/L as $CaCO_3$. In contrast, Mebane *et al*. (2008) exposed rainbow trout embryos to lead in low hardness water (20 mg/L) for about 10 days but only noted mortalities at much higher exposures (≥54 µg/L Pb) than did Birge *et al*. (1978; 1981).

Organic forms of lead appear to be much more toxic than inorganic forms, but are not addressed in the proposed criteria. Wong *et al*'s (1981; experiment 1) data indicated a 7-hour $LC_{10}$ of approximately 3.5 µg/L of tetramethyl lead for rainbow trout fry (at hardness of 135 mg/l) and a time-independent $LC_{50}$ (incipient lethal level, ILL) of approximately 24 µg/L for juveniles. A series of field and laboratory tests with brook trout that had been exposed to environmentally relevant concentrations of lead from combustion of leaded fuel had contrasting results to those of Wong (1981). Trout exposed for 3 weeks in melted snow that was contaminated with lead from snowmobile exhaust showed increased bioaccumulation of lead and decreased swimming stamina at waterborne concentrations as low as 12.5 µg/L dissolved lead (Adams 1975).

**Table 2.4.12.1. Comparison of the most sensitive chronic endpoints (in µg/L, except hardness in mg/L) from relevant studies with dissolved lead and salmonids or benthic invertebrates (i.e., potential prey) for related species**

| Species, test type, Endpoint | Duration | Hardness | NOEC | LOEC | MATC | EC10 | EC20 | Idaho CCC | Note |
|---|---|---|---|---|---|---|---|---|---|
| Mayfly, *Baetis tricaudatus*, EIL-s | 10 d | 20 | 103 | 160 | 130 | 37 | 66 | 0.54 | 1 |
| Midge, *Chironomus dilutus*, LC-g | 55 d | 32 | 57 | 75 | 65 | 15 | 28 | 0.71 | 1 |
| Midge, EIL-s | 27 d | 48 | 109 | 497 | 233 | 108 | 149 | 1.1 | 2 |
| Snail, *Lymnaea stagnalis*, JGS-g | 30 d | 102 | 12 | 16 | 14 | ~1 | <~2 | **2.6** | 2 |
| Amphipod, *Hyalella azteca*, LC | 42 | 138 | 6.3 | 16 | 10 | | 2.8 (EC25) | **3.6** | 3 |
| Daphnid, *Ceriodaphnia dubia* | 7 d | 20 | 51 | 99 | 71 | nc | nc | 0.54 | 4 |
| Rainbow trout, ELS-s | 69 d | 20 | 24 | 54 | 36 | 26 | 34 | 0.54 | 1 |
| Rainbow trout ELS-g | 62 d | 29 | 8 | 18 | 12 | 7 | >87 | 0.64 | 1 |
| Rainbow trout ELS-g | 60 d | 35 | 71 | 146 | 102 | 79 | 99 | 0.75 | 5 |
| Rainbow trout ELS-d | 1.6 yr | 28 | nc | nc | nc | 8.8 | 10.5 | 0.61 | 6 |
| Rainbow trout JGS-d | 1.5 yr | 28 | nc | nc | nc | 8.2 | 10.5 | 0.61 | 6 |
| Brook trout, *Salvelinus fontinalis*, LC-d,g | 3 y | 44 | 39 | 84 | 57 | nc | nc | 1.0 | 7 |
| Fathead minnow, *Pimephales promelas* | 30 d | 19 | 17 | 62 | 32 | nc | 21.6 | 0.54 | 8 |

Table abbreviations: d – days, y – years; nc – not calculable, either because treatments were not replicated precluding statistical NOEC/LOECs; inadequate partial responses occurred or because treatment responses were not reported; *Test type*: EIL – early instar larval test; ELS – early-life stage test; JGS –juvenile growth and survival test; LC – life cycle test; *Most sensitive endpoint responses*: s – survival, g – growth, d – spinal deformity, b – biomass, u – unknown. Hardness in mg/L as $CaCO_3$. To improve comparability, 10th percentile effects concentrations (EC10) were calculated

*Numbered table notes*: 1. Mebane *et al*. 2008; 2. Estimated from data graph in Grosell *et al*. (2006a); 3. Besser *et al*. (2005); 4. Jop *et al*. (1995); 5. Sauter *et al*. (1976); 6. Davies *et al*. (1976); 7. Holcombe *et al*. (1976); 8. Grosell *et al*. (2006b); 9. Davies *et al*. (1993)

***Hardness as a Predictor of Lead Toxicity.*** Water hardness is an important influence on inorganic lead toxicity because lead precipitates out of solution as hardness increases. Lead begins to precipitate when hardness reaches 27 mg/L, and toxicity declines significantly as hardness approaches about 50 mg/L (Sorensen 1991). This response is modified somewhat by variation in pH, but hardness appears to be a primary control on lead bioavailability and toxicity. It is unresolved whether lead precipitation in waters with hardness greater than 27 mg/L to 50 mg/L would be associated with adverse effects to aquatic macroinvertebrates, or to the incubation and overwintering life stages of listed salmonids.

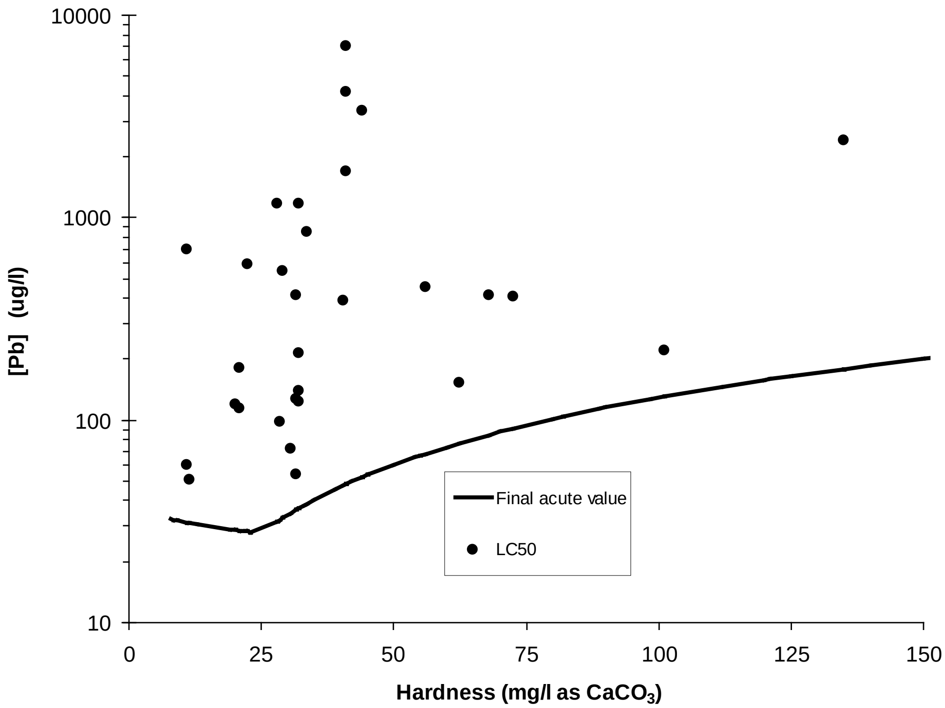



**Figure 2.4.12.1. Acute $LC_{50}$s with salmonids, any life stage vs. the Idaho final acute value for lead.**

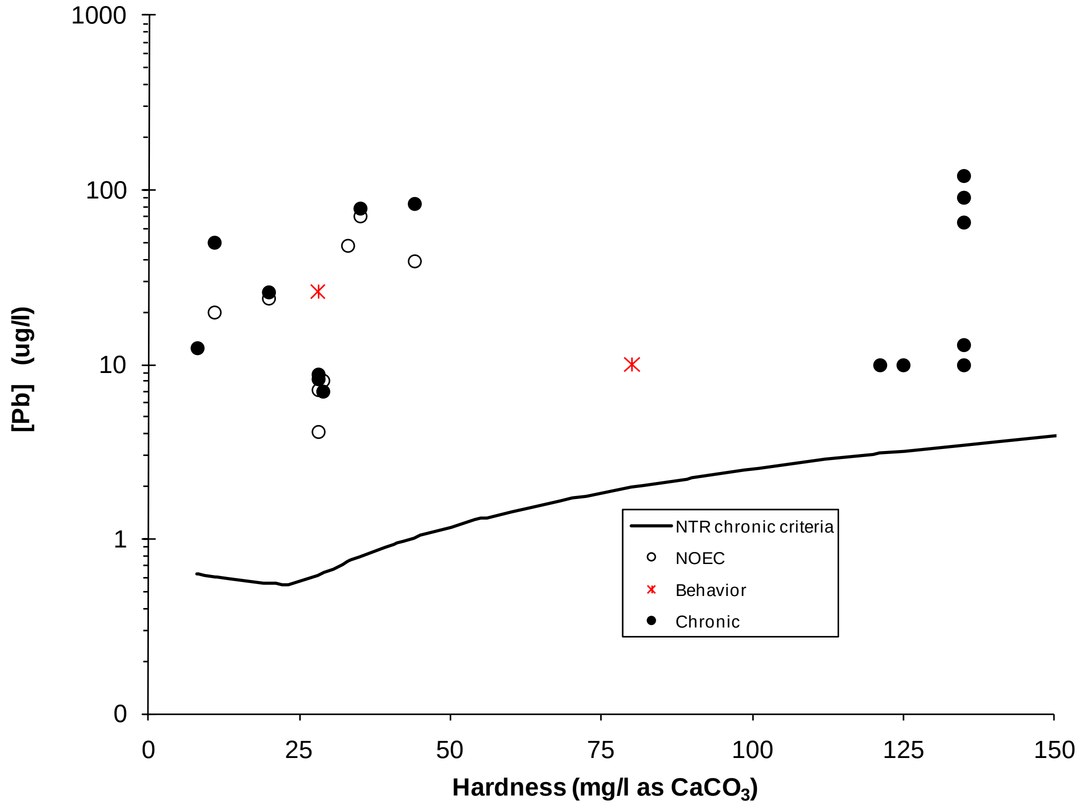



**Figure 2.4.12.2. Chronic effects, no-effects, and avoidance concentrations of lead with salmonids vs. the Idaho chronic criterion concentrations for lead.**

*2.4.12.2. Habitat Effects of Lead Criteria*

***Toxicity to Food Organisms.*** Lead toxicity varies considerably among aquatic macroinvertebrates. Results reviewed in EPA (1985f) and Eisler (1988b) indicate that amphipods are more sensitive than other taxa, and that some freshwater isopods are tolerant of elevated lead levels. Some snail taxa are exceptionally sensitive to lead and suffer reduced growth and mortality at lead concentrations well below the chronic criterion. Amphipods may also be quite sensitive to chronic lead exposures compared to other organisms. The approximate thresholds for adverse effects to the amphipod *Hyalella azteca* were lower than the chronic criterion. The chronic effect threshold for a mayfly and a midge were well above the chronic criterion (Table 1.12.2.1). Salmonids are opportunistic feeders and when snails are abundant in stream, they have sometimes been important food items in salmonid diets (McGrath and Lewis 2007; NCASI 1989). However, less armored prey such as mayflies, midges, aquatic insects and amphipods would be preferred prey items for juvenile salmonids in streams if they are abundant (e.g., Karchesky and Bennett 1999; Muir and Coley 1996; Rader 1997; Sagar and Glova 1987; Suttle *et al*. 2004).

Much data on the acute toxicity of lead to coldwater stream invertebrates under conditions of low hardness and low organic carbon that are representative of much of the Clearwater and Salmon

River habitats in Idaho has recently become available. Testing included several species of mayfly, stonefly, caddisfly, true flies, and snails (Mebane *et al.* 2012). The data indicate that acute mortality of the more sensitive taxa occurred at concentrations that are well above the final acute value for lead. However, whether data from this type of short-term, water-only acute tests with aquatic insects to lead and other metals have any relevance to risks of lead to aquatic insects in nature has been challenged (Buchwalter *et al*. 2007).

Ingersoll *et al*. (1994) determined that while the amphipod *Hyalella azteca* accumulated lead from bed sediments, the level of accumulation was not related to the concentration gradient in the riverbed. Because lead occurs in association with copper, cadmium, and zinc in the field studies reviewed, it is difficult to ascribe a direct adverse chronic effect of lead to aquatic invertebrates at exposure concentrations that are below the chronic criterion.

***Bioaccumulation.*** Lead accumulation is influenced by age, diet, particle size ingested, hardness, pH, water temperature, metal speciation, and presence of other compounds in the water (Eisler 1988b; Sorensen 1991). Bioavailability of lead increases with decreasing pH, organic content, hardness, and metal salt content (Eisler 1988b). Lead precipitation with increasing hardness leads to decreased bioavailability, although the potential for accumulation from precipitated lead still exists (Sorensen 1991). Fish accumulate lead from water or diet but the effects of lead tissue residues is uncertain. Farag *et al*. (1994) determined that adult and juvenile rainbow trout accumulated lead in their gut through their diet, and in gill and kidney tissues when exposed to dissolved lead at concentrations slightly in excess of the chronic AWQC. Other waterborne or dietary lead exposures and field studies have also shown bioaccumulation, but showed few obvious adverse effects at concentrations near chronic criterion (Adams 1975; Davies *et al*. 1976; Holcombe *et al*. 1976; Mount *et al*. 1994; Farag *et al*. 1999; Erickson *et al*. 2010).

#### *2.4.12.3. Summary for Lead*

Potential adverse effects from exposure to lead at concentrations at or below the criterion, are expected to be very minor. The only adverse effects of chronic lead exposures at sub-criteria concentrations were to snails and the amphipod *Hyalella azteca*. In most habitats, listed salmonids would not be expected to be dependent on amphipods and snails for food. Listed salmon and steelhead are unlikely to be injured or killed by exposure to lead concentrations that are at or below the proposed acute or chronic criteria. No evidence of direct adverse sublethal effects occurring at concentrations at or below the chronic criterion to salmonids was found.

### 2.4.13. Organic Pollutants: General Issues

In addition to the general issues (Section 2.4.1) that apply to all contaminants considered in the proposed action, the following issues specific to organic pollutants may create hazards for listed salmon and steelhead.

***Organic Pollutants Toxicity and Exposure.*** Eisler's series of synoptic reviews, EPA's criteria documents, and the World Health Organization's environmental health criteria documents (e.g., WHO 1984) provide a good summary of sources, pathways, and toxic effects of organic pollutants. Most of the organic compounds considered in the proposed action are for organochlorine pesticides (chlordane, dieldrin, aldrin, lindane, heptachlor), used in the past for a variety of agricultural applications, as well as part of measures for controlling insects considered hazardous to human health. The remainder are industrial chemicals (PCBs, PCPs) that have been used widely in the past but are now banned or restricted in the United States. Of the organic contaminants included in the proposed action, only lindane, endosulfan, heptachlor, and PCP are still used at all in the United States, and permitted applications for lindane and heptachlor are very limited. For the most part, these organic contaminants are no longer being released directly into the water column. They generally enter the aquatic environment attached to organic and inorganic particulate matter. However, because they are not highly water soluble and persistent in the environment, they remain sequestered in sediments and provide a continual source of exposure. This is of particular relevance when contaminated streambed sediments are disturbed as part of in-channel work. Organic pollutants may also enter the aquatic environment through non-point surface runoff from contaminated agricultural areas where they have been used in the past. Although the levels of most of these compounds have declined since their use was banned in the 1970s, they are still widely distributed in the environment and found in tissues of aquatic organisms.

Organic contaminants are furthermore rarely found alone in discharges or in the environment. Usually, several compounds are found together in areas where there has been extensive agricultural or industrial activity. In industrialized areas, other classes of contaminants such as metals or aromatic hydrocarbons from petroleum products are also typically present. For instance, the chemical forms of most organic pesticides and PCBs are mixtures that may contain a large number of isomers and congeners of each compound, of which the toxicity and persistence in the environment can vary considerably.

The most direct exposure pathway for dissolved organic compounds to aquatic organisms is via the gills. Dissolved organic compounds are also taken up directly by bacteria, algae, plants, and planktonic and benthic invertebrates. Organic pollutants can also adsorb to particulate matter in the water column and enter organisms through various routes. Planktonic and benthic invertebrates can ingest particulate-bound organic compounds from the water column and sediments and then be eaten by other organisms. Thus, dietary exposure may be a significant source of organic toxic pollutants to aquatic and aquatic-dependent organisms.

Although organic contaminants bound to sediments are generally less bioavailable to organisms, they are nonetheless present, and changes in the environment (e.g., dredging, storm events, temperature, lower water levels, biotic activity) can significantly alter their bioavailability. Feeding habits of fish can determine the amount of uptake of certain organic contaminants, where piscivorous fish are exposed to different levels of organics than are omnivorous or herbivorous fish.

Organic pollutants can have a wide variety of effects on organisms. Exposure to organochlorines can result in damage to gut tissues, disrupt nervous system operation, alter liver and kidney functions, and impair the immune system. Elevated concentrations of many organochlorine compounds can cause growth inhibition, impaired reproduction, and developmental defects that may affect not only the target organisms themselves, but can also impact the growth and survival of predator species further up the food chain. A number of these compounds are promoters that increase the risk of cancer. They may also disrupt immune function and increase the affected animal's susceptibility to infectious disease. Impacts from organic contamination can shift species composition and abundance towards more pollution-tolerant species (e.g., Nimmo 1985; Meador 2006; Rand 1995). Specific examples of these effects are identified for each compound in our analysis.

***Proposed Chronic Criteria Are Based on Maximum Tissue Residues For Human or Wildlife Consumption, Not on Health Effects in Aquatic Organisms.*** For most of the organic contaminants, the chronic ambient water quality criteria are not based on long-term toxic effects in salmonids or other fish species. Instead, they are based on maximum permissible residues for human or wildlife consumption. Numeric criteria that are based on maximum tissue residues considered acceptable for wildlife or human health may not reflect a similar protectiveness of the health of aquatic organisms. Although in some cases these residues may be below those associated with adverse effects in salmonids, adverse effects in fish were not specifically addressed when determining the criteria (EPA 2000d).

***Bioconcentration and Bioaccumulation Factors, Used in Determining and Evaluating Proposed Criteria, Associated With High Variability and Uncertainty.*** An important problem with many chronic criteria for organic pollutants is that the BCFs or BAFs used in their determination may not be accurate. The BCFs determined in the laboratory based on water-borne exposure are typically much lower than field-derived BAFs, and so may significantly underestimate uptake in the natural environment. Even among field-derived bioconcentration factors, estimates can vary by several orders of magnitude. Consequently, it is difficult to determine if BCF-based comparisons of water-borne and tissues concentrations are accurate when evaluating the chronic criteria proposed in this action. However, because a wide range in BCFs appears to exist (EPA 2000d; Meador 2006), such comparisons cannot be discounted and the criteria are evaluated in this Opinion accordingly.

***Insufficient Data for Toxicity of Organic Contaminants That are Mixtures of Different Congeners with Varying Modes of Action.*** Several of the organic contaminants reviewed in this document are not single compounds, but mixtures of a large number of congeners with differing levels of toxicity and modes of action. This is particularly true of PCBs, which are a mixture of over 200 separate congeners, and toxaphene, which is a combination of over 600 isomers. For PCBs, methods such as the calculation of a toxicity equivalence factor (TEF) for those congeners with dioxin-like activity can provide a measure of the overall toxicity of mixtures containing these congeners (Safe *et al*. 1994; Van den Berg *et al*. 1998). For toxaphene, the toxicity of its various isomers is only beginning to be documented (e.g., de Geus *et al*. 1999). However, in neither case are these issues dealt with in existing water quality standards.

***Water-borne Exposure from Contaminated Sediments.*** Because hydrophobic compounds are expected to show a similar or proportional affinity for the lipid of an organism as that for octanol, the degree of partitioning exhibited between water and octanol, as characterized by the partition coefficient $K_{ow}$, can be a useful means for evaluating and predicting bioaccumulation (Mackay 1982; Di Toro *et al*. 1991). For organic compounds that are not metabolized, the relationship between the BCF and $K_{ow}$ is strong (Mackay 1982). The expected ww BCF for a non-metabolized hydrophobic compound is a function of the lipid content of an organism and the value of $K_{ow}$ for the compound. The standard equation for determining the expected BCF is:

$$BCF = 0.046 \times K_{ow}$$

which is derived from fish studies and is based on an average lipid content of 4.6% ww McCarty (1986). This relationship is used in this Opinion for evaluating effects related to exposure and bioconcentration of the toxic organic pollutants addressed by the IWQS.

Sediment concentrations that would result in organic toxic pollutant concentrations in the water column can be calculated using the equation (Di Toro *et al*. 1991):

$$SQC_{oc} = K_{oc} \text{ X } F_{CV}$$

where:

$SQC_{oc}$ = sediment contaminant concentration in µg/kg organic carbon
$K_{oc}$ = partitioning coefficient for sediment organic carbon
$F_{cv}$ = the chronic water quality criterion in µg/L

$K_{oc}$ can be calculated from the octanol/water partitioning coefficient, $K_{ow}$, using the equation:

$$\text{Log}_{10}(K_{oc}) = 0.00028 + 0.983 \text{ X } \text{Log}_{10}(K_{ow})$$

This equation is used in the analysis of effects below for evaluating the potential for water-borne exposure concentrations of organic pollutants that are at or below the Idaho criteria.

***Organic Pollutants: Analysis of Individual Chemicals.*** In the analysis of organic pollutants, the effects of each organic toxic substance of concern are identified, and the proposed criteria are compared with data available to NMFS that describe sample background concentrations and the results of salmonid toxicity tests. Where possible, effects to the food sources of listed salmonids, and effects related to bioaccumulation, are also identified.

The $LC_{50}$s are used in this Opinion to evaluate criteria, rather than the more germane threshold toxicity concentrations. This reflects standard toxicological procedures, which seldom determine toxicity threshold concentrations. The relation between the two measures of toxicity response is not always linear, so use of a consistent, multiplicative CF is precluded.

The following analysis focuses on exceedences of each parameter individually. Where studies indicate an individual contaminant criterion is not likely to harm listed salmonids, the body of evidence may not support such an indication when several contaminants are near or equal to the proposed criteria in the same water sample. As a case in point, Laetz *et al*. (2009) determined that several combinations of organophosphate pesticides were lethal at concentrations that were sublethal in single-chemical trials. As described in the section on mixture toxicity, the proposed criteria cannot be applied individually to assess the effects of additive or greater that additive toxicity, a significant limitation that can adversely affect listed salmonids.

2.4.14. The Effects of EPA Approval of Pentachlorophenol (PCP) Criteria

Pentachlorophenol is a chlorinated hydrocarbon that is used primarily as an insecticide and fungicide, but also secondarily as an herbicide, molluscicide, and bactericide (Eisler 1989). Its primary application is to protect timber from fungal rot and wood-boring insects. According to the EPA, PCP is a Registered Use Product (RUP) in formulations as a wood preservative. A RUP may be purchased only by a certified applicator. Technical grade PCP is approximately 86% pure and historically has been contaminated with dioxins and hexachlorobenzene. Dioxin contamination is the main reason PCP was reclassified as a RUP in 1987 (EPA 2008).

Commercial forms of PCP that include manufacturing impurities resulted in reduced growth and survival in fathead minnow (*Pimephales promelas*) at PCP concentrations that are approximately five times lower than concentrations of purified PCP that caused similar effects to fish (Cleveland *et al.* 1982). The excess toxicity is presumably due to the impurities that occur in the commercial preparations. These differences should be considered when research studies on toxicity are being evaluated and when environmental concentrations from known sources are compared to criteria values.

Pentachlorophenol has a strong propensity to associate with the organic carbon of sediment and lipid of organisms, as represented by a relatively high value octanol-water partition coefficient ($\log_{10}(K_{ow}) = 5$; Eisler 1989). One of the primary toxicity mechanisms of PCP is inhibition of oxidative phosphorylation, which causes a decrease in the production of adenosine triphosphate ATP which is fundamental to metabolism in plants and animals. One consequence of this impairment is increased basal metabolism, resulting in increased oxygen consumption and high fat utilization. The effects of PCP may reduce the availability of energy for maintenance and growth, thus reducing survival of larval fish and ability of prey to escape from a predator (Johansen *et al.* 1985; Brown *et al.* 1985; Eisler 1989).

Pentachlorophenol is known to cause several types of adverse effects in animals including dysfunction of the reproductive, nervous, and immune systems; hormone alterations; and impaired growth. In general, fish growth and behavioral endpoints have been shown to be sensitive indicators of PCP exposure (Webb and Brett 1973; Hodson and Blunt 1981; Dominguez and Chapman 1984; Brown *et al.* 1985). Pentachlorophenol is also considered a probable human carcinogen.

The highest PCP concentrations near the action area were almost three orders of magnitude lower than the most stringent applicable criteria value (0.00047 µg/L in the discharge from Brownlee Dam vs 6.2 µg/L for the fish consumption based water quality criteria) (Table 2.3.1).

*2.4.14.1. Species Effects of Pentachlorophenol Criteria*

The criteria for PCP established by the EPA are pH dependent (EPA 1986b). In general, the toxicity of PCP increases with decreasing pH. At pH 4.74, half of PCP molecules are ionized (anions) and half are non-ionized. At pH 6, the ratio between the ionic and non-ionized forms is 18 (i.e., the concentration of the ionized form is 18 times greater than the non-ionized form), and at pH 7 the ratio is 182. Studies have concluded that the ionic form of PCP is less toxic, primarily because it is less likely to cross membranes (Spehar *et al.* 1985). A correction factor is therefore needed for assessing bioaccumulation and toxicity to account for the effect of pH on the speciation of PCP. To determine the freshwater criteria as a function of pH the following equation is used:

CMC = $\exp^{(1.005 \text{ x pH} - 4.83)}$ (*in µg/L*)
CCC = $\exp^{(1.005 \text{ x pH} - 5.29)}$ (*in µg/L*)

For example, at a pH of 7.0, the corresponding criteria are 9.1 µg/L and 6.7 µg/L for acute and chronic exposures, respectively. At a pH of 8.0, the corresponding criteria are 25 µg/L and 18 µg/L for acute and chronic exposures, respectively.

***Acute Pentachlorophenol Criterion.*** Data contained in the EPA's AQUIRE database indicates that most 96-hour PCP $LC_{50}$s for salmonids are in the 10 µg/L to 80 µg/L range, with the lowest reported for cutthroat trout (Mayer and Ellersieck 1986). Van Leeuwen *et al.* (1985) determined the 96-hour $LC_{50}$ to be 18 µg/L at pH 7.2 for early fry of rainbow trout with 95% confidence intervals ranging between 10 µg/L and 32 µg/L. The acute PCP criterion at pH 7.2 is 11 µg/L, suggesting that some mortality could occur at or close to the acute criterion concentration. Other tests with rainbow trout and coho salmon have produced higher (less sensitive) results. Dominguez and Chapman (1984) obtained a 96-hr $LC_{50}$ of 66 µg/L with steelhead fry at pH 7.4. The acute PCP criterion at pH 7.4 is 14 µg/L. Dwyer *et al.* (2005a) tested rainbow trout, two other "standard" test species (fathead and sheepshead minnows) and 17 endangered or threatened species with PCP. They obtained a 96-hr $LC_{50}$ for rainbow trout of about 160 µg/L and $LC_{50}$s for other listed salmonids ranged from 110 to 170 µg/L. The most sensitive species tested with PCP was the Atlantic sturgeon with an $LC_{50}$ <40 µg/L. All of the salmonid $LC_{50}$s were well above the acute criterion of 20 µg/L, for the pH 7.8 waters tested (Dwyer *et al.* 2005b). Pacific salmon were not among the species tested, and thus the rainbow trout are probably the closest surrogate from Dwyer's study. Hedtke *et al.* (1982) conducted multiple tests of the acute toxicity of PCP to juvenile coho salmon at different life stages. The smallest swim-up fry were the most sensitive, with $LC_{50}$s of about 45 µg/L, compared to the acute criterion of 9 µg/L at pH 7 (Hedtke *et al.* 1982). Thus some tests suggest acute toxicity is possible in salmonids when water concentrations are near the acute criterion, yet other results found acute toxicity only at PCP concentrations three times or more greater than the acute criterion. Additionally, because the available data were $LC_{50}$ values, and did not report the actual lower bounds of lethality, the steepness of the dose-response curve is certain, and the lower limit for water concentrations of PCP that may cause mortality in listed salmonids is thus uncertain.

***Chronic Pentachlorophenol Criterion.*** A review of chronic effects with salmonids indicate that with the exception of one study with sockeye salmon, the thresholds of adverse effects are above the chronic criterion.

With sockeye salmon, Webb and Brett (1973) showed that thresholds for decreased growth rates and food conversion efficiencies of approximately 1.74 to 1.8 µg/L at pH 6.8 following 28-day exposures. These effects occurred at PCP concentrations less than the chronic criterion of 4.7 µg/L for test pH of 6.8. From their Figure 5, at the chronic criterion concentration of 4.7 µg/L

PCP, growth rates and food conversions were about 90% of those in the control treatments. In their abstract, Webb and Brett (1973) listed the 1.74 to 1.8 µg/L values not as thresholds of effect (i.e., highest concentrations with no effects) but rather as $EC_{50}$s for growth and food conversion. A 50% reduction in growth rate or food conversion at concentrations lower than the chronic criterion would be an extremely severe and unacceptable effect. However, further inspection of their results, especially bottom of their p. 504 and their Figure 5, supports a different interpretation. The fish died before growth rates or food conversions were reduced by 50%, and the most severe growth rate and food conversion reductions measured (~ 45% reductions) occurred at about 50 µg/L PCP. No or only low effects were apparent at 1.7 to 1.8 µg/L.

Hodson and Blunt (1981) also observed reduced weight, growth rate, and biomass in rainbow trout exposed 20 µg/L and greater PCP concentrations over 4 weeks from embryo to fry stages, at pH of about 7.8 (chronic criterion = 13 µg/L). Dominguez and Chapman (1984) tested steelhead trout in a 72-day test and found a threshold of effects at about 10 µg/L, which is just above the chronic criterion of 8.6 µg/L for their average test conditions. Besser *et al.* (2005b) tested the effects of PCP on rainbow trout in 60-day tests and found the threshold ($EC_{10}$) for reduced growth was about 40 µg/L at pH 8.3. This was about twice the chronic criterion of 21 µg/L for pH 8.3.

Blood chemistry changes in juvenile Chinook salmon (altered blood urea and glucose levels) occurred following PCP exposures to 3.9 µg/L of PCP (nominal). No differences in survival, growth, feeding, or schooling behavior were noted (Iwama *et al.* 1986). Exposures to 39 µg/L PCP (nominal) killed all fish after 8 days. Chronic criterion were 3.5 to 5.2 µg/L for the conditions of the tests (pH 6.5 to 6.9). Nagler *et al.* (1986) determined the occurrence of oocyte impairment in rainbow trout at 22 µg/L (pH 7.5).

***Behavioral Effects*** Little *et al.* (1990) examined post-exposure behavioral effects in rainbow trout at exposure concentrations that were from ten to 100 times less than the acute criterion of 20 µg/L. A statistically significant reduction in the percent survival by trout that were preyed on by largemouth bass occurred at an exposure concentration of 0.2 µg/L. A similar response may be expected for salmon. Survival of trout was 32% to 55% in these predation studies compared to the control at 72%. This equals reductions in fish numbers of 28% to 55% in treatments compared to the control condition. Statistically significant reductions were also observed in the number of *Daphnia* consumed and swimming activity when fish were exposed to a PCP concentration of 2 µg/L and a significant decrease in the strike frequency by trout on *Daphnia* occurred at 20 µg/L. The exposures in Little *et al.* (1990) were conducted for 96-hours under static test conditions, and were based on nominal concentrations. The authors also expressed some concern about contaminants in the formulation used (technical grade PCP). Acetone was used as a carrier for PCP exposure in treatments and controls, which is very common in such experiments, but it is not likely to have contributed to toxicity. The concentration of acetone was 41 µg/L, which is considered very low. Acetone produces very low toxicity in salmonids (Majewski *et al.* 1978) and it is volatized or biodegraded in a matter of hours (Rathbun *et al.* 1982), implying that acetone was not likely a factor in the observed results.

*2.4.14.2. Habitat Effects of Pentachlorophenol Criteria*

***Toxicity to Food Organisms.*** Eisler (1989) reviewed the effects of PCP on invertebrate growth, survival, and reproduction and reported adverse effects in the range of 3 µg/ to 100 µg/L. It appears that most invertebrates are less sensitive than fish to PCP concentrations in water and therefore may be protected by the proposed criteria. There are, however, studies showing adverse effects to invertebrates exposed to water concentrations below the chronic criterion. Hedtke *et al.* (1986) determined reproductive impairment in a daphnid at 4 µg/L and pH 7.3. Borgmann *et al.* (1989) found that 23 µg/L PCP reduced the amphipod *Gammarus* survival to only 25% of controls, while the amphipod *Hyalella* was only affected at 100 µg/L and above. The chronic criterion for Borgmann's tests ranged from about 19 to 35 µg/L, for pHs ranging from 8.2 to 8.8. Acute responses of amphipods were much higher than acute PCP criteria with $LC_{50}$s ranging between 92 and 790 µg/L. The corresponding acute criterion values were 6 to 41 µg/L for the test pH's of 6.5 to 8.5.

***Bioaccumulation.*** Like other organic pollutants, PCP exhibits a tendency to be bioaccumulated by fish. Van den Heuvel *et al.* (1991) reported BCFs for rainbow trout exposed to PCP (pH 7.6) to be between 411 and 482. Metabolism of PCP is relatively rapid in rainbow trout (McKim *et al.* 1986; Glickman *et al.* 1977), and this is likely true in other salmonids as well. Nevertheless, the elimination rate of this compound is sufficiently slow that it takes 11.7 days for tissue concentrations to reach 95% steady state (McKim *et al.* 1986). According to the data provided in McKim *et al.* (1986) a 96-hour exposure will produce tissue concentrations that are only 63% of steady state. Therefore, any assessment of the maximum attainable tissue concentration and resulting biological response for a given exposure concentration must consider a longer time period (e.g., approximately 12 days) to reach that level. An estimate of the steady-state wet-weight BCF for salmonids is 4,600 (~23,000 dw) using the octanol-water partition coefficient for PCP ($\log_{10}(K_{ow}) = 5$). However, lower than predicted BCF values are common in fish and are likely due to metabolism of PCP.

Bioaccumulation of PCP is pH dependent, because pH determines the proportions of ionized and unionized PCP, which is directly related to bioaccumulation potential. The ionic form of PCP is less likely to bioaccumulate in organisms in large part because it is less likely to be taken up in the first place (Spehar *et al.* 1985). Spehar *et al.* (1985) determined the following regression equation relating BCF (wet weight) and pH for PCP uptake by the fathead minnow:
Log BCF = 4.80 - 0.28 x pH.

2.4.14.3. Summary for Pentachlorophenol

Some studies indicate the proposed acute PCP criterion is at the level where some acute toxicity will occur. Other studies showed that $LC_{50}$s for salmonids were well above the proposed acute water quality standard. Most studies of chronic effects reported the onset of adverse effects occurred at least slightly above the chronic criterion, although a single study found reduced growth in sockeye salmon at lower concentrations than the chronic criterion. Rainbow trout exposed to PCP concentrations far below the chronic criterion showed reduced ability to evade predators, and reduced ability to capture prey. Both the chronic and acute criteria will likely have some effect on listed species or their food sources.

Pentachlorophenol is not likely to be a component of NPDES discharges, but may be used in the treatment of wood that finds its way into inwater or overwater structures so the exposure risk, while very small, is not discountable

## 2.4.15. The Effects of EPA Approval of the Aldrin/Dieldrin Criteria

Aldrin and dieldrin are synthetic cyclic chlorinated hydrocarbons called cyclodienes. Although aldrin has been used in higher quantities than dieldrin, both were used extensively in the 1950s and 1960s as soil insecticides. At that time, they were two of the most widely used domestic pesticides in the United States (EPA 1980a). However, the EPA cancelled the registration for both compounds in 1975 (Biddinger and Gloss 1984).

Once aldrin has been applied to any aerobic and biologically active soil, it rapidly undergoes a metabolic epoxidation reaction that converts it to dieldrin (EPA 1980a; Wolfe and Seiber 1993). In fish, the epoxidation of aldrin to dieldrin occurs via a mixed-function oxidase system, which has been demonstrated in golden shiners, mosquitofish, green sunfish, bluegill sunfish and channel catfish (reviewed in Chambers and Yarbrough 1976). Dieldrin can be further modified when exposed to sunlight, via cyclization to photodieldrin (Wolfe and Seiber 1993).

Dieldrin has extremely low volatility and low solubility in water. It is more environmentally stable than aldrin, and is probably the most stable of the cyclodiene insecticides (EPA 1980a; Wolfe and Seiber 1993). For this reason, dieldrin is more frequently observed in the environment than aldrin (reviewed in Biddinger and Gloss 1984). One study, conducted on the environmental fate and transport of dieldrin in the Coralville Reservoir in eastern Iowa, revealed that of the portion of dieldrin that was present specifically in the water column, 74% occurred in fish, 25% was dissolved in water, and less than 1% was adsorbed to suspended solids (Schnoor 1981).

Acute toxicity of dieldrin reported in rainbow trout and other fish includes effects on cardiac muscles, as well as inhibition of oxygen uptake, the central respiratory center, bronchial muscles, and the central nervous system (Lunn *et al*. 1976). Aldrin and dieldrin are similarly toxic to fish, although aldrin is more toxic to cladocerans than dieldrin (EPA 1980a). Additionally, photodieldrin is more toxic than dieldrin (Wolfe and Seiber 1993).

Because it is extremely hydrophobic, dieldrin that is present in fish has a particularly high affinity for fat. However, although it can be mobilized from tissue when the fish is placed in clean water, the dieldrin that has been eliminated then reenters the water, making it available for subsequent uptake by other organisms (EPA 1980a). In channel catfish, approximately 50% of the dieldrin that had accumulated in dorsal muscle due to water-born exposure was eliminated after 14-days post-exposure, with total depuration by 28-days post-exposure. However, dieldrin that had accumulated in tissue due to dietary exposure was eliminated more slowly; at 28-days post-exposure, approximately one third of the original dieldrin in muscle tissue was still present

(Shannon 1977a). For rainbow trout, the predicted time to eliminate 50% of the dieldrin accumulated via dietary exposure is 40 days (Macek *et al*. 1970). In contrast, *Daphnia* required 4 days to eliminate 50% of the photodieldrin that was accumulated in a water-born exposure study (Khan *et al*. 1975) and goldfish required less than 12 hours (Khan and Khan 1974). For the freshwater mussel *Lampsilis siliquoidea*, the half-life of dieldrin was 4.7 days (Bedford and Zabik 1973). Khan and Khan (1974) noted that the initial elimination of dieldrin or photodieldrin from goldfish or *Daphnia* was due to excretion into the surrounding water.

A study by Van Leeuwen *et al*. (1985) examined the effects of water-borne dieldrin on rainbow trout at various early life stages, including fertilized eggs, early and late eye point eggs, sac fry and early fry. In the egg, the yolk acted as a temporary ‘toxicant sink’, but later in development, during the early sac fry stage, dieldrin was delivered from the yolk and began to accumulate in the fish tissue. The highest concentration in tissue was reached at the end of the sac fry stage. The second highest concentration in tissue was reached at the early fry stage, when susceptibility to dieldrin toxicity is most pronounced in early life stages. The clearance rate was also highest at the early fry stage.

### *2.4.15.1. Species Effects of Aldrin/Dieldrin Criteria*

The proposed acute criterion for aldrin is 3 µg/L; a chronic criterion has not been proposed. However, the most stringent criteria that applies to all critical habitats and waters occupied by listed species is 0.00014 µg/L, based on protecting human health from consuming exposed fish (Table 1.3.1*)*. For dieldrin, the acute criterion proposed for approval is 2.5 µg/L and the proposed chronic criterion is 0.0019 µg/L (Table 1.3.1*)*. Because aldrin and dieldrin have somewhat different toxicities, they are evaluated separately below.

***Acute Aldrin Criterion.*** Few toxicity studies were found in the literature that reported acute effects of aldrin on salmonids. Ninety-six hour $LC_{50}$ values were 7.5 µg/L for Chinook and 45.9 µg/L for coho salmon (Katz 1961). Sublethal effects on juvenile Atlantic salmon in a 24-hour exposure study were observed at 33 times and 50 times the proposed acute criterion (Peterson 1973). Two other toxicity studies were found that tested the acute effects of aldrin on salmonid species:

Macek *et al*. (1969) reported a 96-hour $LC_{50}$ value of 2.2 µg/L for rainbow trout exposed at 12.7°C, pH 7.1 in a static experiment with a nominal (95%) aldrin concentration.

Katz (1961) reported a 96-hour $LC_{50}$ value of 17.7 µg/L for rainbow trout exposed at 20 $^{o}$C, pH 6.8-7.4 in a static experiment with a nominal (88.4%) aldrin concentration.

All of these studies involved exposure in static experiments with nominal aldrin concentrations, a type of experimental design that tends to underestimate the toxicity of a contaminant that can hydrolyze or otherwise degrade (e.g., Stephan *et al*. 1985). Although few studies on the acute effects of aldrin on salmonids could be found in the scientific literature, with none more recent than 1973, the observations that one of them determined acute effects at concentrations below the proposed acute criterion for aldrin, and that all studies likely underestimated the toxicity of this pesticide, suggest that the proposed acute criterion for aldrin can be lethal to listed salmonids.

***Acute Dieldrin Criterion.*** Only two studies were found reporting acute toxicity testing results on the effects of dieldrin on salmon or steelhead, with one showing adverse effects at concentrations below the proposed acute criterion:

Chadwick and Shumway (1969) reported a 50% mortality rate for steelhead trout fry when exposed to 0.39 µg/L dieldrin for 3 to 7 days at 12°C in a flow-through experiment with a measured dieldrin concentration.

Katz (1961) conducted acute toxicity tests on two salmon species, using 90% dieldrin at 20°C, pH 6.8 to 7.4, in static experiments with a nominal (unmeasured) dieldrin concentration. Ninety-six hour $LC_{50}$s equal to 6.1 µg/L and 10.8 µg/L were determined for Chinook and coho salmon, respectively.

Available toxicity tests using other salmonid species showed acute effects to rainbow, cutthroat, and brown trout at concentrations below or near the acute criterion:

Shubat and Curtis (1986) reported a 96-hour $LC_{50}$ value of 0.62 µg/L for juvenile rainbow trout exposed at 12°C to 13°C, pH 7.6-8.1, in a flow-through experiment with measured dieldrin concentrations.

Macek *et al*. (1969) reported a 96-hour $LC_{50}$ value of 1.4 µg/L for rainbow trout exposed at 12.7°C, pH 7.1, in a static experiment with a nominal concentration of 85% dieldrin.

Van Leeuwen *et al*. (1985) reported a 96-hour $LC_{50}$ value of 3.1 µg/L for rainbow trout early fry exposed at 9°C to11°C, pH 7.2, in a static experiment with 99% dieldrin.

Shubat and Curtis (1986) reported a 96-hour $LC_{100}$ value of 3.1 µg/L for juvenile rainbow trout exposed at 12°C to 13°C, pH 7.6 to 8.1, in a flow-through experiment with a measured dieldrin concentrations.

The studies outlined above, while including only limited information from the 1960s on the acute toxicity of dieldrin to salmon or steelhead, nonetheless indicate that lethality occurs at levels which are below or slightly above the acute criterion proposed for dieldrin. The scope of the toxic properties of dieldrin is reinforced by the other studies reported above that involved other salmonid species for which lethality occurred at levels that were also below or slightly above the proposed acute criterion for dieldrin. Two of the trout studies (Van Leeuwen *et al*. 1985; Shubat and Curtis 1986) were more recent than the salmon and steelhead studies. Also, these two trout studies were done in flow-through experiments with measured dieldrin concentrations, which are likely to reflect more accurate estimates of toxicity than static experiments with unmeasured, nominal (target) dieldrin concentrations (Chadwick and Shumway 1969; Macek *et al*. 1969). The more recent and flow-through studies reported lethal concentrations that are below or near the proposed acute criterion for dieldrin, suggesting that this criterion could kill listed salmonid species.

***Chronic Aldrin Criterion*.** The EPA has not determined a chronic freshwater criterion for aldrin, based on a lack of available toxicity information (EPA 1980a), and no chronic criterion has been proposed for aldrin in the current action. However, toxicity testing involving other freshwater fish species suggests the potential exists for adverse effects of chronic exposures. In the freshwater teleost *Puntius conchonius*, exposure to 0.0466 µg/L aldrin for 2 to 4 months resulted in a significant increase in disintegrating oocytes, and reduction in the population of stage 0-II oocytes (Kumar and Pant 1988). Other studies have involved much higher concentrations (Singh and Srivastaya 1992; Singh *et al*. 1996) and thus cannot be used to evaluate effects related to development of a chronic criterion for aldrin.

***Chronic Dieldrin Criterion*.** NMFS did not locate any studies on the chronic toxicity of dieldrin to salmon, but found two studies that reported the results of chronic toxicity experiments on rainbow trout. Concentrations at which adverse effects were noted were 95 and 137 times the proposed chronic criterion of 0.0019 µg/L for dieldrin:

Phillips and Buhler (1979) exposed fingerling rainbow trout to 0.18 µg/L dieldrin for 61 days under flow-through conditions and measured dieldrin concentration. This resulted in a reduction in the rate of fat accumulation in fish that were fed a relatively high fat diet (tubificid worms). Whole wet fish tissue concentration that corresponded to this effect was 0.82 or 1.32 mg/kg dieldrin. The effect of dieldrin exposure on fat accumulation was not apparent when fish were fed a relatively low fat diet (moist pellets), thus demonstrating that dieldrin toxicity can be affected by diet composition.

Shubat and Curtis (1986) reported a 12-day $LC_{50}$ value of 0.26 µg/L for juvenile rainbow trout exposed at 12°C to 13°C, pH 7.6-8.1, in a flow-through experiment with a measured dieldrin concentration.

These limited results suggest that the proposed chronic criterion for dieldrin may avoid harming listed salmon subjected to short-term, water-borne exposure. However, they do not indicate whether the proposed chronic criterion is protective against bioaccumulation-related effects. To address this, several dietary exposure studies were evaluated that reported dieldrin tissue concentrations and chronic effects. If a specific chronic effect is associated with a specific tissue concentration and the BCF for dieldrin is known, then the tissue concentration and BCF can be used to back-calculate an estimate of the aqueous dieldrin exposure concentration resulting in an equivalent tissue concentration (and thus an equivalent chronic effect), in the following manner:

BCF = (µg chemical/g tissue ÷µg chemical/g water)

or,

µg chemical/g tissue = µg chemical/g water x BCF

Two BCF values were identified; 1,700 for early fry rainbow trout (Van Leeuwen *et al*. 1985) and 8,875 whole body BCF for juvenile rainbow trout (calculated from Shubat and Curtis 1986). These BCF values are assumed to represent the low and high range for salmonid BCFs. Using these BCFs and data presented in the following studies, equivalent aqueous (i.e., water-borne only) we estimated dieldrin concentrations to be between 26 and 1,926 times the proposed chronic criterion of 0.0019 µg/L for dieldrin:

Hendricks *et al*. (1979) reported repressed growth in juvenile rainbow trout exposed to 5 mg/kg dieldrin in their diet for 12 months at 12°C, with a corresponding tissue concentration of approximately 1.6 mg dieldrin/kg whole fish. The corresponding concentration for dieldrin in a water-borne-only exposure experiment was estimated here to be between 0.18 µg/L and 0.94 µg/L.

Mehrle *et al*. (1971) reported alteration of the serum concentration of 11 amino acids in rainbow trout exposed to 1 mg dieldrin/kg body weight per week in their diet for 140 days at 16°C, with a corresponding tissue concentration of 1.8 mg dieldrin/kg whole fish. The corresponding concentration for dieldrin in a water-borne-only exposure experiment was estimated here to be between 0.2 µg/L and 1.1 µg/L. The results suggested that the utilization of five of the amino acids was inhibited by dieldrin, possibly due to an effect on enzymes which are responsible for the utilization and energy transformation of these specific amino acids.

Kilbey *et al*. (1972) conducted a 300 day dietary exposure study using rainbow trout held at 17°C. Effects that were observed included increased blood phenylalanine levels, decreased liver phenylalanine hydroxylase activity, and increased concentration of urine phenylpyruvic acid when dieldrin was present in the diet at 14 µg to 430 µg dieldrin/kg body weight/day (0.36 µg to 10.8 µg dieldrin/g of food). The corresponding dieldrin tissue concentration was 0.41 mg/kg to 6.23 mg/kg wet weight. Based on these tissue concentrations, a corresponding concentration for dieldrin in a water-borne only exposure experiment was estimated to be between 0.05 µg/L and 3.66 µg/L. The three effects observed parallel those seen in phenylketonuria, an inherited defect in human phenylalanine metabolism that is also characterized by mental deficiency. Although the study did not address analogous effects, it is possible that fish adaptability, behavior, and survival may be compromised based on biochemical similarities.

There are numerous additional studies on tissue exposure of salmonids to dieldrin. However, they have low utility for the purpose of evaluating the proposed chronic criterion, either because necessary data and findings were not reported, whole body tissue concentration could not be estimated, or test specimens were exposed to a mixture of compounds (e.g., Macek *et al*. 1970; Mehrle and Bloomfield 1974; Poels *et al*. 1980; Shubat and Curtis 1986).

In baseline data from the study area, in fish tissue collected from 33 locations in Idaho, including the lower Snake River below and Salmon River (in the action area), aldrin was <0.005 mg/kg wet weight. The highest concentration found for dieldrin was 0.037 mg/kg ww in carp from Brownlee Reservoir, Idaho (Clark and Maret 1998).

In summary, the reported chronic effects of dieldrin on juvenile salmonids were well above the proposed chronic criterion, suggesting that the chronic criterion for dieldrin is unlikely to injure or kill listed salmonids based on best available information.

***Toxicity in Mixtures.*** Limited information is available on the toxicological interaction of dieldrin with other contaminants. For any interaction between compounds, the route of exposure, the concentration ratio between the compounds, and the presence of additional compounds in a complex environmental mixture can each cause unique variations in toxicological responses. Water-born and dietary exposure studies conducted on rainbow trout and amphipods indicate the occurrence of synergistic, antagonistic, additive, or independent interactions, depending on the compounds included in the mixture or the biological endpoints tested. These are briefly outlined below.

Statham and Lech (1975) noted that dieldrin may interact synergistically with carbaryl. In a water-borne exposure study with fingerling rainbow trout, a 4-hour exposure to dieldrin at 1,000 µg/L caused 16% mortality, but when 1 mg/L carbaryl was added to the mixture, the resulting mortality level was 94%, which was greater than the sum of effects for either compound alone. No mechanism for this interaction was determined or suggested. Based on this information, natural freshwater areas that are known to contain both carbaryl (or other carbamate insecticides) and dieldrin may require special consideration with respect to synergistic toxicity to fish.

Interaction between dieldrin and DDT has been shown to vary depending on the toxicity endpoint considered. Macek *et al*. (1970) conducted an experiment with rainbow trout fed dieldrin and DDT for 140 days. This was sufficient time for equilibrium to be reached with respect to tissue residue accumulation of the two compounds. A significant increase in lipogenesis was seen with either contaminant alone, but, after several months, an additive effect also was apparent in fish that were fed both contaminants. In the pyloric caecae, the accumulation rate of DDT was increased by the presence of dieldrin, while that of dieldrin decreased. Further, elimination of DDT decreased markedly, while elimination of dieldrin remained unchanged. The results from this study suggest the possibility of increased bioaccumulation of DDT when dieldrin and DDT are present together in the environment. In contrast, Mayer *et al*. (1972) noted an antagonistic effect in rainbow trout that were fed dieldrin at non-lethal levels and DDT at lethal levels for 6 days. The fish were found to sustain mortality levels that were about half of that seen with DDT alone. The mechanism of this interaction was not determined in this study. From an environmental perspective, this observation may be important only when high (lethal) levels of DDT are bioavailable.

An antagonistic interaction also was suggested by Hendricks *et al*. (1979) between dieldrin and Aflatoxin $B_1$. In juvenile rainbow trout fed with both compounds for 12 months, the observed growth inhibition of the mixture was similar to that caused by dieldrin alone, thus indicating a reduction in the growth inhibitory effect of Aflatoxin $B_1$.

*2.4.15.2. Habitat Effects of Aldrin/Dieldrin Criteria*

***Acute Toxicity to Food Organisms: Aldrin.*** There is a sizable body of scientific literature that provides details on the adverse effects of aldrin on aquatic macroinvertebrates that may serve as salmonid prey species. Only one study was found where $LC_{50}$ values were below or near the proposed acute criterion of 3 µg/L:

Sanders and Cope (1968) reported a 96-hour $LC_{50}$ value of 1.3 µg/L for the stonefly naiad *Pteronarcys californica* exposed at 15.5°C, and pH 7.1 in a static experiment.

The next highest 96-hour $LC_{50}$s reported were 8 µg/L for the isopod *Asellus brevicaudus* (Sanders 1972), and 9 µg/L for the mayfly *Ephemerella grandis* (EPA 1980a), with both cases involving static experiments using nominal aldrin concentrations. There are numerous additional examples in the toxicological literature that indicate acute toxicity of aldrin to salmonid prey species at much higher levels, with concentrations ranging between 13 to nearly 70,000 times the proposed acute criterion for aldrin (e.g., Jensen and Gaufin 1964; Sanders 1969; Georgacakis *et al*. 1971; Sanders 1972; Khan *et al*. 1973; Kaushik and Kumar 1993). Most salmonid prey species are relatively resistant to the lethal effects of aldrin at the proposed acute criterion.

***Acute Toxicity to Food Organisms: Dieldrin.*** Acute effects of dieldrin on aquatic invertebrates have been noted to occur below or near the proposed acute criterion of 2.5 µg/L in three studies:

Sanders and Cope (1968) reported 96-hour $LC_{50}$ values of 0.5 µg/L for the stonefly naiads *Pteronarcys californica* and *Pteronarcella badia*, and 0.58 µg/L for the stonefly naiad *Claassenia sabulosa*, in static experiments performed at around 15.5°C and pH 7.1.

Karnak and Collins (1974) reported a 24-hour $LC_{50}$ of 0.7 µg/L for the midge larvae *Chironomus tentans*, using 85% dieldrin at 22°C.

Bowman *et al*. (1981) reported an 18-hour $LD_{50}$ value of 3.7 µg/L for the glass shrimp *Palaemonetes kadiakensis* at 23°C in a static experiment with a nominal dieldrin concentration.

Other studies have reported acute effects of dieldrin at concentrations that are considerably higher than the proposed acute criterion (from more than three to 720 times the criterion concentration; e.g., Jensen and Gaufin 1964; EPA 1980a; Sanders and Cope 1966; Sanders 1969; Georgacakis *et al*. 1971; Sanders 1972; Santharam *et al*. 1976; Bowman *et al*. 1981; Daniels and Allan 1981). There is apparently a wide range in the level of sensitivity of salmonid prey organisms to dieldrin, but nonetheless there are several studies which demonstrate toxicity responses at concentrations below or near the acute criterion. Thus, this criterion could result in lethal effects to salmonid food organisms.

***Chronic Toxicity to Food Organisms: Aldrin.*** As stated earlier, neither the EPA nor Idaho have proposed a chronic freshwater criterion for aldrin, based on a lack of toxicity information (EPA 1980a). However, available literature indicates that chronic effects of aldrin may occur on at least two salmonid prey species. For the stoneflies *Pteronarcys californica* and *Aeroneuria pacifica*, Jensen and Gaufin (1966) reported 30-day $LC_{50}$ values of 2.5 µg/L and 22 µg/L, respectively.

***Chronic Toxicity to Food Organisms Dieldrin.*** NMFS did not find any reports in the toxicological literature that indicate adverse effects from dieldrin occur to salmonid prey species at levels below the proposed chronic criterion of 0.0019 µg/L. Results for three aquatic insects and three crustaceans demonstrate that adverse effects manifest at the individual or population level only when dieldrin concentrations are much higher, ranging between 105 to at least 5,000 times the criterion (Jensen and Gaufin 1966; Adema 1978; Daniels and Allan 1981; Phipps *et al*. 1995) . This suggests that the proposed chronic criterion for dieldrin is generally protective of salmonid prey.

***Bioaccumulation of Aldrin.*** The tendency of aldrin to bioconcentrate in aquatic organisms generally has not been documented in the scientific literature, probably because metabolic reactions rapidly convert aldrin to dieldrin. However, one study was found in which *Daphnia magna* were exposed for 1 to 2 days at 1.7 µg/L aldrin, with associated BCFs of approximately 1,800 to 9,100 (Metcalf *et al*. 1973).

***Bioaccumulation of Dieldrin.*** Salmonids and other freshwater fish species have been shown to strongly bioaccumulate dieldrin from the water column in laboratory exposure studies. Van Leeuwen *et al*. (1985) exposed early fry rainbow trout to dieldrin for 24 hours and reported a steady state BCF of 1,700. Chadwick and Shumway (1969) reported a whole body BCF equal to approximately 3,200 for newly hatched steelhead trout alevins after 35 days of exposure.

Whole body or lipid BCF calculated from information provided in other studies on exposure concentration, duration, and tissue residue concentration are also indicative of the tendency of dieldrin to bioaccumulate. Shubat and Curtis (1986) exposed juvenile rainbow trout to 0.04 µg/L dieldrin for 16 weeks in a flow-through experiment with a measured dieldrin concentration, and indicated a whole body tissue residue level of 120 to 320 ng dieldrin/g fish tissue, or 7.1 ng to 11 ng dieldrin/mg lipid. This translates into a whole body BCF of approximately 3,000 to 8,000, or a lipid BCF of 178,000 to 275,000. For fish exposed to 0.08 µg/L, the calculated whole body BCF becomes 2,500 to 8,900, and the lipid BCF 225,000, indicating slightly higher bioaccumulation rates at higher water concentrations.

The only other freshwater fish for which we found laboratory-derived bioaccumulation information is the channel catfish *Ictalurus punctatus*. Shannon (1977a) conducted a 28-day exposure to 0.075 µg/L of an 87% dieldrin formulation in a flow-through experiment with measured concentrations of dieldrin. Based on reported tissue concentrations, the calculated dorsal muscle BCF was 2,333 for smaller fish and 3,653 for larger fish. Although Shannon (1977a) suggests that the higher bioaccumulation observed for the larger fish in this study could be due to a higher fat content, this notion was not supported by results from a field study where larger fish did not consistently harbor higher residue concentrations (Kellogg and Bulkley 1976). In another experiment, a 70-day exposure to 0.013 µg/L dieldrin resulted in a calculated dorsal muscle BCF of 2,385, with equilibrium being reached more rapidly at lower level exposures than at higher levels (Shannon 1977b). These laboratory BCF values for catfish are comparable to BCF determined for salmonids. However, they are approximately 10 fold below the BCF values reported in channel catfish from field studies. Leung *et al*. (1981) sampled fish and water from the Des Moines River in Iowa in June and August 1973, during a time when aldrin was being used on area cropland. The corresponding calculated muscle tissue BCF values range from 2,220 to 22,200. The authors did not discuss the possibility that the tissue residue levels could reflect dieldrin accumulation from food and sediment as well as water. However, Chadwick and Brocksen (1969, cited in Shannon 1977a) noted that, when selected fish were tested for accumulation of dieldrin from food or water, most of the dieldrin in the tissue came from water. The reported information from additional field studies conducted in the Des Moines River can be used to calculate the BCF values for various other freshwater fish, yielding estimated BCFs of up to 1,600 for carpsucker, 10,200 for sand shiner, 15,500 for spotfin shiner, or 7,500 for bluntnose minnow (Kellogg and Bulkley 1976).

No laboratory derived BCF values were available for any aquatic insect species that are prey for salmonids. Reinert (1972) noted a BCF of approximately 14,000 for *Daphnia magna* exposed to dieldrin for 3 days. Kellog and Bulkley (1976) conducted a field study from which reported tissue and water concentrations of dieldrin can be used to calculate BCF values for various insect, crustacean, or fish prey species used by salmonids. Water samples contained 0.004 µg/L to 0.012 µg/L dieldrin, and aquatic organisms had tissue levels ranging from 2 ppb to 61 ppb from the Des Moines River in Iowa in 1973. Corresponding calculations result in BCF values that are on the order of 1,500 for the stonefly *Pteronarcys*, 5,100 for the mayfly *Potamanthus*, 3,500 for chironomidae, 3,600 for trichopterans, and 1,300 for the crayfish *Oronectes rusticus*.

For photodieldrin, BCF values derived from laboratory studies on various freshwater fish are approximately an order of magnitude lower than laboratory dieldrin BCF values determined for salmonids and catfish. For example, after a 1 day exposure to 20 µg/L photodieldrin in a static experiment with measured dieldrin concentrations, BCF values were 133 for bluegill (*Lepomis machrochirus*), 150 for minnow (*Lebistes reticulata*), 609 for goldfish (*Carassius auratus*), and 820 for guppy (*Gambia affinis*) (Khan and Khan 1974). The data of Khan and Khan (1974) also indicated a BCF around 1,200 for a Gammarid exposed for 4 days at 10 µg/L.

Overall, the weight of evidence indicates that both salmonids and their prey bioconcentrate dieldrin from their environment.

***Biomagnification.*** Although no studies could be found that deal directly with salmonids and their prey species, there are a number of published reports involving various food web chains indicating dieldrin does not tend to biomagnify through progressively higher trophic levels. Reinert (1972) conducted a freshwater laboratory study in which they found that direct uptake of dieldrin from water is more likely to occur than uptake through the diet. In the algae-daphnid-guppy (*Poecilia reticulata*) food chain tested, *D. magna* and guppies accumulated more dieldrin directly from water than from their respective food sources exposed to similar water concentrations. Van Sprang *et al*. (1991) determined in another laboratory study using estuarine organisms that biomagnification was not apparent when the estuarine mysid shrimp *Mysidopsis bahia* was fed dieldrin-contaminated *Artemia*. Furthermore, in a field study in the North Sea's Weser Estuary, analysis of dieldrin tissue levels in a cockle-soft clam-brown shrimp-sole food web did not indicate the occurrence of biomagnification of dieldrin in the respective organisms (Goerke *et al*. 1979). This is reflected in literature reviews that have concluded there is little to no evidence to suggest that dieldrin biomagnifies in aquatic food webs (Kay 1984; Suedel *et al*. 1994).

#### *2.4.15.3. Summary for Aldrin/Dieldrin*

***Aldrin.*** The limited information available regarding aldrin toxicity to salmonids indicates that 50% mortality can occur when concentrations are below or slightly above the acute criterion.

Similarly, there is evidence that aldrin is toxic to some salmonid prey species when concentrations are below or close to the criterion. This information suggests that the proposed acute criterion for aldrin if found at these levels is reasonably certain to harm listed salmonids or impact their food sources. The limited information available regarding aldrin toxicity indicates that aldrin is toxic to some salmonid prey species when concentrations are below or close to the criterion and is LAA food sources. However, it is unlikely that discharges of aldrin will occur in the action area as no uses are currently approved and levels currently found in the water column are well below the proposed standards.

***Additional comments on Aldrin.*** Although no chronic criterion for aldrin is proposed, available studies demonstrate that chronic effects do occur to freshwater fish at 0.0466 µg/L, and to prey items at 2.5 µg/L. These results suggest that the absence of a chronic criterion could result in adverse chronic effects to listed salmonids and their food source. However, the human-health based aldrin criteria is also applicable to all waters in the action area that are either designated critical habitat for, or are inhabited by listed salmonids. For aldrin this criterion is 0.00014 µg/L (Table 1.3.1). This value is lower than concentrations causing adverse effects to any aquatic prey species, listed species, or surrogate for a listed species reviewed here. Thus although the aldrin acute aquatic life criteria may not be fully protective to listed species and habitats for long-term exposures the application of the fish consumption based water quality standard to protect recreation is protective and is applicable in all waters of Idaho that contain listed species.

***Dieldrin.*** The scientific literature on effects of dieldrin on salmonids reports acute lethal effects at concentrations below or slightly above the proposed acute criterion. These studies included various salmonid species, such as Chinook and coho salmon, steelhead, and rainbow, cutthroat, or brown trout, as well as toxicological information on juveniles and adults. This available information indicates that the proposed acute criterion for dieldrin will likely adversely affect listed salmonid species. The proposed acute criterion is greater than $LC_{50}$s reported for several important salmonid prey species. However, because acute effects could only come from recent applications, and because the use of dieldrin has been banned since EPA cancelled its registration in 1975, acute effects occurring from release of dieldrin are unlikely. Chronic studies involving juvenile rainbow trout demonstrate that limited adverse effects only occur when ambient concentrations are >95 times the proposed chronic criterion. This information is supplemented by published BCF values and analyses of the results of dietary exposure studies in which estimated aqueous concentrations of dieldrin resulting in reported tissue concentrations was also well above the chronic criterion. These limited studies indicate that the proposed chronic criterion will not result in measurable effects to listed salmonids. Further, no information suggests that prey species may be adversely affected by concentrations below the proposed chronic criterion. Dieldrin was detected in sediment in Brownlee Reservoir of the Snake River (Table 2.3.1). However, levels of dieldrin currently found in Brownlee Reservoir are well below the standard and the reservoir is not occupied by listed species. With no ongoing discharges, the level of dieldrin in sediment in Brownlee Reservoir is likely to decline over time.

2.4.16. The Effects of EPA Approval of the Chlordane Criteria

Chlordane is an organochlorine pesticide that was used in the United States from 1948 to 1988. The commercial formulation is not a single chemical, but a mixture of at least 23 different compounds, the major components being trans-chlordane, cis-chlordane, chlordene, heptachlor, and trans-nonachlor (Kidd and James 1991). For many years it was used as a pesticide on agricultural crops, lawns and gardens, and as a fumigating agent. Because of concerns over cancer risk, evidence of human exposure and accumulation in body fat, persistence in the environment, and danger to wildlife, EPA banned the use of chlordane on food crops in 1978, and phased out other uses over the next 5 years (EPA 1980c). From 1983 to 1988, its only approved use was to control termites in homes. When its application for termite control was banned in 1988, all approved use of chlordane in the United States stopped. However, it continued to be manufactured within the United States for export abroad (ATSDR 1994).

Chlordane is not highly water soluble, and has an octanol/water partition coefficient of $10^{5.54}$. (ATSDR 1992; EPA 1992b). It does not degrade rapidly in water and adsorbs strongly to particles and sediment in the water column, but it can leave aquatic systems by volatilization (Wauchope *et al*. 1992).

Chlordane is absorbed by animals into the body and stored in fatty tissues as well as in the kidneys, muscles, liver, and brain (ATSDR 1989; Smith 1991). It can be released into circulation when these fatty tissues are metabolized, as in the cases of starvation and intense activity (Smith 1991). Excretion of orally administered chlordane is slow and can take days to weeks (ATSDR 1989). Adverse effects of chlordane to mammals occur mainly through the nervous system, the digestive system, and the liver, and can result in convulsions and death (Smith 1991; EPA 1992b; USDHHS 1993a, b; ATSDR 1994). Increased activity of thyroid hormone may also occur (Martin 1971). With chronic exposure, the most frequently observed effects occur to the central nervous system, the liver, and the blood through disorders including aplastic anemia and acute anemia (ATSDR 1989). There is also evidence that exposure to chlordane may be associated with reproductive and developmental effects. Studies indicate that chlordane is weakly or nonmutagenic (Smith 1991), but evidence for carcinogenicity is generally inconclusive (NIOSH 1986; ATSDR 1992, EPA 1993b).

*2.4.16.1. Species Effects of Chlordane Criteria*

The proposed acute criterion is 2.4 µg/L, while the chronic criterion is 0.0043 µg/L. The acute value was derived from $LC_{50}$ values for nine freshwater fish and five invertebrates and represents the fifth percentile of the mean species values for this group of animals, whereas the chronic criterion is based on the 1980 FDA guidelines for marketability of fish for human consumption (EPA 1980c).

***Acute Chlordane Criterion.*** Reported $LC_{50}$ values for salmonids are well above the proposed acute criterion of 2.4 µg/L. Lethal effects (96-hour $LC_{50}$) have been observed for acute water-borne exposures to technical-grade chlordane at 56 µg/L to 57 µg/L in coho and Chinook salmon, 45 µg/L to 47 µg/L in brook trout (Cardwell *et al*. 1977), 8 µg/L to 47 µg/L in rainbow trout (Katz 1961; Mehrle 1974), and higher values have been reported in other studies (e.g., 42 µg/L to 90 µg/L; Kidd and James 1991, HSDB 1995). However, most of these data were from static tests with nominal chlordane concentrations, so it is possible that the compound's actual toxicity was underestimated. In most other species that have been tested, $LC_{50}$s have been in the 25 µg/L to 100 µg/L range (Cardwell *et al*. 1977, EPA 1980c), although $LC_{50}$s as low as 3 µg/L have been reported in carp and bass and 7.1 µg/L in bluegill (EPA 1980c). In a more recent study, Moore *et al*. (1998) tested the toxicity of chlordane to fathead minnow in 48-hour tests and determined a mortality rate gradient of approximately 1.68% of mortality per µg/L, so that at the acute criterion of 2.4 µg/L, the predicted mortality would be about 4%. The $LC_{50}$ reported by Moore *et al*. (1998) was 21.4 µg/L, within the range reported for salmonids. This study therefore suggests that a low level of mortality ~5% or less of the exposed fish could occur in salmonids when chlordane concentrations are at the acute criterion.

***Chronic Chlordane Criterion.*** Since the proposed chronic criterion of 0.0043 µg/L proposed for use in the IWQS is not based on chronic toxicological effects on freshwater fish or invertebrates (EPA 1980c), its utility for protecting listed salmonids is not clear. In general, however, chlordane water concentrations associated with biological effects in laboratory exposures appear to be well above the chronic criterion. Cardwell *et al*. (1977) examined the chronic toxicity of water-borne technical chlordane to brook trout (*Salvelinus fontinalis*) under flow-through conditions, and found that the lowest concentration to cause major chronic effects was 0.32 µg/L. Similarly, in a review of chlordane effects on several species, Eisler (1990) stated that 0.2 µg/L to 3 µg/L chlordane can be harmful with long-term exposure to sensitive fish. Other studies examining larval toxicity or sublethal effects of chlordane, such as changes in blood parameters, somatic indices, and ATPase levels, in non-salmonid fish during exposures of 30 to 60 days reported effects at similar or greater concentrations only (Gupta *et al*. 1995; Verma *et al*. 1978; Bansal *et al*. 1980). The lowest of these values were two orders of magnitude above the proposed chronic criterion.

The proposed criterion can also be evaluated by comparing tissue concentrations of chlordane associated with adverse effects with estimated tissue levels of chlordane at the chronic criterion. According to the criteria development documents (EPA 1980c), this concentration should be approximately 0.3 mg/kg in edible tissue, the FDA action level in place at that time. This tissue concentration estimate assumes a normalized BCF value for chlordane of 4,702 and a tissue lipid concentration of 15%, the default value for freshwater fish (EPA 1980c). For natural-origin salmonids, this value may be rather high, as whole body lipid levels of 5% to10% are more typical of adult salmonids (Meador 2002).

Relatively little information is available on tissue concentrations of chlordane associated with biological effects in salmonids. However, data suggest adverse effects in other species at tissue concentrations below those associated with the chronic criterion. For example, Schimmel *et al*. (1976) report increased mortality ($25^{th}$ percentile effect dose ($ED_{25}$)) for spot at concentrations ranging from 0.16 mg/kg to 0.55 mg/kg, and slightly increased mortality ($ED_5$) in sheepshead minnow at tissue concentrations as low as 0.010 to 0.02 mg/kg. However, other studies report effects at tissue concentrations in the 1 mg/kg to 10 mg/kg range and greater (Parrish *et al*. 1976; Delorme 1998).

It should be noted that BCF values used in calculating the chronic criterion were based primarily on data from fathead and sheepshead minnow, not on studies with salmonids, and thus may not reflect uptake in the species of concern. Also, because these BCFs were determined in the laboratory, they may underestimate chlordane uptake by animals in the field. In the natural environment, major routes of chlordane uptake are likely to be though sediments and diet, and the water quality criteria and laboratory derived BCF do not account for this additional exposure. More realistic assessments of exposure might be obtained from field derived BAFs. Oliver and Niimi (1998) determined a chlordane BAF of about 106 (562,000), or 51,000 normalized for lipid content for salmonid species from Lake Ontario. Using this value, predicted chlordane tissue concentrations would be 1.1 mg/kg to 3.3 mg/kg for the aquatic life criteria and 0.07 mg/kg to 0.44 mg/kg for human health based criteria from Table 1.3.1. For tissue lipid levels in the 5% to 15% range more typical of adult salmonids (Meador 2002), the value would be 0.05 mg/kg to 0.1 mg/kg. This approach is problematic; however, as these values may greatly overestimate exposure for species like anadromous salmon with short residence times in freshwater environments. Whatever the case, the variability in reported BCF and BAF values indicates that there is considerable uncertainty concerning the tissue concentrations that might be expected in fish at the proposed criteria.

These data suggest that exposure to water borne chlordane at concentrations below the chronic criterion should not cause mortality to threatened and endangered salmon. However, even when water concentrations of chlordane are very low, sediment and prey concentrations may be elevated, causing salmonids to accumulate these compounds to levels that are considered adverse (see below). Other harmful effects were not evaluated in the literature.

***Behavioral Effects.*** Behavioral effects may occur at concentrations near or below the acute criterion. Little *et al*. (1990) examined spontaneous swimming activity, swimming capacity, feeding behavior, and vulnerability to predation in rainbow trout (*Oncorhynchus mykiss*) in 96-hour exposures to sublethal concentrations of chlordane. Behavioral changes were consistently demonstrated at concentrations of 2 µg/L.

***Factors Affecting the Toxicity of Chlordane.*** The toxicity of chlordane can vary with temperature, sediment loading, age, condition, and nutritional history of the exposed organism and the formulation and isomer of the chemical. Toxicity is typically greater at higher temperatures (Rai and Mandal 1993). Specific mixtures of chlordanes in the environment may vary from the original mixture of technical-grade chlordane and thus also vary in toxicity. For example, Gooch *et al*. (1990) compared the toxicity of technical-grade chlordane with a mixture isolated from tissues of lake trout. When the toxicity of the residue was evaluated using an acute bioassay and a neuroreceptor binding affinity assay, it was found to be three to five times more toxic than the technical mixture. Chlordane may also interact with other contaminants present in the environment. Gupta *et al*. (1994) for example, found that the fish *Notopterus notopterus* exhibited additive and synergistic toxicity effects when chlordane was combined with the pesticide furadan.

*2.4.16.2. Habitat Effects of Chlordane Criteria*

***Toxicity to Food Organisms.*** Data for acute and chronic toxicity of chlordane to salmonid prey species are not extensive. Cardwell *et al*. (1977) examined acute and chronic toxicity of water-borne technical chlordane to the cladoceran, *Daphnia magna*, the amphipod *Hyallela azteca*, and the midge *Chironimus sp*. under flow-through conditions. The concentrations of technical chlordane causing 50% immobilization in the cladoceran was 38.4 µg/L. By 96 hours, the amphipod was only slightly affected by the chlordane concentrations tested, and the 168-hour $EC_{50}$ was 97.1 µg/L. While Hall *et al*. (1986) report a much higher $LC_{50}$ of 270 µg/L, the reported $LC_{50}$ values in other studies for these species are more similar to Cardwell *et al*. (1977). For example, EPA (1980c) reported a 48hour $LC_{50}$ of 35 µg/L for *D. magna*. Chlordane 96 hour $LC_{50}$ values of 26 mg/kg and 40 mg/kg have been reported for the amphipods *Gammarus lacustris* and *G. fasciatus*, respectively (Sanders and Cope 1966; Sanders 1969). More recently, Moore *et al*. (1998) performed aqueous 48-hour toxicity tests of chlordane on several invertebrate species, including *D. magna*, *H. azteca*, and *Chironomus tentans Fabricius*. Mortality rate gradients varied from 0.88% mortality per µg/L for *H. azteca* to 2.54% mortality per µg/L for *C. tentans*, with $LC_{50}$s of 20 µg/L to 57 µg/L. Thus, at the acute criterion of 2.4 µg/L, predicted mortality for these prey species would be between 1% and 6%.

In terms of chronic toxicity, Cardwell *et al*. (1977) reported that the lowest concentrations of technical chlordane found to cause chronic effects on long-term survival, growth, and reproduction were 1.7 µg/L for midges, 11.5 µg/L for amphipods, and 21.6 µg/L for cladocerans, values all well above the chronic criterion.

***Bioaccumulation.*** Chlordane bioconcentrates in both marine and freshwater fish and invertebrates, and studies conducted in the late 1970s showed that the fatty tissues of both land and water wildlife contained large amounts of chlordane and other cyclodiene insecticides (Gobas *et al*. 1988; Isnard and Lambert 1988; ATSDR 1989; HSDB 1995). Bioaccumulation factors in marine waters have been reported to range between 3,000 and 12,000 (Zaroogian *et al*. 1985), and may be as high as 18,500 in freshwater for rainbow trout (Oliver and Niimi 1985). There is some evidence of biotransformation of chlordane in rainbow and cutthroat trout (Albright *et al*. 1980; Pyysalo *et al*. 1981). Albright *et al*. (1980) measured residues in cutthroat trout (*Salmo clarki*) from a lake treated with technical chlordane, and found that trans-nonachlor was the most persistent constituent, accounting for about 50% of the total chlordane remaining in fish collected 3 years after the lake was treated. Other measured constituents (heptachlor, heptachlor epoxide, and chlordene) were non-detectable in less than a year after treatment. The study also indicated that animals appeared to produce oxychlordane from chlordane.

Other studies suggest that the bioconcentration and bioaccumulation of chlordane in nature are complex and may not always follow predictions of octanol-water partitioning. For example, Swackhamer and Hites (1988) examined uptake of chlordane and several other pesticides in different size classes of lake trout and compared the bioconcentration factors with the octanol-water partition coefficient ($K_{ow}$). However, the correlation was weak ($r^2 = 0.73$) when compared to published relationships based on laboratory data. Factors that can influence bioaccumulation of chlordane in fish include lipid content and trophic positioning (Kidd *et al*. 1998).

In baseline data from the study area, in fish tissue collected from 33 locations in Idaho, including the lower Snake River below and Salmon River (in the action area), chlordane was <0.005 mg/kg wwin fish, except for demersal fish from Brownlee Reservoir. There, using data from carp, largescale suckers, and channel catfish chlordane was detected with a maximum concentration of 0.020 mg/kg ww (Clark and Maret 1998). Chlordane was not detected in any salmonid.

***Uptake and Toxicity Through Alternate Routes of Exposure.*** Because chlordane is no longer in use in the United States, the major source of this compound will not be through point source discharges into surface water bodies, but from repositories of the contaminant that are persistent in sediments. This means that chlordane will not be taken up only through the water column, but also through direct contact with sediments or through the diet. Thus, studies evaluating the effects of water-borne exposure alone are likely to underestimate actual exposure of organisms in the field.

Because sediments are likely the primary source of chlordane, the sediment chlordane concentration that would result in chlordane concentrations in the water column at or below the proposed criteria can be calculated per Section 2.4.13. For chlordane, $\log_{10}K_{ow} = 5.54$, $\log_{10}K_{oc} = 5.45$, and the aquatic life criterion $F_{CV} = 0.0043$, resulting in an estimated $SQC_{oc}$ = 1.21 µg/g organic carbon. This would mean that for sediment total organic carbon (TOC) levels ranging between 1% to 5%, the chronic aquatic life criterion would be associated with sediment chlordane concentrations ranging between 12 ng/g to 61 ng/g sediment. This exceeds the sediment screening guideline of 10 ng/g dry wet established by the Army Corps of Engineers (COE) for in-water disposal of dredged sediment (COE 1998), and are above the interim Canadian freshwater sediment guidelines of 2.26 ng/g to 4.79 ng/g dry wet sediment. The higher of these values is a probable effect level, based on spiked sediment toxicity testing and associations between field data and biological effects (CCME 2001). These data suggest that chlordane could adversely affect the salmonid prey base at concentrations below the proposed criteria, as the COE and the Canadian sediment quality criteria are based primarily on tests with benthic invertebrates. The most stringent applicable criterion in the action area, the fish consumption based AWQC of 0.00057 µg/L that are also applicable to waters occupied by listed species and designated critical habitats, are about eight times lower than the chronic criterion of 0.0043 µg/L for chlordane (Table 1.3.1). When extrapolated to predict sediment concentrations in the same fashion as the chronic criterion, the resulting sediment concentration would be about 1.6 ng/g to 8 ng/g dw sediment, which is less than the COE screening criteria and overlap the Canadian guidelines.

Because there has been very little research on the toxicity of sediment-associated chlordane to salmonids, the sediment concentrations that can cause adverse effects are not well defined. There are a few estimates of biota-sediment accumulation factors (BSAFs) for salmonids. For example, for trans-chlordane, Oliver and Niimi (1988) determined a BSAF of 2.22 for salmonid species from Lake Ontario, with 11% lipid and sediment TOC of 2.7%.

*2.4.16.3. Summary for Chlordane*

Lethal effects from short-term exposures of salmonids or salmonid invertebrate prey species to chlordane only occurred at concentrations above the acute criterion. There are no current approved uses of chlordane in the United States and no manufacturing of chlordane takes place in Idaho. The levels of chlordane in Idaho detected in Brownlee Reservoir (Table 2.3.1) are well below the proposed criteria and no listed salmon or steelhead are located in or above the reservoir.

Data generally indicate that the proposed chronic criterion for chlordane is likely to avoid harm to listed salmonids. However, many sublethal effects of chronic exposure to chlordane that have been documented in mammals (i.e., neurological damage, altered immune and reproductive function, and increased cancer risk) have not been studied in salmonid species subjected to long-term chlordane exposure at concentrations near or below the criterion. Similarly, few data are available on the sublethal effects of long-term exposure to chlordane on salmonid prey. There are also a few studies suggesting that a risk of increased long-term mortality or sublethal effects at chlordane tissue concentrations close to those that might be expected in fish exposed to chlordane at levels allowed under the chronic aquatic life criteria. Additionally, bioaccumulation can occur in salmonids with chronic exposure to chlordane at levels allowable under the proposed criteria, and exposure is likely to occur not only through the water column but also through diet and contact with sediments. The proposed criteria do not account presently for these other sources of exposure. There is some evidence of risk to benthic invertebrates or through food web uptake associated with bioaccumulation and exposure from sources other than the water column. Based on the strength of evidence considered, the chronic criterion does not appear likely to harm salmonids through water column exposure. If exposure occurs the different exposure pathways may pose some risk for salmon and steelhead, but appear unlikely to result in injury or death. Additionally, there will be no new sources of chlordane and so exposure is unlikely to occur.

2.4.17. The Effects of EPA Approval of the Dichlorodiphenyltrichloroethane Criteria

Dichlorodiphenyltrichloroethane (DDT) is a waxy, odorless or slightly aromatic solid that has been used extensively as an insecticide throughout the world. DDT occurs in three isomeric forms o,p’, o,o’, and p,p’. The technical product consists primarily of p,p’-DDT (60% to 85%) and o.p’-DDT (15% to 21%), with small amounts of other impurities (NTP 2001). DDT is metabolized to dichlorodiphenylethylene (DDE) and dichlorodiphenyldichloroethane (DDD).

The insecticidal properties of DDT were first discovered in the early 1940s, and the pesticide was used extensively on crops in the United States over the period 1945 to 1972. It was also used as a mosquito larvacide, as a spray for eradication of malaria in dwellings, and as a dust in human delousing programs for typhus control. The EPA banned the use of DDT in food in 1972 and banned non-food uses in 1988, except as an insecticide for public health emergencies. Currently, no United States companies report the production of DDT, but major producers and users of DDT exist outside the country (ATSDR 1994; SRI 1997).

Recent studies (e.g., ATSDR 1994; EPA 1992b) report that DDT (usually expressed as the sum of DDT and its metabolites) can be found at concentrations in the hundreds of ppb in sediment and at ppm levels in fish in many urban areas in the United States. DDT is highly persistent in the environment, with a reported half-life between 2 and 15 years. Volatilization, photolysis, and biodegradation are the main processes for breakdown, but they appear to act very slowly on this compound. This pesticide exhibits a log10 $K_{ow}$ of approximately 6.19, indicating that it has a strong tendency to bioaccumulate in the lipid of organisms. Even though it has been several decades since it was banned in the United States, DDT still persists in the environment and can be found in aquatic sediments.

Chronic exposure to DDT can affect the mammalian nervous system, liver, kidneys, and immune system (ATSDR 1994; WHO 1989). Immunological effects observed in test animals include reduced antibody formation and reduced levels of immune cells in rats and mice at doses ranging from 1 mg/kg/day to 13 mg/kg/day for 3 to 12 weeks. There is also evidence that DDT causes reproductive effects, including sterility and developmental problems, and it is thought that many of these observed effects may be the result of disruptions in the endocrine system. DDT may also be associated with teratogenic effects (ATSDR 1994). The evidence for mutagenicity and genotoxicity of DDT is contradictory (NTP 2001). There is some evidence that DNA exposure may be associated with chromosomal damage, but overall studies suggest that although DDT may have the potential to cause genotoxic effects, it is not strongly mutagenic (ATSDR 1994). Similarly, the evidence regarding the carcinogenicity of DDT is equivocal. It is classified by EPA and International Agency for Research on Cancer as "reasonably anticipated to be a human carcinogen," based on sufficient evidence of carcinogenicity in experimental animals (IARC 1974, 1982). These effects are assumed here to be similar in fish.

### *2.4.17.1. Species Effects of DDT Criteria*

The proposed acute criterion for dissolved concentrations of 4,4'-DDT (p,p'-DDT) is 1.1 µg/L. The proposed chronic criterion for 4,4'-DDT (p,p'-DDT) is 0.001 µg/L, and the also applicable fish consumption based criterion is 0.00059 µg/L *(*Table 1.3.1) No criteria for the DDT metabolites, DDE and DDD, are proposed in this action.

***Acute DDT Criterion.*** The proposed acute criterion is based on toxicity data from 18 freshwater invertebrate species and 24 fish species (EPA 1980f). For invertebrates, $LC_{50}$ values ranged from 0.18 µg/L to 1800 µg/L, while for fish, $LC_{50}$ values ranged from 0.6 µg/L for yellow perch to 180 µg/L for goldfish. The acute criterion of 1.1 µg/L is a value that would be protective of 95% of the species tested. Available data suggest that the acute criterion could expose listed salmonids to lethal DDT concentrations. Studies involving cutthroat trout reported $LC_{50}$s ranging from 0.85 µg/L to 1.32 µg/L, below or very close to the proposed acute criterion. For rainbow trout, reported $LC_{50}$ values range from 1.7 µg/L to 42 µg/L (Katz 1961; Macek and McAllister 1970; Macek and Sanders 1970; Post and Schroder 1971; Marking 1966). For brown trout, values range from 2 µg/L to 17.5 µg/L (Macek and McAllister 1970; Marking 1966). Other reported 96-hour $LC_{50}$ values range from 4 µg/L to 44 µg/L in coho salmon (Katz 1961; Macek and McAllister 1970; Post and Schroder 1971; Schaumberg *et al*. 1967), 8 µg/L to 20 µg/L for brook trout, and 9.1 µg/L to 9.5 µg/L for lake trout (Marking 1966, Post and

Schroeder 1971). An $LC_{50}$ of 11.5 µg/L was reported for Chinook salmon (Katz 1961). These values are all based on static tests with nominal DDT concentrations.

The only data NMFS found for a flow-through test involved rainbow trout fry, and yielded an $LC_{50}$ of 2.4 µg/L (Tooby *et al*. 1975). In cases where both flow through and static tests were conducted with the same species (shiner perch and dwarf perch), $LC_{50}$ values for the static tests were approximately 20 times higher than those from the flow through tests (4.6 µg/L to 7.6 µg/L vs. 0.26 µg/L to 0.45 µg/L; Earnest and Benville 1971). This suggests that the acute $LC_{50}$s for salmonids based on static tests could underestimate the toxicity of DDT, and testing values more relevant to a natural stream environment in critical habitat could be an order of magnitude lower, below the proposed acute criterion.

***Chronic DDT Criterion.*** Most available information on DDT effects is based on mammals, or fish species other than salmonids. Chronic exposure of mammals to DDT is known to cause physiological effects in the nervous system, liver, kidneys, endocrine system, and immune system (ATSDR 1994; WHO 1989). There is also evidence that DDT may have the potential to cause genotoxic effects, but it does not appear to be strongly mutagenic (ATSDR 1994).

There are few long-term studies on the effects of water-borne exposure to DDT in salmon, and it is difficult to know how to interpret a number of studies because they were conducted in static systems at nominal DDT concentrations above reported $LC_{50}$ levels. For example, Allison *et al*. (1963) conducted a long-term study in which Snake River cutthroat trout were exposed to DDT in the water for 28 days at concentrations ranging from 10 µg/L to 1000 µg/L. Above 30 µg/L, fish showed increased cumulative mortality and effects on fry survival. However, since acute $LC_{50}$ values for cutthroat trout are reported well below 30 µg/L, the results of Allison *et al*. (1963) do not provide a clear indication of the lower limits of concentrations where chronic effects might occur.

Early life stages of salmonids may be more susceptible to DDT effects than smolts or adults (Hudson *et al*. 1984; WHO 1989), but the reported concentrations where mortality occurred from water-borne exposure were well above the proposed chronic criteria. For example, Halter and Johnson (1974) reported that mean survival times of early life stages of coho salmon were considerably reduced by DDT concentrations above 0.5 µg/L. In another study, Atlantic salmon eggs were exposed to water containing 5 µg/L, 10 µg/L, 50 µg/L, or 100 µg/L of DDT (Dill and Saunders 1974). The hatched fry had balance problems and impaired behavioral development at 50 and 100 µg/L. In a more recent study, Glubokov (1990) reported increased mortality (0.7% to 10% above baseline) of coho salmon during early ontogeny when exposed to DDT over the range of 0.1 µg/L to 10 µg/L.

For studies with water-borne DDT conducted with other species, effect concentrations were also well above the 0.001 µg/L chronic criterion. For example, Pandey *et al*. (1996) exposed the estuarine mullet, *Liza parsia* to DDT at a concentration of 100 µg/L for 15 days, and observed dilation of blood sinusoids, as well as vacuolization, granular degeneration, necrosis and fibrosis in the liver. Weis and Weis (1974) observed increased individual activity and increased school size in goldfish exposed to DDT at 1 µg/L for 7 days. More recently, studies have been conducted to evaluate the estrogenicity of o,p'-DDT, o,p'-DDE, and p,p'-DDE by assessing their potential to cause the production of estrogen-inducible proteins such as vitellogenin (yolk). Metcalfe *et al*. (2000) exposed medaka embryos to o,p'-DDT for 100 days and found males with testis-ova at nominal concentrations as low as 5 µg/L, or average measured concentrations as low as 1.2 µg/L. Exposure of female medaka to nominal water concentrations of 2.5 µg/L for 2 weeks resulted in progeny with longer hatching times, earlier ovarian development in females, and enhanced vitellogenic response in males exposed to estrogens. Cheek *et al*. (2001) conducted a similar study with medaka in which fish were exposed to water-borne o,p'-DDT in a flow through system. After 2 weeks of exposure, percent hatch and fertilization were reduced at exposure concentrations as low as 0.23 µg/L, while after 8 weeks, vitellogenin induction and effects on fertilization and hatching success were observed at 0.30 µg/L. Because o,p'-DDT typically accounts for about 20% of total DDTs in commercial DDT mixtures, the total DDT concentrations associated with such effects reported by Metcalf *et al*. (2000) and Cheek *et al*. (2001) would probably be in the range of 6 µg/L to 25 µg/L, well above the proposed criteria.

As noted below, the chronic criterion was determined by EPA as an ambient water concentration that would result in fish tissue DDT levels at or below 0.15 mg/kg. There are some problems with this analysis, particularly regarding uncertainty in applying the standardized BCF of 17,870 to salmonids. The range of reported BCFs for salmonids in EPA's water quality documents for DDT include much higher values (EPA 1980f). Reported laboratory-derived BCFs for salmonids for whole body DDT concentrations range from 38,600 in rainbow trout (Reinert *et al*. 1974) to 47,400 in lake trout (Reinert and Stone 1974). Field derived BCFs are higher. Examples are 1,560,000 for coho salmon (Lake Michigan Interstate Pesticide Commission 1972), 1,170,000 for lake trout (Reinert 1970). For muscle tissue only, BCFs range from 11,600 in rainbow trout (Miles and Harris 1973) to 45,400 in brown trout (Miles and Harris 1973) to 458,000 in lake trout (Miles and Harris 1973). These data suggest that the BCF may be unrealistically low for field-collected salmonids. If only salmonid data are used, the geometric mean of the lipid-normalized BCFs in the EPA criteria for DDT (EPA 1980f) is 28,298. Using this BCF, the predicted DDT tissue concentrations in a salmonid at the proposed AWQC of 0.001 mg/L would range from 0.14 mg/kg to 0.42 mg/kg for lipid levels of 5% to15%. Similarly, using the most stringent applicable fish consumption based criterion provides lower values, 0.08 to 0.25 mg/kg. On the other hand, however, these salmonid BCFs could overestimate exposure of listed salmonids with short residence times in Idaho waters.

A number of studies have been conducted in which salmonids and other fish were exposed to DDTs in the diet or through injection, and in some of these, whole body DDT concentrations associated with adverse effects have been measured. Most reported effects of DDTs on salmonids are associated with whole body tissue concentrations in the 1 mg/kg to 3 mg/kg ww range or greater, with some effects on early life stages (e.g., eggs, embryos, and fry) at tissue concentrations in the 0.5 mg/kg ww range (Johnson and Pecor 1969; Poels *et al*. 1980; Burdick *et al*. 1964; Buhler *et al*. 1969; Allison *et al*. 1964, Macek 1968). These concentrations are somewhat higher than the 0.08 to 0.4 mg/kg that were calculated by EPA under the proposed chronic aquatic life or fish consumption criteria. Effects at lower tissue concentrations have been reported in field studies. For example, Vuorinen *et al.* (1997) found correlations between DDT concentrations in muscle of female Baltic salmon and mortality of yolk sac fry. Muscle DDT concentrations in this study ranged from 0.00134 mg/kg to 0.0277 mg/kg, with an average of 0.00541 mg/kg. However, these are muscle, not whole body residues, and the data are difficult to interpret because PCBs and other organochlorine pesticides were also present.

The estrogenicity of various DDT isomers (o,p'-DDT, o,p'-DDE, and p,p'-DDE) has been tested in salmonids exposed to DDTs through injection or in the diet (Arukwe *et al*. 1998, 2000; Donohoe and Curtis 1996; Celius and Walther 1998). These compounds appear to be estrogenic, but relatively high exposure concentrations were required for effects to be observed. For example, Donohoe and Curtis (1996) observed vitellogenin induction in juvenile rainbow trout after injecting trout at 14 day intervals with single or triplicate doses of o,p'-DDT, or o,p'-DDE (0 mg/kg, 5 mg/kg, 15 mg/kg or 30 mg/kg). Plasma vitellogenin and hepatic estrogen binding site concentrations were significantly elevated by o,p'-DDT and o,p'-DDE (total dose 45 mg/kg and 90 mg/kg). Celius and Walther (1998) and Arukwe *et al*. (1998, 2000) observed induction of eggshell (zona radiata) proteins in Atlantic salmon after injection with o,p'-DDT at a dose of 25 mg/kg body weight twice a week for 3 weeks. If we assume an uptake rate of 50% for dietary exposure, which is the typical value observed in feeding studies with salmonids (Allison *et al*. 1963; Meador 2002), and a 75% uptake rate for injection (Meador 2002), associated tissue concentrations of DDT in the fish in these studies would be approximately 7.5 mg/kg to 19 mg/kg. This is far above the estimated tissue concentration resulting from water-borne exposure under the proposed chronic criterion (0.15 mg/kg).

One non-salmonid study, performed on Atlantic croaker, suggests that DDT concentrations below the chronic criterion could be associated with adverse health effects. Khan and Thomas (1998) reported a stimulatory effect of o,p'-DDT on gonadotropin release and gonadal growth in Atlantic croaker after 3 weeks at dietary concentrations of 0.02 mg/kg and 0.1 mg/kg, or an estimated tissue concentrations of 0.01 mg/kg to 0.05 mg/kg, assuming a 50% uptake rate (Allison *et al*. 1963). However, this result was obtained with o,p'-DDT alone, which accounts for only about 20% of typical DDT mixtures, so tissue body burdens associated with such a result in the environment would probably be closer to the 0.05 mg/kg to 0.25 mg/kg range. These results suggest the potential for subtle effects of DDT on fish reproductive physiology at concentrations below the 0.15 mg/kg concentration allowed under the proposed criterion.

Some additional studies show that chronic exposure to DDTs can threaten fish health through other modes of action, but there is insufficient information to determine the effective doses for these health effects. For example, chronic exposure to DDT may contribute to cancer risk. Nunez *et al*. (1988) determined that DDT in the diet enhanced the risk of hepatocarcinogenesis in rainbow trout treated with carcinogenic PAHs aflatoxin B1 and N-methyl-N'-nitro-N-nitrosoguanidine. However, the concentration in the diet was relatively high (100 mg/kg for 12 months), so would likely result in tissue concentrations well above those associated with the chronic criterion. The minimum DDT exposure associated with increased cancer risk in fish is unknown. There is some evidence that DDT may disrupt cortisol secretion and stress response in salmonids (Benguira and Hontela 2000; Hontela 1997) from *in vitro* experiments, but it is difficult from these studies to determine concentrations of DDTs in ambient water or tissue that would be associated with such effects.

The chronic criterion for DDT of 0.001 µg/L is not based on species effects on aquatic life, but on the highest permissible value for wildlife protection. At the time when these criteria were developed, chronic toxicity data for DDT were available for only one freshwater fish species, the fathead minnow (Jarvinen *et al*. 1977). The chronic value from this study, which was a life cycle test, was 0.74 µg/L. The value of 0.001 µg/L was obtained by using a maximum permissible tissue concentration of 0.15 mg/kg, based on reduced reproductive output of the brown pelican. The value of 0.15 mg/kg is the lowest reported DDT concentration in the pelican's major food source, the northern anchovy, associated with reduced egg shell thickness and low productivity in the pelican (C). The water quality criterion was calculated by dividing the target tissue concentration (0.15 mg/kg) by geometric mean (17,870) of a group of 80 normalized BCF values derived from field and laboratory studies in freshwater fish and invertebrates, and by an estimated percent lipid value of 8 in the pelican diet (EPA 1980f). Consequently, the data used for the development of this criterion have little bearing on the chronic toxicity of DDTs to listed salmonids or their prey.

***Behavioral Effects.*** A variety of behavioral effects, including changes in temperature selection and exploratory behavior have been observed in salmonids and other fish species following short-term exposure to DDTs (Davy *et al*. 1973; Peterson 1973; Gardner 1973), but exposure concentrations were substantially above the proposed acute criterion (10 µg/L to 50 µg/L).

***Factors Affecting DDT Uptake and Toxicity.*** Several reports indicate that smaller-sized salmonids take up relatively more DDT from the water column and are more sensitive to the action of DDT compared to larger individuals (Buhler and Shanks 1972; WHO 1989; Murphy 1971). Uptake of DDT also increases with temperature (Reinert *et al*. 1974), and decreases with increased salinity (Murphy 1970). In rainbow trout exposed to 0.33 µg/L DDT at temperatures of 5°C, 10°C and 15°C, whole body residues after 12 weeks were 3.8 mg/kg, 5.9 mg/kg, and 6.8 mg/kg respectively (Reinert *et al*. 1974). Murphy (1970) determined that increasing salinity from 0.15% to 10% decreased DDT uptake over 24 hours from 22% of the dose to 18% of the dose (body residues decreased from 658g to 328g).

Some studies suggest that DDT and organophosphate (cholinesterase inhibiting) pesticides can act synergistically to produce greater toxicity to the nervous system and cause higher mortality than either contaminant can alone (WHO 1989). DDT and PCB appear to have an additive relationship that impacts other vertebrate populations, such as contributing to avian eggshell thinning (WHO 1989).

#### *2.4.17.2. Habitat Effects of DDT Criteria*

***Toxicity to Food Organisms.*** DDT is highly toxic to many aquatic invertebrate species. Johnson and Finley (1980) reported 96-hour $LC_{50}$s in various aquatic invertebrates (e.g., stoneflies, midges, crayfish, sow bugs) ranging from 0.18 µg/L to 7.0 µg/L, and 48-hour $LC_{50}$s of 4.7 µg/L for daphnids. Other reported 96-hour $LC_{50}$s for various aquatic invertebrate species have been from 1.8 mg/L to 54 mg/L (WHO 1989). In a more recent study, Lotufo *et al*. (2000) examined the relative toxicity of DDTs to several species of freshwater amphipods in water-borne exposures. For *Hyalella azteca*, the $LC_{50}$ for DDT was 0.17 µg/L for a 4-day exposure and 0.1 µg/L for a 10-day exposure. For *Diporeia* spp., the $LC_{50}$ was 2.16 µg/L for 10 days and 0.26 µg/L for 28 days. Using narcosis as an endpoint, the $EC_{50}$ was 0.67 µg/L for 10 days and

0.07 for 28 days. In general, early developmental stages are more susceptible than adults to DDT's effects (WHO 1989). At sub-lethal concentrations, DDT may cause reproductive, developmental, cardiovascular, and neurological changes in aquatic invertebrates (WHO 1989). The reversibility of some effects, as well as the development of some resistance, may be possible in some aquatic invertebrates (Johnson and Finley 1980).

These results suggest that the acute (1.1 µg/L) criterion is probably not protective of gammarid amphipods and related invertebrate salmonid prey, but the chronic aquatic life (0.001 µg/L) standard would likely be protective if the major source of DDT exposure were through the water column. However, because DDTs tend to accumulate in sediment, some reduction in available prey species will likely occur in areas with contaminated sediments.

***Bioaccumulation.*** The chronic exposure to DDTs results in bioaccumulation of these compounds in fish, with most accumulating in the liver and other fatty tissues and relatively little in muscle tissues (WHO 1989). This occurs mainly through the diet from eating contaminated prey, and by uptake from sediment and water (WHO 1989). Developing embryos have been documented to take up DDTs from maternal yolk (WHO 1989).

Bioaccumulation rates vary among fish species. Reported BCFs for DDT range from 1,000 to 1,000,000 in various aquatic species (EPA 1989b), and bioaccumulation may occur in some species at very low environmental concentrations (< 100 pg/L; Johnson and Finley 1980; Oliver and Niimi 1988). The BCFs for salmonids range from ~10,000 to over 1,000,000; Oliver and Niimi (1988) reported field-derived BCFs of over 4,000,000 and over 11,000,000 for salmonid species from Lake Ontario exposed to p,p'-DDT and p,p'-DDE, respectively. The half-life for elimination of DDT from rainbow trout has been estimated to be about 160 days (WHO 1989).

Uptake of DDT in salmon and other fishes can be influenced by a variety of factors. It tends to be greater with increased trophic status and lipid content (Berglund *et al*. 1997, Bentzen *et al*. 1996). Fish uptake of DDT from the water is also size-dependent with smaller fish taking up relatively more than larger fish (WHO 1989). Eutrophication and nutrient loading also tend to increase uptake, probably because of the higher concentration of organic matter and bound DDT in the water (Berglund *et al*. 1997). Muir *et al*. (1994) studied uptake and bioconcentration of p,p'-DDT by rainbow trout at differing levels of DOC. The equilibrium BCFs ranged from 33,300 to 91,000, and bioconcentration tended to be lower with addition of unfiltered humic acid.

***Uptake and Toxicity Through Sediments.*** Because DDT is no longer in use in the United States, the primary source of this compound will not be through point source discharges into surface water bodies, but rather from repositories of the contaminant that are persistent in sediments. This means that DDT will not be taken up only through the water column, but also through direct contact with sediments or through the diet. Thus, studies evaluating the effects of water-borne exposure alone are likely to underestimate actual exposure of organisms in the field.

Because sediments are the likely the primary potential source of DDT, the sediment DDT concentration that would result in DDT concentrations in the water column at or below the proposed criteria can be calculated per Section 2.4.13. For DDT, $\log_{10}(K_{ow}) = 6.19$, $\log_{10}(K_{oc})$

= 6.08, and $F_{cv}$ = 0.001, resulting in an estimated $SQC_{oc}$ = 1.2 mg/kg organic carbon. This would mean that for sediment TOC levels of 1% and 5%, the proposed criteria would be associated with sediment DDT concentrations ranging from 12 ng/g to 60 ng/g sediment. This level exceeds the sediment screening guideline of 6.9 ng/g dw established by the COE for in-water disposal of dredged sediment (COE 1998), and are above the interim Canadian freshwater sediment guidelines of 1.19 ng/g to 4.77 ng/g dw sediment. The higher of these values is a probable effect level, based on spiked sediment toxicity testing and associations between field data and biological effects (CCME 2001). This suggests the potential for impacts on the salmonid prey base, as these guidelines are based primarily on tests with benthic invertebrates.

Because there has been very little research on the toxicity of sediment-associated DDT, the sediment concentrations that can cause adverse effects are not well defined. The BSAFs have not been determined for salmonids, so it is difficult to estimate the likely tissue concentrations of DDT that would be associated with sediment DDT concentrations permissible under the proposed criteria. Without site-specific BSAFs for DDTs in salmonids, it is difficult to determine whether the proposed chronic criterion would be sufficiently protective.

As noted earlier, salmonid invertebrate prey are also likely to take up DDTs from sediments. Results of laboratory and field investigations suggest that thresholds for chronic effects generally occur at total DDT concentrations in sediments of about 2 ng/g dw (Long *et al*. 1995). Similarly, equilibrium partitioning methods predict that chronic effects may occur at DDT concentrations in sediment as low as 0.6 ng/g to 1.7 ng/g dw (Pavlou *et al*. 1987). Chapman (1996) estimated no observed effect levels for sediment DDTs at 8.5 mg/kg dw sediment based on full life cycle tests with the marine polychaete worm *Neanthes arenaceodentata*. If the sediment DDT concentrations associated with the proposed water column concentrations were associated with sediment DDT concentrations of 7 µg/kg to 60 µg/kg, these results suggest that they may not be adequate to protect invertebrate prey species from potential injury.

### *2.4.17.3. Summary for DDTs*

Sediment and fish tissue DDT concentrations from Brownlee Reservoir tended to be the highest found in sampling in various locations in Idaho (Table 2.3.1; Clark and Maret 1998). In water, baseline concentrations of DDT found in Brownlee Reservoir in 2011 were <0.00066 µg/L, which is below the levels where effects would be expected to listed salmon and steelhead. DDT is a banned substance in the United States and so no new or ongoing discharges are expected to occur.

Concentrations of DDT in the action area at the proposed action acute criterion could cause harm to listed fish; however, because there will be no new discharges of DDT and no known hotspots of DDT occur in the action area where listed fish are present these effects are unlikely to occur.

The chronic criteria have risk of sublethal health effects in salmonids if bioconcentration results in tissue concentrations that are higher than those expected by EPA. The proposed chronic criterion may allow substantial bioaccumulation to occur because DDTs are taken up not only from the water column but also from sediments and prey organisms. No reports of direct adverse effects to listed salmonids were located at concentrations lower than the chronic criterion. While some data are equivocal and there are quite a few uncertainties in interpreting DDT risks to fish, we found no persuasive evidence of adverse effects from DDT at concentrations lower than the chronic criterion concentrations.

### 2.4.18. The Effects of EPA Approval of the Endosulfan Criteria

Endosulfan is a broad spectrum polychlorinated cyclodiene insecticide. It is used to control over 100 agricultural pests and 60 food and non-food crops, and does not occur naturally in the environment. It was first developed in Germany by Hoechst in 1954 under the registered trade name Thiodan. Endosulfan use is highly restricted in the United States. The EPA cancelled its registration for home and garden use in 2000, and in 2012 banned all uses in the United States[6].

Endosulfan is virtually insoluble in water, but is readily dissolved in organic solvents before its addition to aqueous formulations (Naqvi and Vaishnavi 1993; Goebel *et al*. 1982). In its pure form, endosulfan exists in two different conformations: I (alpha) and II (beta). Technical endosulfan, the form which is most often used in laboratory toxicity studies, is 94% to 96% pure, with an approximate ratio of 7:3 alpha:beta isomers (Naqvi and Vaishnavi 1993). In alkaline water, hydrolysis is the primary process for degradation, with the beta isomer hydrolyzing more rapidly than the alpha isomer (Peterson and Batley 1993). Endosulfan diol is the main product of chemical hydrolysis, but it is also oxidized to endosulfan sulfate (Naqvi and Vaishnavi 1993). In solution, the alpha isomer is more abundant than the beta isomer or endosulfan sulfate. Also, in the aquatic environment, endosulfan beta and endosulfan sulfate are more likely to be bound to sediment and particulates than endosulfan alpha (Peterson and Batley 1993).

Endosulfan acts as a central nervous system poison (Naqvi and Vaishnavi 1993). Of the organochlorine insecticides, it is one of the most toxic to aquatic organisms (EPA 1976; EPA 1980g). In general, freshwater fish are more sensitive to endosulfan than freshwater invertebrates (EPA 1980g), and marine organisms are more sensitive than freshwater ones (Naqvi and Vaishnavi 1993). The toxicities of endosulfan and endosulfan sulfate are roughly equivalent (Naqvi and Vaishnavi 1993). However, comparisons of the toxicity of individual isomers of endosulfan indicate that the alpha form is generally more toxic than the beta. The other biological metabolites of endosulfan that do not contain sulfur, such as endosulfan diol, endosulfan ether, and endosulfan lactone, are considerably less toxic than either the sulfur-containing endosulfan sulfate or alpha or beta isomers.

Most endosulfan toxicity studies on aquatic organisms that have been conducted have evaluated direct water-borne exposure. Studies reported by Barry *et al*. (1995) indicated that, for the cladoceran *Daphnia carinata*, water-borne exposure is in fact the most toxic route. Toxicity towards *D. carinata* was also found to increase at higher food concentrations. This may be due to a higher level of persistence of endosulfan in the water column, or increased uptake of the compound by the test organisms due to elevated metabolism. Similar toxicity studies that assessed food concentration or route of exposure for fish were not found in the literature.

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[6] http://www.epa.gov/pesticides/reregistration/endosulfan/endosulfan-cancl-fs.html

However, there are other aspects of study design that can influence toxicity outcome. Static flow or semi-static assay conditions are more likely to underestimate toxicity when compared with the more environmentally relevant constant flow assays. Studies that include nominal, or unmeasured, test compound concentrations during the exposure period also are more likely to underestimate toxicity compared with those with measured concentrations (Naqvi and Vaishnavi 1993).

The toxic effects of endosulfan on fish are influenced by water temperature, with increased toxicity generally observed at higher temperatures. The influence of temperature is discussed further below.

### *2.4.18.1. Species Effects of Endosulfan Criteria*

The proposed acute criterion for endosulfan is 0.22 µg/L and the chronic criterion is 0.056 µg/L.

***Acute Endosulfan Criterion.*** NMFS found only one study that reported acute lethal effects of endosulfan on salmonids during a 96-hour exposure period at a concentrations roughly 0.8 times the proposed acute criterion for endosulfan. These results were reported as $LC_{50}$ values, suggesting that the proposed acute criterion could be lethal to listed salmonids.

Lemke (1980, cited in EPA 1980g) reported a 96-hour $LC_{50}$ value of 0.17 µg/L for rainbow trout exposed in a flow-through experiment in which endosulfan concentration was measured.

Two other studies reported 96-hour $LC_{50}$s that were near the acute criterion:

> Nebeker *et al*. (1983) reported a value of 0.3 µg/L for rainbow trout exposed at 12$^{o}$C in a flow-through experiment in which endosulfan concentration was measured.
>
> Schoettger (1970) also reported a value of 0.3 µg/L for rainbow trout exposed at 10$^{o}$C andpH 7.4 in a static experiment with a nominal endosulfan concentration.

Most other studies that were found reported 96-hour $LC_{50}$s greater than the acute criterion, including:

> Sunderam *et al*. (1992) reported a value of 0.7 µg/L for rainbow trout exposed at 12$^{o}$C, pH 7.5 in a static experiment in which endosulfan concentration was measured.
>
> Faggella *et al*. (1990, cited in Fujimura *et al*. 1991) reported a value of 0.74 µg/L for Chinook salmon fry.
>
> Johnson and Finley (1980) reported a value of 1.2 µg/L for rainbow trout exposed at 13$^{o}$C, pH 7.2 to 7.5 in a static experiment with a nominal endosulfan concentration.
>
> Macek *et al*. (1969) reported a value of 1.5 µg/L for rainbow trout exposed at 12.7$^{o}$C, pH 7.1 in a static experiment with a nominal endosulfan concentration.

In rainbow trout exposures to endosulphan lasting minutes to hours, ventilation frequencies were increased; however, the exposure concentrations were more than an order of magnitude above the median $LC_{50}$ for rainbow trout (Patra *et al*. 2009).

It should be noted that half of these studies were performed with nominal concentrations of endosulfan, and most studies were performed under static conditions, both of which tend to underestimate toxicity. Lemke (1980, cited in EPA 1980g) noted that flow-through assays with rainbow trout resulted in three times higher toxicity at the same measured concentration of endosulfan as in static assays.

***Chronic Endosulfan Criterion*.** The available information on the chronic effects of endosulfan on salmonids or other freshwater fish is limited. NMFS found only one study in the literature that reported chronic effects of endosulfan on salmonids. Arnold *et al*. (1996) observed sublethal effects at concentrations between 0.2 times and 1.8 times the proposed chronic criterion. Mature male rainbow trout that were exposed for 28 days to 0.01 µg/L endosulfan (measured) in a flow-through assay at 14.5°C developed qualitative hepatic cytological ultrastructural alterations. This dose was the LOEC. At 0.05 µg/L and 0.1 µg/L, degenerative effects such as dilation of intermembranous spaces in mitochondria and deformation of mitochondria were observed. Other effects included proliferation of smooth endoplasmic reticulum (SER), circular arrays of rough endoplasmic reticulum (RER), and an increase in lysosomal elements. The SER and RER effects were probably an indication of the activity of mixed-function oxygenases. These types of structural alterations have been shown by many investigators to be highly selective and sensitive biomarkers of chronic toxicity, although specific effects on fish health have not been elucidated.

Toxicity studies on other freshwater fish species have indicated adverse effects when exposure concentrations ranged between 0.8 times and 3.6 times the chronic criterion:

> Verma *et al*. (1981) exposed the freshwater catfish *Mystus vittatus* to 0.045, 0.067, and 0.13 µg/L endosulfan for 30 days at 24°C in a nominal, static renewal assay. This treatment caused alterations in acid phosphatase, alkaline phosphatase, and glucose-6-phospatase in liver, kidney, and gills. Although the reason for these alterations is not clear, they may be due to uncoupling of oxidative phosphorylation or structural alterations on lysosomes.
>
> Sastry and Siddiqui (1982) exposed the freshwater murrel *Channa punctatus* to 0.2 µg/L endosulfan for 15 and 30 days at 20°C, pH 7.4 in a static renewal assay. This resulted in a reduction in the rate of glucose absorption by the intestine, possibly due to structural damage to the intestinal mucosa, or a decrease in the activity of enzymes that are involved in nutrient absorption, such as Na+-K+ ATPase and alkaline phosphatase.

The results of several studies indicate adverse effects can occur when concentrations are below or near the proposed chronic criterion after an exposure period less than 96 hours. Effects were evident at concentrations that were between 0.9 times and 1.8 times the proposed chronic criterion, suggesting that chronic toxic effects could occur to salmonids under the proposed criterion.

Murty and Devi (1982) exposed the freshwater snakehead fish *Channa punctata* (Bloch) to 0.05 µg/L endosulfan alpha for 4 days at 27°C in a nominal, continuous flow assay. The lipid content and glycogen concentration of liver, muscle, and brain were significantly altered, as was the protein content of muscle and kidney.

Nowak (1996) exposed the freshwater catfish *Tandanus tandanus* to 0.1 µg/L endosulfan for 24 hours in a nominal, static assay. Effects observed included dark atrophied hepatocytes (usually a sign of cell necrosis resulting from chronic injury); structural (necrotic) changes in liver; proliferation, dilation, and vesiculation of the RER (possibly due to inhibition of protein synthesis); concentric bodies (a possible sign of cytologic regeneration); and residue levels in liver up to 80 ppb.

Nowak (1992) exposed *Tandanus tandanus* to 0.1 µg/L endosulfan for 24 hours in a measured, static assay. This resulted in the presence of edema and lifting and hyperplasia of lamellar epithelium in the gills, and also led to an increase in the respiratory diffusion distance. Although this may allow separation of blood from the toxicant, it can also damage gills, having deleterious effects on fish physiology.

Rao *et al*. (1980) exposed the Indian major carp *Labeo rohita* to 0.1 µg/L endosulfan for 1 hour at 28°C, pH 8.4 in a nominal, static assay. An increase in oxygen consumption was observed.

These studies collectively indicate the possibility for adverse effects to occur to listed salmonid species under the chronic and acute criteria proposed for endosulfan. Adverse effects of this nature will likely result in appreciable mortality depending on the nature of the exposure. NMFS assumes this will reduce abundance and productivity of any listed salmon and steelhead that are exposed.

***Other Water Quality Parameters as Predictors of Endosulfan Toxicity.*** Schoettger (1970) tested various water quality parameters to determine their effect on the toxicity of endosulfan to several fish species. Variations in calcium and magnesium salts did not alter the acute toxicity to western white suckers, nor did changes in pH between 6.4 and 8.4. However, experiments with rainbow trout indicated that temperature changes did have an effect on toxicity. In three different studies, endosulfan toxicity was found to increase with increasing temperature. Two other studies using rainbow trout also reported a temperature effect. Sunderam *et al*. (1992) determined that the 96-hour $LC_{50}$ changed from 1.6 µg/L at 4°C to 0.7 µg/L at 12°C, using static conditions, pH 7.5, and measured concentrations of endosulfan. Macek *et al*. (1969) reported 96-hour $LC_{50}$s of 2.6 µg/L, 1.7 µg/L, and 1.5 µg/L at 1.6°C, 7.2°C, or 12.7°C, respectively, under static conditions at pH 7.1 and nominal endosulfan concentrations.

### *2.4.18.2. Habitat effects of Endosulfan Criteria*

***Toxicity to Food Organisms.*** NMFS found three studies that reported lethal toxicity of endosulfan to aquatic macroinvertebrates at concentrations that were 0.5 to 10 times the acute criterion, suggesting this criterion might not be protective for some salmonid prey species:

Magdza (1983, cited in Sunderam *et al*. 1994) reported a 48-hour $EC_{50}$ value of 0.3 µg/L for the South African freshwater cladoceran *Daphnia longispina*.

Leonard *et al*. (1999) conducted acute toxicity tests on three insect species in static experiments using river water at 26°C and a nominal concentration of endosulfan. A 48-hour $LC_{50}$ value of 0.4 µg/L was determined for a trichopteran larvae, with an LOEC of 0.3 µg/L. Seventy-two hour $LC_{50}$s of 1.0 µg/L and 0.6 µg/L were determined for two ephemeropteran nymphs, with a corresponding LOEC of 0.3 µg/L.

Sanders and Cope (1968, cited in EPA 1980g) reported an $LC_{50}$ value of 2.3 µg/L for a stonefly species under static conditions with nominal endosulfan concentrations.

However, most toxicity studies indicate lethal effects from endosulfan do not occur on salmonid prey species until concentrations are between 19 to 2,200 times the proposed acute criterion. These species include the freshwater scud *Gammarus lacustris*, with 96-hour $LC_{50}$ values of 4.1 µg/L or 5.8 µg/L (Johnson and Finley 1980; Sanders 1969, cited in EPA 1980g); the cladoceran *Daphnia magna*, with 96-hour $LC_{50}$ values of 56 µg/L to 271 µg/L (Schoettger 1970; Nebeker *et al*. 1983; EPA 1976); damselfly naiad 96-hour $LC_{50}$ of 71.8 µg/L to 107 µg/L (Schoettger 1970); and a 48-hour $LC_{50}$ of 215 µg/L for *Moinodaphnia macleayi* or 491 µg/L for *Ceriodaphnia dubia* (Sunderam *et al*. 1994).

Chronic exposure studies reported in the scientific literature appear to include only cladocerans, and all of these studies report chronic effects at concentrations well above the proposed chronic criterion. For example, *D. magna* exhibited reduced survival after 22 days of exposure to 7 µg/L endosulfan or reduced reproduction in the second generation at 37.7 µg/L (EPA 1976); the LOEC for decrease in number of young for *C. dubia* was 20 µg/L after 14 days exposure, or 40 µg/L for *M. macleay* (Sunderam *et al*. 1994); and reduction of brood size and body length for *Daphnia carinata* was observed after 6 days at 320 µg/L (Barry *et al*. 1995).

In summary, available toxicity data suggest that the proposed chronic criterion may be protective of salmonid prey species relevant to Idaho waters. However, because this collection of reports does not represent the range of salmon prey species, it is impossible to know for certain whether the chronic criterion would avoid population impacts on important prey items, such as insects, copepods, gammarid amphipods, other crustaceans, and molluscs.

***Bioaccumulation.*** Information on uptake, metabolism, and elimination of endosulfan was not available for any salmonid species. However, the following is a brief overview of information available for other freshwater fish species, including *Channa punctata* (Devi *et al*. 1981), *Labeo rohita* (Rao *et al*. 1980), the Indian carp *Catla catla* (Rao 1989), *Anabus testudineus* (Bloch) (Rao and Murty 1980), and goldfish and western white sucker (Schoettger 1970).

The unaltered alpha and beta forms of endosulfan were detected in *Channa punctata*, *Anabus testudineus*, and *Catla catla* in one or more tissues, including brain, gills, kidney, liver, and muscle. In *Catla catla* in particular, muscle was found to be the principle storage site of unaltered endosulfan.

The principal metabolites of endosulfan in *Catla catla*, *Channa punctata*, or *Labeo rohita* were reported to be endosulfan alcohol, endosulfan ether, or endosulfan lactone. Other metabolites that were detected in various fish included endosulfan alpha-hydroxyether and endosulfan sulfate. The liver was cited as either the principal detoxifying organ or the site where uptake appeared to be considerably higher than for other tissues in *Labeo rohita*, western white sucker, and goldfish. This differed somewhat from *Anabus testudineus*, in which both the liver and kidneys were reported as being the principal sites of detoxification.

Reports on the bioconcentration of endosulfan in salmonids were not available, although limited information for other freshwater fish was found, indicating that the BCF can vary greatly between species. Ramaneswari and Rao (2000) exposed *Channa punctata* to 0.141 µg/L endosulfan (alpha or beta isomers) for 1 month and measured a whole body BCF of 13. A similar exposure of *Labeo rohita* yielded a BCF of 37 for alpha endosulfan and 55 for beta endosulfan. The exposure concentration used (0.141 µg/L) was 2.5 times the proposed chronic criterion. These BCF values were much lower than those obtained for yellow tetra (*Hyphessobrycon bifasciatus*), in which the whole body BCF was 11,600 after a 21-day exposure to 0.3 µg/L endosulfan at 22$^{o}$C, pH 7.1 under static-renewal conditions (Jonsson and Toledo 1993). In this study, the total residues in fish increased with increasing time, and the authors indicated that a steady state had not been reached. The biological half-life was estimated at 1.8 days, which is similar to goldfish (Oeser *et al*. 1971, cited in Geobel *et el*. 1982).

NMFS found only two reports of endosulfan bioaccumulation for salmonid prey species. Sabaliunas *et al*. (1998) exposed the lake mussel *Anodonta piscinalis* to 1.5 µg/L endosulfan in a continuous flow experiment at 10$^{o}$C with measured contaminant concentration. They noted a whole body concentration factor of 750 under conditions that may not have reached steady state. Finally, a field study was conducted by the Mussel Watch Project (part of the National Oceanic and Atmospheric Administration's National Status and Trends Program) using paired oyster whole body tissue samples and water samples from the Patuxent River, which discharges into the Chesapeake Bay in Maryland (Lehotay *et al*. 1999). They found that in oyster tissue, more endosulfan sulfate was present compared to the alpha or beta isomers. In the water samples, more of the beta isomer was present than the alpha isomer or endosulfan sulfate (even though beta is less soluble than alpha and constitutes only 30% of the endosulfan mixture that is commonly used). Based on the average concentration of endosulfan alpha, beta, or sulfate in oyster tissue (0.037 ng/g to 0.13 ng/g) or in water samples (0.5 ng/L to1.0 ng/L), one can calculate the BCF range as 37 to 260.

#### *2.4.18.3 Summary for Endoculfan*

Endosulfan has not been found in Idaho waters or sediments at levels that approach the standards as proposed and future discharges of endosulfan are unlikely to occur because the product use has been banned so an acute exposure scenario from an authorized release is unlikely. The proposed acute lethal criterion for endosulfan would likely result in some mortality of listed salmonids. Reported rainbow trout $LC_{50}$s near or below the proposed acute criterion indicate that appreciable mortality can occur in waters meeting the proposed criterion. Evaluation of the proposed chronic criterion was restricted by the absence of relevant toxicity testing data involving salmonid species. The limited information that could be gathered on rainbow trout and two other freshwater fish suggests that the proposed chronic criterion can allow chronic physiological damage to listed salmonid species. The physiologic damage was not directly related to "clinically significant" fish health changes. Although there is a paucity of toxicity testing data, the available information suggests that the proposed acute and chronic criteria may protect some invertebrate prey species. Little test data exists for specific salmonid prey species

### 2.4.19. The Effects of EPA Approval of the Endrin Criteria

Endrin is a chlorinated pesticide that is a steroisomer of dieldrin. It is no longer manufactured in the United States. Endrin ketone and endrin aldehyde are variants that occur as impurities or degradation products of endrin in commercial preparations of the insecticide. Endrin was first used in 1951 to control insects and rodents on cotton, apples, sugarcane, tobacco, and grain (IARC 1974; EPA 1980h; HSDB 1995). Its toxicity to migrant populations of migratory birds was the main reason for its cancellation as a pesticide in 1986 (EPA 1992a). It was still used as a toxicant on bird perches for several years, but this use was also banned in 1991 (EPA 1992a). There are no current authorized uses of endrin in the United States

Exposure of rodents to endrin has been noted to result in adverse neurologic, liver, kidney, and miscellaneous endocrine and tissue weight effects (Kavlock *et al*. 1981; Hassan *et al*. 1991; Deichmann *et al*. 1970). There are some indications that endrin may have genotoxic effects, including increased DNA damage in hepatocytes due to oxidative injury (Bagchi *et al*. 1992a,b, 1993c; Hassoun *et al*. 1993). However, most studies suggest that endrin is not carcinogenic (EPA 1980h).

#### *2.4.19.1. Species Effects of Endrin Criteria*

The acute criterion for dissolved concentrations of endrin is 0.18 µg/L. The chronic criterion is 0.0023 µg/L (Table 1.3.1) and is based on tissue residue values associated with adverse effects in wildlife (EPA 1980h).

***Acute Endrin Criterion.*** The proposed acute criterion of 0.18 µg/L is below values associated with adverse effects in fish in most studies, but there is evidence in some studies of mortality occurring at concentrations below or near the proposed criterion. Reported $LC_{50}$s for salmonids range from 0.113 µg/L to 343 µg/L (Post and Schroeder 1971; Katz 1961; Bennett and Wolke 1987a; 1987b; EPA 1980h). While the majority of available studies showed effects at concentrations well above the criterion, in many cases they were nominal concentrations only, not measured concentrations, so their accuracy is not assured. Other fish species have also been found to be sensitive to acute effects when concentrations of endrin that are close to the acute criterion. For example, Jarvinen *et al*. (1988) reported a 96-hour $LC_{50}$ of 0.7 µg/L for fathead minnow larvae (*Pimphales promelas*). They also found that a 48-hour exposure at the same concentration led to a reduction in growth that was detectable within 28 to 30 days. Similarly, Hansen *et al*. (1977) reported an $LC_{50}$ of 0.3 µg/L for juvenile sheepshead minnow.

***Chronic Endrin Criterion.*** There are few data available regarding chronic effects of water-borne exposure to endrin in salmonids. In other species, adverse effects have not been reported unless water concentrations were more than 10 times the proposed chronic criterion of 0.0023 µg/L (e.g., Hansen *et al*. 1977; Jarvinen and Tyo 1978; Jarvinen *et al*. 1988). However, there are some data available on tissue concentrations of endrin associated with a variety of sublethal effects in rainbow trout. Grant and Mehrle (1973) determined that tissue levels associated with effects in rainbow trout included: alteration of plasma parameters, suppression of cortisol secretion and inhibited carbohydrate metabolism after a swim challenge at 0.01 mg/kg to 0.02 mg/kg; hyperexcitability at 0.12 mg/kg; and hyperglycemia and reduction in growth at 0.12 mg/kg to 0.22 mg/kg. No effects were seen at tissue concentrations at or below 0.00025 mg/kg (Grant and Mehrle 1973).

It is difficult to estimate the likely tissue concentrations of endrin in salmonids exposed at ambient water concentrations equivalent to the chronic criterion of 0.0023 µg/L, because no specific BCFs could be found for salmonids. However, for other fish species, reported BCFs range from 1,340 for spot to 15,000 for flagfish, with exposure periods ranging from 4 days to 300 days (EPA, 1980h). Many of these values were derived from field exposures, and thus likely incorporated dietary as well as water uptake. Assuming that this range of BCFs is accurate for salmonids would mean that a water concentration at the chronic criterion would result in estimated tissue concentrations ranging from 0.0033 mg/kg to 0.0345 mg/kg. Data from Grant and Mehrle (1973) suggest the potential for some effects on metabolism, stress response, and growth at water concentrations of endrin at or within 10 times the chronic criterion.

Laboratory exposure studies also suggest that exposure to endrin may affect immune responsiveness in rainbow trout. Bennet and Wolke (1987a,b) exposed rainbow trout for 30 days to sublethal concentrations of endrin that were greater than criteria concentrations (0.12 µg/L to 0.15 µg/L) and found that several immune responses (migration inhibition factor assay), plaque forming cell assay, and serum agglutination titres were inhibited when fish were exposed to *Yersinia ruckeri* O-antigen. Serum cortisol concentrations were found to be significantly elevated in endrin-exposed fish. Fish receiving cortisol in the diet also showed reduced immune responsiveness, suggesting that elevated serum cortisol concentration obtained in endrin-exposed fish has a central role in repression of the immune response. Fish were exposed to only one dose of endrin in this experiment; however, so there is no information on the threshold endrin concentration for immunosuppresive effects. Exposure to water-borne endrin from agricultural runoff has been associated with an increased prevalence of parasitic infections in cultured sand goby (Supamataya 1988), but the fish were also exposed at the same time to dieldrin, DDTs, and possibly stress due to changes in dissolved oxygen and water temperature.

Singh and Singh (1980) reported total lipid levels in ovary and liver and cholesterol concentrations in ovary, liver and blood serum in the Asiatic catfish *Heteropneustes fossilis* after 4 weeks exposure to endrin at concentrations of 0.0006 µg/L and 0.008 µg/L during different phases of the annual reproductive cycle. Even the lower concentrations of pesticides induced a significant decrease in liver lipid during the preparatory and late post-spawning phases. An appreciable increase in ovarian cholesterol was noticed during the pre-spawning and spawning. Serum cholesterol values demonstrated a significant increase in the preparatory and late post-spawning phases after exposure to pesticides at all concentrations. This study suggests that exposure to endrin concentrations below the proposed chronic criterion could affect lipid and cholesterol balance in other gravid fish, including presumably salmon.

***Factors Affecting Toxicity.*** Studies by Dalela *et al*. (1978) suggest that increases in temperature and pH may increase endrin toxicity, and that smaller fish were more susceptible to adverse effects from a given exposure concentration than larger fish.

### *2.4.19.2. Habitat Effects of Endrin Criteria*

***Toxicity to Food Organisms.*** Invertebrates tend to be more tolerant of endrin than fishes. When Anderson and DeFoe (1980) exposed stoneflies, caddisflies, isopods, and snails to endrin in a flowing-water test system for 28 days, increased mortality was observed at concentration in the 30,000 µg/L to 150,000 µg/L range. These values are at least two orders of magnitude above the acute criterion and at least four orders of magnitude above the chronic criterion, suggesting that both criteria would likely be protective of salmonid prey species. However, the available information is limited and may not account for exposure through other routes, such as sediments (see below).

***Bioaccumulation.*** Studies show that endrin is bioaccumulated significantly by fish and other aquatic organisms (ATSDR 1996; EPA 1980h; Metcalf *et al*. 1973). Although specific bioconcentration factors are not available for salmonids, for other fish they range from 1,640 to 15,000 (EPA 1980h; Hansen *et al*. 1977). Endrin is also taken up by invertebrate prey species of salmonids, although bioconcentration factors are typically lower than those for fish. Anderson and DeFoe (1980) report pesticide accumulation in stoneflies, an invertebrate prey species, of 350 to 1150 times greater than the water concentrations after a 28-day exposure. However, biomagnification of endrin with increasing trophic level is less than that for some other chlorinated pesticides (Leblanc 1995; Metcalf *et al*. 1973). For example, in a model laboratory aquatic ecosystem containing algae, snails, water fleas, mosquito larvae, and mosquito fish, Metcalf *et al*. (1973) reported a ratio of biomagnification through the aquatic food chain to bioconcentration by direct uptake from water of 2.0 for endrin compared to 2.5 for DDT.

***Uptake and Toxicity Through Alternate Routes of Exposure.*** Endrin in the diet may be an important source of uptake for fish species. Jarvinen and Tyo (1978) found that endrin in the food at a concentration of 0.63 mg/kg significantly reduced survival of fathead minnows in whole life cycle exposure tests, and residues contributed by food-borne endrin appeared to be additive to those contributed by water. Based on available BCF estimates for endrin; however, prey items would not accumulate endrin at this level under the proposed criterion. For a water concentration of 0.0023 µg/L, the proposed chronic criterion, and a BCF of 15,000, the highest reported for aquatic organisms in EPA's criteria documents (EPA 1980h), the predicted tissue concentration would be only 0.035 mg/kg.

Because endrin is no longer in use in the United States, the primary source of this compound will be from repositories of the contaminant that are persistent in sediments, not through point source discharges into surface water bodies. This means that endrin exposure can occur through the water column, through direct contact with sediments, or through the diet. Thus, studies evaluating the effects of water-borne exposure alone are likely to underestimate actual exposure of organisms in the field.

Because sediments are likely the primary source of endrin, the sediment endrin concentration that would result in endrin concentrations in the water column at or below the proposed criteria can be calculated per Section 2.4.13. For endrin, where the maximum reported $\log_{10}(K_{ow})$ is estimated at 5.6, $\log_{10}(K_{oc})$ would equal 5.5. A value of $F_{cv} = 0.0023$ results in $SQC_{oc}$ = 736 µg/kg organic carbon. This would mean that for sediment TOC levels of 1% to 5%, the proposed criteria would be associated with sediment endrin concentrations ranging from 7.36 µg/kg to 36.8 µg/kg dw sediment. These levels are within the range of the interim Canadian freshwater sediment guidelines of 2.67 to 62.4 ng/g dw sediment. The higher of these values is a probable effect level, based on spiked sediment toxicity testing and associations between field data and biological effects (CCME 2001). This suggests that the proposed criteria are unlikely to reduce the quality or quantity of listed salmon food items, although the data used to develop the criteria may not have been specific to salmon or their prey items.

Because there has been very little research on the toxicity of sediment-associated endrin to salmonids, the sediment concentrations that can cause adverse effects are not well defined. Similarly, BSAFs have not been determined for salmonids, so it is difficult to estimate the likely tissue concentrations of endrin that would be associated with sediment endrin concentrations permissible under the proposed criteria. Without this information, it is difficult to determine whether the proposed chronic criterion would be sufficiently protective. Data on effects of sediment-associated endrin to known salmonid prey species are also lacking. Some marine invertebrates show behavioral effects, such as changes in sediment reworking rates, at sediment endrin concentrations within the 7 µg/kg to 38 µg/kg range (Keilty *et al*. 1988a,b,c). In contrast, effects on mortality or burrowing occurred at much higher concentrations (15 mg/kg to 60 mg/kg dw for burrowing avoidance and about 2500 mg/kg for mortality) (Keilty *et al*. 1988a).

### *2.4.19.3. Summary for Endrin*

Endrin is a banned product in the United State and so new discharges are unlikely to occur. In Idaho levels of endrin in Brownlee Reservoir have been detected at less than the chronic criteria. Most reports of mortality following short-term endrin exposures produced $LC_{50}$s greater than the acute criterion, although some effects occurred at lower concentrations. Evidence indicates that concentrations at the acute criterion will not harm salmonid prey species.

While data are sparse, most reports of adverse effects from chronic exposures to salmonids or other fish occurred at concentrations higher than the chronic criterion. A report of subclinical reductions in cholesterol and lipids in gravid Asiatic catfish are of ambiguous importance to salmon. Food chain exposure via diet or sediment was estimated by NMFS to mostly result in tissue residues lower than those shown to be harmful to fish.

2.4.20. The Effects of EPA Approval of the Heptachlor Criteria

Heptachlor is an organochlorine cyclodiene insecticide first isolated from technical chlordane in 1946 (ATSDR 1993). During the 1960s and 1970s, it was commonly used for crop pest control and by exterminators and home owners to kill termites. In 1976, it was prohibited from home and agricultural use, although commercial applications to control insects continued. In 1988, its use for termite control was banned, and currently its only permitted commercial use in the United States is fire ant control in power transformers (ATSDR 1993).

The principal metabolite of heptachlor is heptachlor epoxide, an oxidation product formed by many plant and animal species and through breakdown of heptachlor in the environment. The epoxide degrades more slowly and, as a result, is more persistent than heptachlor. Both heptachlor and heptachlor epoxide adsorb strongly to sediments, and both are bioconcentrated in terrestrial and aquatic organisms (EPA 1980i; ATSDR 1993). Uptake may also occur through the diet or through exposure to contaminated sediments. Heptachlor is readily taken up through the skin, lungs or gills, and gastrointestinal tract (ATSDR 1993). Once absorbed, it is distributed systemically and moves into body fat and is readily converted to its most persistent and toxic metabolite, heptachlor epoxide, in mammalian livers (Smith 1991; ATSDR 1993). Heptachlor is also metabolized to some extent by fish, although most evidence points to it being stored in the body predominantly as heptachlor rather than heptachlor epoxide (Feroz and Khan 1979).

Heptachlor and heptachlor epoxide are considered highly to moderately toxic to mammals, birds, and fish. The primary adverse health effects associated with acute exposure are central nervous system and liver effects (Smith 1991; ATSDR 1993; Buck *et al*. 1959). Chronic exposure to heptachlor may cause some of the same neurological effects as acute exposure. An increased prevalence of neurological symptoms in humans has been associated with environmental exposure to heptachlor in epidemiological studies (Dayal *et al*. 1995), and in laboratory exposure where effects were noted on functional observational ability and motor activity (Moser *et al*. 1995). There is also evidence from epidemiological and laboratory studies that heptachlor alters the expression and function of dopamine transporters (Miller *et al*. 1999). Heptachlor may also affect immune function by inhibiting normal chemotactic responses of neutrophils and monocytes (Miyagi *et al*. 1998) or promoting necrosis of lymphocytes in the spleen and thymus (Berman *et al*. 1995). There is other evidence that heptachlor and heptachlor epoxide are associated with infertility and improper development of offspring (ATSDR 1993; Amita Rani and Krishnakuman 1995; Mestitzova 1967; Oduma *et al*. 1995a,b). On the other hand, heptachlor appears to have limited developmental toxicity, and shows few teratogenic effects in most studies (WHO 1984; ATSDR 1993; Narotsky and Kavlock 1995). Heptachlor does not appear to be a primary carcinogen, and laboratory tests indicate that neither heptachlor nor heptachlor epoxide are mutagenic (WHO 1984; ATSDR 1993).

Heptachlor toxicity can be influenced by the presence of other compounds in the environment, but its interactions with other contaminants have not been well-studied.

*2.4.20.1. Species Effects of Heptachlor Criteria*

The proposed acute criterion for dissolved concentrations of heptachlor is 0.52 µg/L and the chronic criterion is 0.0038 µg/L. The chronic criterion is based on marketability of fish for human consumption, and is the water concentration of heptachlor estimated to ensure that tissue concentrations are below the FDA action level of 0.34 mg/kg for edible fish (EPA 1980i). This expected tissue concentration is unlikely to represent concentrations that would occur in salmon or steelhead tissues because the BCF of 5,220 was not derived from data on salmon or steelhead. It should also be noted that the most stringent heptachlor criteria that are applicable critical habitats are the human health criteria based on fish consumption rather than the chronic aquatic life criteria. The fish consumption based criteria are 10 times more restrictive than the aquatic life criteria and are applicable to all waters with listed salmon and steelhead.

***Acute Heptachlor Criterion.*** The acute heptachlor criterion of 0.52 µg/L was derived from acute $LC_{50}$ values for 18 species of freshwater fish and invertebrates, based primarily on static laboratory exposure tests, and represents the 5th percentile of the mean species values for this group of animals (EPA 1980i). Heptachlor concentrations in water were not measured in any of these tests; reported exposure concentrations were nominal. Acute toxicity to salmonids occurs generally when concentrations are at least an order of magnitude greater than the proposed acute criterion. For example, $LC_{50}$s have been reported as 81.9 µg/L, 24.0 µg/L, and 7.4 µg/L to 26.9 µg/L for coho salmon, chinook salmon, and rainbow trout, respectively (Johnson and Finley 1980; EPA 1980i; Macek *et al*. 1969; Katz 1961). Reported 96-hour $LC_{50}$ values in other fish species have ranged from 5 µg/L to 25 µg/L (Johnson and Finley 1980).

Although measured $LC_{50}$ values for salmonids appear to be substantially above the proposed criterion, there is evidence that the corresponding tests, which involved static exposures at nominal concentrations, may have significantly underestimated the toxicity of heptachlor. In the EPA criteria documents for heptachlor (EPA 1980i), $LC_{50}$ values for saltwater fish ranged from 0.85 µg/L to 10.5 µg/L in flow-through, measured concentration tests (Hansen and Parrish 1977; Schimmel *et al*. 1976; Korn and Earnest 1974), but were as high as 194 µg/L in static, unmeasured tests (Eisler 1970). Notably, the saltwater criterion, based on both types of tests, is 0.053 µg/L which is an order of magnitude lower than the freshwater criterion. Thus, the acute toxicity data for salmonids may underestimate actual toxicity of heptachlor. Still, the criterion of 0.52 µg/L is substantially lower than the lowest reported $LC_{50}$ concentration of 10 µg/L, and this difference probably provides an adequate margin of safety against acutely lethal effects of heptachlor.

***Chronic Heptachlor Criterion.*** Little information is available on the sublethal effects of heptachlor in salmonid species. Carr *et al*. (1999) reported that in channel catfish, heptachlor epoxides, and to a lesser extent heptachlor, bind to the gamma-aminobutyric acid receptor and may thus suppress the activity of inhibitory neurons in the central nervous system. However, because this was an in vitro study, the exposure concentrations associated with this effect in live animals are not clear. Hiltibran (1982) investigated the effects heptachlor on the metal-ion-activated hydrolysis of ATP by bluegill (*Lepomis macrochirus*) liver mitochondria and found that it significantly inhibited ATP hydrolysis in an in-vitro assay. The lowest effective concentration was 0.00056 g/ml of reaction medium, but how that would compare to water concentrations affecting a live animal is not clear.

Chronic toxicity data are correspondingly limited for evaluating the protectiveness of the chronic criterion for salmonids. Exposure studies conducted with other species generally report effects at concentrations well above the proposed chronic criterion. For example, a study conducted on fathead minnow (Macek *et al*. 1976) showed 100% mortality after 60 days at 1.84 µg/L, with effects on sublethal endpoints at 0.86 µg/L. Similarly, Goodman *et al*. (1976) found effects of heptachlor on growth and survival of embryos and fry of the saltwater sheepshead minnow to occur when heptachlor concentrations exceeded 1.2 µg/L. Hansen and Parrish (1977) tested the chronic toxicity of heptachlor to sheepshead minnow in an 18 week partial life cycle exposure begun with juveniles, and observed decreased embryo production at 0.71 µg/L, but dose-response relationships were not consistent for this study so the data may not be accurate. The histological studies revealed conspicuous pathological changes in the liver. Other studies with non-salmonids report pathological effects on the liver and kidney, altered enzyme levels, inhibited fin regeneration, and mortality at higher concentrations (3 µg/L to 70 µg/L) with exposures ranging from 5 to 60 days (EPA 1980g; Radhiah, *et al*. 1986; Radhaiah 1987; Azharbig *et al*. 1990; Konar *et al*. 1970; Rao *et al*. 1980).

In contrast to studies involving strictly water-borne exposure, other evidence suggests that adverse effects may occur when tissue concentrations are below the 0.34 mg/kg limit used to develop the chronic criterion. Tests with non-salmond species also suggest that some effects could occur at tissue residue levels in the 0.016 mg/kg to 0.3 mg/kg range. In spot (*Leistomus xantharus*), tissue concentrations of 0.654 mg/kg were associated with 25% mortality in test fish, and there are reports of increased long-term mortality at concentrations as low as 0.022 mg/kg in sheepshead minnow and 0.01 mg/kg in spot (Schimmel *et al*. 1976). It should be noted that there are some problems with analyses on which fish tissue heptachlor concentrations associated with the chronic criterion were based, particularly with respect to uncertainty about the applicability of a standardized BCF of 5,220 to salmonids.

### *2.4.20.2. Habitat Effects of Heptachlor Criteria*

***Toxicity to Food Organisms.*** There is little data available on the effects of long-term exposures of heptachlor to salmonid prey. Heptachlor is acutely toxic to freshwater aquatic invertebrates at concentrations comparable to those that are lethal to fish (Johnson and Finley 1980; HSDB 1995). Reported $LC_{50}$ values for freshwater invertebrate species have included 0.9 to 2.8 µg/L for stoneflies (Sanders and Cope 1968), 29 mg/kg to 47 mg/kg for gammarid amphipods (Sanders 1969, 1972), and 42 µg/L to 78 µg/L for daphnids (Macek *et al*. 1976; Sanders and Cope 1966). These values were derived from static tests in which heptachlor concentrations were unmeasured. Tests using saltwater species using flow-through tests yielded lower $LC_{50}$ values for grass shrimp and pink shrimp (0.03 µg/L to 0.11 µg/L) than static tests for shrimp and crayfish (1.8 µg/L to 7.8 µg/L; Sanders 1972; Schimmel *et al*. 1976), suggesting that the static tests underestimate the toxicity of heptachlor to aquatic invertebrates.

Sublethal effects of acute exposure have also been reported for some invertebrate species at concentrations close to the proposed criteria, although these studies were not conducted in salmonid prey. Naik *et al*. (1997) determined that heptachlor induced changes in the rate of oxygen consumption and acetylcholinesterase activity in the central nervous system of a freshwater leech *Poecilobdella viridis* within 2 hours, at concentrations ranging from 0.7 µg/L to 3.5 µg/L, the lowest of which is very close to the current acute criterion of 0.52 µg/L.
When the criteria for heptachlor were developed (EPA 1980i), no data were available on chronic effects of this compound on invertebrate species, and little additional information has been generated since that time. Lowest heptachlor concentrations at which effects are reported have been above 0.01 µg/L. For example, a concentration of 0.04 µg/L was associated with increased mortality in the pink shrimp, *Penaeus duoraum* (Schimmel *et al*. 1976), which is well above the proposed chronic criterion.

***Bioaccumulation.*** Both heptachlor and heptachlor epoxide have been shown to bioconcentrate in aquatic organisms such as fish, mollusks, insects, plankton, and algae (ATSDR 1989). They have been found in the fat of fish, mollusks, and other aquatic species at concentrations of 200 to 37,000 times the concentration of heptachlor in the surrounding waters (WHO 1984; ATSDR 1989). A wide range of BCFs have been determined in laboratory studies using fish (EPA 1980i). No BCF values are available for salmonids, but values for fathead minnow range from 9,500 to 14,400 (Veith *et al*. 1979; EPA 1980i). Goodman *et al*. (1976) reported average bioconcentration factors for heptachlor of 3,600 for sheepshead minnow. Uptake of heptachlor by aquatic organisms is influenced by a number of environmental and water quality factors (Vanderford and Hamelick 1977) including concentrations of organic particulate matter in the water column, turbidity, and season of the year. Residue concentrations may also vary considerably between fish species.

***Uptake and Toxicity through Alternate Routes of Exposure.*** Because heptachlor is no longer in use in the United States, except for selected special applications, the primary potential source of this compound will be from repositories of the contaminant that are persistent in sediments not from point source discharges into surface water bodies. This means that if present, heptachlor and heptachlor epoxide would likely be taken up through direct contact with sediments or through the diet not only through the water column. Thus, studies evaluating the effects of water-borne exposure alone are likely to under-estimate actual exposure of organisms in the field.

Because sediments are likely the primary source of heptachlor, the sediment heptachlor concentration that would result in heptachlor concentrations in the water column at or below the criteria is of interest and can be calculated per Section 2.4.13. For heptachlor, $\log_{10}(K_{ow}) = 6.26$, $\log_{10}(K_{oc}) = 6.15$, and $F_{cv} = 0.0038$, resulting in $SQC_{oc} = 5.37$ mg/kg organic carbon. This would mean that for sediment TOC levels of 1% to 5%, the sediment heptachlor concentrations would range from 54 ng/g to 269 ng/g sediment. These levels are higher than the sediment screening guideline of 10 ng/g dw established by the COE for in-water disposal of dredged sediment (COE 1998), and are above the interim Canadian freshwater sediment guidelines of 0.6 ng/g to 2.74 ng/g dry wet sediment. The higher of these values is a probable effect level, based on spiked sediment toxicity testing and associations between field data and biological effects (CCME 2001). This indicates a potential for adverse effects on aquatic life.

Because there has been very little research on the toxicity of sediment-associated heptachlor to salmonids, the sediment concentrations that cause adverse effects are not well defined. The BSAFs have not been determined for salmonids, so it is difficult to estimate the likely tissue concentrations of heptachlor that would be associated with sediment heptachlor concentrations permissible under the proposed criteria. Van der Oost *et al*. (1996) examined biota-sediment ratios of heptachlor in feral eel (*Anguilla anguilla*) and found a large variation in BSAF values between different sites, suggesting that inter-site differences in contaminant bioavailability or in the diets of resident fish could have a strong influence on heptachlor uptake. Without site-specific BSAFs for heptachlor in salmonids, it is difficult to determine if the proposed chronic water quality criterion would be sufficiently protective. The highest levels found of heptachlor in Idaho were in Brownlee Reservoir with sediment levels of <.001ng/g.

*2.4.20.3. Summary for Helptchlor*

Available evidence indicates that listed salmon or steelhead experience acute lethal effects at concentrations much higher than the proposed acute criterion. However, all such evidence is derived from static tests with nominal heptachlor concentrations, a methodology that tends to underestimate toxicity. There is a greater likelihood that heptachlor could harm salmon or steelhead through lethal effects on aquatic invertebrates; however, little information is available on the effects on invertebrate prey species.

Data on chronic effects of heptachlor are sparse, but suggest that the risk of adverse effect through water-borne exposure is likely to be low. Some studies suggest that tissue concentrations that are possible under the chronic criterion could have sublethal or lethal effects on alevins or fry. Bioaccumulation can occur in salmonids with chronic exposure to heptachlor, and when exposure occurs, it is likely to be not only through the water column but through diet and contact with sediments.

2.4.21. The Effects of EPA Approval of Lindane (gamma-BHC) Criteria

On August 2, 2006, EPA announced that the registrants of lindane requested to voluntarily cancel all remaining pesticide registrations of lindane and so there are no remaining uses in the United States.

Lindane is moderately water soluble and may accumulate in sediments. It is relatively persistent and experiences significant degradation only under anaerobic conditions. Lindane is readily absorbed into the body, but in mammals is metabolized to some extent through conversion to tri- and tetrachlorophenols and conjugation with sulfates or glucuronides. Other pathways involve the ultimate formation of mercapturates which are water soluble end-products eliminated via the urine (Smith 1991). Of the isomers, g-HCH is stored to the greatest extent in fat (Smith 1991).

In mammals, the major effects of acute exposure to lindane include central nervous system stimulation, mental and motor impairment, excitation, convulsions, increased respiratory rate or respiratory failure, pulmonary edema, and dermatitis. Effects on the gastrointestinal, musculoskeletal, liver, kidney, and immune systems have also been reported (Smith 1991; Kidd and James 1991). Chronic exposure to lindane has been associated with effects on the blood (decrease in numbers of red and white blood cells); on the musculoskeletal, immune, and nervous systems; and on the liver and kidney (Smith 1991; Matsumura 1985). Reproductive effects such as decreased sperm count may also be possible (Smith 1991). Available data on the mutagenicity and carcinogenicity of lindane are somewhat contradictory (Smith 1991).

### *2.4.21.1. Species Effects of Proposed Lindane Criteria*

The proposed acute criterion for lindane is 2 µg/L. The proposed chronic criterion is 0.08 µg/L (Table 1.3.1).

***Acute Lindane Criterion.*** Johnson and Finley (1980) reported an $LC_{50}$ value of 1.7 µg/L for brown trout, indicating that the acute criterion could allow mortality to salmonids. For most salmonids and other fish species, however, $LC_{50}$ values are more than an order of magnitude greater than the proposed acute criterion of 2 µg/L. Johnson and Finley (1980) reported 96-hour $LC_{50}$ values of 23 µg/L, 27 µg/L, and 32 µg/L, for coho salmon, rainbow trout, and lake trout, respectively, in static exposure tests. Values for other fish species (goldfish, carp, fathead minnow, black bullhead, channel catfish, green sunfish, bluegill, largemouth bass, and yellow perch) range from 32 µg/L to 131 µg/L (Johnson and Finley 1980). Schimmel *et al*. (1977) conducted flow-through, 96-hour bioassays to determine the acute toxicity of technical grade BHC and lindane to sheepshead minnow (*Cyprinodon variegatus*), and pinfish (*Lagodon rhomboides*). The respective 96-hour $LC_{50}$ values were 104 µg/L and 30.6 µg/L. A few studies show sublethal effects after acute exposure to lindane, but at concentrations well above the proposed acute criterion (e.g., Rozados *et al*. 1991; Soengas *et al*. 1997).

Most data determine $LC_{50}$ values above the proposed acute criteria, although the low $LC_{50}$ for brown trout reported by Johnson and Finley (1980) suggests the need for further testing. This is especially true in light of the fact that Johnson and Finley's (1980) values were based on static exposure tests with nominal (unmeasured) lindane concentrations, which could have under- or overestimated toxicity.

***Chronic Lindane Criterion.*** The proposed chronic criterion for lindane is 0.08 µg/L. This was based on acute:chronic ratios calculated from $LC_{50}$ data and whole life cycle tests fathead minnow, and did not incorporate data on chronic toxicity of lindane to salmonids (EPA 1980q). Few chronic toxicity data are available for salmonids exposed to lindane in the water column. Macek *et al*. (1976) exposed brook trout for 261 days to 16.6 µg/L lindane. While survival was not affected, a reduction was observed in fish weight and length. Some disruption in reproductive activity was also recorded during the same experiment (Macek *et al*. 1976). Mendiola *et al*. (1981) determined decreased efficiency of protein utilization in rainbow trout exposed to lindane at concentrations of 1 µg/L to 10 µg/L for 21 days.

Some additional information is available on the effects of lindane associated with specific measured tissue residues in test fish. For example, in immature brook trout, Macek *et al*. (1976)

found that growth rates were decreased, and observed abnormal spawning behavior in females, when muscle tissue concentrations were 1.2 mg/kg. However, there was no effect on survival.

Other fish species also show effects of lindane at relatively low tissue concentrations. For example, in the gudgeon (*Gobio gobio*) the lowest tissue concentration at which a significant increase in mortality could be observed within 96 hours was 0.19 mg/kg in muscle (Marcelle and Thorne 1983). Similarly, in bluegill, the proposed NOEL for growth and mortality was 0.297 mg/kg (Macek *et al*. 1976). For other fish species, adverse biological effects occur at somewhat higher levels. Macek *et al*. (1976) observed decreased growth and increased mortality of fathead minnow at a concentration of 9.53 mg/kg in the carcass. In pinfish, the dose causing 50% effects ($ED_{50}$) for growth effects was 5.22 mg/kg (Schimmel *et al*. 1976).

Tissue concentrations of lindane in fish exposed to the concentrations of lindane in the water column at the proposed criteria concentration can be calculated from EPA's estimated BCFs for lindane. Multiplying the proposed chronic criterion by the geometric mean of BCF values for lindane (1400; EPA 1980q) and a percent lipid of 15% (default value for freshwater fish) results in an estimated maximum allowable tissue concentration of 1.68 mg/kg lindane. For lower lipid values (5% to 10%) the values would be on the order of 0.56 mg/kg to 1.12 mg/kg. It should be noted that the normalized BCF value is based primarily on data for fathead and sheepshead minnow, not on studies with salmonids, so it may not reflect uptake in the species of concern. Also, because these BCFs were determined in the laboratory, they may underestimate lindane uptake by animals in the field. Assuming that the BCF values are in a reasonable range, it appears that tissue concentrations of lindane associated with biological effects (Macek *et al*. 1976) in salmonids could be relatively close to those predicted based on the proposed chronic criterion (1.68 mg/kg). However, despite this calculations using mean BCFs, the water concentration that actually produced Macek *et al.*'s (1976) tissue residues were far higher than the chronic criterion (16 vs. 0.08 µg/L, above).

Some studies have also been conducted in which lindane was administered through feeding or injection. For example, Dunier *et al*. (1994, 1995) report that lindane modified non-specific immune responses in rainbow trout fed lindane for 30 days at a dose of 1 mg/kg.

Aldegunde *et al*. (1999) observed lower body weights, increased serum cortisol levels and changes in the serotonergic brain activity after 18 days in rainbow trout implanted with 0.005 mg/kg body weight of lindane in coconut oil. These studies suggest the potential for sublethal effects on growth, metabolism, and immune function at tissue concentrations comparable or lower than those associated with the water quality criteria being reviewed.

***Factors affecting the Toxicity of Lindane.*** Water hardness does not seem to alter the toxicity of lindane to fish. In some experiments, increased temperature caused increased toxicity for some species and decreased toxicity for others (Johnson and Finley 1980).

*2.4.21.2. Habitat Effects of Proposed Lindane Criteria*

***Toxicity to Food Organisms.*** Available data on the acute toxicity of lindane to aquatic invertebrates suggest that the proposed criterion of 2.0 µg/L may be protective of most types of salmonid invertebrate prey. Reported 96-hour $LC_{50}$ values are on the order of approximately two to three times the criteria, including 4.5 µg/L for stoneflies (*Pteronarcys*) and 6.3 µg/L for mysids (*Mysidopsis bahia*; Johnson and Finley 1980). For other prey species, such as *Daphnia*, $LC_{50}$ values are substantially higher, e.g., 460 µg/L to1460 µg/L (Ferrando *et al*. 1995), or as high as 20,000 µg/L for rotifers (Janssen *et al.* 1994). For amphipods, reported $LC_{50}$ values have ranged from 5 µg/L to 80 µg/L (*Gammarus pulix*, McLoughlin *et al*. 2000; Abel 1980; Stephenson 1983; Taylor *et al*. 1991; *Gammarus lacutris and G fasciatus*, Sanders 1972; *Hyalella azteca*; Blockwell *et al*. 1998).

Only one study was found that reported effects on aquatic macroinvertebrates at lindane concentrations that were below the chronic criterion; Schulz and Liess (1995) reported reduced emergence of caddisfly larvae after 90 days of exposure to targeted (unmeasured) concentrations of lindane as low as 0.0001 µg/L. However, most studies of the chronic effects of lindane exposure on aquatic invertebrates have reported effects occurring at levels that are more than 25 times the proposed criterion of 0.08 µg/L. For example, for the amphipod, *Hyallela azteca*, Blockwell *et al*. (1998) reported 240-hour $LC_{50}$s of 26.9 µg/L and 9.8 µg/L for adults and neonates, respectively. In the amphipod *Gammarus pulix*, growth was reduced after a 14-day exposure to concentrations between 2.7 µg/L and 6.1 µg/L range (Blockwell *et al*. 1996). Taylor *et al*. (1998) reported alterations in haeme biosynthesis in *Gammarus pulex* after a 240-hour exposure to lindane at 4.5 µg/L. Similarly, in mesocosm experiments involving exposures of 2 to 4 weeks, some zooplankton species, such as copepod and cyclopod nauplii and midge larvae, experienced significant mortality at lindane concentrations in the 2 µg/L to 12 µg/L range (Fliedner and Klein 1996; Peither *et al*. 1996). In contrast, effects were not observed on survival, reproduction and growth of *Daphnia magna* after 21 days of exposure until concentrations were 250 µg/L or higher (Ferrando *et al*. 1995).

***Bioaccumulation.*** Lindane will accumulate slightly in fish and shellfish. Uptake of lindane by aquatic organisms is influenced by a number of environmental and water quality factors, including concentrations of organic particulate matter in the water column, turbidity, pH, and season of the year. Residue concentrations may also vary considerably between fish species. Lindane bioconcentrates to some extent in aquatic organisms such as fish, mollusks, insects, plankton, and algae (ATSDR 1989). Lindane has been found in the fat of fish, mollusks, and other aquatic species at concentrations up to 1400 times the concentration in the surrounding waters (WHO 1991; ATSDR 1989). Bioconcentration factors determined in laboratory studies with fish have ranged from 35 to 486, with the 486 value determined for rainbow trout (EPA 1980q). No BCF values were found for salmon.

***Uptake and Toxicity Through Alternate Routes of Exposure.*** Because there are no registered uses of lindane in the United States, the only sources of this compound will be from repositories of the contaminant that are persistent in sediments. These means that lindane will be taken up not only through the water column, but also through direct contact with sediments or through the diet. Thus, studies evaluating the effects of water-borne exposure alone are likely to under estimate actual exposure of organisms in the field. However, because the value of the octanol/water partitioning coefficient of lindane ($\log_{10}(K_{ow}) = 3.3$) is relatively low in comparison to compounds such as DDTs and PCBs, adsorption and accumulation in sediments is also generally lower.

Because sediments are likely the primary source of lindane, the sediment lindane concentration that would result in lindane concentrations in the water column at or below the proposed criteria can be calculated per Section 2.4.13. For lindane, $\log_{10}(K_{ow}) = 3.3$, $\log_{10}(K_{oc}) = 3.24$, and $F_{cv} = 0.08$, resulting in $SQC_{oc} = 0.14$ mg/kg organic carbon. This would mean that for sediment TOC levels of 1% to 5%, the sediment lindane concentrations would range from about 1 ng/g to 7 ng/g sediment. These values are about an order of magnitude below the sediment screening guideline of 10 ng/g dry wet established by the COE for in-water disposal of dredged sediment (COE 1998), and are approximately at the level of the interim Canadian Sediment Quality Guidelines (SQG) of 0.32 ng/g to 0.99 ng/g dry wt sediment. The higher of these values is a probable effect level, based on spiked sediment toxicity testing and associations between field data and biological effects (CCME 2001). This suggests that the proposed criterion is reasonably likely not to harm salmonids or impact their prey items, although there is some uncertainty since tests used to establish these sediment guidelines were not specific to salmon and their prey.

Data on sediment toxicity of lindane are limited. Most studies suggest that adverse effects to benthic invertebrates that could serve as salmonid prey occur at much higher concentrations. For example, studies show effects on larval growth and adult emergence in chironomids at 2000 ng/g dry wt sediment (Watts and Pascoe 2000). Similarly, Ciarelli *et al*. (1997) reported 10-day $LC_{50}$ values of 780 ng/g to 1490 ng/g dw sediment for the amphipod, *Corophium valutator*.

#### *2.4.21.3. Summary for Lindane*

There are not current registered used of lindane in the United States and no known contamination of sites in Idaho at levels that may impact listed salmonids. Most of the available data tend to show adverse effects to listed salmonid species, or their close relatives, or their prey at greater than criteria concentrations. The reliability of a single acute test reporting mortalities at concentrations lower than the acute criterion is uncertain since targeted exposure concentrations were not verified by chemical analysis (i.e., were nominal concentrations).

### 2.4.22. The Effects of EPA Approval of the Polychlorinated Biphenyl Criterion

Polychlorinated biphenyl (PCBs) were produced by the Monsanto Company and were marketed under the trade name of “Aroclor” using a numbering designation of four digits to identify the different commercial mixtures. For example, "12" was used as the first 2 digits for PCB mixtures and the last two digits identified the percent chlorine by weight of the mixture (e.g., the PCB mixture Aroclor 1254 contains 54% chlorine by weight). Aroclor 1254 is one of the most common PCB mixtures that persists widely as a gobal pollutant. Polychlorinated biphenyls are common in urban waterways and can occur in high concentrations in biota and cause a variety of biological effects. Polychlorinated biphenyl production in the United States was banned by Congress in 1979.

Many biological responses in laboratory animals have been reported for PCBs, including mortality, impaired growth and reproduction, immune dysfunction, hormonal alterations, enzyme induction, neurotoxicity, behavioral responses, disease susceptibility, and mutagenicity. While some biological responses, such as mortality, growth inhibition, and reproductive impairment, have measurable impacts on a population (Forbes and Calow 1999), other endpoints, such as altered hormone levels or induced enzyme systems, also have adverse physiological effects on species, thereby reducing their fitness. For example, thyroid function is associated with many physiological processes in fish metabolism. As noted by Mayer *et al*. (1977), thyroid metabolism plays a role in respiration, carbohydrate and ammonia metabolism, oxygen consumption, nervous system function, and behavior.

Impairment of these vital functions may affect the ability of fish to tolerate normal environmental fluctuations, including the physiologically demanding process of smoltification. A few studies have demonstrated that PCBs affect the thyroid hormones important for smoltification in salmon (Mayer *et al*. 1977, Folmar *et al*. 1982). Several physiological parameters (e.g., ATPase levels in the gill, thyroid and pituitary hormones, liver glycogen, blood glucose, and lipid metabolism) change during the parr to smolt transformation in salmonids (Wedemeyer *et al*. 1980). Alteration of any associated physiological functions may substantially reduce the chances of successful smoltification and the individual's ability to survive, thrive, and mature in the marine environment.

Variation in the PCB mixture is associated with variation in toxicity response, which is likely due to variable congener makeup and interspecies variation in uptake and elimination rates of the different congeners. Mayer *et al*. (1977) tested three fish species exposed to four different Aroclor mixtures and found a large (10- to 100-fold) range in $LC_{50}$ values depending on the period of exposure and species. This observation was somewhat different from that reported by DeFoe *et al*. (1978) who showed similar $LC_{50}$ values for fathead minnows exposed to Aroclors 1248 and 1260, which may be indicative of the range of species-related differences.

### *2.4.22.1. Species Effects of PCB Criterion*

Idaho has defined a chronic AWQC of (0.014 µg/L), but not an acute criterion. A recreational use criteria based on fish consumption criteria is also applicable to all waters in Idaho with anadromous fish and is more than100 times more restrictive than the chronic aquatic life criterion of (0.000045 µg/L).

***Acute PCB Criterion.*** There is no acute criterion for PCBs.

***Chronic PCB Criterion.*** The proposed chronic criterion for PCBs is 0.014 µg/L in freshwater (Table 1.3.1). Data in the AQUIRE database (EPA 2001) and presented in literature reviews (Niimi 1996; Monosson 2000) indicate that water concentrations in the 0.1 µg/L to 10 µg/L range can be associated with sublethal, adverse effects in fish. One of the lowest response concentrations for a salmonid was reported by Mauck *et al*. (1978) who demonstrated that backbone composition of phosphorus and hydroxy-proline was altered significantly in brook trout fry exposed to Aroclor 1254 at a concentration of 0.4 µg/L. A slightly higher concentration (0.69 µg/L) also affected collagen and calcium levels in the backbone of fry. In the case of non-salmonids, a study on reproduction in fathead minnows found that larvae were the most sensitive life stage (DeFoe *et al*. 1978). Additionally, when the second generation of fish were examined, mortality and growth were significantly affected at 0.4 µg/L indicating greater sensitivity for offspring of adult fish subjected to chronic exposure.

***Factors Affecting the Toxicity of PCBs.*** In recent work it has been shown that some PCB congeners are considerably more toxic than others, which is primarily a function of the position of the chlorine atoms and their ability to interact with the aryl hydrocarbon (Ah) receptor. This is more a concern for vertebrates, including fish, than invertebrates which generally lack this receptor and are not sensitive to the "dioxin-like" effects of PCBs. The most toxic PCBs are the non-ortho and mono-ortho substituted congeners, which tend to be planar compounds. Some toxicological responses such as developmental and reproductive abnormalities, enzyme induction, and immunosuppression can occur at extremely low concentrations and are likely caused by "dioxin-like" PCB congeners (planar congeners). These planar congeners can occur in the Aroclor mixtures, but usually at low concentrations. The responses caused by the non-planar congeners ("non-dioxin-like") are likely due to different modes of action and include neurotoxicity, hypothyroidism, carcinogenicity, behavioral alteration, and endocrine disruption (Giesy and Kannan 1998).

The TEF approach has been used to determine the relative toxicity of the planar PCB congeners as a fraction of that elicited by 2,3,7,8 Tetrachlorodibenzo-p-dioxin (TCDD). Tissue concentrations of PCB congeners are multiplied by the TEF to generate a toxicity equivalence quantity (TEQ) concentration in terms of its "dioxin-like" potency. These TEQs are then summed to generate a total TEQ concentration for the sample that can be compared to dioxin toxicity results. Ideally, the TEFs should be species and endpoint-specific because of the observed variability (Giesy and Kannan 1998). The TEF approach is not applicable for those "non-dioxin-like" biological responses caused by the non-planar PCB congeners, primarily due to the different modes of action. The TEF approach is not valid for invertebrates because they generally do not contain the aryl hydrocarbon receptor that would cause dioxin-like toxicity.

Most TEFs have been developed for mammals and birds, and only recently have any been developed for fish (Walker and Peterson 1991). The TEFs for fish are somewhat limited because they apply only to ELS mortality in salmonids and enzyme induction (Giesy and Kannan 1998). There are no TEFs for biological effects occurring beyond the embryo/alevin state. For fish, TEFs have been developed for non-ortho PCBs, but not for the ortho-substituted congeners because of a general lack of biological activity (Giesy and Kannan 1998; Van den Berg *et al*. 1998). Table 2.4.22.2 lists TEFs for fish based on ELS mortality due to injection of congeners into eggs (Van den Berg 1998).

**Table 2.4.22.2. Reported Toxicity Equivalent Factors (TEFs) for the early life stage of fish**

| PCB Conger | IUPAC No. | TEF |
|---|---|---|
| 3,3',4,4' | 77 | 0.0001 |
| 3,4,4',5- | 81 | 0.0005 |
| 3,3',4,4',5- | 126 | 0.005 |
| 3,3',4,4',5,5' | 169 | 0.00005 |

The values in Table 2.4.22.2 are generally one to two orders of magnitude lower than those reported for mammals and birds (Van den Berg *et al*. 1998). Polychlorinated biphenyl congener data are not available for fish tissue samples, especially eggs. Application of TEFs therefore provides less accurate toxicity response information for this life stage. Walker and Peterson (1991) conducted a dose response study with rainbow trout eggs and determined the TCDD $LD_{50}$ to be 0.23 ng/g, with very low mortality occurring at 0.1 ng/g. Based on this work, a TEQ value above 0.1 ng/g egg may therefore not be protective against ELS mortality. It is uncertain if concentrations below 0.1 ng/g in eggs may lead to adverse effects. This approach could be valid for many fish species, although differences may exist between species (Monosson 2000).

This information could be used to assess toxicity when congener-specific toxicity information becomes available for biological responses relevant to salmonid life stages beyond the early life stages. For example, a recent study demonstrated a significant increase in mortality for adult rainbow trout exhibiting a muscle tissue concentration of 2,3,7,8 TCDD of only 0.00044 ng/g ww (Jones *et al*. 2001).

*2.4.22.2. Habitat Effects of PCB Criterion*

***Toxicity to Food Organisms.*** One comprehensive study of PCB toxicity to freshwater invertebrates found responses at relatively low concentrations. Nebeker and Puglisi (1974) examined eight Aroclor mixtures and their effects on survival and reproduction in *Daphnia magna*, *Gammarus pseudolimnaeus* (amphipod), and *Tanytarsus dissimilis* (midge), which are all potential prey for salmonids. The midge was the most sensitive invertebrate studied, with 21-day $LC_{50}$ values at 0.63 µg/L for larvae and 0.45 µg/L for pupae (Aroclor 1254). Data contained in the EPA's AQUIRE database for toxic effects of PCBs on aquatic organisms indicate that invertebrates are affected by water concentrations of PCBs in the 0.5 µg/L to 5 µg/L range, which is at least an order of magnitude above the chronic AWQC.

***Bioaccumulation.*** With very high BCFs, it takes only a few µg/L in water to cause tissue concentrations of PCB in the range considered lethal. In addition, many studies have demonstrated that salmonids absorb about 50% of PCBs available in their diet. Madenjian *et al*. (1999) reported the efficiency of retention by coho salmon through dietary uptake of various PCB congeners ranged from 38% to 56%. Similar results were also reported by Gruger *et al*. (1975, 1976) for coho salmon and by Opperhuizen and Schrap (1988) for guppies and other fish species. In a long-term study with rainbow trout, Lieb *et al*. (1974) fed trout PCB-laden pellets for 32 weeks. Fish grew from 0.8 grams to approximately 75 grams and the percent retention of

PCBs was determined to be 68%. The authors also determined that the ratio between the ww PCB concentration in fish and the PCB concentration in dry food was 0.54.

The BCFs, which indicate the relative partitioning between water and tissue, are governed by the balance between the rates of uptake and elimination and can be altered by changes in either rate. Mackay *et al*. (1992) reported an average $\log_{10}$ (BCF) value equal to 4.9 for fish exposed to PCBs (Aroclor 1254). However, because of the large variability in congener hydrophobicity, BCF values for fish range almost four orders of magnitude. The determination of tissue residues from water exposure is consequently extremely uncertain because of the large variability in BCFs for PCB congeners, and there is no one BCF suitable for Aroclor mixtures (Bremle *et al*. 1995).

For example, Berlin *et al*. (1981) reported significantly more mortality in lake trout fry exposed to Aroclor 1254 when tissue concentrations were 1.5 ppm wet weight. Folmar *et al*. (1982) found altered thyroid hormones in coho salmon exposed to 0.1 mg/L of Aroclor 1254 in tissue, which would influence the smoltification process and a smolt's ability to osmoregulate in marine waters. Another study with coho salmon also found effects on thyroid activity, as determined by uptake of iodine, when the whole-body tissue concentration of Aroclor 1254 reached 0.6 mg/kg ww (Mayer *et al*. 1977). If the average BCF noted above (i.e., $\log_{10}$ (BCF) = 4.9) were applied to the data reported in Mayer *et al*. (1977), a water concentration of 0.007 µg/L would be estimated to produce a tissue concentration of 0.6 mg/kg. This water concentration is half the proposed chronic criterion.

The BAF may be modified as a BSAF to include lipid-normalized tissue and organic carbon normalized sediment concentrations with the following equation:

$$\text{BSAF} = [\text{tissue}]/[\text{sediment}] \times f_{oc}/f_{lip}$$

where:

> [tissue] and [sediment] are respective concentrations
> $f_{oc}$ is the fraction of organic carbon (g/g)
> $f_{lip}$ is the fraction of lipid (g/g) (Meador 2006)

Equilibrium partitioning theory was developed to explain and predict the partitioning behavior between sediment, water, and tissue for neutral hydrophobic organic compounds (HOCs), such as PCBs and polycyclic aromatic hydrocarbons (PAHs; McFarland 1984; Di Toro *et al*. 1991). At equilibrium, the BSAF, which represents partitioning between these phases, has a theoretical maximum value of unity (1.0), whereas empirical maximum values range from 2 to 4 (Di Toro *et al*. 1991; EPA/USACOE 1991; Boese *et al*. 1995). For invertebrates and some fish, especially those associated with sediment, the PCB BSAF values are generally close to expected values (2 to 4; Bierman 1990; Tracey and Hansen 1996). The BSAF values for fish can also be close to expected values depending on the exposure time and variability in exposure concentration (Bierman 1990). However, few studies have reported PCB BSAFs for salmon because, for such a highly migratory species, it is exceedingly difficult to determine a relevant exposure concentration. The BSAFs were estimated for migrating juvenile chinook in the Duwamish estuary system where individuals spend days to weeks feeding on abundant invertebrate populations (Meador *et al*. 2002). The BSAFs over years, location, and natural-origin versus hatchery fish were determined to be relatively consistent, ranging 0.10 to 0.16.

***Uptake and Toxicity from Alternate Routes of Exposure.*** Polychlorinated biphenyls are typically not found in the water column at concentrations of concern because of their high affinity for sediment and biological tissues. It is possible for high sediment concentrations, and water concentrations that are below the chronic criterion or are undetectable, to co-occur in streams. Water quality criteria in such areas thus may have little relevance for assessing impacts to organisms that can accumulate high concentrations of these compounds from their diet. It is more relevant to assess impacts to biota based on tissue or sediment concentrations. Only a few studies have examined this approach for protecting aquatic life based on sediment concentrations, and fewer for tissue concentrations.

There are a number of empirical methods for assessing effects of sediment-associated contaminants and generating SQG (MacDonald *et al*. 2000b). One such method is the "Effects Range" approach, which ranks sediment concentrations associated with adverse effects observed in bioassays. Using a large database of bioassay experiments, the concentration associated with the 10$^{th}$ percentile of all studies is termed the "Effects Range-low". The 50$^{th}$ percentile is called the "Effects Range-median" (Long *et al*. 1995). These values are often used to determine the potential for a sediment concentration to cause adverse effects in biota. MacDonald *et al*. (2000b) have recently reviewed all such approaches and proposed unifying them into a consensus-sediment effect concentration (SEC), using total PCBs as an example. They proposed dividing SECs into three groups, the threshold, midrange, and extreme effect concentrations (TEC, MEC, and EEC, respectively). The TECs are concentrations below which adverse effects on sediment dwelling organisms are not expected, MECs are concentrations above which adverse effects are frequently observed, and EECs are concentrations above which adverse effects are usually or always observed. For freshwater ecosystems the following SECs were generated by MacDonald *et al*. (2000b) for total PCBs in sediment: TEC = 35 µg/kg, MEC = 340 µg/kg, and EEC = 1,600 µg/kg /kg (all in dry weights). Because these values are based on hundreds of bioassay experiments, they should be useful in identifying sediment concentrations in Idaho that may cause adverse effects in sediment-dwelling organisms exposed to total PCBs. In sediment surveys in various locations in the Snake River basin in Idaho, PCBs were less than 50 µg/kg (Table 2.3.1, Clark and Maret 1998). This indicates that PCB concentrations in sediment are likely close to, or below the TEC.

Characterizing the toxic effects caused by PCBs can be simplified by examining tissue concentrations associated with adverse effects. Variation in the toxic response can be reduced because of large differences in time of exposure, makeup of Aroclor mixtures, and differences in toxicokinetic abilities of species. Niimi (1996) and Monosson (2000) provide summaries showing the range in tissue concentrations associated with several different biological responses Niimi (1996) reported that fish tissue concentrations of PCBs greater than 50 mg/kg were associated with adverse effects to growth and reproduction. Monosson (2000) focused on reproductive and developmental effects of Aroclor 1254 and associated tissue concentrations, and determined that concentrations associated with adverse effects ranged from 5 ppm in whole bodies of larvae to 25 mg/kg in liver of adult fish (all wet weights). Additional analyses with congener 77 indicated an effective concentration of 0.3 mg/kg in eggs. In a critical review of the literature, Meador *et al*. (2002) examined the toxic effects of total PCBs in salmonids and determined that the $10^{th}$ percentile value of 15 studies considered valid in the determination of a residue effect threshold for salmonids was 2.4 mg/kg lipid. Tissue residues below this were considered to be generally protective of salmonids.

#### *2.4.22.3. Summary for PCBs*

In the studies reviewed (above) water borne PCB concentrations close to, or below, the proposed chronic criterion, in concert with predicted bioaccumulation rates, were projected to result in impaired thyroid function in coho salmon and embryo mortality in lake trout. Even though the proposed chronic criterion may result in some effects to listed species, this appears unlikely to occur because the product is banned and no known contamination exists at levels of concern in Idaho areas with listed salmon and steelhead. If discharges do occur the most stringent controlling ambient water quality criterion applicable in designated critical habitats is the fish consumption based human-health criteria, rather than the chronic aquatic life criteria (Table 1.3.1). The fish consumption based criterion is more than100 times more restrictive than the aquatic life criteria. Therefore any effects from the proposed approval of the PCB criterion will have only very small effects on listed species and designated critical habitat.

### 2.4.23. The Effects of EPA Approval of the Toxaphene Criteria

Toxaphene is a trade name for a man-made organochlorine pesticide that consists of between approximately 177 and 670 congener compounds and has a chlorine content of 67% to 69%. Toxaphene is produced by combining camphene (a pine tree extract) with chlorine, and activating the mixture with ultraviolet radiation and catalysts. Only 26 congeners have been isolated, of which 10 have been identified. The 26 isolated congeners comprise approximately 40% of the toxaphene mixture. Toxaphene is also known as chlorinated camphene and was listed under other trade names including Alltex, Estonox, Motox, Anatox, Penphene, and Geniphene. Toxaphene was first introduced in 1947 and used extensively as an insecticide in the 1970s after DDT was banned. The pesticide was used primarily in the southern United States to control insects on cotton and livestock, and to kill undesirable fish in lakes. Toxaphene was banned for most uses in 1982 and all uses in 1990 in the United States, but is still used on fruit crops in other countries.

Toxaphene exhibits a relatively low $\log_{10}$ octanol-water partition coefficient at 3.3, but is very persistent in the environment, with a reported half-life in soil between 1 and 14 years. In water it will not appreciably hydrolyze, undergo photolysis, or biodegrade. Degradation is faster under anaerobic than aerobic conditions. Evaporation from the aqueous phase is a significant process for toxaphene dispersion, with a half-life of approximately 6 hours. Once it has volatilized, toxaphene can be carried far from the original site.

The EPA has determined that exposure of animals to toxaphene potentially affects the central nervous system, and that chronic exposure also has the potential to affect the liver and kidney, suppress the immune system, and cause cancer and endocrine disruption. There are reports that it may also have antiestrogenic activity and inhibit the binding of estrogen, progesterone, dexamethasone, and testosterone to their respective receptors (Yang and Chen 1999; ; Hood *et al*. 2000).

*2.4.23.1. Species Effects of Toxaphene Criteria*

The proposed acute criterion for toxaphene is 0.73 µg/L in freshwater, and the chronic criterion is 0.0002 µg/L.

***Acute Toxaphene Criterion***. The BA provided by EPA only reported acute toxicity studies on fish that showed effects at concentrations above the proposed acute criterion. Acute effects reported by EPA occurred at 2 µg/L in bass, 2.4 to 29 µg/L in bluegill, 3.1 µg/L in brown trout, and 18.0 µg/L in fathead minnows. Studies not reported in the BA suggest that fish mortality may occur at toxaphene concentrations that are relatively close to the proposed acute criterion, due to the proximity of the $LC_{50}$ values to the proposed criterion. Schimmel *et al*. (1977) reported 96-hour $LC_{50}$s of 1.1 µg/L and 0.5 µg/L for the sheepshead minnow (*Cyprinodon variegatus*) and pinfish (*Lagodon rhomboides*), respectively. Macek and McAllister (1970) listed 96-hour $LC_{50}$s for 12 fish species from five families exposed to toxaphene, ranging from 2 µg/L to 14 µg/L. Two salmonids were tested, the brown trout and the coho salmon, with $LC_{50}$ values (and 95% confidence intervals) of 3 µg/L (2 µg/L to 5 µg/L) and 8 µg/L (6 µg/L to 10 µg/L), respectively. Johnson and Finley (1980) reported 96-hour $LC_{50}$s for toxaphene to coho salmon, rainbow trout, and brown trout of 8.0 µg/L, 10.6 µg/L, and 3.1 µg/L, respectively. Schoettger (1970) reported a 96-hour $LC_{50}$ for Chinook salmon of 1.5 µg/L, which is within a factor or two of the proposed acute criterion. These results are very similar to those reported by Katz (1961) for rainbow trout, chinook and coho salmon.

***Chronic Toxaphene Criterion.*** No studies were found that documented chronic effects when toxaphene concentrations were below the chronic criterion of 0.0002 µg/L. Mayer and Mehrle (1977) and Mayer *et al*. (1975) reported that water concentrations of 0.039 µg/L had significant effects on survival and growth in brook trout fry. The tissue concentrations associated with these responses were only 0.4 mg/kg dw. Other treatments in these studies (0.068 µg/L, 0.14 µg/L, 0.29 µg/L, and 0.5 µg/L) also caused adverse effects in this species. Tissue concentrations for these treatments ranged from 0.2 mg/kg to 8 mg/kg dw. Similar studies by these authors also found adverse effects in fathead minnow at similar water exposure concentrations (Mayer *et al*. 1977; Mayer and Mehrle 1977). An examination of the AQUIRE database on sublethal effects from toxaphene exposure indicated several other studies showing sublethal effects on fish in the 0.03 µg/L to 1 µg/L range (EPA 2001).

Sublethal effects reported by EPA in the BA suggest that reduced reproduction, growth inhibition and histopathology of the kidney and intestinal tract could occur from acute exposure, based on occurrence of these effects in fish at concentrations as low as 0.054 µg/L. Growth inhibition and reduced reproduction were reported in brook trout exposed for 161 days to 0.288 µg/L and 0.068 µg/L, respectively (Mayer *et al.* 1975, 1977a).

***Factors Affecting Toxicity.*** Like PCBs, toxaphene is made up of a large number of congeners, which may vary in toxicity and mode of action, and whose biological effects are not well-characterized. Stelzer and Chan (1999) found differences in the estrogenic activity of a technical toxaphene mixture compared with two congeners that are prominent in humans. Toxaphene may also show interactive effects with other pesticides or environmental contaminants (Chaturvedi 1993).

### *2.4.23.2. Habitat Effects of Toxaphene Criteria*

***Toxicity to Food Organisms.*** Based on available literature, it appears that invertebrates are less sensitive to toxaphene exposure than fish. Results from the AQUIRE database (EPA 2001) show $LC_{50}$s for freshwater invertebrates ranging from 5 µg/L to 20 µg/L, which is similar to values for saltwater studies in which oysters and shrimp exhibited 96-hour $LC_{50}$s ranging from 1.4 µg/L to 16 µg/L (Schimmel *et al*. 1977). However, one study with stonefly naiads reported $LC_{50}$ values in the low µg/L range. The 96-hour $LC_{50}$ for *Pteronarcys californica*, *P. badia*, and *Claassenia sabulosa* were 2.3 µg/L, 3.0 µg/L, and 1.3 µg/L, respectively (Sanders and Cope 1968). These values are within a factor of 2 or 3 of the proposed acute criterion, and even closer when the 95% confidence intervals are considered. This study also reported data for the 24- and 48-hour $LC_{50}$s and found that these values generally decreased by a factor of two for each time point (e.g., 24-hour, 48-hour, and 96-hour $LC_{50}$s), indicating that steady state had not been reached and that lower $LC_{50}$ values would likely occur with more exposure time. In general, sublethal effects occur at much lower concentrations than those causing mortality. Sanders (1980) reported adverse effects to growth of the amphipod (*Gammarus pseudolimnaeus*) in exposure concentrations of 0.2 µg/L, and a 96-hour $LC_{50}$ of 24 µg/L. Sanders (1980) also reported a reduction in reproduction in *Daphnia magna* at 0.12 µg/L. These chronic values are orders of magnitude higher than the chronic criterion (0.0002 µg/L), indicating that no chronic effects from long-term exposure to toxaphene are expected for invertebrate prey. Toxaphene may cause endocrine-disrupting effects in invertebrate prey species (Hood *et al*. 2000), but the exposure levels associated with these effects have not been quantified. However, based on the acute response data and the small differences between 96-hour $LC_{50}$s and the CMC value, adverse effects to invertebrates are possible for short-term exposures similar to the CMC.

***Bioaccumulation.*** Bioconcentration factors are very high for toxaphene. Mayer *et al*. (1975) reported BCFs for brook trout ranging from 5,000 to 76,000, and Terriere *et al*. (1966) determined BCFs for rainbow trout to range from 10,000 to 20,000. The AQUIRE database reported BCFs for Atlantic salmon ranging from 4,400 to 11,000 (EPA 2001). Environment Canada (1997) summarized several studies listing BCFs from 3,000 to 76,000 for fish, 400 to 1,200 for some crustaceans, and 7,000 to 10,000 for other groups such as algae and snails. Schimmel *et al*. (1977) also reported BCFs up to 60,000 for juvenile killifish *Fundulus similis*, and a range of 3,100 to 20,600 for other fish.

Toxaphene is biomagnified up the food web by several species (Eisler 2000) and it has been demonstrated in several studies that tissue concentrations increase with trophic level. Evans *et al*. (1991) reported a biomagnification factor of five from plankton to fish.

*2.1.23.3. Summary for Toxaphene*

Based on available information, toxaphene, under most circumstances, appears unlikely to cause lethal or sublethal effects from direct exposure at toxaphene concentrations in water equal to or below the proposed acute or chronic criteria.

## 2.5. Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act.

According to the most recent census, between 2000 and 2010, the cumulative population in the nine central Idaho counties with anadromous fish increased by 5.8%.[7] NMFS therefore assumes that future private and state actions will continue within the action area with a slight increase from their current rate. Seventy-one percent of the action area is Federally-owned, which somewhat limits possible cumulative effects from private and state actions. However, private land is often clustered in valley bottoms, adjacent to occupied habitat for ESA-listed species.

NMFS is aware of several potential future state and private actions in the action area that may benefit ESA-listed species. The *Draft Recovery Plan for Idaho Snake River Spring/Summer Chinook and Steelhead* recommends habitat restoration projects on private lands throughout the action area. The current draft is posted online at *http://www.westcoast.fisheries.noaa.gov/protected_species/salmon_steelhead/recovery_planning_and_implementation/snake_river/snake_river_salmon_recovery_subdomain.html*. Idaho Department of Lands is working with NMFS to develop a proposed Idaho Forestry Program, which aims to reduce the impacts of state and private forestry on stream habitat through road maintenance and stream buffer measures. The state of Idaho is also working with NMFS and irrigators on measures to reduce the impacts of water withdrawals on stream habitat in watersheds in the Salmon River.

It is reasonable to assume that future mining or municipal development will occur on state, private or tribal lands within the action area that may result in discharges of arsenic, copper, cyanide, mercury, nickel, selenium, silver, chromium III, chromium VI, lead, and zinc to waters of the state that contain listed species. However, many of these activities will be subject to section 7 consultation and are therefore not considered cumulative effects. Additionally, cleanup and closure activities for contaminated sites may also occur in the future and some of these will be on private and state lands and are considered cumulative effects.

Cyanide discharges may also occur as a result of future activities on state, private or tribal lands within the action area from activities like road salting.

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[7] U.S. Census Bureau, State and County Quickfacts. Available http://quickfacts.census.gov/qfd/states/16000.html.

Mercury discharges may also occur as a result of future atmospheric deposition.

Pentachlorophenol discharges may occur when treated wood is used in or near water for construction activities or when it is used as a restricted use pesticide for activities on private, state or tribal lands.

Continued agriculture and forestry activities on private land are also likely to occur in the future. This will result in continued use of pesticide and fertilizers. It will also result in continued water diversions for agriculture that reduce flow rates and alter habitat throughout freshwater systems. The above non-Federal actions are likely to pose continuous unquantifiable negative effects on listed species addressed in this Opinion. These effects include increases in sedimentation, increased point and non-point pollution discharges, and decreases in summer low flows.

Non-Federal actions likely to occur in or near surface waters in the action area may also have beneficial effects on listed species addressed in this Opinion. They include implementation of riparian improvement measures and fish habitat restoration projects, for example. Coupled with EPA's approval of the proposed water quality standards for aquatic life, the effects from anthropogenic growth on the natural environment will continue to allow toxic discharges to affect and influence the overall distribution, survival, and recovery of listed species in the Columbia River Basin.

NMFS also expects the natural phenomena in the action area (*e.g.*, ongoing and future climate change, storms, natural mortality) will continue to influence listed species. Climate change effects are expected to be evident as alterations of water yield, peak flows, and stream temperature. Other effects, such as increased vulnerability to catastrophic wildfires, may occur as climate change alters the structure and distribution of forest and aquatic systems.

Although these factors are ongoing to some extent and likely to continue in the future, past occurrence is not a guarantee of a continuing level of activity. That will depend on whether there are economic, administrative, and legal impediments or safeguards in place. Therefore, although NMFS finds it likely that the cumulative effects of these activities will have adverse effects commensurate with or greater than those of similar past activities; it is not possible to quantify these effects.

## 2.6. Integration and Synthesis

The Integration and Synthesis section is the final step of NMFS' assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (Section 2.4) to the environmental baseline (Section 2.3) and the cumulative effects (Section 2.5) to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) Result in appreciable reductions in the likelihood of both survival and recovery of the species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and critical habitat (Section 2.2).

## Hardness Floor

Exposure of listed Snake River salmon and steelhead to levels of metals in discharges at proposed criteria levels will result in adverse effects. Many of the streams in the Salmon River and Clearwater River drainages of Idaho also have hardness concentrations that average less than 25 mg/L which is the current floor in the hardness equation. For copper and lead, hardness is less important than DOC, but if DOC is low, toxicity does increase below the hardness floor. For nickel, and zinc, acute toxicity to fish rises as hardness declines below the 25 mg/L. For silver, acute toxicity increases modestly in early life stages, below the hardness floor.

The use of a hardness floor of 25 mg/l in calculating metals discharge limits will allow for increased exposures of listed fish to levels of metals that result in adverse effects. These effects range from a direct increase in mortality to decreases in growth and survival of juvenile Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River Sockeye salmon and Snake River Basin steelhead.

It reasonable to assume that listed Snake River spring/summer Chinook salmon and Snake River Basin steelhead will be exposed to levels of metals that are harmful to fish based on exposures to metals that are currently occurring in the action area. These exposures are also described in more detail in the sections that follow for each metal. However, is not possible to estimate within the ESU the number of locations where future metals discharges will overlap with areas that also have low water hardness values. Some examples of current discharges that meet both criteria are shown in Table 2.4.2.1. Two of these discharges are into the mainstem of Panther Creek and Yankee Fork of the Salmon River and have the potential to affect nearly all of the fish that occupy the population due to their location low in the watershed. It is reasonable to assume that future discharges may be located similarly in these areas or in a location that affects a different population in a similar fashion. Because of this, it is reasonable to assume that a large percentage of at least one population of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon or Snake River Basin steelhead within their respective ESUs will be exposed to levels of metal toxicity in early life phases which may reduce egg survival in redds or reduce growth and survival of smolts in the exposed population.

## Arsenic

Arsenic occurs in waters in Idaho both naturally and as a discharged pollutant. Arsenic is likely to be discharged in the future through mining or municipal sources so exposure to listed fish and critical habitat is likely to occur.

If only direct water exposures were considered, arsenic would be of minimal concern to listed salmonids at typical ambient concentrations or at the criteria concentrations under review. The risk of harm from short-term water-only exposures to arsenic concentrations at the acute criterion is unlikely enough to be considered a minor risk for short-term exposures.

The chronic criterion appears to avoid chronic adverse effects to the adult and juvenile salmonid life stages from water-only exposures; however, arsenic concentrations below the chronic criterion have been reported to cause mortality in salmonid embryos. The chronic arsenic criterion is far higher than concentrations of arsenic sufficient to bioaccumulate in invertebrates to concentrations that cause harm to the salmonids that feed on them. Bioaccumulation of arsenic in prey organisms to concentrations that could be harmful to salmonids has occurred in streams at exposures less than 10 µg/L. As such, adverse effects are likely to occur at the chronic criterion, through reduced growth of juveniles via food web transfer.

It is reasonable to assume that listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead will be exposed to levels of arsenic that are harmful to fish based on the possibility of future mining or municipal activities in the state. However, is not possible to estimate within the ESU the number of locations where future arsenic discharges and exposure may occur. Most likely these locations will be associated with a mine or a municipal discharge. It is also likely that one or more these discharges may be located within the area used by the majority of the fish in a single population, for example Panther Creek is discussed in the analysis and it had multiple discharges into river sections that are used by the entire Panther Creek population of Chinook at some point in their life cycle. Because of this it is reasonable to assume that a large percentage of at least one population of listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead will be exposed to levels of arsenic approaching the chronic criteria during early life phases which will reduce egg survival in redds and reduce growth and survival of smolts in the exposed population.

Copper

Sources of copper such as mines, municipalities and stormwater runoff from highways exist in the action area currently and will likely be present in the future. It is also likely that copper will be found in new discharges or will be present in water bodies related to past activities.

The results of this analysis suggest that concentrations below the proposed acute and chronic criteria for copper can cause acute and chronic toxicity to salmon and steelhead. At the lower range of hardness values encountered in Idaho streams and lakes the acute standard could result in injury and death.

Listed salmon and steelhead can experience a variety of adverse effects at or below the chronic Idaho copper criterion. These include:

- Deprivation of chemosensory function which in turn causes maladaptive behaviors including the loss of ability to avoid copper, and the loss of ability to detect chemical alarm signals. Appreciable adverse effects can be expected with increases as small as 0.6 µg/L above background concentrations.

- Reduced growth in juvenile Chinook salmon and rainbow trout under conditions of low hardness and low organic carbon.

- Because survival of juvenile salmon and steelhead in their migration to sea is strongly size-dependent, small reductions in size will result in disproportionately larger reductions in survival during migration to sea. Using population modeling, growth reductions at the chronic copper criterion were projected to result in slight increases in extinction risk and pronounced delays in recovery time in a model Chinook salmon population.

- The diversity and abundance of the macroinvertebrate food base for rearing juvenile salmon and steelhead could be reduced at copper concentrations near or below the Idaho chronic criterion.

While a variety of adverse effects relevant to listed salmonids have been demonstrated at copper concentrations less than the copper criteria under consultation, the most important issue is that the hardness-toxicity equation embedded into the criteria commonly results in fundamentally inaccurate and misleading indications of risk in critical habitats. This is because the best available science indicates that organic carbon is a more important mediator of copper risks than water hardness. During late summer or fall base flow conditions, copper would be expected to be most toxic because organic carbon tends to be low. Yet this is the time of year that hardness tends to be highest, and the hardness-based copper criteria wrongly indicate that copper would be of least risk at this time of year (Conclusion Section; Appendix C).

It reasonable to assume that listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead will be exposed to levels of copper that are harmful to fish based on the possibility of future mining or municipal activities in the state. However, is not possible to estimate within the ESU the number of locations where future copper discharges and exposure may occur. Most likely these locations will be associated with a mine or a municipal discharge. It is also likely that one or more of these discharges may be located within the area used by the majority of the fish in a single population, for example Panther Creek is discussed in the analysis and it had multiple discharges into river sections that are used by the entire Panther Creek population of Chinook at some point in their life cycle. Because of this it is reasonable to assume that a large percentage of at least one population of Snake River spring/summer Chinook, Snake River fall Chinook salmon, Snake River sockeye salmon or Snake River Basin steelhead within their respective ESUs will be exposed to levels of copper approaching the acute or chronic criteria during early life phases which will reduce growth and survival of smolts in the exposed population.

Cyanide

It is likely that cyanide will be found in new discharges or will already be present in water bodies. Potential sources such as mines and forest fires exist in the action area currently and will be present in the future.

The proposed acute and chronic criteria can expose listed salmonids to harmful cyanide concentrations under specific situations. The acute criterion is not reliably protective when water temperatures drop to about 6°C or lower. Further, Leduc (1984) found that cyanide concentrations at the chronic criterion in water colder than 6°C may be associated with chronic toxicity effects. Temperatures in streams within the action area routinely drop below 6°C.

It reasonable to assume that listed Snake River spring/summer Chinook salmon and Snake River Basin steelhead will be exposed to levels of cyanide that are at or near the proposed standard.

However, it is not possible to estimate the number of locations where future cyanide discharges and exposure may occur. Most likely these locations will be associated with mining activities but other sources may also occur. It is also likely that one or more these discharges may be located within the area used by the majority of the fish in a single population; for example, Jordan Creek is discussed in the analysis and it had multiple discharges into river sections that were used by the entire Yankee Fork population of spring/summer Chinook at some point in their life cycle. Because of this it is reasonable to assume that a large percentage of at least one population of Snake River spring/summer Chinook or Snake River Basin steelhead within their respective ESUs will be exposed to levels of cyanide approaching the chronic criteria during early life phases which will reduce survival or juveniles as they overwinter and this will reduce abundance of smolts in the exposed population.

In separate reviews, USFWS (2010) and NMFS (2010b) evaluated the same cyanide criteria from a national perspective. Both described scenarios in which impaired reproduction from diverse species was extrapolated to effects on listed anadromous salmonids, through the use of interspecies correlation estimates of acute toxicity. Under these scenarios, adverse effects were considered by USFWS and NMFS as likely to jeopardize the continued existence of a variety of species, including Snake River salmon and steelhead. The findings and conclusions in the earlier draft biological opinions are similar to those reached here.

Mercury

Mercury toxicity in fish occurs by bioaccumulation through the food web. Direct toxicity from exposure to mercury in the water column is unlikely in the natural environment.

The chronic mercury criterion in the proposed action is based upon EPA's 1984 chronic criterion value (EPA 1985g). The 1984 chronic mercury criterion was back calculated from the FDA limit for allowable mercury content in commercially marketed seafood (1.0 mg/kg ww), using a bioconcentration factor derived from a laboratory water-only (aquaria) methylmercury exposures with fathead minnow (EPA 1985g). Thus, the criterion derivation had no consideration of ecological effects of mercury or effects of mercury to sensitive species. In the 25 plus years since this fish marketability-based criterion was developed, much new information on the effects of mercury on the fish themselves, not just their marketability, has been developed. The newer information both reflects that: (1) The older bioconcentration values considered in the 1984 chronic criterion were about four times lower than the average bioaccumulation factors obtained in field settings; and (2) that adverse developmental effects in fish occur at <1 mg/kg.

Severe adverse effects have been observed in fish that accumulated mercury in their muscle tissue, including brain damage, behavioral abnormalities, and reproductive failure. However, effects of methylmercury on fish are not limited to neurotoxicity, but also include histological changes in the spleen, kidney, liver and gonads. These effects have been observed in multiple species of freshwater fish at tissue concentrations of methylmercury well below 1.0 mg/kg ww (Sandheinrich and Wiener 2010).

The EPA has developed a fish-tissue based water quality criterion of 0.3 mg/kg for mercury to reduce human risks of eating mercury-tainted fish. Idaho has adopted this criterion, and is implementing it as a 0.24 mg/kg a triggering residue concentration for existing dischargers, using an uncertainty (safety factor) of 0.8 times (IDEQ 2007a). This fish tissue-based criterion is unlikely to result in adverse effects to listed salmon and steelhead and their habitats. However, if mercury concentrations in rivers or lakes were allowed to approach the chronic water-based criteria of 12 ng/L, resulting mercury residues in fish could be about an order of magnitude higher than the selected threshold (~3 mg/kg ww).

It is reasonable to assume that listed Snake River salmon and steelhead will be exposed to levels of mercury that are harmful based on fish tissue information collected from other fish species within the state. Most likely these locations will be associated with mining or atmospheric deposition. Because of this it is reasonable to assume that a large percentage of at least one population within the ESU will be exposed to levels of mercury that will bioaccumulate and cause severe adverse effects including neurotoxicity and histological changes resulting in reduced abundance and productivity.

Nickel

Although nickel is rarely found in Idaho waters it does occur in some streams large enough to contain listed salmon and steelhead like the Panther Creek watershed. Therefore it is reasonable to assume that it has the potential to occur in other areas of the state where mining activities are likely to occur.

A striking feature of the information reviewed for nickel toxicity is the tremendous range of effects concentrations. Much work, particularly short-term exposures, has shown adverse effects from nickel at concentrations in the milligrams per liter range, which are hundreds or even thousands of times higher than environmentally relevant concentrations. Yet other work has shown nickel to be about as toxic or more toxic, in long-term exposures than metals more commonly considered to pose a risk to sensitive organisms, such as copper or cadmium. No reports were located of adverse effects from short-term (96-hr) toxicity tests using salmonids at concentrations below the final acute value (two times the acute criterion) for nickel.

During this consultation, EPA revised the proposed chronic criterion for nickel resulting in a level that is considerably more protective of listed salmon and steelhead. Potential adverse effects from exposure to nickel at concentrations at or below the criterion in the revised action are expected to be primarily to sensitive invertebrates which may be a food source for listed species. This affect is expected to be small.

Selenium

The acute criteria for selenium is of little relevance because selenium in the water column is not expected to affect listed salmon and steelhead directly through ventilation. The primary concern with selenium is build up in the muscle tissues as trophic transfer from prey species.

If water concentrations were near the chronic selenium criterion of 5 µg/L indefinitely, selenium would likely be transferred through the food web resulting in selenium concentrations in juvenile salmonids greater than twice as high as a concentration estimated to be low risk for appreciable effects in juvenile salmon or steelhead (~7.6 mg/kg dw in whole bodies). Fish tissue residues resulting from stream food web transfer from a constant water concentration of about 5 µg/L were projected to exceed about 19.5 mg/kg dw in juvenile salmonids. This selenium tissue burden would be projected to result in growth reductions and increased mortality in juvenile anadromous salmonids, on the order of about a 50% reduction in weight, a 10% reduction in length, and about a 25% reduction in survival. Lesser reductions in growth (e.g., a 7.5% reduction) were projected to appreciably increase extinction risks and delay recovery in a modeled Chinook salmon population (Mebane and Arthaud 2010). While their modeling was specific to a Snake River spring/summer Chinook salmon populations from the upper Salmon River, NMFS assumes that the relations between size and survival during downstream migration would also hold for steelhead and sockeye salmon,

It is reasonable to assume that listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead will be exposed to levels of selenium that are harmful based future mining activities. However, it is not possible to estimate within the ESU the number of locations where future selenium discharges or exposure may occur. Most likely these locations will be associated with a mine or highly mineralized areas. Because of this it is reasonable to assume that a large percentage of at least one population within the ESU will be exposed to levels of selenium approaching the chronic standard during some phase of life which could result in mortality primarily due to reduced growth and survival of juveniles. This could significantly reduce the abundance and productivity of that population of salmon or steelhead which will prevent the ESU from achieving recovery.

Silver

No concentrations of silver at or near the acute or chronic criteria have been identified in Idaho, even in areas where silver mining occurred for extended periods of time and significant environmental damage was caused by other substances related to the mining activities.

In natural waters silver is likely much less toxic than in most published laboratory experiments because of the strong modifying influence of naturally occurring ligands in ambient waters. Because of this, it appears unlikely that acute toxicity to salmonids at criterion concentrations will occur.

Unlike other criteria considered in this Opinion that had two part values to protect against short term and longer exposures, for silver only a short-term (acute) criterion is proposed. However, adverse chronic effects, including premature hatching, growth inhibition, and chronic mortality, have been observed at in laboratory settings at concentrations below the proposed single silver criterion. Thus, using a single criterion value that was derived using short-term toxicity data to also protect aquatic life from indefinite exposures may be under-protective. The acute criterion is derived as a function of hardness, which is not supported by more current literature which shows chloride, DOC, and sulfide to be more important factors in mitigating silver toxicity. The potential inadequacies and underprotectiveness of the silver criterion are mitigated by the fact that in the environment, silver occurs in a less toxic form than that used in most of the toxicity tests published in the literature. Significant food chain biomagnification by fish is also possible, but all of these effects appear unlikely to occur because of the low silver concentrations typically encountered in the aquatic environment.

Zinc

Zinc is primarily an acute toxin to salmonids, hence the acute criterion is of greater environmental relevance than the chronic criteria. A confusing aspect of the literature on zinc toxicity to salmonids is the great disparity in reported effects between studies. Across different studies, $EC_{50}$ values for rainbow trout with zinc at similar test hardnesses varied by an order of magnitude. Said differently, zinc at criteria concentrations has been found to be highly toxic and killed most of the fish exposed (Figure 2.4.10.2), but in other tests, concentrations well in excess of the criteria killed no fish. This disparity may be due to differences in the sensitivity of fish at different sizes as they develop. While it is commonly assumed that the smallest organisms will be most sensitive (e.g., ASTM 1997), this is clearly not always the case with zinc. Instead for salmonids, the likely pattern is that the newly hatched, smallest fish appear resistant to zinc, lose resistance as they grow during the first and second months after hatching, and then regain resistance as the fish become older and larger. This suggests that even though most of the studies reviewed that addressed zinc toxicity to listed Snake River salmon and steelhead did not show adverse effects below criteria values (Figure 2.4.10.1 and 2.4.10.3c) the risk from exposure to zinc may have been underestimated because the studies did not distinguish between sensitive life stages, and did not examine effects to listed steelhead and salmonids at their most vulnerable post-hatch stages.

Adverse effects were found at sub-criteria values in tests conducted at hardnesses less than 25 mg/L, a few other tests at moderately low hardness of 35 mg/L with the most sensitive size fish tested (Figure 2.4.10.2), and multiple tests reported by Hansen *et al*. (2002c) with rainbow trout. The preponderance of the information reviewed indicate that in waters with hardness less than about 25 mg/L as $CaCO_3$ the Idaho zinc criteria would not be sufficiently protective of listed Snake River salmon and steelhead if they were exposed at their most sensitive life stages. If alternatively, the current IDEQ zinc criteria were determined using the actual water hardness, instead of the assumed hardness of 25 mg/L, most of those data indicate that the criteria would then be sufficient to avoid harm in most of the studies reviewed. This would be sufficient to avoid population level effects to Snake River salmon and steelhead.

Pentachlorophenol

Some studies indicate the proposed acute PCP criterion is at the level where some acute toxicity will occur. Other studies showed that $LC_{50}$s for salmonids were well above the proposed acute water quality standard. Most studies of chronic effects reported the onset of adverse effects occurred at least slightly above the chronic criterion, although a single study found reduced growth in sockeye salmon at lower concentrations than the chronic criterion. Rainbow trout exposed to PCP concentrations far below the chronic criterion showed reduced ability to evade predators, and reduced ability to capture prey. Both the chronic and acute criteria will likely have some effect on listed species or their food sources.

Pentachlorophenol is not likely to be a component of NPDES discharges, but may be used in the treatment of wood that finds its way into inwater or overwater structures so the exposure risk, while small, is not discountable.

Chromium III and VI

There are no known concentrations of chromium that approach the proposed standards in water bodies in the action area, and no current discharges of chromium into water bodies in the action area. Because new permits are also unlikely to reach concentrations of chromium where effects to listed species have been identified, adverse effects are unlikely to occur.

Data reviewed by NMFS indicate few direct adverse effects to listed salmonids at concentrations less than the chronic trivalent or hexavalent chromium criteria. Studies on the effects of hexavalent chromium to salmon sperm are contradictory with one test indicating it is toxic at concentrations below the chronic criteria, and a more recent study showing no effects at criteria concentrations. Because the more recent study that showed no effects appeared to use a more relevant exposure duration, it is relied upon in concluding that direct adverse effects of chromium to listed salmonids are unlikely at or below criteria.

The amphipod *Hyalella azteca* suffered adverse effects at a test concentration below the chronic criterion in one study but not in another. Because so few data on long-term effects of chromium to benthic invertebrates are available, this test is interpreted as suggesting adverse effects to food sources are possible. Bioaccumulation of chromium clearly occurs when water concentrations are high, but relevant data are absent regarding the effects to salmonids when water-borne concentrations are below the chronic criterion. Because adverse effects to the species or critical habitat should never reach the scale where take occurs, the effects of the proposed action for chromium are very minor.

Lead

Potential adverse effects from exposure to lead at concentrations at or below the acute or chronic criteria, to listed salmon and steelhead and their critical habitat are likely to be very minor. The only adverse effects of chronic lead exposures at sub-criteria concentrations were to snails and the amphipod *Hyalella azteca*. In most habitats, listed salmonids would not be expected to be dependent on amphipods and snails for food. Listed salmon and steelhead are unlikely to be injured or killed by exposure to lead concentrations that are at or below the proposed acute or chronic criteria. No evidence of direct adverse sublethal effects occurring at concentrations at or below the chronic criterion to salmonids was found.

Aldrin/Dieldrin

***Aldrin.*** The limited information available regarding aldrin toxicity to salmonids indicates that 50% mortality can occur when concentrations are below or slightly above the acute criterion. Similarly, there is evidence that aldrin is toxic to some salmonid prey species when concentrations are below or close to the criterion. This information suggests that the proposed acute criterion for aldrin if found at these levels is reasonably certain to harm listed salmonids or impact their food sources. The limited information available regarding aldrin toxicity indicates that aldrin is toxic to some salmonid prey species when concentrations are below or close to the criterion and is likely to adversely affect food sources. However, it is unlikely that discharges of aldrin will occur in the action area as no uses are currently approved and levels found in the water column are well below the proposed standards. Because of this NMFS finds that adverse affects are unlikely to occur.

***Additional comments on Aldrin.*** Although no chronic criterion for aldrin is proposed, available studies demonstrate that chronic effects do occur to freshwater fish at 0.0466 µg/L, and to prey items at 2.5 µg/L. These results suggest that the absence of a chronic criterion could result in adverse chronic effects to listed salmonids and their food source. However, the human-health based aldrin criteria is also applicable to all waters in the action area that are either designated critical habitat for, or are inhabited by listed salmonids. For aldrin this criterion is 0.00014 µg/L (Table 1.3.1). This value is lower than concentrations causing adverse effects to any aquatic prey species, listed species, or surrogate for a listed species reviewed here so lack of a chronic criteria does not pose a risk to listed salmon and steelhead.

***Dieldrin.*** The scientific literature on effects of dieldrin on salmonids reports acute lethal effects at concentrations that are below or slightly above the proposed acute criterion. These studies included various salmonid species, such as Chinook and coho salmon, steelhead, and rainbow, cutthroat, or brown trout, as well as toxicological information on juveniles and adults. This available information indicates that the proposed acute criterion for dieldrin will likely adversely affect listed salmonid species. The proposed acute criterion is greater than $LC_{50}$s reported for several important salmonid prey species. However, because acute effects could only come from recent applications, and because the use of dieldrin has been banned since EPA cancelled its registration in 1975, acute effects occurring from an authorized release are unlikely. Chronic studies involving juvenile rainbow trout demonstrate that limited adverse effects only occur when ambient concentrations are >95 times the proposed chronic criterion. This information is supplemented by published BCF values and analyses of the results of dietary exposure studies in which estimated aqueous concentrations of dieldrin resulting in reported tissue concentrations was also well above the chronic criterion. These limited studies indicate that the proposed chronic criterion will not result in measurable effects to listed salmonids. Further, no information suggests that prey species may be adversely affected by concentrations below the proposed chronic criterion. Dieldrin was detected in sediment in Brownlee Reservoir of the Snake River (Table 2.3.1) and these are likely the highest levels that will be found in the state based on the location of the reservoir. However, levels of dieldrin currently found in Brownlee Reservoir are well below the standard and the reservoir is not occupied by listed species. With no ongoing discharges, the level of dieldrin in sediment in Brownlee Reservoir is likely to decline over time.

<u>Chlordane</u>

Lethal effects from short-term exposures of salmonids or salmonid invertebrate prey species to chlordane only occurred at concentrations above the acute criterion. There are no current approved uses of chlordane in the United States and no manufacturing of chlordane takes place in Idaho. Chlordane was detected in Brownlee Reservoir (Table 2.3.1) and these are likely the highest levels that will be found in the state based on the location of the reservoir. The levels detected were well below the proposed criteria and no listed salmon or steelhead are located in or above the reservoir.

Data generally indicate that the proposed chronic criterion for chlordane is likely to avoid harm to listed salmonids. However, many sublethal effects of chronic exposure to chlordane that have been documented in mammals (i.e., neurological damage, altered immune and reproductive function, and increased cancer risk) ; we found no studies of salmonid species subjected to long-term chlordane exposure at concentrations near or below the criterion. Similarly, few data are available on the sublethal effects of long-term exposure to chlordane on salmonid prey. There are also a few studies suggesting that a risk of increased long-term mortality or sublethal effects at chlordane tissue concentrations close to those that might be expected in fish exposed to chlordane at levels allowed under the chronic aquatic life criteria. Additionally, bioaccumulation can occur in salmonids with chronic exposure to chlordane at levels allowable under the proposed criteria, and exposure is likely to occur not only through the water column but also through diet and contact with sediments. There is some evidence of risk to benthic invertebrates or through food web uptake associated with bioaccumulation and exposure from sources other than the water column. Based on the strength of evidence considered, the chronic criterion does not appear likely to harm salmonids through water column exposure. The other exposure pathways may pose some risk for salmon and steelhead, but appear likely to result in only minor effects.

DDTs

Sediment and fish tissue DDT concentrations from Brownlee Reservoir tended to be the highest found in sampling in various locations in Idaho (Table 2.3.1; Clark and Maret 1998). In water, baseline levels for DDT found in Brownlee Reservoir in 2011 were <0.00066 µg/L, which is below the levels where effects would be expected to listed salmon and steelhead. DDT is a banned substance in the United States and so no new or ongoing discharges are expected to occur.

Concentrations of DDT in the action area at the proposed action acute criterion could cause harm to listed fish; however, the effects of the EPA approval of the acute criterion is discountable because DDT is extremely unlikely to occur at that concentration because there will be no new discharges of DDT and no known hotspots of DDT occur in the action area where listed fish are present.

The chronic criteria have risk of sublethal health effects in salmonids if bioconcentration results in tissue concentrations that are higher than those expected by EPA. The proposed chronic criterion may allow substantial bioaccumulation to occur because DDTs are taken up not only from the water column but also from sediments and prey organisms. No reports of direct adverse effects to listed salmonids were located at concentrations lower than the chronic criterion. While some data are equivocal and there are quite a few uncertainties in interpreting DDT risks to fish, NMFS found no persuasive evidence of appreciable adverse effects from DDT at concentrations lower than the chronic criterion concentrations.

Endosulfan

Endosulfan has not been found in Idaho waters or sediments at levels that approach the standards as proposed and future discharges of endosulfan are unlikely to occur because the product use has been banned so an acute exposure scenario from an authorized release is unlikely. The proposed acute lethal criterion for endosulfan would likely result in some mortality of listed salmonids. Reported rainbow trout $LC_{50}$s near or below the proposed acute criterion indicate that appreciable mortality can occur in waters meeting the proposed criterion. Evaluation of the proposed chronic criterion was restricted by the absence of relevant toxicity testing data involving salmonid species. The limited information that could be gathered on rainbow trout and two other freshwater fish suggests that the proposed chronic criterion can allow chronic physiological damage to listed salmonid species. The physiologic damage was not directly related to “clinically significant” fish health changes. Although there is a paucity of toxicity testing data, the available information suggests that the proposed acute and chronic criteria may protect some invertebrate prey species. Little test data exist for specific salmonid prey species.

Endrin

Endrin is a banned product in the United State so new discharges are likely to occur. Endrin was detected in Brownlee Reservoir which would likely contain the highest levels in the state due to its location. Concentrations elsewhere in Idaho are likely to be lower than those in Brownlee Reservoir (Table 2.3.1). The levels detected were lower than the chronic criteria. Most reports of mortality following short-term endrin exposures produced $LC_{50}$s greater than the acute criterion, although some effects occurred at lower concentrations. Evidence indicates that concentrations at the acute criterion will not harm salmonid prey species.

While data are sparse, most reports of adverse effects from chronic exposures to salmonids or other fish occurred at concentrations higher than the chronic criterion. A report of subclinical reductions in cholesterol and lipids in gravid Asiatic catfish are of ambiguous importance to salmon. Food chain exposure via diet or sediment was estimated by NMFS to mostly result in tissue residues lower than those shown to be harmful to fish.

Heptachlor

Currently heptachlor is not used in Idaho because the only remaining use is to control fire ants which are not present in Idaho. The only information regarding existing concentrations of heptachlor in Idaho is from Brownlee Reservoir which is the reservoir most likely to contain contaminated sediments. The levels measured at Brownlee are well below levels found to cause acute or chronic effects and should decline over time.

Available evidence indicates that listed salmon or steelhead experience acute lethal effects at concentrations much higher than the proposed acute criterion. However, all such evidence is derived from static tests with nominal heptachlor concentrations, a methodology that tends to under estimate toxicity. There is a greater likelihood that heptachlor could harm salmon or steelhead through lethal effects on aquatic invertebrates; however, little information is available on the effects on invertebrate prey species.

Data on chronic effects of heptachlor are sparse, but suggest that the risk of adverse effect through water-borne exposure is likely to be low. Some studies suggest that tissue concentrations that are possible under the chronic criterion could have sublethal or lethal effects on alevins or fry. Bioaccumulation can occur in salmonids with chronic exposure to heptachlor, and when exposure occurs, this is could occur through the water column, diet and contact with sediments.

Lindane

There are no current registered uses of lindane in the United States and no known contamination of sites in Idaho at levels that may impact listed salmonids. Most of the available data tended to show adverse effects to listed salmonid species, or their close relatives, or their prey at levels greater than the proposed criteria concentrations. The reliability of a single acute test reporting mortalities at concentrations lower than the acute criterion is uncertain since targeted exposure concentrations were not verified by chemical analysis (i.e., were nominal concentrations).

PCBs

Water borne PCB concentrations close to, or below, the proposed chronic criterion, in concert with predicted bioaccumulation rates, were projected to result in impaired thyroid function in coho salmon and embryo mortality in lake trout. However, polychlorinated biphenyls are no longer manufactured in the United States, and PCB contamination of surface waters in Idaho is not recorded in the impaired waters list for Idaho and no known cleanup or sediment concerns that might impact listed fish were identified. This makes the risk of exposure to listed salmon or steelhead unlikely. If discharges do occur the most stringent controlling ambient water quality criterion applicable in designated critical habitats is the fish consumption based human-health criteria, rather than the chronic aquatic life criteria (Table 1.3.1). The fish consumption based criterion is more than100 times more restrictive than the aquatic life criteria. Therefore any effects from the proposed approval of the PCB criterion will be very small on listed species and designated critical habitat.

Toxaphene

Toxaphene appears unlikely to cause adverse effects to habitat or listed salmon or steelhead from exposure at concentrations in water equal to or below the proposed acute or chronic criteria. The risk of exposure is also very small.

### 2.6.1. Integration and Synthesis Summary for Each Affected Species

For Snake River A run steelhead the existing populations may have achieved “maintained” or “moderate risk” status based population estimates using an aggregate of the returns over LGD. However, none of the populations have attained the “low risk” or “viable” status needed to attain the recovery goal for the ESU. To achieve recovery goals, improvement in abundance and productivity is necessary in a least half of the populations. For Snake River B run steelhead populations all of the populations remain at high risk for abundance and productivity.

Abundance and productivity for all populations must improve for the DPS to attain its recovery goal (Ford 2011). The proposed action for the hardness floor, arsenic, copper, cyanide, selenium and mercury is likely to cause adverse modification to critical habitat or lethal and sublethal effects to a large portion of one or more populations, and the reduction in, or loss of, that population will result in jeopardy for the Snake River steelhead DPS.

Snake River spring/summer Chinook populations all remain at high risk for abundance and productivity. Abundance and productivity for all populations must improve for the ESU to attain its recovery goal (Ford 2011). The proposed action for the hardness floor, arsenic, copper, cyanide, selenium and mercury is likely to cause adverse modification to critical habitat or lethal and sublethal effects to a large portion of one or more populations, and the reduction in, or loss of, that population will result in jeopardy for the Snake River spring summer Chinook salmon ESU.

Snake River fall Chinook salmon are at moderate risk for abundance and productivity and have not attained the recommended level of "very low risk" for abundance and productivity necessary for a single population MPG to achieve its recovery goal (Ford 2011). The proposed action for the hardness floor, arsenic, copper, cyanide, selenium and mercury is likely to cause adverse modification to critical habitat or lethal and sublethal effects the population and will result in jeopardy for the Snake River fall Chinook salmon ESU.

Snake River sockeye salmon are currently at high risk for abundance and productivity the Redfish Lake Population must improve for the ESU to achieve its recovery goal (Ford 2011). The proposed action for the hardness floor, arsenic, copper, cyanide, selenium and mercury is likely to cause adverse modification to critical habitat or lethal and sublethal effects to a large portion of the population, and will result in jeopardy for the Snake River sockeye salmon ESU.

## 2.7. Conclusion

After reviewing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, NMFS has made the follow determinations in its Opinion.

Hardness Floor

Metal limits for discharges that are calculated using the current equation with the 25 mg/L hardness floor may result unacceptable declines in abundance and productivity for any exposed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead population and prevent the population from achieving the minimum level of abundance and productivity needed for the ESU or DPS to achieve its recovery goal. NMFS concludes that the potential effects of using the hardness floor in applying the proposed IWQS will rise to the level of jeopardizing the Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead.

Permitting of new or existing discharges using the current equations containing a floor of 25mg/L for calculating metal discharge limits will allow some metals to reduce water quality, accumulate in sediments, periphyton, and in aquatic macroinvertebrate tissues in concentrations that will be detrimental to aquatic macroinvertebrate communities. NMFS concludes this will result in the adverse modification of habitat within designated critical habitat for Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon, or Snake River Basin steelhead.

Arsenic

If sustained concentrations of arsenic in a surface water within the action area exceeds 10 µg/L, which is much lower than the proposed chronic criterion of 150 µg/L dissolved arsenic in water, then accumulation of arsenic would be expected in sediments, periphyton, and in aquatic macroinvertebrate tissues at concentrations that would be harmful in diets of salmonids. This would also likely create a reserve of arsenic in sediment resulting in a contaminated food source for listed species over an extended timeframe. This will result in unacceptable declines in population abundance and productivity for an exposed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead population and will prevent the populations from achieving the minimum level of abundance and productivity needed for the ESU or DPS to achieve its recovery goals. NMFS concludes that the potential effects from the proposed chronic arsenic criteria would jeopardize Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead.

If new or existing discharges containing concentrations of arsenic in surface water within the action area approach the chronic criterion of 150 µg/L dissolved arsenic in water the accumulation of arsenic is expected in sediments, periphyton, and in aquatic macroinvertebrate tissues to concentrations that will be detrimental to aquatic macroinvertebrate communities. Continued exposure may result in a reserve of arsenic in sediment that may take years to dissipate resulting in an ongoing effect to prey species over a number of years. Because of the likelihood of a new discharge, or continuation of existing discharges, being located within critical habitat for listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead, NMFS has determined the chronic standard for arsenic is likely to result in an adverse modification of designated critical habitat for these species.

The proposed acute criterion for arsenic is not likely to result in adverse effects to listed salmon and steelhead in Idaho because the exposure to concentrations of dissolved arsenic at the proposed standards is not expected to result in significant toxic effects to individual fish or populations.

Copper

Continued exposure to copper at the proposed acute or chronic criteria levels will result in adverse effects including mortality and reduced growth in juvenile fish. NMFS concludes that these potential effects are likely to jeopardize the four listed salmon and steelhead ESUs or DPSs because of predicted effects to growth, reproduction and survival that would increase the extinction risk for a DPS or ESU.

New or existing discharges containing concentrations of copper in surface water that approach either the acute or chronic criterion will compromise the diversity and abundance of the macroinvertebrate food base for rearing juvenile salmon and steelhead. It is likely that a new discharge will be located within critical habitat for listed species, thus NMFS has determined the proposed acute and chronic criteria for copper are likely to result in an adverse modification of critical habitat for Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead.

Cyanide

Juvenile Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon or Snake River steelhead will likely die if exposed to cyanide concentrations at the proposed chronic criteria (in water temperatures below 6°C). NMFS concludes that the loss of juveniles will negatively affect the exposed population, and prevent attainment of viability criteria for the exposed DPS or ESU. Therefore, NMFS concludes that the proposed cyanide criteria will jeopardize Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead.

Cyanide in the water column at the proposed chronic criteria concentrations during the colder seasons will result in the water quality being unsuitable for listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead resulting in an adverse modification of designated critical habitat for these species.

The proposed acute criterion for cyanide is not likely to result in adverse effects to listed salmon and steelhead in Idaho because the exposure to concentrations of cyanide at the proposed standards is not expected to result in significant toxic effects to individual fish or populations.

Mercury

Risks of mercury toxicity result primarily from bioaccumulation occurring from exposure to mercury in the diet. In reviewing the proposed chronic mercury criterion, NMFS concludes that these potential dietary effects impair the ability of listed fish to locate, capture, and ingest prey, and to avoid predators, as well as impaired reproduction. These effects can reduce survival of individual fish and reduce the viability of a population. Therefore, NMFS concludes that the proposed chronic criteria for mercury will jeopardize Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead. Because the nature of effects is through ingestion of prey with a body burden of mercury, NMFS also concludes the proposed chronic criterion will adversely modify designated critical habitat for rearing Snake River salmon and steelhead.

NMFS concludes that exposure of listed salmon and steelhead to mercury at the acute criterion is unlikely to result in death or sub-lethal effects that result in injury or reduced survival.

Selenium

Continued exposure to selenium at the chronic criterion level will result in transfer of selenium through prey species to listed juvenile Snake River Basin steelhead, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River spring/summer Chinook salmon creating a selenium tissue burden that reduces growth and survival. These effects will translate to reduced viability of the exposed population. Thus NMFS concludes that the proposed chronic criterion will jeopardize Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead. NMFS also concludes that these potential dietary effects will adversely modify critical habitat for rearing Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead.

The proposed acute criterion for selenium is not likely to result in adverse effects to listed salmon and steelhead in Idaho because the exposure to concentrations of selenium at the proposed standards is not expected to result in significant toxic effects to individual fish or populations.

Nickel

The available information indicates that the risk of adverse effects from exposure to the chronic criterion for nickel to listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River Basin steelhead and Snake River Sockeye or their habitat at concentrations at or below the criterion, are likely to adversely affect listed species because it may result in reduced survival and growth of salmonid embryos. However, the effect is not expected to reduce the viability of the exposed population. Therefore, NMFS concludes that the proposed chronic criterion for nickel is not likely to jeopardize the species or adversely modify designated critical habitat for listed Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River Basin steelhead or Snake River Sockeye.

The proposed acute criterion for nickel is not likely to result in adverse effects to listed salmon and steelhead or their designated critical habitat in Idaho because the exposure to concentrations of nickel at the proposed standards is not expected to result in significant effects.

Silver

The available information indicates that when salmonids or their habitat are exposed to silver (as silver nitrate) over the long-term, mortality and reduced reproduction could occur at concentrations below the acute silver criteria. However, because silver in the environment is expected to form complexes with chloride, DOC, or sulfide that have less toxicity than silver nitrate, these effects are unlikely to affect the viability of populations in the listed ESUs, and therefore, are unlikely to jeopardize Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead or adversely modify designated critical habitat for these species.

Zinc

Some available studies have shown some adverse effects from exposure to zinc by listed salmon and steelhead at concentrations at or below the proposed acute or chronic criteria. However, many other studies show adverse effects only at concentrations higher than the criteria. NMFS concludes that any effects that may occur are unlikely to decrease the viability of affected populations, and therefore are not likely to jeopardize the Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead.

Exposure to zinc at concentrations below the proposed criteria for zinc may change the composition of prey species for listed salmon. However, the overall abundance of prey availability is unlikely to result in adverse effects to listed populations, and therefore, exposure to zinc at the proposed criteria is unlikely to result in the adverse modification of designated critical habitat for these species.

Pentachlorophenol

Discharges of PCP in Idaho are only expected from the use of treated wood in construction in or around surface water. Therefore any effects from the proposed approval of the PCP chronic or acute criteria are expected to be short-term events and have only minor effects on listed species and designated critical habitat. Therefore, NMFS concludes that the proposed criteria for PCP are not likely to jeopardize Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead, or result in the adverse modification of designated critical habitat.

Chromium III and Chromium VI

The proposed criterion for chromium III and chromium VI are not likely to result in adverse effects to Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead in Idaho because the exposure to concentrations of chromium III and chromium VI at the proposed standards are not expected to result in significant toxic effects to individual fish or fish populations.

The proposed criterion for chromium III and chromium VI are not likely to result in adverse effects to designated critical habitat for Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead in Idaho.

Lead

The proposed criterion for lead are not likely to result in adverse effects to Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead in Idaho because the exposure to concentrations of lead at the proposed standards are not expected to result in significant toxic effects to individual fish or fish populations.

The proposed criterion for lead are not likely to result in adverse effects to designated critical habitat for Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead in Idaho.

Aldrin

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to Aldrin is very unlikely. This is based on the inability to use Aldrin as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead. An additional safety factor is provided by the applicability of a lower recreation criterion based on fish consumption.

Dieldrin

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to Dieldrin very unlikely. This is based on the inability to use Deildrin as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead. An additional safety factor is provided by the applicability of a lower recreation criterion based on fish consumption.

Chlordane

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to Chlordane very unlikely. This is based on the inability to use Chlordane as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead. An additional safety factor is provided by the applicability of a lower recreation criterion based on fish consumption.

DDT

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to DDT is unlikely. This is based on the inability to use DDT as a pesticide and the lack of other new discharges. An additional safety factor is provided by the applicability of a lower recreation criterion based on fish consumption. The more meaningful exposure scenario for DDT is chronic exposure to low level concentrations that have persisted in sediments of rivers, reservoirs, and lakes. On the whole, the available information indicates that the risk of adverse effects from exposure to DDT by listed Snake River salmon and Snake River Basin steelhead at concentrations at or below the chronic criterion is very low and because no new discharges will occur and any potential exposure from existing contamination will be reduced over time.

Endosulfan

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to Endosulfan is very unlikely. This is based on the inability to use Endosulfan as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead.

Endrin

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to endrin is very unlikely. This is based on the inability to use endrin as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead. Even if fish are exposed, few effects at sub-criterion concentrations have been documented.

Heptachlor

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to heptachlor is very unlikely. This is based on the inability to use heptachlor as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead. An additional safety factor is provided by the applicability of a lower recreation criterion based on fish consumption.

Lindane

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to lindane is very unlikely. This is based on the inability to use lindane as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead.

PCBs

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to PCBs is very unlikely. There are no registered uses of PCBs in the state, and no known contamination in waters containing listed salmon or steelhead. If discharges do occur the most stringent controlling ambient water quality criterion that is applicable to designated critical habitats are the fish consumption based human-health criteria, rather than the chronic aquatic life criteria (Table 1.3.1). The fish consumption based criteria are more than100 times more restrictive than the aquatic life criteria. Therefore any effects from the proposed approval of the PCB criteria will have only very small effects on listed species and designated critical habitat.

Toxaphene

Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to toxaphene is very unlikely. This is based on the inability to use toxaphene as a pesticide, the lack of other discharges and the lack of known contamination in waters containing listed salmon or steelhead. Even if fish are exposed few effects at sub-criterion concentrations have been documented.

## 2.8. Reasonable and Prudent Alternatives (RPAs) and Analysis of Effects of the RPAs

"Reasonable and prudent alternatives" (RPAs) refer to alternative actions identified during formal consultation that can be implemented in a manner consistent with the intended purpose of the action, that can be implemented consistent with the scope of the Federal agency's legal authority and jurisdiction, that are economically and technologically feasible, and that would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat (50 CFR 402.02).

The EPA's authorities include the responsibility to review and approve or disapprove state revisions of their water quality standards; states are to review their water quality standards, at least once every 3 years (40 CFR sections 131.20 through 131.21). If EPA disapproves a state's new or revised water quality criteria and the state does not adopt specified changes, the EPA Administrator has the responsibility and authority to promptly propose and promulgate such standard 40 CFR section 131.22). The water quality standards considered in this action are implemented in part through wastewater discharge permits, administered by EPA through the National Pollutant Discharge Elimination System (NPDES). Monitoring, including biological monitoring, may be required of dischargers as part of their permit conditions (40 CFR 122.48). When the ESA is applicable and requires consideration or adoption of particular permit conditions, those requirements must be followed (40 CFR 122.49).

### 2.8.1. The RPA for the Hardness Floor

#### *2.8.1.1. New Aquatic Life Criteria*

The EPA shall recommend that the state of Idaho adopt, and EPA will promulgate if necessary, removal of the low end hardness floor on the hardness dependent metals criteria equations within 3 years of the date of this Opinion.

### 2.8.2. The RPAs for Arsenic

#### *2.8.2.1. Interim Protection for Listed Species*

Until a new chronic criterion for arsenic is adopted, EPA shall ensure that the 10 µg/L recreational use standard is applied in all Water Quality Based Effluent Limitations (WQBELs) and Reasonable Potential to Exceed Calculations using the human health criteria and the current methodology for developing WQBELs to protect human health. The recreational use standard is interpreted to apply as inorganic, unfiltered, arsenic.

##### *2.8.2.2. New Chronic Aquatic Life Criterion for Arsenic*

The EPA shall ensure, either through EPA promulgation of a criterion or EPA approval of a state-promulgated criterion, that a new chronic criterion for arsenic is in effect in Idaho within 7 years of the date of this Opinion. The new criterion shall be protective of listed salmon and steelhead, consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criterion, EPA shall provide an adequate biological evaluation to NMFS and initiate consultation within 6 years of the date of this Opinion, unless NMFS and EPA mutually agree to a different time-frame, to allow for consultation to be completed prior to EPA progmulgation or approval of the new criterion.

#### 2.8.3. The RPAs for Copper

##### *2.8.3.1. Interim Protection for Listed Species*

Until new criteria are adopted, a zone of passage must be maintained around any mixing zone for discharges that include copper, sufficient to allow unimpeded passage of adult and juvenile salmonids as defined in Appendix F Salmonid Zone of Passage Considerations.

Permits for new discharges must ensure a zone of passage persists under seasonal flow conditions (see Appendix D). If the regulatory mixing zone is limited to less than or equal to 25% of the seasonal flow conditions, then a sufficient zone of passage is presumed to be present.

Permits reissued for existing discharges must ensure a zone of passage persists under seasonal flow conditions. If the regulatory mixing zone is limited to less than or equal to 25% of the volume of a stream, then sufficient zone of passage is presumed to be present. If existing discharges were calculated using greater than 25% of the seasonal flow conditions for applying aquatic life criteria the mixing zone must be reduced to 25% unless one of the following conditions exists:

1. An evidence-based "Salmonid Zone of Passage Demonstration" (see Appendix F) indicates that impeding fish movements is unlikely, or;

2. Biological monitoring of aquatic communities in the downstream receiving waters shows no appreciable adverse effects relative to reference conditions as described in Appendix E Biomonitoring of Effects, and biological whole-effluent toxicity testing is consistently negative, defined as follows:

a. Whole effluent toxicity (WET) testing shall be required, using at least the 7-day *Ceriodaphnia dubia* 3-brood test and the 7-day fathead minnow growth and survival test. If previous testing of a facility's effluents have demonstrated that one test is more sensitive, at EPA's discretion it is acceptable to base further testing on only the more sensitive test. Toxicity trigger concentrations for WET tests shall also be established using dilution series based upon no more than 25% of the applicable critical flow volume. The dilution series for WET testing (7Q10) shall be designed such that one treatment consists of 100% effluent, and at least one treatment is more dilute than the targeted critical flow conditions. Receiving waters upstream of the effluent discharge should be used as dilution water.

   The "critical concentration" is defined here as the condition when the smallest permitted dilution factor occurs, modified by a 25% mixing zone fraction. For example, if the minimum effluent dilution occurring at a site is a 1:4 ratio (one part effluent to four parts streamwater), then because only 25% of the measured streamflow is authorized for dilution; then the dilution factor for effluent testing is likewise reduced to 1:1. The critical concentration would then be 50% effluent, i.e., one part each effluent and dilution water.

   WET tests results need to be consistently negative to indicate the absence of appreciable instream toxicity in test conditions that reflect the critical effluent concentration, above. A "negative test result" is produced by a test meeting the performance objectives of a passing test according to EPA (2002c) or EPA (2010c). Test results are considered to be consistently negative if the failure rate is less than one in 20.

b. If instream biological monitoring shows adverse effects or if WET tests are not consistently negative, then a toxicity identification evaluation and toxicity reduction evaluation (TIE/TRE) must be undertaken to identify and remedy the causes of toxicity, which may include reducing effluent limits as warranted. Because considerable judgment may be involved in designing and carrying out a TIE/TRE, and because the results are performance-based (no detectable toxicity per A.2), more specific guidance is inappropriate to provide here. Mount and Hockett (2000) provide one example of a TIE/TRE.

#### *2.8.3.2. New Acute and Chronic Aquatic Life Criteria for Copper*

The EPA shall ensure, either through EPA promulgation of criteria or EPA approval of a state-promulgated criteria, that new acute and chronic criteria for copper are in effect in Idaho within 3 years of the date of this Opinion. The new criteria shall be no less stringent than the Clean Water Act section 304(a) 2007 national recommended aquatic life criteria (i.e. the BLM Model) for copper. NMFS does not anticipate that additional consultation will be required if the 2007 national recommended aquatic life criteria for copper are adopted.

### 2.8.4. The RPAs for Mercury

#### *2.8.4.1. Interim Protection for Listed Species*

1. Until a new chronic criterion is adopted EPA will use the 2001 EPA/2005 Idaho human health fish tissue criterion of 0.3 mg/kg wet weight for WQBELs and reasonable potential to exceed criterion calculations using the current methodology for developing WQBELs to protect human health. Implementation of the Idaho methylmercury criterion shall be guided by EPA's (EPA 2010a) methylmercury water quality criteria implementation guidance or IDEQ's (IDEQ 2005) methylmercury water quality criteria implementation guidance, (or)

2. For water bodies for which appropriate fish tissue data are not available, if the geometric mean of measured concentrations of total mercury in water is less than 2 ng/L, then the water body will be presumed to meet the fish tissue criterion of 0.3 mg/kg wet weight. If the water column concentration is greater than 2 ng/L, fish tissue data shall be collected.

#### *2.8.4.2. New Chronic Aquatic Life Criteria for Mercury*

The EPA shall ensure, either through EPA promulgation of a criterion or EPA approval of a state-promulgated criterion, that a new chronic criterion for mercury is in effect in Idaho within 7 years of the date of this Opinion. The new criterion shall be protective of listed salmon and steelhead, consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criterion, EPA shall provide an adequate biological evaluation to NMFS and initiate consultation within 6 years of the date of this Opinion, unless NMFS and EPA mutually agree to a different time-frame, to allow for consultation to be completed prior to EPA progmulgation or approval of the new criterion.

### 2.8.5. The RPA for Cyanide

Calculation of effluent limits for cyanide shall be made using the receiving water mixing zone limitations described in "RPAs for Copper", in the *Interim Measures, Zone of Passage* section.

### 2.8.6. The RPAs for Selenium

#### *2.8.6.1. Interim Protection for Listed Species*

Until a new chronic criterion is adopted, EPA shall ensure that all effluent discharges located within critical habitats or habitats of Snake River listed salmonids that are regulated under the NPDES program apply the following terms:

1. At discharge locations where at the edge of the mixing zone, selenium concentrations are measured or projected to be higher than natural background for the locale and annual

geometric mean concentrations are higher than 2 µg/L, whole body fish tissue should be monitored in locations downstream of the discharge and in reference locations. The results shall be reported as an NPDES permit condition.

2. If fish tissue concentrations exceed the screening risk concentration of 7.6 mg/kg dw and are higher than reference concentrations, then the issuance of an NPDES permit shall include provisions to reduce selenium loading in order to reduce impairment of aquatic life uses. These provisions are not required if fish population surveys using surrogate species such as rainbow trout show that appreciable adverse effects are not present, as defined in Appendix E Biomonitoring of Effects.

#### *2.8.6.2. New Chronic Aquatic Life Criterion for Selenium*

The EPA shall ensure, either through EPA promulgation of a criterion or EPA approval of a state-promulgated criterion, that a new chronic criterion for selenium is in effect in Idaho within 4 years of the date of this Opinion. The new criterion shall be protective of listed salmon and steelhead, consistent with the discussion and analysis in this Opinion. If ESA consultation is required for the new criterion, EPA shall provide an adequate biological evaluation to NMFS and initiate consultation within 3 years of the date of this Opinion, unless NMFS and EPA mutually agree to a different time-frame, to allow for consultation to be completed prior to EPA progmulgation or approval of the new criterion.

### 2.8.7. Notification of EPA Final Decision

Because this Opinion has found jeopardy and destruction or adverse modification of critical habitat, the EPA is required to notify NMFS of its final decision on the implementation of the reasonable and prudent alternatives.

### 2.8.8. Analysis of the RPAs

A reasonable and prudent alternative to the proposed action is one that avoids jeopardy by ensuring that the action’s effects do not appreciably increase the risks to the species’ potential for survival or to the species’ potential for recovery. It also must avoid destruction or adverse modification of designated critical habitat. A detailed analysis of how the RPA avoids jeopardy and destruction or adverse modification of critical habitat is set out in sections below.

In determining the time frame for implementing the RPAs in this Opinion NMFS recognized that EPA needs to complete consultation with USFWS on these water quality standards to make sure that they are protective of other listed species. This consultation is scheduled to be completed in early 2015. After that, promulgation of rules under either the state or Federal process will require a minimum of 2 years to complete. For most water quality standards the state of Idaho will likely take the lead and promulgate state rules that require approval by the Idaho Board of Environmental Quality. Additionally, before becoming effective the rules will be reviewed by the Idaho Legislature. Finally, EPA approval of the new rules must also occur. Based on this process we have assumed that the soonest new rules can be completed is 3 years and have used 3 years for the implementation time frame for the RPAs that will not require additional analysis to derive new criteria (i.e., hardnes floor, 2007 BLM copper criteria). The RPA for cyanide can be implemented immediately and therefore does not include an implementation period.

For the other RPAs, EPA and/or the state will likely require additional time to conduct the analyses necessary to support new criteria (arsenic, mercury, selenium). These RPAs therefore provide a longer implementation period of 4 to 7 years. To ensure that the listed species are not adversely affected during the implementation period, these RPAs include interim protective meaures that NMFS expects will adequately reduce any interim risk of harm to the species or their critical habitats. In addition, EPA consults with NMFS over each new or reissued NPDES permit in Idaho to ensure that it will not cause jeopardy to the species or adverse modification to critical the habitat. These factors, when considered together, will minimize any adverse during the implementation period while new criteria are developed and adopted.

#### *2.8.8.1. Analysis of the Reasonable and Prudent Alternative for the Hardness Floor*

Use of rules and guidance that require hardness-dependent metals criteria to be implemented using ambient water hardness without a hardness floor was analyzed as being protective in this Opinion. The RPA requires that the hardness floor be removed within 3 years. In the interim, within the action area, only one major discharger is known to be permitted to discharge metals into a water body with water hardness values that are consistently lower than the 25 mg/L hardness floor. This facility, the Beartrack Mine discharges to Napias Creek upstream of a waterfall which is considered to be impassible by Snake River Chinook salmon and thus excluded from critical habitat for Snake River salmon or steelhead (50 CFR §226.205, 226.212). The facility discharges high in the Napias Creek watershed; in lower Napias Creek, where it becomes designated as critical habitat downstream of Napias Creek falls, streamflows are estimated to increase by a factor of about 1.9 (USGS 2012). Thus, assuming discharges from the Beartrack Mine resulted in instream metals concentrations that approached adverse effects thresholds, i.e., criteria constrained by the hardness floor, this increase in dilution would effectively result in reducing metals concentrations by 0.54 times, assuming no intervening sources in the Napias Creek drainage. Because the amount of critical habitat downstream of Napias Creek Falls is small (less than 2 linear miles), in the interim time before the hardness floor is removed it is unlikely to result in appreciable reductions of any of the listed species' survival or recovery. The likelihood of new, major facilities coming online and discharging metals into low-hardness waters within this 3 year time-frame is considered unlikely.

#### *2.8.8.2. Analysis of the Reasonable and Prudent Alternative for Arsenic*

An interim protection for arsenic is available through use of the human health criterion, which is 10 µg/L. This criterion is applicable to all waters in the action area. Because it is more stringent than the chronic criterion of 150 µg/L, the criterion for the protection of human health is the controlling criterion for permitting actions. While bioaccumulation has been found in salmonid prey from exposures at concentrations at or near 10 µg/L the application of this lower standard, coupled with biological monitoring, will provide adequate information to review effects in a site specific manner. Because any new or reissued permits will be subject to individual ESA consultation to assure they avoid jeopardy or adverse modification of habitat, EPA will make adjustments as necessary during the NPDES permitting cycle taking into account local conditions to avoid measureable direct effects to the listed species. Some minor adverse effects, as described in the effects section, may still occur during the early life history phases for all listed Snake River salmon and steelhead. Use of the human health criterion will provide adequate protection in the interim to avoid jeopardy.

The adoption of a new chronic aquatic life criterion within the next 7 years will be subject to ESA consultation to ensure that the new criterion will be adequately protective. Therefore, NMFS concludes that the arsenic RPA will not jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats.

*2.8.8.3. Analysis of the Reasonable and Prudent Alternative for Copper*

For the next 3 years, the interim requirement of assuring an adequate zone of passage for any permits that contain copper discharge limits as described in the copper RPA will minimize adverse effects to listed salmon and steelhead. Any new permits will also be subject to individual consultation to assure they avoid jeopardy or adverse modification of habitat. NMFS found five existing permits that contain copper limits and these will be updated to the new criteria when they are reissued. Over the next permitting cycle this should reduce the adverse effects described in the effects section to acceptable levels.

In Appendix C of this Opinion, we analyze implementation of the 2007 BLM EPA copper criteria and conclude that they will be adequately protective to avoid jeopardy to the listed species or critical habitat considered in this Opinion.

Therefore, NMFS concludes that the copper RPA will not jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats.

*2.8.8.4. Analysis of the Reasonable and Prudent Alternative for Cyanide*

The propsed cyanide criteria are likely to advesley affect the listed salmonids species in specific situations, primarily where water temperatures are at of below 6°C. Implementation of the more restrictive practices in developing cyanide discharge limits as described in the cyanide RPA will suffice to minimize the adverse effects to listed salmon and steelhead. These practices will assure an adequate zone of passage exists for the fish, under all flow conditions, and provide biological monitoring and whole-effluent toxicity testing to assure the permit limits are protective of listed fish and prey species. This will be done at each discharge site by taking into account the localized conditions that affect toxicity of cyanide. Based on development of these site specific limits and the associated monitoring of discharge levels, combined with the fact that NMFS consults with EPA over each new or reissued NPDES permit, we expect only minor adverse effects. Therefore, NMFS concludes that the cyanide RPA will not jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats.

*2.8.8.5. Analysis of the Reasonable and Prudent Alternative for Mercury*

The interim requirement of using a human health criterion that consists of a fish-tissue based water quality criterion of 0.3 mg/kg for mercury to determine permit limits will be followed. Idaho has adopted this criterion, and is implementing it as a 0.24 mg/kg a triggering residue concentration for existing dischargers, using an uncertainty (safety factor) of 0.8 times (IDEQ 2007a). This fish tissue-based criterion is close to being a threshold below which adverse effects are unlikely and is sufficient to protect listed salmon and steelhead species and their habitats.

The adoption of a new chronic aquatic life criterion for mercury within the next 7 years will be subject to ESA consultation to ensure that the new criterion will be adequately protective. Therefore, NMFS concludes that the mercury RPA will not jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats.

*2.8.8.6. Analysis of the Reasonable and Prudent Alternative for Selenium*

The interim requirement of monitoring fish tissues and taking corrective action when fish tissues exceed 7.6 mg/kg dw or 2 µg/L in the water column will be sufficiently protective to minimize any food web transfer to concentrations in listed salmon and steelhead. There is one known existing facility within the action area that currently discharges selenium to a stream within critical habitat for Snake River salmon or steelhead at concentrations that approach those described in the RPA (Thompson Creek Mine's discharges to Thompson Creek). As evaluated in Section 2.4.8 (Analysis of Effects), as of 2012, conditions had not resulted in appreciable harm to aquatic life. Any new permits containing discharges of selenium will be subject to individual consultation to assure that jeopardy or adverse modification do not occur. Based on these protective interim practices and the low number of discharges, the continued use of the existing selenium standard during the implementation period will result in only minor adverse effects.

The adoption of a new chronic aquatic life criterion within the next 4 years will be subject to ESA consultation to ensure that the new criterion will be adequately protective. Therefore, NMFS concludes that the selenium RPA will not jeopardize any of the listed species considered in this Opinion or adversely modify their critical habitats

## 2.9. Incidental Take Statement

Section 9 of the ESA and Federal regulation pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. For purposes of this consultation, we interpret "harass" to mean an intentional or negligent action that has the potential to injure an animal or disrupt its normal behaviors to a point where such behaviors are abandoned or significantly altered.[8] Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA, if that action is performed in compliance with the terms and conditions of this ITS.

### 2.9.1. Amount or Extent of Take

The NPDES permits issued or approved under the rules evaluated in this action are reasonably certain to affect the water quality within critical habitats and subsequently result in incidental take. As such, the proposed action creates the framework through which incidental take will occur. This take, however, is indirect and will only occur through implementation of the WQS analyzed in this Opinion through NPDES permits, TMDLs, and clean-up actions. Because of the future and indirect nature of this take, and due to the large degree of variability in effects caused by future implementation of the criteria, it is not possible for NMFS to attempt to quantify the amount of take with any accuracy. However, the extent of critical habitats with foreseeable water quality changes can be described, which can serve as a surrogate measure of the extent of take (as habitat) rather than the amount of take (as fish). Additionally, a more precise measure of the amount and extent of take will be assessed with each individual NPDES consultation, TMDL or CERCLA cleanup, which occurs subsequent to this consultation.

Although calculating the amount of take has substantial inherent uncertainties, it is reasonably certain that some incidental take will occur. Fish will be present in waters that are affected by discharges permitted under the standards reviewed in this consultation and, in some instances, harm to those fish will occur. As described above, the extent of take likely to occur as a result of the proposed action will be evaluated in individual consultations on a site-specific basis for each NPDES permit issued in the action area. NMFS anticipates the upper bounds of the extent of take through the following assumptions and calculations as follows:

1. Incidental take will occur in the immediate proximity of discharges, i.e., in mixing zones, from permitted active or inactive mining facilities. Some smaller amount of incidental take will also likely occur downstream of the mixing zones but the amount of take will decrease in a downstream direction. The level of take downstream of the mixing zones will be proportional to the level of take within the mixing zones.

[8] NMFS has not adopted a regulatory definition of harassment under the ESA. The World English Dictionary defines harass as "to trouble, torment, or confuse by continual persistent attacks, questions, etc." The U.S. Fish and Wildlife Service defines "harass" in its regulations as an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3). The interpretation NMFS adopts in this consultation is consistent with our understanding of the dictionary definition of harass and is consistent with the U.S. Fish and Wildlife interpretation of the term.

2. As of 2014, there were five such facilities located in the action area (EPA 2010a; NMFS 2011a).

3. To be conservative we doubled the existing number of operating NPDES dischargers that could be operating at one time and assume they are all mines. This would mean up to 10 mining facilities could discharge into critical habitats.

4. Each facility is assumed to have three outfalls with mixing zones located in critical habitats, and each mixing zone persists for 50 m and thus impinges on 50 m of habitat.

The rationales for these assumptions include the following: (1) Mixing zones are a place where criteria are allowed to be exceeded. Take resulting from exposure to toxic substances would be from metals or cyanide because the organic chemicals considered in this Opinion are extremely unlikely to be present in discharges; (2) the conclusion that discharges at criteria concentrations of metals will occur at metals mining facilities, and not the urban or other industrial facilities was based on discussions and analyses regarding EPA's separate consultation on revised cadmium criteria (EPA 2010a; NMFS 2011a) and independent review of recent NPDES monitoring and permitting results that were available online (*epa.gov/r10earth/waterpermits.htm*); (3) it would be conservative to assume an increase in discharges. Since EPA's (2010a) review, at least one new mining discharge has been permitted for the Idaho Cobalt Project, which will discharge into a tributary upstream of critical habitat into Big Deer Creek, a tributary to Panther Creek. Further, active exploration in advance of a potential new mine is going on at the old Stibnite Mine, on the East Fork of the South Fork Salmon River, and NMFS presumes for the purpose of this estimate of potential take that there could be other new exploration and development; and (4) the distance it takes for effluent plumes released from wastewater outfalls to fully mix will vary based on various factors such as the relative flows of the receiving water and effluents, temperature, configuration of the outfall, and channel and substrate characteristics. However, in mountain streams, dye studies and simulation studies have shown that thorough mixing usually occurs within 50 m (or 0.05 km) (IDEQ 1999; Mebane 2000).

Following these four assumptions, an estimate of the extent of take, as stream habitat is 10 facilities with three mixing zones per facility for a total of 30 mixing zones. Each mixing zone is 50 m long and 2.5 meters wide or 125 square meters. This results in a total area of 3,750 square meters. The extent of take authorized under this Opinion will be exceeded if total mixing zone areas exceed 3,750 square meters.

No additional take is authorized for new anthropogenic nonpoint sources of contaminents. These are most likely to occur in highly mineralized areas as a result of mining activities and the most likely response is to require removal, isolation, or treatment of water prior to a discharged occurring. Water treatment may lead to a new NPDES permit and based on that permit may be included in the mixing zone take analysis above.

## 2.9.2. Effect of the Take

After reviewing the status of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon, and Snake River Basin steelhead, the status of designated critical habitats, the environmental baseline for the action area, the effects of the proposed action as revised by the RPA, and cumulative effects, NMFS concludes that this level of anticipated take is not likely to result in jeopardy to these species.

## 2.9.3. Reasonable and Prudent Measures and Terms and Conditions

### *2.9.3.1. Reasonable and Prudent Measures*

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR 402.02). "Terms and conditions" implement the reasonable and prudent measures (50 CFR 402.14). These must be carried out for the exemption in section 7(o)(2) to apply.

NMFS believes the RPMs and terms and conditions described below, are necessary and appropriate to minimize the likelihood of incidental take of ESA-listed species due to implementation of the proposed action.

The EPA shall:

1. Minimize the potential for mixture toxicity in discharges.

2. Minimize the potential adverse effects that occur when discharging under NPDES permits.

3. Minimize exposure of aquatic life to PCP.

4. Use updated procedures for calculating any WERs developed for determining discharge limits.

5. Ensure completion of a monitoring and reporting program to confirm that the terms and conditions in this ITS are effective in avoiding and minimizing incidental take from permitted activities and ensure the amount of incidental take is not exceeded.

*2.9.3.2. Term and Conditions*

1. To implement RPM No. 1 (minimize the effects of toxicity resulting from simultaneous exposure to mixtures), the EPA shall:

   a. For all discharges that are expected to simultaneously contain two or more toxic substances evaluated in this opinion, or cadmium, this section shall apply to prevent mixture toxicity.

      If discharges and the permit limits are authorized such that >1 cumulative criterion units (CCU) would be calculated to be allowed in receiving waters, then WET testing and biomonitoring shall be included in the permit provisions as described in Appendix E Biomonitoring of Effects. Cumulative criterion units are defined for this purpose as CCU $=\sum(C_d \div CCC)$ where $C_d$ is the projected authorized concentration in the fully mixed receiving waters downstream of the effluent discharge, the CCC is the applicable chronic criterion concentration of each regulated constituent calculated for that location.

2. To implement RPM No. 2 (minimize the potential adverse effects when discharging under NPDES Permits.) for discharges that include silver, nickel and zinc, the EPA shall:

   a. For new discharges: Ensure a zone of passage exists under seasonal flow conditions (see Appendix D). If the regulatory mixing zone is limited to less than or equal to 25% of the volume of a stream, then sufficient zone of passage is presumed to be present.

   b. For existing discharges: When permits are renewed, ensure a zone of passage under seasonal flow conditions. If the regulatory mixing zone is limited to less than or equal to 25% of the volume of a stream, then sufficient zone of passage is presumed to be present. If existing discharges were calculated using greater than 25% of the applicable seasonal flow conditions for applying aquatic life criteria the mixing zone must be reduced to 25% unless one of the following conditions exist:

      (1) An evidence-based "Salmonid Zone of Passage Demonstration" indicates that impeding fish movements is unlikely, or;

      (2) Biological monitoring of aquatic communities in the downstream receiving waters show no appreciable adverse effects relative to reference conditions as described in the Appendix E Biomonitoring of Effects, and biological whole-effluent toxicity testing is consistently negative, defined as follows:

         (a) Whole effluent toxicity (WET) testing shall be required, using at least the 7-day *Ceriodaphnia dubia* three-brood test and the 7-day fathead minnow growth and survival test. If previous testing of a facility's effluents have demonstrated that one test is more sensitive than the other, at EPA's discretion it is acceptable to base further testing on only the more sensitive test. Toxicity trigger concentrations for WET tests shall also be established using dilution series based upon no more than 25% of the applicable critical flow volume. The dilution series for WET testing (7Q10) shall be designed such that one treatment consists of 100% effluent, and at least one treatment is more dilute than the targeted seasonal flow conditions. Receiving waters upstream of the effluent discharge should be used as dilution water.

The "critical concentration" is defined here as the condition when the smallest permitted dilution factor occurs, modified by a 25% mixing zone fraction. For example, if the minimum effluent dilution occurring at a site is a 1:4 ratio (one part effluent to four parts streamwater), then because only 25% of the measured streamflow is authorized for dilution; then the dilution factor for effluent testing is likewise reduced to 1:1. The critical concentration would then be 50% effluent, i.e., one part each effluent and dilution water.

The WET tests results need to be consistently negative to indicate the absence of appreciable instream toxicity in test conditions that reflect the critical effluent concentration, above. A "negative test result" is produced by a test meeting the performance objectives of a passing test according to EPA (2002c) or EPA (2010c). Test results are considered to be consistently negative if the failure rate is less than one in 20.

(b) If instream biological monitoring shows adverse effects or if WET tests are not consistently negative, then a TIE/TRE must be undertaken to identify and remedy the causes of toxicity, which may include reducing effluent limits as warranted. Because considerable judgment may be involved in designing and carrying out a TIE/TRE, and because the results are performance-based (no detectable toxicity per this subsection, more specific guidance is inappropriate to provide here. Mount and Hockett (2000) provide one example of a TIE/TRE.

3. To implement RPM No. 2 (minimize the potential adverse effects that occur when discharging under NPDES permits.)

   a. For purposes of calculating effluent limits, the effluent discharge volumes and receiving streamflows shall apply the "conservative assumptions" described in Appendix D.

4. To implement RPM No.3 (minimize exposure to pentachlorophenol) the EPA shall:

   a. Whenever possible require wood structures being installed with treated wood should be installed in accordance with the BMPs described in *The Use of Treated Wood Products in Aquatic Environments: Guidelines to West Coast NOAA Fisheries staff for Endangered species Act and Essential Fish Habitat Consultations in Alaska, Northwest and Southwest regions*. October 12, 2009.

5. To implement RPM No. 4 (use WERs conservatively) EPA shall:

   a. Calculate the WER using the lesser of: (1) The site water $EC_{50}$/ lab water $EC_{50}$ ratios; or, (2) the ratio of site water $EC_{50}$ divided by the (SMAV) for that test organism (e.g., Ceriodaphnia dubia, fathead minnow, or rainbow trout) from an updated criteria dataset as described by EPA (2001a); or,

   b. In the case of copper, the WER should be calculated as the site water BLM-based copper criterion ÷ Hardness-based copper criterion for the same hardness.

6. To implement RPM No. 5 (monitoring and reporting), the EPA shall monitor and report as described below. The goal of the monitoring program is to assure that the level of take described in this opinion in not exceeded by monitoring the extent of take.

   a. Monitoring and Reporting the Extent of Take. To insure that the amount of take is not exceeded EPA shall monitor and report on the amount of take as a term and condition of this Incidental Take Statement. The reporting shall be done each time a new NPDES permit is issued that discharges a toxic substance evaluated in this Opinion into waters containing Snake River salmon or steelhead or their critical habitat.

      The reporting shall include the following:

      (1) A copy of the NPDES permit issued to the facility either as a draft or final permit.

      (2) A calculation of the total area of mixing zones granted for the new permit and for existing permits that discharge a toxic substance into waters occupied by listed salmon or steelhead.

7. To implement RPM No. 5 (monitoring and reporting) Biomonitoring of Effects (in situ biological monitoring and assessment) EPA shall require biomonitoring as described in Appendix E Biomonitoring of Effects. The goal of this monitoring is to assure that the nature of the effects occurring are not greater that those described in the effects section of the Opinion.

## 2.10. Conservation Recommendations

Section7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Specifically, conservation recommendations are suggestions regarding discretionary measures to minimize or avoid adverse effects of a proposed action on listed species or critical habitat or regarding the development of information (50CFR 402.02).

2.10.1. Conservation Recommendation for Arsenic

With an emphasis on arsenic, develop an approach to assess the risk to fish at metals-contaminated sites which addresses exposure to multiple contaminants via both water and dietary routes, and which define samples (e.g., invertebrates, sediment, water), metals, and analyses (total metal, speciated metal) necessary to appropriately quantify risk.

2.10.2. Conservation Recommendations for Silver

Publish updated aquatic life criteria for silver that include a chronic criterion value, using a biotic ligand model (BLM) to account for factors that modify toxicity. Much of the fundamental research into the proof of principal, refinement and validation of the BLM-approaches to define metals bioavailability and toxicity was with silver (Di Toro *et al.* 2001; Paquin *et al.* 2002). As result, the BLMs available for silver may be more mature than those for any other metal except for copper (Niyogi and Wood 2004; this Opinion).

Based on the material reviewed to prepare this opinion, NMFS also believes that adequate data exist to derive BLM-based aquatic life criteria for silver including the development of a chronic criterion.

2.10.3. Conservation Recommendation for Cyanide

Revise the cyanide aquatic life criteria to include temperature dependence. Doing so could alleviate the concern about under protectiveness at temperatures less than 6°C and would be consistent with the EPA Guidelines: "*If the acute toxicity of the material to aquatic animals apparently has been shown to be related to a water quality characteristic such as hardness or particulate matter for freshwater animals or salinity or particulate matter for saltwater animals, a Final Acute Equation should be derived based on that water quality characteristic*" (Stephan *et al*. 1985).

2.10.4. Conservation Recommendation for use of bioassessment data in permitting decisions

NMFS recognizes that EPA's WQBEL strategy and biocriteria efforts have long appreciated that well informed field bioassessments complement single-chemical numerical criteria and whole-effluent toxicity testing. In fact due to the nearly infinite permutations of chemical mixtures possible, field assessments may be one of the few practical means for addressing the issue of interactions, mixture effects and multiple stressors. However, there has been little implementation of bioassessment into permitting decisions. Guidance on how bioassessments could be used with point or non-point source implementation has been developed through EPA's Stressor Identification manual, and the State of Idaho has developed extensive biological monitoring databases and interpretive assessment methodologies. Bioassessment of receiving waters has been required as a monitoring element for receiving waters in NPDES permits issued by EPA in Idaho; however, to our knowledge, the data collected has never been a factor in determining the adequacy of permit limits in renewal applications. NMFS recommends that EPA develop an approach to effectively use bioassessment data in permitting decisions.

### 2.11. Reinitiation of Consultation

As provided for in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) The amount or extent of take is exceeded; (2) new information reveals effects of the agencies action on listed species or designated critical habitat in a manner or to an extent not considered in this opinion; (3) the agencies action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat is designated that may be affected by the action.

### 2.12. Summary of Conclusions

Tables 2.12.1 and 2.12.2 provide a summary conclusions, reasonable and prudent alternatives and reasonable and prudent measures. Table 2.12.3 provides a summary of conclusions for organic chemicals.

**Table 2.12.1. Aspects of the action that would or would not likely contribute to "adverse modifications" of critical habitat or "jeopardy".**

| Category | Provisions or chemicals (unless otherwise specified, applies to both acute and chronic criteria) |
|---|---|
| **A. Adverse modifications or jeopardy:** Elements of the action that will likely contribute to jeopardizing the continued existence of listed Snake River anadromous salmonids or adversely modify critical habitat. | |
| General aspects | "*Hardness floor*" |
| Specific chemical criteria | *Arsenic (chronic), Copper, Cyanide, Selenium (chronic), Mercury (chronic),* |
| **B. No jeopardy or adverse modifications of critical habitats:** Elements of the action that are likely to adversely affect listed species, but the magnitude of potential take is unlikely to reach levels that jeopardize the continued existence of listed Snake River anadromous salmonids or adversely modify critical habitat. | |
| Chemical criteria | *Zinc*, *PCP*, *Silver* , *Nickel*, *Chromium III*, *Chromium VI*, *Lead*, *Aldrin*, *Dieldrin*, *Chlordane*, *DDTs*, *Endosulfan*, *Endrin*, *Heptachlor*, *Lindane*, *PCBs*, *Toxaphene* |

**Table 2.12.2. Summary of conclusions on the protectiveness of the Idaho Toxics aquatic life criteria for inorganic chemicals.**

| Constituent | Criteria | EPA's BA Conclusion for Salmonids | Biological Opinion Conclusion | Likely outcomes if water quality in critical habitats were allowed to be at criteria | Synopsis of reasonable and prudent alternatives or measures (see full text for details) |
|---|---|---|---|---|---|
| Arsenic | Acute | Not likely to adversely affect listed species (NLAA) | Unlikely to jeopardize the continued existence of listed species or to result in an adverse modification of critical habitat ("No jeopardy or adverse mod.") | Appreciable adverse effects unlikely | (1) Until a new chronic criterion for arsenic is adopted, ensure that the 10 µg/L recreational use standard is applied in all Water Quality Based Effluent Limitations; and (2) develop a new aquatic life criteria for arsenic that incorporates dietary exposure; |
| | Chronic | NLAA | Likely to jeopardize the continued existence of listed species and to result in an adverse modification of critical habitat ("Adverse mod. and jeopardy") | Chronic criterion concentrations could lead to food chain contamination and adversely affect growth and survival of salmonids | |
| Chromium (III) & (VI) | Acute | NLAA | No jeopardy or adverse mod. | Appreciable adverse effects unlikely | None |
| | Chronic | NLAA | No jeopardy or adverse mod. | Risk to sensitive benthic invertebrates, but effects of a magnitude that would fundamentally alter benthic communities and food webs seems very low. | |
| Copper | Acute | NLAA | Adverse mod. and jeopardy | Adverse effects predicted from abnormal behavior resulting from loss of sense of smell; | (1) Ensure appropriate biological monitoring is conducted and that an adequate zone of |

| Constituent | Criteria | EPA's BA Conclusion for Salmonids | Biological Opinion Conclusion | Likely outcomes if water quality in critical habitats were allowed to be at criteria | Synopsis of reasonable and prudent alternatives or measures (see full text for details) |
|---|---|---|---|---|---|
| Copper | Chronic | NLAA | Adverse mod. and jeopardy | Adverse effects predicted from abnormal behavior resulting from loss of sense of smell; reduced growth is predicted to result in reduced survival during migration. Habitat effects possible from altered invertebrate communities. Adverse effects of copper at concentrations lower than criteria are more likely in high-calcium waters with low organic carbon concentrations. | passage exists; and (2) adopt EPA's 2007 national recommended aquatic life criteria |
| Cyanide | Acute | NLAA | Adverse mod. and jeopardy | Lethality to listed salmonids possible under winter conditions. | Conduct appropriate biological monitoring and ensure an adequate zone of passage exists |
| | Chronic | NLAA | Adverse mod. and jeopardy | Criterion is close to threshold for adverse effects to salmonid reproduction, and swimming ability. | |
| Lead | Acute | NLAA | No jeopardy or adverse mod. | Appreciable adverse effects unlikely | None |
| | Chronic | NLAA | No jeopardy or adverse mod. | Appreciable direct adverse effects unlikely; some effects to snails and sensitive benthic invertebrates exposed through both diet and water. | None |
| Mercury | Acute | NLAA | No jeopardy or adverse mod. | Appreciable adverse effects unlikely | Use EPA's 2001 fish tissue criterion as adopted by IDEQ in 2005. |
| | Chronic | Likely to adversely affected listed species (LAA) | Adverse mod. and jeopardy | Idaho's aquatic life criterion predicted to result in bioaccumulation in salmonids to levels impairing reproduction and neurological/behavioral problems | |

| Constituent | Criteria | EPA's BA Conclusion for Salmonids | Biological Opinion Conclusion | Likely outcomes if water quality in critical habitats were allowed to be at criteria | Synopsis of reasonable and prudent alternatives or measures (see full text for details) |
|---|---|---|---|---|---|
| Nickel | Acute | NLAA | No jeopardy or adverse mod. | Appreciable adverse effects unlikely | Conduct appropriate biological monitoring and ensure an adequate zone of passage exists. |
| | Chronic | NLAA | No jeopardy or adverse mod. | Appreciable direct adverse effects unlikely; some effects to snails and sensitive benthic invertebrates exposed through both diet and water. | |
| Selenium | Acute | NLAA | No jeopardy or adverse mod. | Appreciable adverse effects unlikely | If receiving water concentrations are >2 µg/L then fish tissue monitoring is needed. |
| Selenium | Chronic | LAA | Adverse mod. and jeopardy | Predicted to bioaccumulate via food chain transfer to burdens linked to reduced growth and survival of juvenile salmonids. | If whole-body fish tissues in juvenile salmonids or adult sculpin are greater than 7.6 mg/kg dw, then remedial steps to reduce Se loading are needed. |
| Silver | Acute | NLAA | No jeopardy or adverse mod. | Reduced survival from short or long-term exposures. | Conduct appropriate biological monitoring and ensure an adequate zone of passage exists. |
| | Chronic | | | No separate chronic criterion; acute criterion was assumed by EPA to protect against chronic effects | |
| Zinc | Acute | NLAA | No jeopardy or adverse mod. | Some risk of lethality for sensitive life stages and sizes under some water chemistry conditions | Conduct appropriate biological monitoring and ensure an adequate zone of passage exists. |
| | Chronic | NLAA | No jeopardy or adverse mod. | Species adverse effects unlikely, Habitat indirect effects to co-occurring species possible | |

**Table 2.12.3. Summary of conclusions on the protectiveness of Idaho aquatic life criteria for organic chemicals**

| Chemical | EPA's BA Conclusion for Salmonids | Biological Opinion Conclusion | Reasonable and Prudent Measures to Minimize Take (RPMs) | Notes |
|---|---|---|---|---|
| **Organics** | | | | |
| Aldrin | NLAA | No jeopardy or adverse mod. | None | |
| Dieldrin | NLAA | No jeopardy or adverse mod. | None | |
| Chlordane | NLAA | No jeopardy or adverse mod. | None | |
| DDTs | NLAA | No jeopardy or adverse mod. | None | |
| Endosulfan | NLAA | No jeopardy or adverse mod. | None | Sub-criteria adverse effects documented, but no registered uses. |
| Endrin | NLAA | No jeopardy or adverse mod. | None | |
| Heptachlor | NLAA | No jeopardy or adverse mod. | None | Some uncertainty about chronic criterion protectiveness from bioaccumulation but fish consumption based criteria is also applicable and is sufficiently low to make risks of harm via bioaccumulation unlikely. |
| Lindane | NLAA | No jeopardy or adverse mod. | None | |

| Chemical | EPA's BA Conclusion for Salmonids | Biological Opinion Conclusion | Reasonable and Prudent Measures to Minimize Take (RPMs) | Notes |
|---|---|---|---|---|
| PCBs | NLAA | No jeopardy or adverse mod. | None | Product not manufactured in the USA. Human-health (HH) based criteria is lower than aquatic life criterion (ALC). HH criterion is likely sufficiently protective for all life stages and prey |
| PCP | NLAA | No jeopardy or adverse mod. | Use appropriate BMPs for construction in or around water. | At subcriteria concentrations, maladaptive behavior in rainbow trout occurred, and reduced growth in sockeye salmon |
| Toxaphene | NLAA | No jeopardy or adverse mod. | None | |

# 3. MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT ESSENTIAL FISH HABITAT CONSULTATIONS

The consultation requirement of section 305(b) of the MSA directs Federal agencies to consult with NMFS on all actions or proposed actions that may adversely affect EFH. The MSA (section 3) defines EFH as “those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity.” Adverse effects include the direct or indirect physical, chemical, or biological alterations of the waters or substrate and loss of, or injury to, benthic organisms, prey species and their habitat, and other ecosystem components, if such modifications reduce the quality or quantity of EFH. Adverse effects on EFH may result from actions occurring within EFH or outside EFH, and may include site-specific or EFH-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810). Section 305(b) also requires NMFS to recommend measures that can be taken by the action agency to conserve EFH.

This analysis is based, in part, on the BA provided by the EPA and descriptions of EFH for Pacific coast salmon (PFMC 1999) contained in the fishery management plans developed by the Pacific Fishery Management Council (PFMC) and approved by the Secretary of Commerce.

## 3.1. Essential Fish Habitat Affected by the Project

The proposed action and action area for this consultation are described in Section 1.4 of this document. Juvenile (rearing and migratory) and adult (migratory and spawning) spring/summer Chinook salmon EFH will be affected by the proposed action. Based on information provided in the BA and the analysis of effects presented in the ESA portion of this document, NMFS concludes that the proposed action would adversely affect Pacific Coast salmon EFH. The affected habitat potentially includes all of the critical habitat in Idaho as described in Table 1.4.1. NMFS has considered areas designated as critical habitat under the ESA to be synonymous with EFH. However, as a practical matter our EFH discussion in this section we will limited our description to existing mixing zones where discharges are currently occurring as described in the incidental take statement in Section 2.9. Together these mixing zones represent approximately 1 acre of EFH.

## 3.2. Adverse Effects on Essential Fish Habitat

Because the action area’s designated critical habitat is nearly identical to EFH for the effects are also the same. Effects to critical habitat were discussed in the previous Opinion (Section 2.4) and are incorporated by reference for the effects to EFH. In the preceding opinion, NMFS determined the action’s effects to critical habitat, and thus to EFH, will have the following adverse effects on EPH designated for salmon.

1. Disharges of toxic substances into EFH will result in reduced water quality in the water column and accumulate in sediments at levels that directly affects the suitability of the Habitat for listed species.

2. Discharges of toxic substances into EFH will result increased concentrations in macroinvertebrate tissues at concentrations that make them harmful in the diets of salmonids.

3. Discharges of toxic substances into EFH will result will result in fewer macroinvertebrates in the habitat resulting in reduced food sources in the habitat.

### 3.3. Essential Fish Habitat Conservation Recommendations

1. For mercury and arsenic use the recreational use standard, human health criteria for all water quality based effluent limitations in permits until new criteria can be promulgated.

2. For copper, cyanide and selenium assure that any permitting decisions allowing discharges contain an adequate zone of passage and include any other provisions available to reduce contaminant loading in the discharges until new criteria can be promulgated.

3. Provide adequate monitoring in NPDES permits to assure that mixture toxicity and bioaccumulation is not occurring in either the habitat or species.

NMFS expects that full implementation of these EFH Conservation Recommendations will protect, by avoiding or minimizing the adverse effects described in Section 3.2 above, on approximately 1 acre of designated EFH for Pacific coast salmon.

### 3.4. Statutory Response Requirement

As required by section 305(b)(4)(B) of the MSA, the Federal agency must provide a detailed response in writing to NMFS within 30 days after receiving an EFH Conservation Recommendation from NMFS. Such a response must be provided at least 10 days prior to final approval of the action if the response is inconsistent with any of NMFS' EFH Conservation Recommendations, unless NMFS and the Federal agency have agreed to use alternative time frames for the Federal agency response. The response must include a description of measures proposed by the agency for avoiding, mitigating, or offsetting the impact of the activity on EFH. In the case of a response that is inconsistent with NMFS Conservation Recommendations, the Federal agency must explain its reasons for not following the recommendations, including the scientific justification for any disagreements with NMFS over the anticipated effects of the action and the measures needed to avoid, minimize, mitigate, or offset such effects [50 CFR 600.920(k)(1)].

In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many conservation recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, NMFS asks that in your statutory reply to the EFH portion of this consultation, you clearly identify the number of conservation recommendations accepted.

### 3.5. Supplemental Consultation

The EPA must reinitiate EFH consultation with NMFS if the proposed action is substantially revised in a way that may adversely affect EFH, or if new information becomes available that affects the basis for NMFS' EFH conservation recommendations [50 CFR 600.920(l)].

## 4. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

Section 515 of the Treasury and General Government Appropriations Act of 2001 (Public Law 106-554) (Data Quality Act [DQA]) specifies three components contributing to the quality of a document. They are utility, integrity, and objectivity. This section of the Opinion addresses these DQA components, documents compliance with the DQA, and certifies that this Opinion has undergone pre-dissemination review.

### 4.1. Utility

"Utility" principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users. This ESA consultation concludes that the proposed project will not jeopardize the affected Snake River Basin steelhead and Snake River spring/summer Chinook salmon. Therefore, the EPA can implement this action in accordance with their authorities under the CWA and CERCLA. The intended users of this Opinion are the EPA and any of their cooperators, contractors, or permittees. Individual copies of this Opinion were provided to the agencies. This Opinion will be posted on the NMFS West Coast Region web site (http://www.westcoast.fisheries.noaa.gov) . The format and naming adheres to conventional standards for style.

### 4.2. Integrity

This consultation was completed on a computer system managed by NMFS in accordance with relevant information technology security policies and standards set out in Appendix III, "Security of Automated Information Resources," Office of Management and Budget Circular A-130; the Computer Security Act; and the Government Information Security Reform Act.

### 4.3. Objectivity

Information Product Category: Natural Resource Plan

***Standards:*** This consultation and supporting documents are clear, concise, complete, and unbiased; and were developed using commonly accepted scientific research methods. They adhere to published standards including the NMFS ESA Consultation Handbook, ESA regulations, 50 CFR 402.01, et seq., and the MSA implementing regulations regarding EFH, 50 CFR 600.

***Best Available Information:*** This consultation and supporting documents use the best available information, as referenced in the References section. The analyses in this Opinion/EFH consultation contain more background on information sources and quality.

***Referencing:*** All supporting materials, information, data and analyses are properly referenced, consistent with standard scientific referencing style.

***Review Process****:* This consultation was drafted by NMFS staff with training in ESA and MSA implementation, and reviewed in accordance with West Coast Region ESA quality control and assurance processes.

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# Appendix A

## A Review of Water Hardness Data for Idaho — 1979-2004

### Introduction

Water hardness is an important water-quality parameter, not only because it affects the quality of domestic water, but also because it affects the toxicity of metals to fish. Hardness mitigates metals toxicity, because $Ca^{2+}$ and $Mg^{2+}$ help keep fish from absorbing metals such as lead, arsenic, and cadmium into their bloodstream through their gills. The greater the hardness, the harder it is for toxic metals to be absorbed through the gills. NMFS retrieved hardness data collected in Idaho during the last 25 years from the USGS database in order to assess the relevant data, identify potential water quality problems, and locate regions where further investigation may be warranted.

The USGS National Water Quality Laboratory generally provides hardness analyses in terms of an equivalent concentration of calcium carbonate ($CaCO_3$). Approximately 3600 water samples from 324 sites on Idaho rivers and streams have been analyzed for hardness, defined as equivalent quantities of CaCO3, in milligrams per liter (mg/L), since 1979. Most of these samples were analyzed as part of the Statewide Water Quality Network; other samples were analyzed in the course of other water-quality investigations by the USGS Idaho District office. NMFS used these data to construct a general overview of water hardness over the last 25 years in the State of Idaho. A summary of the results of this preliminary analysis are given below.

A list of the 324 water-quality sampling sites, with USGS station identification numbers (STAID), descriptive names, and the number of samples from 1979-2004, is given in Table 1. A map showing locations of sites is given in figure 1a. The sites are located in 10 different hydrologic units (6-code HUCs). Approximately 75 percent (246) of the sites are within HUCs 170103 (Spokane, 128 sites) and 170402 (Upper Snake, 118 sites). The Middle Snake-Boise (170501), Salmon (170602), and Clearwater (170603) are represented by 33, 20, and 16 sites, respectively. The remaining HUCs have 11 or fewer sites.

The size and color of the symbols in Figure 1b represent the number of samples collected at each site since 1979. The number of samples per site ranges from 1 to 179; approximately 75 percent have been sampled fewer than 10 times. One site had no reported hardness value because calcium was below reporting limits on the single occasion it was sampled (station 12413850, Evans Creek near St. Maries, ID).

Forty sites were sampled intermittently over the course of 20 or more years, but only a few sites were regularly monitored over that period of time.

### Overview of Water Hardness in Idaho

Regional variation in maximum water hardness (defined here as variation in the highest value measured at each site between 1979 and 2004) is displayed symbolically in the map in Figure 2. (Recall that some of the sites are represented by only one sample). The color of the symbol indicates the maximum hardness value. Sites where the maximum hardness exceeded 140 mg/L are shown in shades of green. In general, the highest hardness values were found in southern and southeastern Idaho. Many of these sites are located in areas where carbonate rocks are present (Figure 2). This relation is predictable because rocks containing calcium carbonate are an obvious source of water-soluble $Ca^{+}$ ions, which contribute to hardness.

For domestic water use, water with hardness over 120 mg/L generally is considered "hard" (many different hardness classifications exist), and over 180 is "very hard". "Soft" water has hardness less than 60 mg/L. According to this classification, 56 of the sites would be considered to have soft water and 72 sites would be considered to have very hard water.

Maximum and minimum hardness values data for the 324 sites in Idaho are shown in Figure 2 and 5. A cumulative distribution plot of the data is given in Figure 3.

In addition to important effects of hardness on metals toxicity, the different major ions that contribute to hardness may affect toxicity differently (Naddy *et al.* 2002). Therefore it is useful to examine the Ca:Mg ratio. The Ca:Mg ratio in these data ranges over two orders of magnitude, from 0.9 to 90. The average ratio is 4.9; the median ratio is 3.9. About 53% of the sites have a maximum Ca:Mg ratio less than 4. The sites having ratios greater than 4 are mainly in central Idaho and in the Boise River Basin. Some of the lowest ratios are found in the Coeur d'Alene region and in south-central and southeastern Idaho (figure 4).

## Metals

Because low, not high, water hardness directly contributes to metals toxicity for aquatic biota, the minimum hardness value measured at each site is shown on the map in figure 5. This map clearly indicates that potential metals toxicity problems exist in northern, western, and central Idaho, where minimum hardness measurements generally were less than approximately 50 mg/L. Of particular concern are sites in HUC 170103 (Spokane), a region where high metal concentrations in rivers and streams are known to exist.

The EPA has established national recommended water quality criteria (EPA, 2002) to help States and Tribes to establish water quality standards under the Clean Water Act. EPA lists the water quality criteria for 158 pollutants. Because the toxicity of certain dissolved metals, including cadmium, lead, nickel, and zinc, is hardness-dependent, the EPA recommended criteria for these metals, in µg/L are calculated by using equations of the form

$$\mathrm{EXP}^{(m_{a,c}(\ln H)+b_{a,c})},$$

where $m_{a,c}$ and $b_{a,c}$ are empirically-determined constants (a = acute, c = chronic), different for each metal, and H is the hardness of the water. A conversion factor for fresh water is also applied. Two criteria are commonly used: the CMC (criterion maximum concentration; for acute toxicity) and the CCC (criterion continuous concentration; for chronic toxicity).

The EPA does not recommend using a low-end hardness "cap" for calculating CMC and CCC in cases where hardness is unusually low, asserting that doing so may provide less protection for aquatic organisms than intended by Guidelines given in EPA 822/R-85-100. Nevertheless, some agencies have used a low-end floor (e.g. 25 mg/L) for establishing standards, substituting 25 mg/L for measured values less than 25 mg/L.

As a demonstration of how hardness is used to establish water quality criteria, we retrieved available data for Cd for the 324 sites. In all cases, the water sample analyzed for Cd was taken at the same date and time as the hardness sample.

Among the 324 sites for which we compiled hardness data, 167 sites had at least one sample that was analyzed for Cd (1,287 total samples analyzed for Cd). The minimum reporting limit for Cd by the National Water Quality Lab has varied throughout the last 25 years; 1 µg/L and 0.04 µg/L were the two reporting limits encountered in the data set. Cadmium was detected at 90 of these sites, in a total of 758 samples. The Cd values for these sites are shown on the map in Figure 6a.

NMFS calculated CMCs for Cd for each of the 90 sites using the EPA equation and parameters given by EPA (2002). In cases where more than one sample was available at a site, we used the minimum hardness value measured at the site with the corresponding Cd value (not necessarily the maximum measured Cd value), assuming this represented the potentially most toxic "instantaneous" situation.

To assess the effect of using different lower floor for hardness, NMFS calculated CMCs for these sites three ways: 1) Using actual measured values for samples having H < or equal to 25 mg/L; 2) substituting H = 10 mg/L for actual values < or = 10 mg/L; and 3) substituting H = 25 mg/L for actual values < or = 25 mg/L.

In the first case (no cap, using measured hardness values to calculate the CMC), 70 of the 90 sites exceeded the criterion. The criterion values ranged from 0.09 µg/L to 36 µg/L (fig. 6b; table 1).

Using a low-end floor of H =10 mg/L (that is, changing all hardness values less than 10 mg/L to 10 mg/L) affected the CMC of 9 sites, and resulted in 69 of the samples exceeding the criterion. In other words, one site that had previously exceeded the criterion now met the criterion. The measured Cd value at this site was 0.11 µg/L; the unadjusted minimum hardness was 5 mg/L. Setting the cap at H = 25 mg/L affected the CMC of 48 out of 90 sites. In this case, 67 samples exceeded the criterion (74%).

When there was no lower hardness floor, the criterion was exceeded in 70 of 90 of the samples (78%) (Figure 6c). This cursory analysis suggests that setting low-end floor for hardness when calculating the CMC and CCC for Cd could make a difference in whether or not a site met the CMC, albeit in a small number of cases.

At 5 of the sites, measured Cd was less than or equal to 0.6 µg/L, but low hardness resulted in potentially toxic situations for aquatic biota. All these samples are in the Spokane-Coeur d'Alene region. On the other hand, a sample from Bannock County contained 20 µg/L Cd but had a minimum hardness of 280 mg/L. The CMC for this site was approximately 5 µg/L Cd, ordinarily considered a high concentration of Cd; the water's high hardness mitigated the toxicity of Cd to some degree, but not sufficiently to meet the criterion.

One site in the Spokane/Coeur d'Alene region contained 2.42 µg/L of Cd, but met the CMC of 36.6 µg/L because the hardness was 2100 mg/L. Clearly this demonstrates a need for an upper cap for hardness as well as a lower floor. EPA provides guidance for hardness > 400 mg/L by recommending two options: 1) calculate the criterion using a Water Effect Ratio (WER) of 1.0 and use a hardness of 400 mg/L in the equation, or 2) calculate the criterion using a WER and the actual hardness of the water. If this sample had been calculated using H = 400 mg/L, the CMC would have been 7.44 µg/L and the sample still would have met the criterion.

A "quick lookup" graph showing the CMC for Cd (EPA, 2002), in µg/L, as a function of water hardness is shown in Figure 7. This graph permits an estimate of the Cd CMC to be easily determined if the hardness of the water is known.  Graphs such as these are easily constructed for other metals by entering the appropriate equations in a spreadsheet.

## Trends in water hardness

For this analysis, NMFS selected sites with long-term hardness records (10 or more readings over at least 13 years) to test for temporal trends in hardness.  The maximum hardness measured within each year in the period of record was chosen in an effort to compensate for changes in hardness related to seasonal discharge.  The Mann-Kendall trend test was integrated into an Excel spreadsheet, which performed the test by comparing each measurement with all the other previous measurements, one at a time, and assigning a “+1” or “-1“, depending on whether that measurement is larger or smaller.  We compared the sum of all the pluses and minuses  (the “S-statistic”) with values in a table to determine if a statistically significant (in this case, $p \leq 0.05$) trend was present.

The results of the trend analysis are given in Table 1.  Of the 38 tested sites, 3 showed statistically significant positive trends (increasing hardness), 7 showed negative trends, and 28 showed no significant trends.  Note that even though some significant trends were found, the magnitude of the changes relative to the total water hardness does not appear to be relevant from a water quality management point of view.

## Summary

NMFS analyzed data from 324 water quality sampling sites from 1979 to 2004 to gain insight to water hardness in Idaho.  Sites for which water hardness data were collected in the last 25 years are clustered in northern Idaho (Spokane/Coeur d’Alene region), the Snake River and its tributaries in south-central Idaho, Big Lost River basin, the Portneuf River and its tributaries, and the Lower Boise River.  A wide range of hardness values was found; some of the statewide variation is apparently related to the bedrock geology.  Approximately 38% of the maximum hardness values are classified as “hard” (>120 mg/L).  High hardness values were found mainly in south-central and southeastern Idaho.  Very low hardness waters were predominant in northern Idaho and were commonly associated with high metal concentrations.

Cd was detected at 90 of the sites for which hardness data were compiled (approximately 27%).  Concentration Maximum Criteria for Cd were determined for these 90 sites, using concurrent hardness values; 70 sites failed to meet the EPA-recommended criterion.  Most of the noncompliant sites are located in the Spokane/Coeur d’Alene region, but one noncompliant site occurs in each of the following counties: Ada, Bannock, Canyon, Idaho, Nez Perce, Owyhee, and Valley.

The above discussion suggests that the hardness level of receiving waters is an important water quality parameter that needs to be considered in the development of water quality criteria for some metals.  There are areas in Idaho with hardness lower that the current floor used in calculating discharge limits of 25mg/L and that results increasing risk and harm to listed salmon and steelhead.

## References for Appendix A

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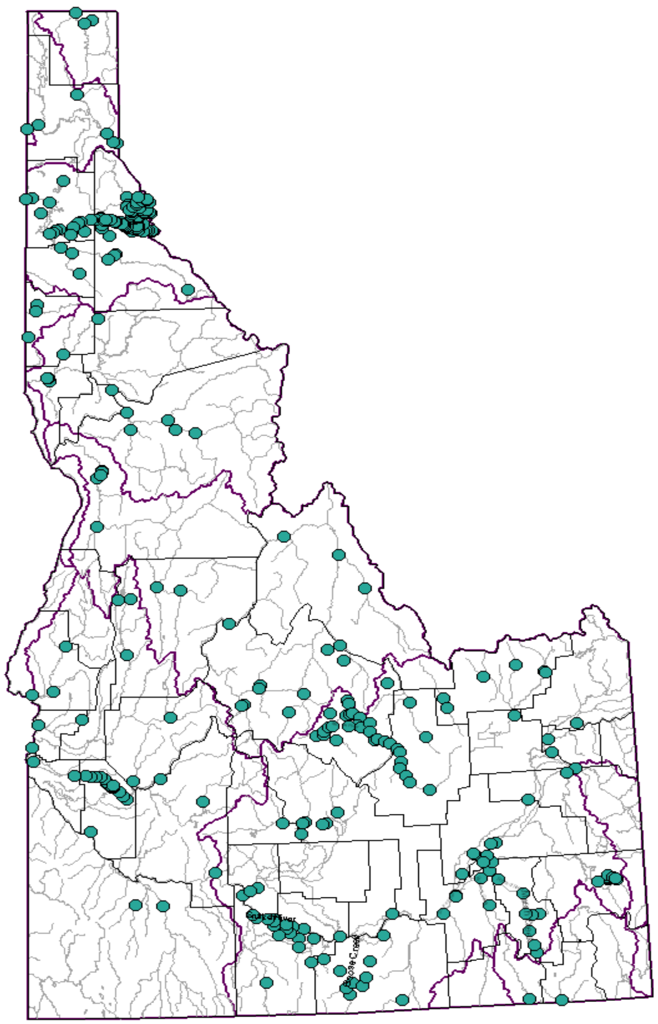


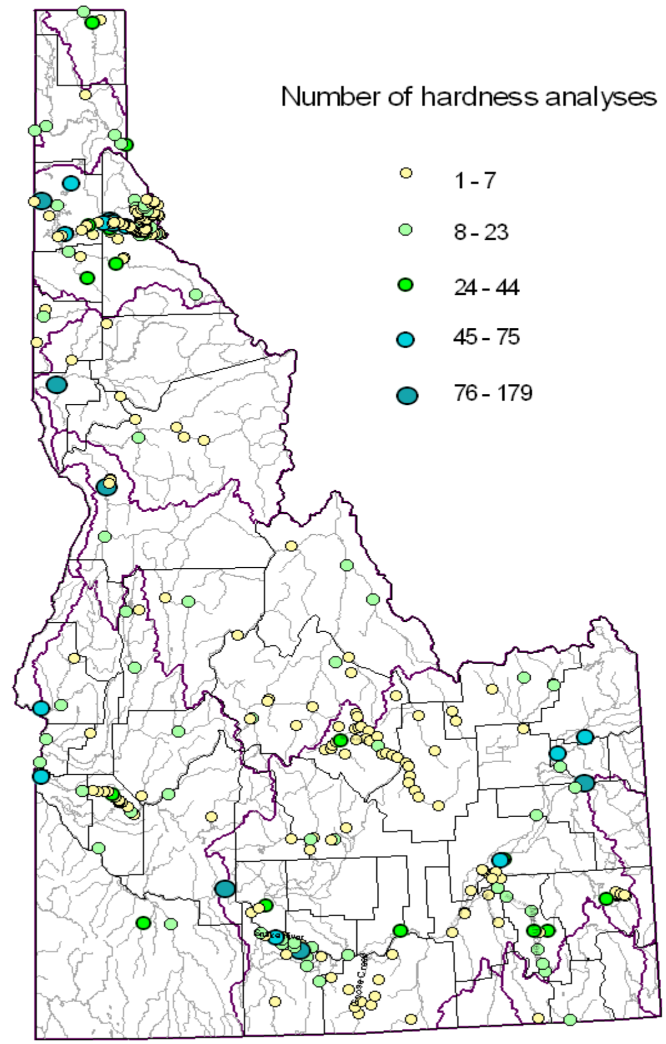



Figure 1a.  USGS surface water sampling sites where at least one sample was analyzed for water hardness from 1979 to 2004.

Figure 1b.  Number of hardness analyses performed from 1979 to 2004.

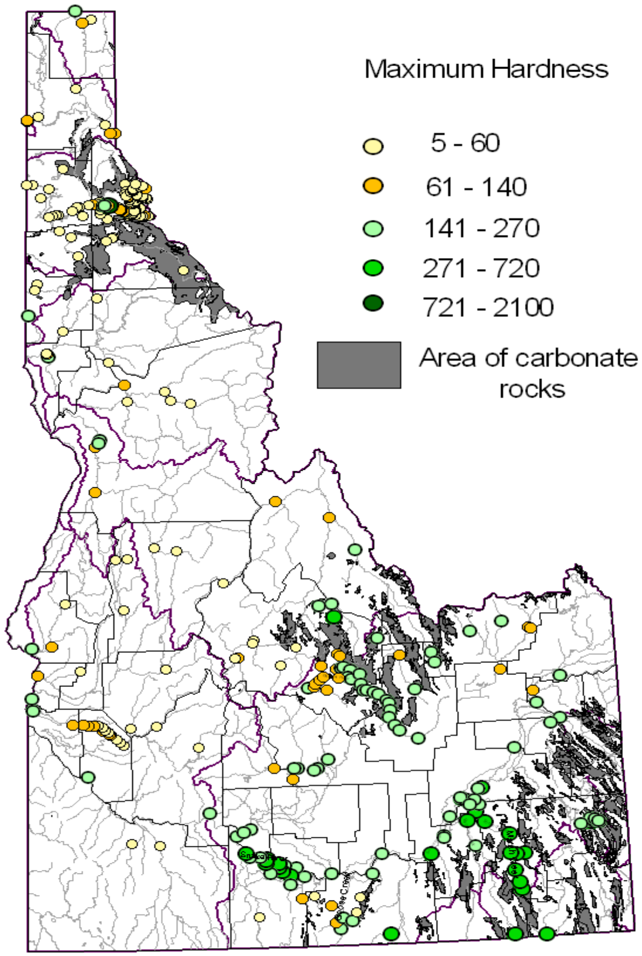



Figure 2.  Map showing maximum hardness measured from 1979 to 2004 and areas of carbonate rock in Idaho.

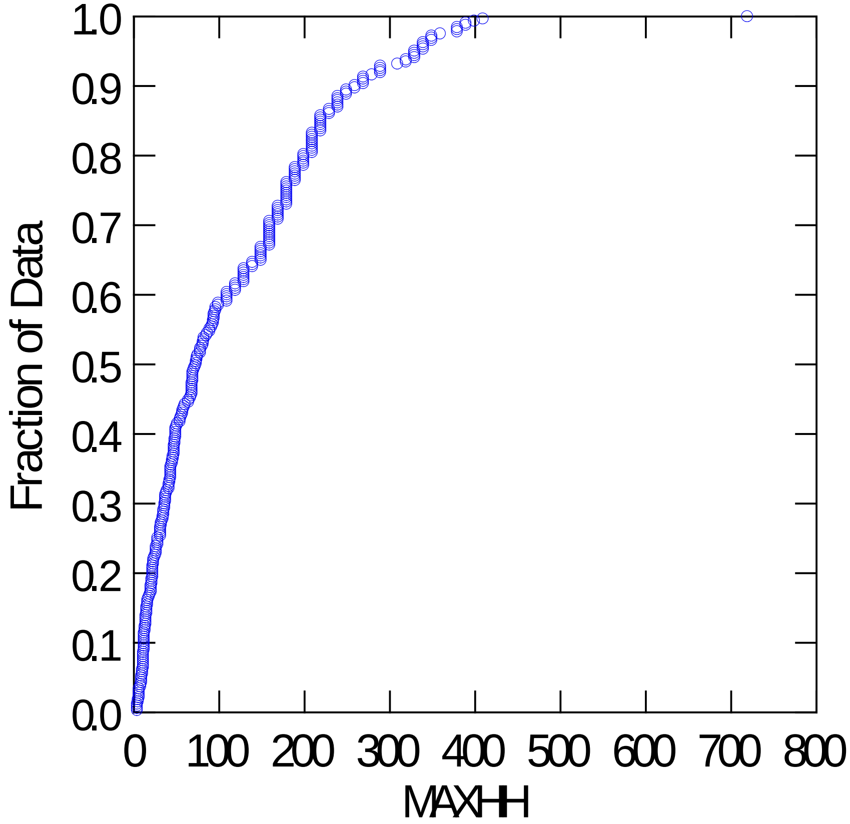



Figure 3.  Cumulative distribution plot of maximum hardness (n = 323). Plot does not include one sample of maximum hardness equal to  2100 mg/L.

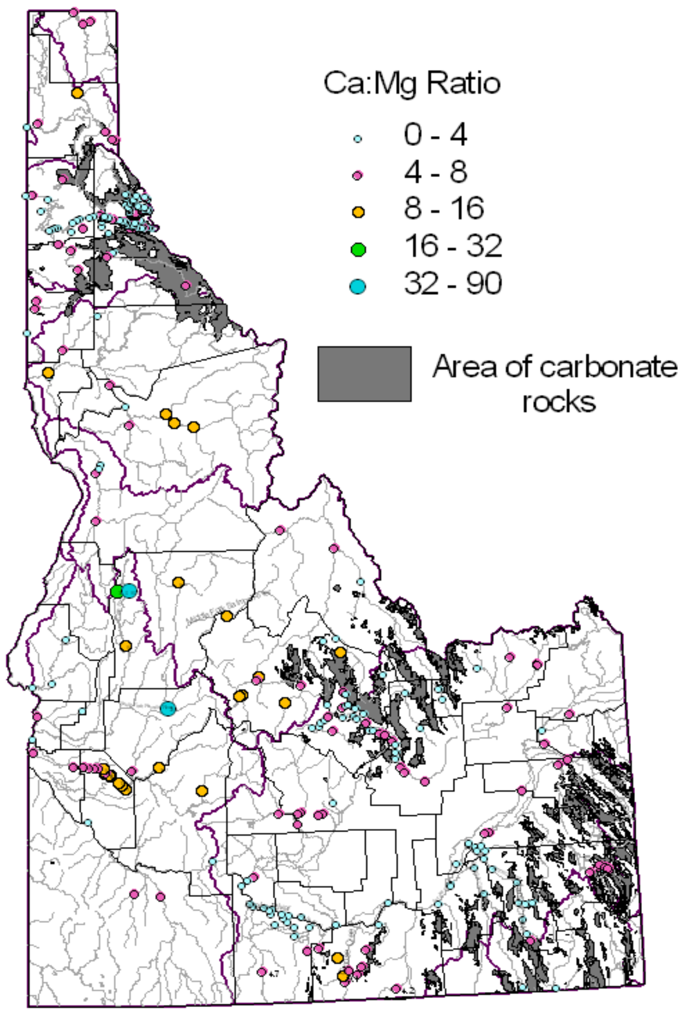
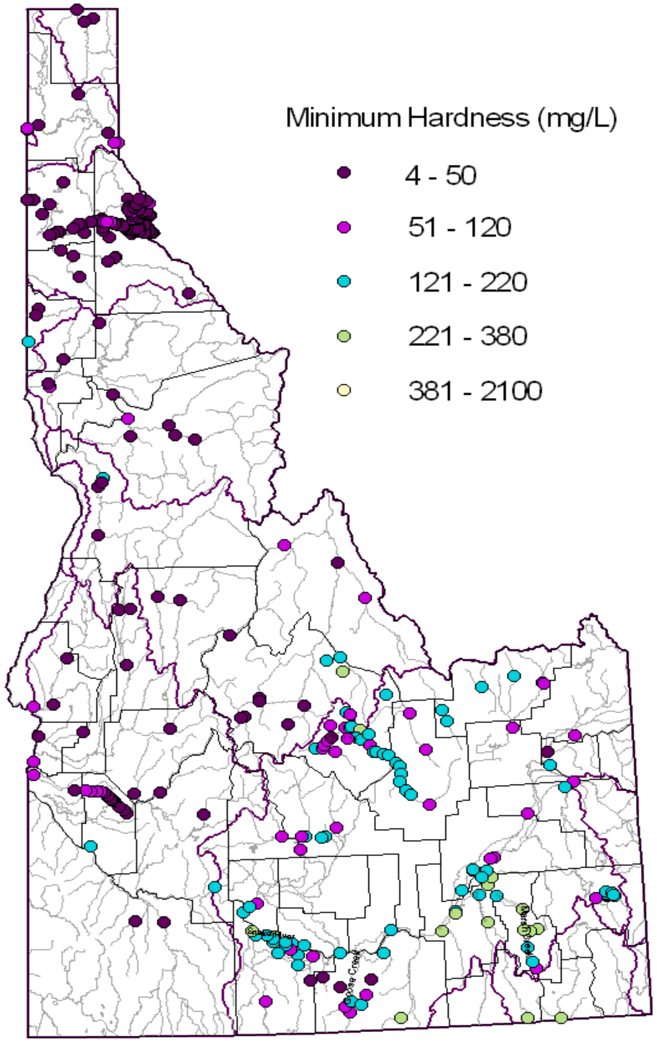



Figure 4.  Map showing maximum Ca:Mg ratio at 324 water quality sites in Idaho.



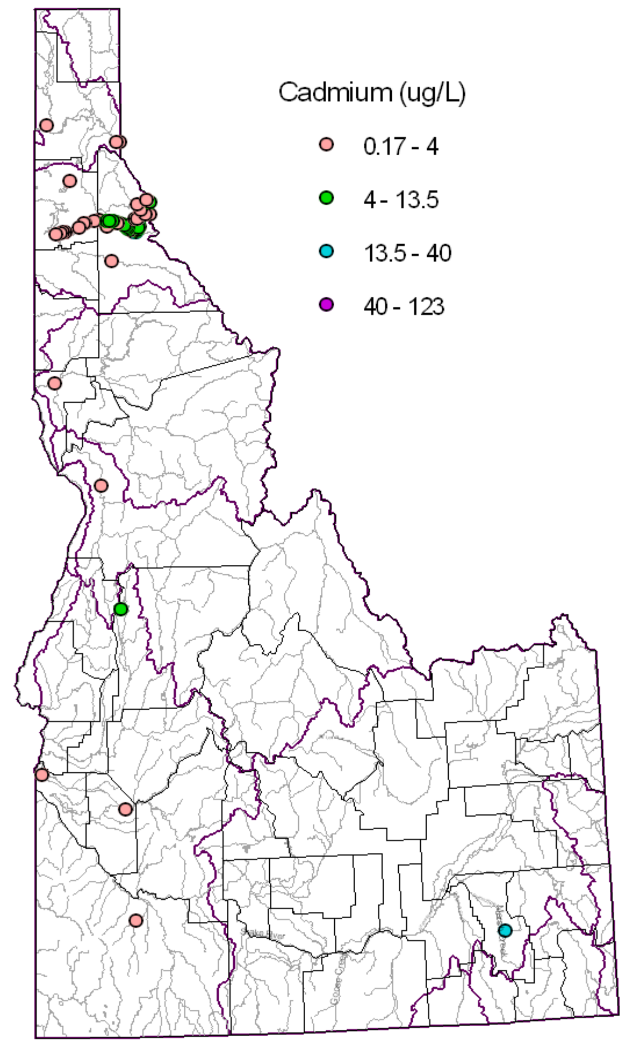




Figure 5. Minimum hardness measured at sampling sites from 1979 to 2004.

Figure 6a. Cadmium value associated with minimum hardness value (n = 90)

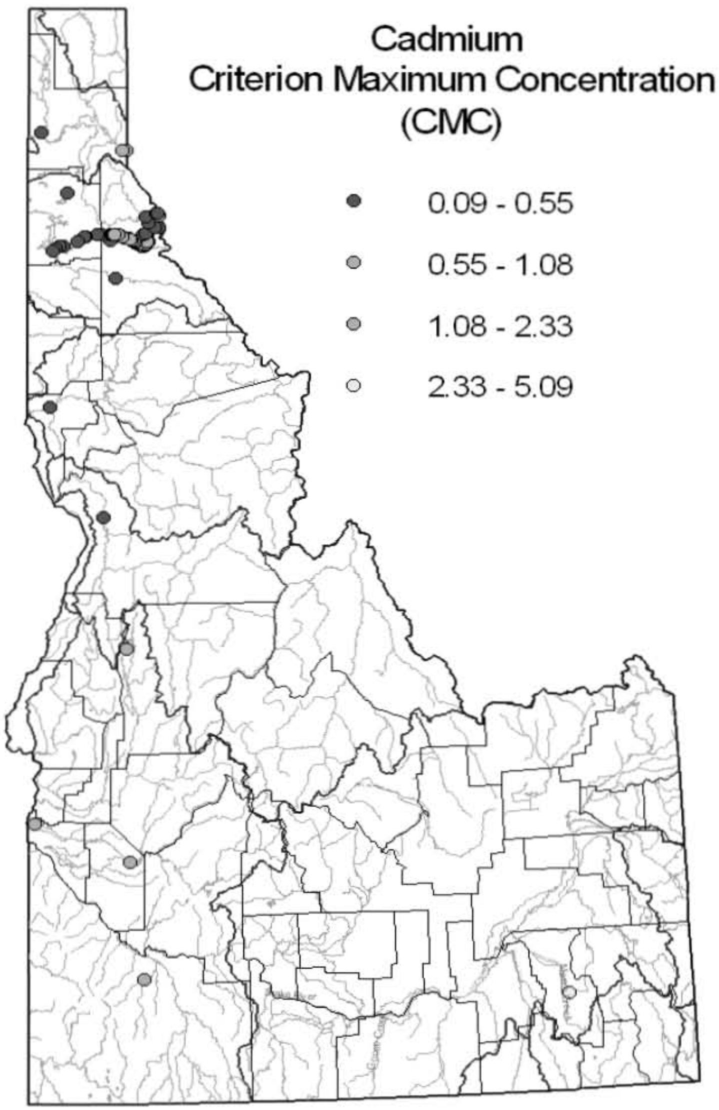



Figure 6b.  Criterion Maximum Concentration (CMC) for cadmium calculated with  no lower floor on hardness (n = 90).


Figure 6c.  Sites where Cd CMC was met (n = 20) and exceeded (n = 70).

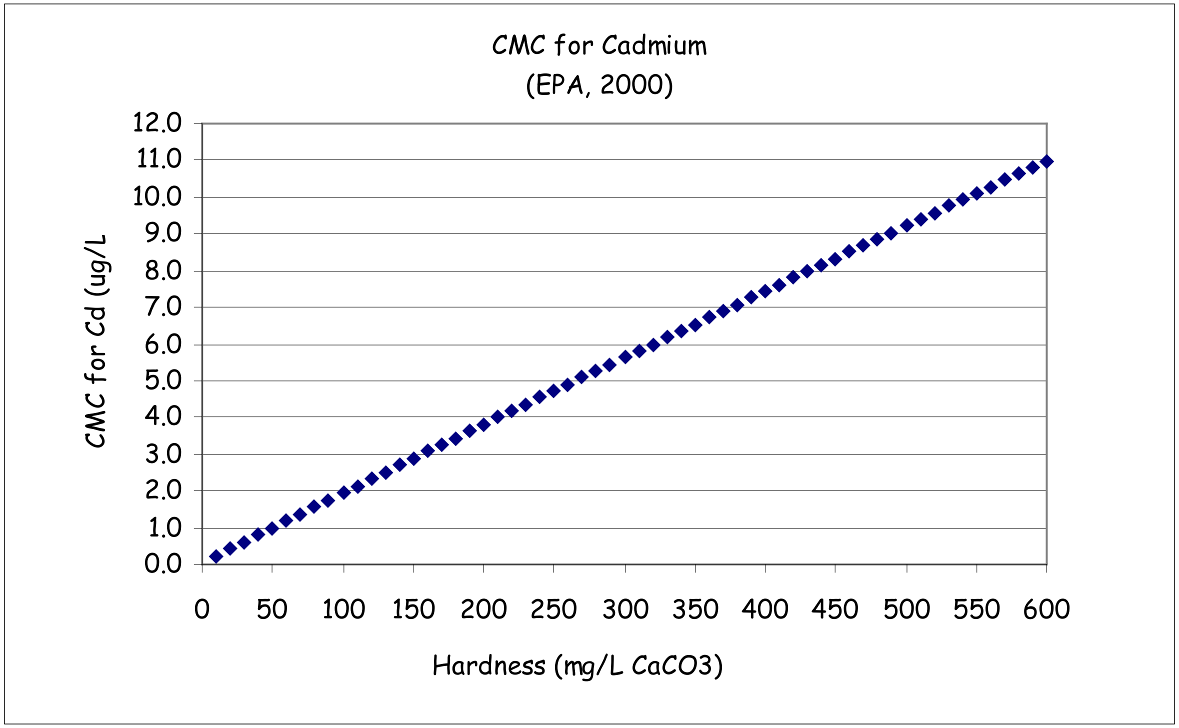



Figure 7.  "Quick lookup" graph for the Cd CMC.  If water hardness is known, the CMC can be visually estimated.  Similar graphs for other metals can easily be constructed.

**Table A.1.** Summary of hardness values in Idaho, downloaded from the USGS National Water Information System 1979-2004 (http://nwis.waterdata.usgs.gov) [H – water hardness in mg/L as $CaCO_3$; Ca – calcium; Mg – magnesium, Ave-average. All concentrations in mg/L

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 10092700 | Bear River at Idaho-Utah State Line | 42.013 | -111.919 | 11 | 300 | 330 | 390 | 48 | 63.2 | 76 | 33 | 41.6 | 52 | 0.9 | 1.6 | 2.2 |
| 10125500 | Malad River at Woodruff ID | 42.03 | -112.229 | 4 | 370 | 562.5 | 720 | 68 | 127.0 | 160 | 49 | 59.0 | 83 | 1.4 | 2.2 | 3.2 |
| 12316800 | Mission Creek nr Copeland ID | 48.932 | -116.333 | 5 | 8 | 8.6 | 10 | 2.3 | 2.5 | 2.7 | 0.33 | 0.5 | 0.73 | 3.7 | 5.2 | 7.9 |
| 12318500 | Kootenai River nr Copeland ID | 48.912 | -116.416 | 34 | 36 | 97.6 | 140 | 10 | 27.0 | 40 | 2.7 | 7.2 | 10 | 3.5 | 3.8 | 4.1 |
| 12322000 | Kootenai River at Porthill ID | 48.996 | -116.508 | 23 | 46 | 99.3 | 150 | 13 | 27.7 | 40 | 3.3 | 7.3 | 11 | 3.5 | 3.8 | 4.1 |
| 12391950 | Clark Fork River Below Cabinet Gorge Dam ID | 48.088 | -116.073 | 39 | 64 | 86.2 | 97 | 17.7 | 24.0 | 27.5 | 4.8 | 6.3 | 7.54 | 3.4 | 3.8 | 4.2 |
| 12392000 | Clark Fork at Whitehorse Rapids nr Cabinet ID | 48.093 | -116.118 | 16 | 70 | 88.3 | 96 | 20 | 24.8 | 27 | 4.8 | 6.4 | 7 | 3.6 | 3.9 | 4.2 |
| 12392155 | Lightning Creek at Clark Fork ID | 48.151 | -116.182 | 16 | 4 | 9.3 | 15 | 1.29 | 2.6 | 3.94 | 0.272 | 0.7 | 1.18 | 3.0 | 4.1 | 4.8 |
| 12392300 | Pack River nr Colburn ID | 48.42 | -116.501 | 6 | 5 | 10.2 | 17 | 1.5 | 3.3 | 5.6 | 0.004 | 0.4 | 0.7 | 6.0 | 8.3 | 12.0 |
| 12395000 | Priest River nr Priest River ID | 48.209 | -116.914 | 18 | 18 | 25.3 | 36 | 5 | 7.2 | 10.7 | 1.4 | 1.8 | 2.32 | 3.3 | 3.9 | 4.7 |
| 12395500 | Pend Oreille River at Newport Wa | 48.182 | -117.033 | 6 | 71 | 78.7 | 83 | 20 | 22.0 | 23 | 5.1 | 5.8 | 6.2 | 3.7 | 3.8 | 3.9 |
| 12395502 | Pend Oreille River at Us Hwy No.2 at Newport Wa | 48.185 | -117.033 | 9 | 62 | 78 | 87 | 17 | 21.4 | 24 | 4.8 | 5.9 | 6.6 | 3.4 | 3.7 | 3.9 |
| 12411000 | Nf Coeur D Alene R ab Shoshone Ck nr Prichard ID | 47.707 | -115.977 | 17 | 13 | 21.2 | 27 | 2.99 | 4.8 | 6.26 | 1.35 | 2.2 | 2.89 | 2.0 | 2.1 | 2.2 |
| 12411935 | Prichard Creek at Mouth at Prichard ID | 47.657 | -115.968 | 18 | 6 | 10.8 | 14 | 1.68 | 2.8 | 3.77 | 0.519 | 0.9 | 1.2 | 3.0 | 3.1 | 3.2 |
| 12411950 | Beaver Cr ab Carpenter Gulch nr Prichard, ID | 47.633 | -115.979 | 2 | 21 | 27 | 33 | 5.49 | 6.9 | 8.26 | 1.67 | 2.3 | 2.93 | 2.8 | 3.1 | 3.3 |
| 12413000 | Nf Coeur D Alene River at Enaville ID | 47.569 | -116.252 | 102 | 10 | 18.6 | 25 | 2.48 | 4.5 | 6.2 | 0.937 | 1.7 | 2.3 | 2.3 | 2.6 | 3.2 |
| 12413025 | Little Nf Sf Coeur D Alene Riv Abv Mouth nr Mullan | 47.465 | -115.722 | 1 | 7 | 7 | 7 | 1.68 | 1.7 | 1.68 | 0.677 | 0.7 | 0.677 | 2.5 | 2.5 | 2.5 |
| 12413030 | Sf Coeur D Alene R Bl Obrien Gulch nr Larson, ID | 47.467 | -115.733 | 2 | 8 | 10.5 | 13 | 2.29 | 2.9 | 3.43 | 0.672 | 0.9 | 1.05 | 3.3 | 3.3 | 3.4 |
| 12413040 | Sf Coeur D Alene R Abv Deadman Gulch nr Mullan ID | 47.473 | -115.766 | 20 | 12 | 39.7 | 69 | 3.25 | 10.7 | 19.2 | 0.963 | 3.2 | 5.11 | 2.8 | 3.3 | 3.9 |
| 12413100 | Boulder Creek at Mullan ID | 47.469 | -115.796 | 1 | 20 | 20 | 20 | 6.2 | 6.2 | 6.2 | 1 | 1.0 | 1 | 6.2 | 6.2 | 6.2 |
| 12413103 | Sf Coeur D Alene R ab Slaughterhse Gulch at Mullan | 47.466 | -115.813 | 1 | 18 | 18 | 18 | 5.16 | 5.2 | 5.16 | 1.32 | 1.3 | 1.32 | 3.9 | 3.9 | 3.9 |
| 12413104 | Sf Coeur D Alene R Bl Trowbridge Gulch nr Wallace | 47.474 | -115.869 | 1 | 23 | 23 | 23 | 6.32 | 6.3 | 6.32 | 1.8 | 1.8 | 1.8 | 3.5 | 3.5 | 3.5 |
| 12413118 | Canyon Creek at Burke, ID | 47.521 | -115.818 | 16 | 4 | 9.2 | 12 | 1.19 | 2.4 | 3.31 | 0.338 | 0.7 | 1 | 3.1 | 3.3 | 3.6 |
| 12413120 | Canyon Creek at Gem ID | 47.508 | -115.867 | 2 | 10 | 13 | 16 | 2.71 | 3.5 | 4.38 | 0.683 | 0.9 | 1.21 | 3.6 | 3.8 | 4.0 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 12413123 | Canyon Creek at Woodland Park ID | 47.489 | -115.889 | 18 | 9 | 31 | 45 | 2.49 | 8.7 | 12.7 | 0.623 | 2.2 | 3.2 | 3.8 | 3.9 | 4.0 |
| 12413125 | Canyon Creek ab Mouth at Wallace, ID | 47.473 | -115.914 | 43 | 10 | 36 | 58 | 2.77 | 10.2 | 16.2 | 0.692 | 2.6 | 4.25 | 3.5 | 3.9 | 4.2 |
| 12413126 | Ninemile Cr ab Mouth Of Ef Ninemile Cr nr Blackcld | 47.514 | -115.898 | 1 | 95 | 95 | 95 | 22.6 | 22.6 | 22.6 | 9.43 | 9.4 | 9.43 | 2.4 | 2.4 | 2.4 |
| 12413127 | Ef Ninemile Creek Abv Mouth nr Blackcloud ID | 47.513 | -115.893 | 17 | 8 | 24.8 | 42 | 2.56 | 8.0 | 13.8 | 0.396 | 1.2 | 1.9 | 6.0 | 6.9 | 7.3 |
| 12413130 | Ninemile Creek ab Mouth at Wallace, ID | 47.479 | -115.919 | 44 | 16 | 50.2 | 75 | 4.42 | 14.0 | 21.4 | 1.05 | 3.7 | 5.89 | 3.1 | 3.8 | 4.3 |
| 12413131 | Sf Coeur D Alene R Abv Placer Cr at Wallace ID | 47.475 | -115.928 | 1 | 21 | 21 | 21 | 5.77 | 5.8 | 5.77 | 1.63 | 1.6 | 1.63 | 3.5 | 3.5 | 3.5 |
| 12413140 | Placer Creek at Wallace ID | 47.463 | -115.937 | 18 | 19 | 36 | 48 | 5.85 | 10.8 | 14.1 | 1.07 | 2.2 | 3.12 | 4.5 | 4.9 | 5.6 |
| 12413150 | Sf Coeur D Alene River at Silverton ID | 47.492 | -115.954 | 19 | 18 | 46.1 | 69 | 4.96 | 12.5 | 18.9 | 1.3 | 3.6 | 5.53 | 3.2 | 3.5 | 3.8 |
| 12413151 | Lake Creek ab Mouth nr Silverton, ID | 47.49 | -115.952 | 1 | 27 | 27 | 27 | 7 | 7.0 | 7 | 2.29 | 2.3 | 2.29 | 3.1 | 3.1 | 3.1 |
| 12413168 | Twomile Creek ab Mouth at Osburn, ID | 47.51 | -115.995 | 2 | 23 | 31.5 | 40 | 6.87 | 9.0 | 11.1 | 1.42 | 2.2 | 3.01 | 3.7 | 4.3 | 4.8 |
| 12413169 | Sf Coeur D Alene R Blw Twomile Cr nr Osburn ID | 47.51 | -115.996 | 10 | 19 | 39.1 | 70 | 5.25 | 10.6 | 18.6 | 1.38 | 3.1 | 5.63 | 3.3 | 3.5 | 3.8 |
| 12413174 | Terror Gulch Creek ab Mouth nr Osburn, ID | 47.514 | -116.021 | 2 | 35 | 41.5 | 48 | 7.44 | 9.0 | 10.6 | 3.88 | 4.6 | 5.22 | 1.9 | 2.0 | 2.0 |
| 12413175 | Sf Coeur D Alene R at Terror Gulch at Osburn ID | 47.514 | -116.022 | 1 | 22 | 22 | 22 | 6.24 | 6.2 | 6.24 | 1.66 | 1.7 | 1.66 | 3.8 | 3.8 | 3.8 |
| 12413179 | Sf Coeur D Alene R ab Big Creek nr Big Creek, ID | 47.527 | -116.049 | 1 | 23 | 23 | 23 | 6.29 | 6.3 | 6.29 | 1.69 | 1.7 | 1.69 | 3.7 | 3.7 | 3.7 |
| 12413185 | Big Creek ab Mouth nr Big Creek, ID | 47.529 | -116.051 | 1 | 12 | 12 | 12 | 3.2 | 3.2 | 3.2 | 0.946 | 0.9 | 0.946 | 3.4 | 3.4 | 3.4 |
| 12413190 | Moon Creek Abv Mouth at Elk Creek ID | 47.533 | -116.058 | 16 | 15 | 27.4 | 38 | 3.56 | 6.2 | 8.49 | 1.49 | 2.9 | 4.07 | 2.0 | 2.2 | 2.4 |
| 12413204 | Montgomery Creek ab Mouth nr Elizabeth Park, ID | 47.531 | -116.088 | 1 | 13 | 13 | 13 | 3.21 | 3.2 | 3.21 | 1.16 | 1.2 | 1.16 | 2.8 | 2.8 | 2.8 |
| 12413209 | Elk Creek ab Mouth at Elizabeth Park, ID | 47.53 | -116.09 | 1 | 16 | 16 | 16 | 4.3 | 4.3 | 4.3 | 1.36 | 1.4 | 1.36 | 3.2 | 3.2 | 3.2 |
| 12413210 | Sf Coeur D Alene at Elizabeth Park nr Kellogg ID | 47.531 | -116.092 | 64 | 18 | 50 | 79 | 5.12 | 13.4 | 20.9 | 1.38 | 4.0 | 6.5 | 3.1 | 3.3 | 3.7 |
| 12413250 | Sf Coeur D Alene T at Kellogg, ID | 47.545 | -116.134 | 10 | 19 | 54.3 | 71 | 5.32 | 14.4 | 19 | 1.43 | 4.5 | 6 | 3.0 | 3.3 | 3.8 |
| 12413290 | Government Gulch nr Mouth at Smelterville ID | 47.545 | -116.166 | 16 | 11 | 35.1 | 61 | 2.96 | 10.1 | 17.7 | 0.783 | 2.4 | 4.03 | 3.8 | 4.1 | 4.4 |
| 12413300 | Sf Coeur D Alene River at Smelterville ID | 47.549 | -116.174 | 23 | 22 | 74 | 180 | 6 | 19.3 | 45.2 | 1.65 | 6.3 | 16.7 | 2.7 | 3.2 | 3.8 |
| 12413360 | Ef Pine Creek Abv Gilbert Cr Near Pinehurst ID | 47.44 | -116.174 | 1 | 5 | 5 | 5 | 1.31 | 1.3 | 1.31 | 0.361 | 0.4 | 0.361 | 3.6 | 3.6 | 3.6 |
| 12413440 | Pine Creek ab Mouth Of Ef Pine Cr at Pine, ID | 47.487 | -116.241 | 1 | 6 | 6 | 6 | 1.5 | 1.5 | 1.5 | 0.441 | 0.4 | 0.441 | 3.4 | 3.4 | 3.4 |
| 12413445 | Pine Creek Blw Amy Gulch nr Pinehurst ID | 47.516 | -116.24 | 44 | 4 | 10.5 | 16 | 1.14 | 2.7 | 4.26 | 0.348 | 0.9 | 1.46 | 2.8 | 3.0 | 3.3 |
| 12413460 | Pine Creek ab Mouth nr Pinehurst, ID | 47.547 | -116.227 | 1 | 5 | 5 | 5 | 1.41 | 1.4 | 1.41 | 0.427 | 0.4 | 0.427 | 3.3 | 3.3 | 3.3 |
| 12413470 | Sf Coeur D Alene River nr Pinehurst ID | 47.552 | -116.237 | 115 | 17 | 70.2 | 190 | 4.59 | 18.4 | 50 | 1.34 | 5.9 | 18 | 2.3 | 3.2 | 4.7 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 12413490 | Sf Coeur D Alene River at Enaville ID | 47.56 | -116.251 | 1 | 42 | 42 | 42 | 12 | 12.0 | 12 | 2.9 | 2.9 | 2.9 | 4.1 | 4.1 | 4.1 |
| 12413500 | Coeur D Alene River nr Cataldo ID | 47.555 | -116.323 | 75 | 12 | 35.4 | 65 | 3.02 | 9.1 | 16 | 1.12 | 3.1 | 6.1 | 2.6 | 3.0 | 3.7 |
| 12413700 | Latour Creek Abv Baldy Creek nr Cataldo ID | 47.469 | -116.439 | 3 | 4 | 4.7 | 5 | 1.2 | 1.2 | 1.3 | 0.27 | 0.4 | 0.5 | 2.4 | 3.7 | 4.4 |
| 12413755 | Coeur D Alene River Blw Latour Creek nr Cataldo ID | 47.551 | -116.367 | 3 | 18 | 38.7 | 50 | 4.56 | 9.8 | 12.7 | 1.57 | 3.4 | 4.35 | 2.8 | 2.9 | 2.9 |
| 12413810 | Coeur D Alene River at Rose Lake ID | 47.537 | -116.472 | 31 | 12 | 28.9 | 48 | 3.06 | 7.3 | 12.1 | 1.07 | 2.6 | 4.2 | 2.5 | 2.8 | 3.2 |
| 12413815 | Coeur D Alene River Blw Rose Creek nr Rose Lake ID | 47.535 | -116.499 | 4 | 19 | 36 | 51 | 5.05 | 9.1 | 13 | 1.62 | 3.2 | 4.53 | 2.6 | 2.9 | 3.1 |
| 12413825 | Coeurdalene Riv ab Kilarney Lk Outlet nr Rose Lake | 47.506 | -116.554 | 4 | 18 | 34 | 48 | 4.65 | 8.5 | 12.1 | 1.57 | 3.1 | 4.22 | 2.6 | 2.8 | 3.0 |
| 12413850 | Evans Creek nr St Maries ID | 47.449 | -116.567 | 0 | | | | 0.02 | 0.02 | 0.02 | 0.004 | 0.004 | 0.004 | 5.0 | 5.0 | 5.0 |
| 12413858 | Coeur D Alene River Blw Blue Lake nr Harrison ID | 47.48 | -116.699 | 7 | 17 | 26.1 | 42 | 4.3 | 6.5 | 10.8 | 1.51 | 2.4 | 3.77 | 2.5 | 2.7 | 2.9 |
| 12413860 | Coeur D Alene River nr Harrison ID | 47.479 | -116.732 | 61 | 12 | 29.7 | 50 | 2.88 | 7.4 | 12.7 | 1.16 | 2.7 | 4.71 | 2.4 | 2.7 | 3.2 |
| 12413862 | Coeur D Alene River at Harrison Bridge nr Harrison | 47.465 | -116.765 | 4 | 17 | 32.3 | 44 | 4.35 | 8.1 | 11 | 1.5 | 3.0 | 4.1 | 2.5 | 2.7 | 2.9 |
| 12413875 | St. Joe River at Red Ives Ranger Station ID | 47.056 | -115.352 | 14 | 13 | 18.9 | 24 | 3.75 | 5.6 | 7.16 | 0.831 | 1.2 | 1.52 | 4.5 | 4.7 | 4.9 |
| 12414350 | Big Creek ab East Fork nr Calder ID | 47.306 | -116.116 | 7 | 12 | 16.3 | 21 | 3 | 4.3 | 5.6 | 1 | 1.3 | 1.8 | 2.7 | 3.2 | 3.8 |
| 12414400 | Ef Big Creek nr Calder ID | 47.302 | -116.118 | 2 | 17 | 19 | 21 | 4.3 | 4.9 | 5.4 | 1.4 | 1.6 | 1.7 | 3.1 | 3.1 | 3.2 |
| 12414500 | St Joe River at Calder ID | 47.275 | -116.188 | 26 | 14 | 23.5 | 32 | 4.14 | 6.9 | 9.62 | 0.898 | 1.5 | 1.93 | 4.3 | 4.6 | 5.2 |
| 12414900 | St Maries River nr Santa ID | 47.176 | -116.492 | 26 | 10 | 16.4 | 23 | 2.79 | 4.8 | 6.8 | 0.686 | 1.1 | 1.5 | 3.5 | 4.4 | 6.0 |
| 12415075 | St Joe River at St Maries ID | 47.317 | -116.561 | 1 | 19 | 19 | 19 | 6 | 6.0 | 6 | 0.86 | 0.9 | 0.86 | 7.0 | 7.0 | 7.0 |
| 12415140 | St Joe River Near Chatcolet ID | 47.36 | -116.691 | 13 | 14 | 20.5 | 29 | 4.03 | 6.0 | 8.42 | 0.888 | 1.4 | 1.9 | 3.8 | 4.3 | 4.8 |
| 12415300 | Mica Creek nr Coeur D Alene ID | 47.6 | -116.883 | 2 | 13 | 14 | 15 | 3.3 | 3.6 | 3.9 | 1.2 | 1.2 | 1.2 | 2.8 | 3.0 | 3.3 |
| 12416000 | Hayden Creek Bl North Fork nr Hayden Lake ID | 47.823 | -116.654 | 66 | 17 | 30.1 | 49 | 4.1 | 8.0 | 13 | 1.4 | 2.5 | 4 | 2.3 | 3.2 | 4.1 |
| 12417598 | Spokane River at Lake Outlet at Coeur D Alene ID | 47.676 | -116.801 | 20 | 17 | 20.6 | 27 | 4.49 | 5.6 | 7.4 | 1.39 | 1.6 | 2 | 3.1 | 3.5 | 3.9 |
| 12419000 | Spokane River nr Post Falls ID | 47.703 | -116.977 | 97 | 16 | 20.6 | 29 | 4.21 | 5.5 | 7.9 | 1.23 | 1.6 | 2.6 | 2.9 | 3.4 | 4.4 |
| 12419495 | Spokane River at Stateline Br nr Greenacres, Wa | 47.699 | -117.044 | 5 | 18 | 19.6 | 21 | 4.68 | 5.3 | 5.78 | 1.46 | 1.6 | 1.74 | 3.2 | 3.3 | 3.4 |
| 13037500 | Snake River nr Heise ID | 43.613 | -111.659 | 112 | 120 | 188 | 270 | 35 | 53.3 | 76 | 8.2 | 13.4 | 20 | 3.4 | 4.0 | 4.8 |
| 13038500 | Snake River at Lorenzo ID | 43.735 | -111.876 | 12 | 140 | 192.5 | 240 | 40 | 54.9 | 68 | 9.7 | 13.6 | 17 | 3.8 | 4.1 | 4.4 |
| 13055000 | Teton River nr St Anthony ID | 43.927 | -111.615 | 53 | 67 | 132.4 | 180 | 19 | 35.8 | 50 | 4.7 | 10.3 | 15 | 2.7 | 3.5 | 4.2 |
| 13056500 | Henrys Fork nr Rexburg ID | 43.826 | - | 52 | 34 | 64.9 | 95 | 9.7 | 17.6 | 26 | 2.4 | 5.1 | 7.4 | 3.0 | 3.5 | 4.0 |

Appendix A: Water Hardness in Idaho

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | 111.904 | | | | | | | | | | | | | |
| 13058000 | Willow Creek nr Ririe ID | 43.583 | -111.746 | 10 | 190 | 197 | 210 | 51 | 52.6 | 55 | 13 | 16.3 | 19 | 2.8 | 3.3 | 4.2 |
| 13060000 | Snake River nr Shelley ID | 43.414 | -112.135 | 17 | 69 | 133.4 | 160 | 20 | 37.5 | 47 | 4.5 | 9.6 | 11 | 3.7 | 4.0 | 4.4 |
| 13062690 | Blackfoot River at Bridge Abv Angus Cr nr Henry,ID | 42.824 | -111.323 | 1 | 180 | 180 | 180 | 53 | 53.0 | 53 | 10.7 | 10.7 | 10.7 | 5.0 | 5.0 | 5.0 |
| 13062692 | Angus Creek Bl Angus Creek Reservoir | 42.827 | -111.4 | 1 | 290 | 290 | 290 | 78.8 | 78.8 | 78.8 | 22.1 | 22.1 | 22.1 | 3.6 | 3.6 | 3.6 |
| 13062693 | Angus Creek 1.3 Mi Bl Angus Creek Reservoir | 42.843 | -111.414 | 1 | 220 | 220 | 220 | 64.2 | 64.2 | 64.2 | 15.2 | 15.2 | 15.2 | 4.2 | 4.2 | 4.2 |
| 13062695 | Angus Creek Near Henry ID | 42.854 | -111.411 | 1 | 200 | 200 | 200 | 57.4 | 57.4 | 57.4 | 13 | 13.0 | 13 | 4.4 | 4.4 | 4.4 |
| 13062698 | Angus Creek at Road 121 Xing nr Henry ID | 42.842 | -111.359 | 1 | 170 | 170 | 170 | 53.3 | 53.3 | 53.3 | 9.53 | 9.5 | 9.53 | 5.6 | 5.6 | 5.6 |
| 13062700 | Angus Creek nr Henry ID | 42.828 | -111.338 | 1 | 160 | 160 | 160 | 49.7 | 49.7 | 49.7 | 9.34 | 9.3 | 9.34 | 5.3 | 5.3 | 5.3 |
| 13063000 | Blackfoot River ab Reservoir nr Henry ID | 42.817 | -111.51 | 30 | 120 | 168.7 | 210 | 25.3 | 50.6 | 62 | 7.35 | 10.2 | 14.1 | 1.9 | 5.1 | 6.0 |
| 13068500 | Blackfoot River nr Blackfoot ID | 43.131 | -112.476 | 36 | 120 | 203.1 | 340 | 34.2 | 52.3 | 82 | 8.3 | 17.7 | 34 | 2.1 | 3.1 | 4.3 |
| 13069500 | Snake River nr Blackfoot ID | 43.125 | -112.518 | 53 | 81 | 145.1 | 170 | 23 | 40.5 | 50 | 5.7 | 10.6 | 14 | 3.3 | 3.8 | 4.5 |
| 13069515 | Mctucker Creek nr Pingree ID | 43.034 | -112.626 | 5 | 230 | 290 | 320 | 57 | 66.0 | 73 | 22 | 30.4 | 35 | 2.0 | 2.2 | 2.6 |
| 13069532 | Crystal Waste nr Springfield ID | 43.052 | -112.686 | 3 | 320 | 326.7 | 330 | 73 | 74.0 | 75 | 32 | 34.0 | 35 | 2.1 | 2.2 | 2.3 |
| 13069540 | Danielson Creek nr Springfield ID | 43.059 | -112.69 | 6 | 210 | 225 | 250 | 51 | 55.3 | 61 | 20 | 21.0 | 23 | 2.5 | 2.6 | 2.9 |
| 13069565 | Aberdeen Waste nr Aberdeen ID | 42.924 | -112.811 | 3 | 190 | 213.3 | 230 | 49 | 53.7 | 57 | 16 | 19.3 | 21 | 2.6 | 2.8 | 3.1 |
| 13073000 | Portneuf River at Topaz ID | 42.625 | -112.088 | 42 | 230 | 336 | 410 | 59 | 80.2 | 97 | 18 | 32.8 | 43 | 1.9 | 2.5 | 3.4 |
| 13073120 | Portneuf/Marsh Valley Canal nr Mccammon ID | 42.615 | -112.166 | 5 | 280 | 302 | 340 | 67 | 74.2 | 81 | 21 | 28.8 | 37 | 2.1 | 2.7 | 3.9 |
| 13073743 | Marsh Creek at Red Rock Pass nr Downey ID | 42.356 | -112.126 | 9 | 98 | 206.4 | 290 | 30 | 62.9 | 95 | 5.6 | 12.0 | 15 | 4.1 | 5.3 | 7.9 |
| 13073750 | Marsh Creek at Hwy 191 Xing nr Downey ID | 42.408 | -112.156 | 10 | 150 | 267 | 330 | 43 | 70.2 | 90 | 11 | 22.4 | 29 | 2.5 | 3.2 | 3.9 |
| 13074810 | Marsh Creek ab Hawkins Creek nr Virginia ID | 42.506 | -112.192 | 9 | 150 | 276.7 | 400 | 39 | 67.2 | 100 | 13 | 26.6 | 37 | 1.9 | 2.6 | 3.1 |
| 13075000 | Marsh Creek nr Mccammon ID | 42.63 | -112.225 | 27 | 250 | 321.1 | 350 | 59 | 74.7 | 85 | 22 | 32.7 | 38 | 2.0 | 2.3 | 3.0 |
| 13075050 | Marsh Creek ab Mouth nr Inkom ID | 42.767 | -112.232 | 9 | 250 | 287.8 | 330 | 60 | 68.4 | 79 | 24 | 28.3 | 35 | 2.1 | 2.4 | 2.8 |
| 13075500 | Portneuf River at Pocatello ID | 42.872 | -112.468 | 20 | 170 | 280 | 340 | 45 | 64.4 | 80 | 13 | 28.7 | 35 | 1.8 | 2.3 | 3.5 |
| 13075910 | Portneuf River nr Tyhee ID | 42.945 | -112.544 | 15 | 260 | 282 | 310 | 64 | 68.1 | 75 | 23 | 27.1 | 30 | 2.3 | 2.5 | 2.8 |
| 13075960 | Ross Fork nr Fort Hall ID | 43.001 | -112.516 | 3 | 230 | 243.3 | 270 | 57 | 60.3 | 67 | 22 | 23.3 | 26 | 2.6 | 2.6 | 2.6 |
| 13075983 | Spring Creek at Sheepskin Rd nr Fort Hall ID | 43.043 | -112.555 | 3 | 210 | 213.3 | 220 | 58 | 59.7 | 62 | 16 | 16.0 | 16 | 3.6 | 3.7 | 3.9 |

Appendix A: Water Hardness in Idaho

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13075985 | Spring Creek nr Fort Hall ID | 43.003 | -112.6 | 5 | 200 | 210 | 220 | 56 | 57.8 | 60 | 15 | 15.8 | 17 | 3.5 | 3.7 | 4.0 |
| 13076100 | Rattlesnake Creek nr Pocatello ID | 42.7 | -112.561 | 1 | 240 | 240 | 240 | 60 | 60.0 | 60 | 23 | 23.0 | 23 | 2.6 | 2.6 | 2.6 |
| 13076200 | Bannock Creek nr Pocatello ID | 42.886 | -112.642 | 4 | 200 | 275 | 320 | 52 | 66.5 | 79 | 17 | 26.8 | 31 | 2.3 | 2.5 | 3.1 |
| 13076500 | American Falls Res at American Falls ID | 42.779 | -112.879 | 2 | 200 | 210 | 220 | 47 | 53.5 | 60 | 18 | 18.5 | 19 | 2.5 | 2.9 | 3.3 |
| 13076600 | Reuger Springs nr American Falls ID | 42.767 | -112.882 | 5 | 250 | 252 | 260 | 63 | 64.4 | 68 | 22 | 22.4 | 23 | 2.8 | 2.9 | 3.1 |
| 13077650 | Rock Creek nr American Falls ID | 42.652 | -113.014 | 1 | 360 | 360 | 360 | 78 | 78.0 | 78 | 41 | 41.0 | 41 | 1.9 | 1.9 | 1.9 |
| 13078205 | Raft River Bl Onemile Creek nr Malta ID | 42.07 | -113.444 | 1 | 380 | 380 | 380 | 110 | 110.0 | 110 | 26 | 26.0 | 26 | 4.2 | 4.2 | 4.2 |
| 13081500 | Snake R nr Minidoka ID (at Howells Ferry) | 42.673 | -113.5 | 43 | 150 | 186.3 | 210 | 42 | 47.9 | 55 | 12 | 16.3 | 20 | 2.5 | 3.0 | 3.5 |
| 13082500 | Goose Creek ab Trapper Creek nr Oakley ID | 42.125 | -113.939 | 5 | 89 | 161.8 | 210 | 27 | 48.4 | 62 | 5.2 | 10.0 | 13 | 4.6 | 4.9 | 5.2 |
| 13083000 | Trapper Creek nr Oakley ID | 42.169 | -113.972 | 5 | 58 | 107.6 | 130 | 19 | 36.0 | 43 | 2.5 | 4.1 | 4.8 | 7.6 | 8.7 | 9.1 |
| 13084000 | Goose Creek nr Oakley ID | 42.203 | -113.911 | 1 | 160 | 160 | 160 | 48.2 | 48.2 | 48.2 | 9.86 | 9.9 | 9.86 | 4.9 | 4.9 | 4.9 |
| 13084400 | Birch Creek ab Feeder Canal nr Oakley ID | 42.178 | -113.819 | 1 | 150 | 150 | 150 | 45 | 45.0 | 45 | 8.7 | 8.7 | 8.7 | 5.2 | 5.2 | 5.2 |
| 13084590 | Mill Creek 14s 23e 04 | 42.237 | -113.777 | 1 | 55 | 55 | 55 | 17 | 17.0 | 17 | 3.1 | 3.1 | 3.1 | 5.5 | 5.5 | 5.5 |
| 13084650 | Willow Creek nr Burley ID | 42.348 | -113.729 | 1 | 37 | 37 | 37 | 12 | 12.0 | 12 | 1.6 | 1.6 | 1.6 | 7.5 | 7.5 | 7.5 |
| 13087995 | Snake River Gaging Station at Milner ID | 42.528 | -114.018 | 7 | 140 | 171.4 | 220 | 29 | 42.8 | 57 | 11.5 | 15.8 | 19 | 1.6 | 2.8 | 3.4 |
| 13088000 | Snake River at Milner ID (Total Flow) | 42.528 | -114.018 | 9 | 170 | 195.6 | 230 | 46 | 50.2 | 59 | 13 | 17.1 | 21 | 2.6 | 3.0 | 3.5 |
| 13088020 | Wrong No - Twin Falls Main Canal - See 13087800 | 42.518 | -114.275 | 7 | 100 | 168.6 | 190 | 26 | 42.6 | 48 | 9 | 15.1 | 18 | 2.3 | 2.8 | 3.2 |
| 13088400 | Dry Creek nr Artesian City ID | 42.372 | -114.186 | 3 | 24 | 25 | 26 | 7 | 7.2 | 7.3 | 1.6 | 1.7 | 1.8 | 4.1 | 4.2 | 4.4 |
| 13088510 | Cottonwood Creek nr Oakley ID | 42.294 | -114.022 | 4 | 19 | 34 | 69 | 6.1 | 11.2 | 23 | 0.91 | 1.5 | 2.7 | 6.5 | 7.3 | 8.5 |
| 13090000 | Snake River nr Kimberly ID | 42.591 | -114.36 | 20 | 150 | 203 | 240 | 39.5 | 49.1 | 58 | 11.9 | 19.4 | 24 | 2.0 | 2.6 | 3.3 |
| 13091500 | Blue Lakes Outlet nr Twin Falls ID Mv 15 | 42.608 | -114.476 | 5 | 210 | 230 | 240 | 53 | 58.4 | 62 | 18.6 | 19.9 | 21 | 2.8 | 2.9 | 3.1 |
| 13092000 | Rock Creek nr Rock Creek ID | 42.356 | -114.303 | 9 | 29 | 61.7 | 94 | 8.7 | 19.1 | 29 | 1.7 | 3.4 | 5.3 | 5.1 | 5.6 | 6.7 |
| 13092710 | Rock Creek Near 3200 East nr Twin Falls ID | 42.523 | -114.42 | 9 | 150 | 255.6 | 290 | 41 | 68.3 | 80 | 12 | 20.4 | 24 | 3.0 | 3.4 | 3.8 |
| 13092747 | Rock Creek ab Hwy 30/93 Xing at Twin Falls ID | 42.563 | -114.494 | 107 | 91 | 247.5 | 340 | 25 | 61.4 | 83 | 6.82 | 22.8 | 50.7 | 1.1 | 2.8 | 3.7 |
| 13093000 | Rock Creek Below Poleline Road nr Twin Falls ID | 42.594 | -114.529 | 12 | 130 | 272.5 | 350 | 33 | 66.4 | 83 | 11 | 25.6 | 34 | 2.4 | 2.7 | 3.0 |
| 13093095 | Rock Creek nr Mouth nr Twin Falls ID | 42.624 | -114.533 | 10 | 210 | 286 | 350 | 53 | 70.1 | 86 | 20 | 27.3 | 34 | 2.5 | 2.6 | 2.7 |
| 13093394 | Crystal Spring at Head nr Buhl ID | 42.659 | -114.642 | 1 | 280 | 280 | 280 | 64 | 64.0 | 64 | 28 | 28.0 | 28 | 2.3 | 2.3 | 2.3 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13093470 | Cedar Draw ab Low Line Canal nr Filer ID | 42.52 | -113.595 | 2 | 180 | 185 | 190 | 46 | 46.5 | 47 | 17 | 17.0 | 17 | 2.7 | 2.7 | 2.8 |
| 13093475 | Cedar Draw Bl Low Line Canal nr Filer ID | 42.544 | -114.613 | 7 | 170 | 195.7 | 270 | 43 | 49.0 | 62 | 15 | 17.9 | 27 | 2.3 | 2.8 | 3.1 |
| 13093500 | Cedar Draw nr Filer (Old Station) | 42.623 | -114.654 | 9 | 210 | 286.7 | 390 | 52 | 66.3 | 86 | 20 | 29.1 | 42 | 2.0 | 2.3 | 2.6 |
| 13093530 | Cedar Draw ab Mouth nr Filer ID | 42.649 | -114.659 | 9 | 220 | 280 | 380 | 52 | 65.2 | 85 | 21 | 28.6 | 40 | 2.0 | 2.3 | 2.6 |
| 13094000 | Snake River nr Buhl ID | 42.666 | -114.711 | 49 | 170 | 223.7 | 260 | 43 | 54.5 | 63 | 14.4 | 21.4 | 26 | 2.3 | 2.6 | 3.0 |
| 13095200 | Briggs Creek nr Buhl ID | 42.672 | -114.817 | 1 | 190 | 190 | 190 | 43 | 43.0 | 43 | 19 | 19.0 | 19 | 2.3 | 2.3 | 2.3 |
| 13106000 | Salmon River Canal Co Canal nr Rogerson ID | 42.221 | -114.738 | 1 | 58 | 58 | 58 | 17.2 | 17.2 | 17.2 | 3.67 | 3.7 | 3.67 | 4.7 | 4.7 | 4.7 |
| 13108150 | Salmon Falls Creek nr Hagerman ID | 42.696 | -114.854 | 21 | 240 | 282.9 | 330 | 60 | 70.9 | 82 | 21.3 | 25.7 | 30 | 2.5 | 2.8 | 3.0 |
| 13108500 | Camas Creek at 18mi Shearing Corral nr Kilgore ID | 44.3 | -111.905 | 2 | 66 | 67.5 | 69 | 19 | 19.5 | 20 | 4.4 | 4.6 | 4.7 | 4.3 | 4.3 | 4.3 |
| 13108900 | Camas Creek at Red Road nr Kilgore ID | 44.289 | -111.894 | 9 | 57 | 72 | 82 | 16 | 20.6 | 23 | 4.2 | 5.0 | 6 | 3.8 | 4.1 | 4.4 |
| 13112000 | Camas Creek at Camas ID | 44.003 | -112.22 | 3 | 59 | 67.3 | 72 | 17 | 19.3 | 21 | 3.9 | 4.6 | 5 | 4.0 | 4.2 | 4.4 |
| 13113000 | Beaver Creek at Spencer ID 12n-36e-23a | 44.355 | -112.18 | 13 | 180 | 213.8 | 230 | 53 | 61.6 | 67 | 12 | 14.5 | 16 | 3.6 | 4.3 | 4.8 |
| 13116000 | Medicine Lodge Creek at Ellis Ranch nr Angora ID | 44.291 | -112.503 | 1 | 220 | 220 | 220 | 59 | 59.0 | 59 | 18 | 18.0 | 18 | 3.3 | 3.3 | 3.3 |
| 13117020 | Birch Creek at Blue Dome Inn nr Reno ID | 44.153 | -112.909 | 2 | 180 | 180 | 180 | 44 | 44.5 | 45 | 16 | 16.0 | 16 | 2.8 | 2.8 | 2.8 |
| 13117030 | Birch Creek at Eight-Mile Canyon Rd nr Reno ID | 44.08 | -112.876 | 2 | 170 | 170 | 170 | 40 | 41.5 | 43 | 15 | 15.5 | 16 | 2.5 | 2.7 | 2.9 |
| 13117390 | Summit Cr ab Barney H Sp nr Goldburg 11n 25e 22aaa | 44.276 | -113.456 | 1 | 160 | 160 | 160 | 36 | 36.0 | 36 | 16 | 16.0 | 16 | 2.3 | 2.3 | 2.3 |
| 13118700 | Little Lost River Bl Wet Creek nr Howe ID | 44.139 | -113.244 | 5 | 64 | 110.8 | 130 | 15 | 26.2 | 34 | 6.4 | 10.9 | 13 | 1.9 | 2.4 | 3.1 |
| 13119000 | Little Lost River nr Howe ID | 43.886 | -113.1 | 7 | 110 | 158.6 | 200 | 28 | 38.4 | 47 | 9.2 | 14.9 | 19 | 2.3 | 2.6 | 3.0 |
| 13119800 | Nf Big Lost River nr Chilly ID | 43.926 | -114.183 | 1 | 170 | 170 | 170 | 41 | 41.0 | 41 | 17 | 17.0 | 17 | 2.4 | 2.4 | 2.4 |
| 13120000 | Nf Big Lost River at Wild Horse nr Chilly ID | 43.934 | -114.113 | 2 | 110 | 110 | 110 | 29 | 29.0 | 29 | 8.4 | 8.7 | 8.9 | 3.3 | 3.4 | 3.5 |
| 13120240 | Ef Big Lost R at Rosenkance Rch nr Chilly ID | 43.896 | -113.983 | 1 | 65 | 65 | 65 | 20 | 20.0 | 20 | 3.6 | 3.6 | 3.6 | 5.6 | 5.6 | 5.6 |
| 13120420 | Twin Bridges Creek nr Chilly ID 07n 20e 9b | 43.953 | -114.103 | 1 | 90 | 90 | 90 | 24 | 24.0 | 24 | 7.4 | 7.4 | 7.4 | 3.2 | 3.2 | 3.2 |
| 13120450 | Garden Creek nr Chilly ID 08n 20e 35d | 43.979 | -114.06 | 1 | 110 | 110 | 110 | 31 | 31.0 | 31 | 8.9 | 8.9 | 8.9 | 3.5 | 3.5 | 3.5 |
| 13120500 | Big Lost River at Howell Ranch nr Chilly ID | 43.998 | -114.021 | 39 | 44 | 81.9 | 110 | 13 | 24.1 | 31 | 2.6 | 5.3 | 6.9 | 4.2 | 4.6 | 5.4 |
| 13121500 | Big Lost River at Chilly Bridge nr Chilly ID | 44.059 | -113.878 | 2 | 88 | 90.5 | 93 | 26 | 26.5 | 27 | 5.6 | 6.0 | 6.3 | 4.3 | 4.5 | 4.6 |
| 13121580 | Sage Creek ab Div nr Mackay ID 09n 20e 25ad | 44.083 | -114.033 | 1 | 110 | 110 | 110 | 24 | 24.0 | 24 | 12 | 12.0 | 12 | 2.0 | 2.0 | 2.0 |
| 13121700 | Willow Creek Bl Freighter Spring 10n 22e 28 | 44.168 | -113.861 | 1 | 130 | 130 | 130 | 37 | 37.0 | 37 | 8.6 | 8.6 | 8.6 | 4.3 | 4.3 | 4.3 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13121900 | Cedar Creek ab Div nr Dickey ID 10n 22e 34cc | 44.149 | -113.839 | 2 | 120 | 125 | 130 | 32 | 33.0 | 34 | 9.6 | 9.8 | 10 | 3.3 | 3.4 | 3.4 |
| 13122000 | Thousand Springs Creek nr Chilly ID | 44.067 | -113.84 | 3 | 210 | 233.3 | 270 | 48 | 54.7 | 60 | 20 | 23.0 | 28 | 2.1 | 2.4 | 2.8 |
| 13122100 | Elkhorn Creek nr Chilly ID 09n 22e 26c | 44.075 | -113.819 | 1 | 160 | 160 | 160 | 26 | 26.0 | 26 | 22 | 22.0 | 22 | 1.2 | 1.2 | 1.2 |
| 13122400 | Long Cedar Creek ab Div nr Chilly ID | 44.044 | -113.744 | 1 | 250 | 250 | 250 | 62 | 62.0 | 62 | 23 | 23.0 | 23 | 2.7 | 2.7 | 2.7 |
| 13122500 | Big Lost River Bl Chilly Sinks nr Chilly ID | 43.996 | -113.771 | 2 | 130 | 145 | 160 | 34 | 36.0 | 38 | 12 | 14.0 | 16 | 2.4 | 2.6 | 2.8 |
| 13123400 | Big Lost River at Goddard Bridge 07n 23e 33cca1 | 43.977 | -113.739 | 1 | 160 | 160 | 160 | 38 | 38.0 | 38 | 15 | 15.0 | 15 | 2.5 | 2.5 | 2.5 |
| 13124030 | Hamilton Springs nr Mackay ID | 43.991 | -113.865 | 3 | 110 | 116.7 | 120 | 32 | 32.3 | 33 | 8.3 | 8.5 | 8.6 | 3.7 | 3.8 | 3.9 |
| 13125800 | Upper Cedar Creek ab Div nr Mackay 08n 24e 19cb | 44.008 | -113.658 | 1 | 220 | 220 | 220 | 57 | 57.0 | 57 | 19 | 19.0 | 19 | 3.0 | 3.0 | 3.0 |
| 13127000 | Big Lost River Bl Mackay Res nr Mackay ID | 43.939 | -113.648 | 16 | 120 | 140 | 160 | 34 | 39.6 | 44 | 8.6 | 9.9 | 11 | 3.6 | 4.0 | 4.3 |
| 13127700 | Big Lost River at Mackay ID | 43.886 | -113.616 | 1 | 150 | 150 | 150 | 42 | 42.0 | 42 | 11 | 11.0 | 11 | 3.8 | 3.8 | 3.8 |
| 13127780 | Big Lost River at Alder Cr Rd Brdg nr Mackay ID | 43.887 | -113.578 | 1 | 160 | 160 | 160 | 44 | 44.0 | 44 | 11 | 11.0 | 11 | 4.0 | 4.0 | 4.0 |
| 13130200 | Big Lost River Bl Alder Creek nr Mackay ID | 43.871 | -113.511 | 1 | 180 | 180 | 180 | 53 | 53.0 | 53 | 12 | 12.0 | 12 | 4.4 | 4.4 | 4.4 |
| 13130300 | Big Lost River nr Leslie ID | 43.859 | -113.466 | 1 | 180 | 180 | 180 | 51 | 51.0 | 51 | 12 | 12.0 | 12 | 4.3 | 4.3 | 4.3 |
| 13130847 | Big Lost R at Darlington Rd Xing nr Darlington ID | 43.813 | -113.392 | 1 | 170 | 170 | 170 | 50 | 50.0 | 50 | 12 | 12.0 | 12 | 4.2 | 4.2 | 4.2 |
| 13132050 | Big Lost River ab Moore Div nr Moore ID | 43.787 | -113.358 | 1 | 190 | 190 | 190 | 54 | 54.0 | 54 | 14 | 14.0 | 14 | 3.9 | 3.9 | 3.9 |
| 13132150 | Big Lost River at Moore ID | 43.729 | -113.359 | 1 | 200 | 200 | 200 | 55 | 55.0 | 55 | 14 | 14.0 | 14 | 3.9 | 3.9 | 3.9 |
| 13132310 | Big Lost River ab Arco ID | 43.682 | -113.366 | 1 | 200 | 200 | 200 | 55 | 55.0 | 55 | 15 | 15.0 | 15 | 3.7 | 3.7 | 3.7 |
| 13132375 | Big Lost River at Arco-Minidoka Rd Xing at Arco ID | 43.624 | -113.311 | 1 | 210 | 210 | 210 | 61 | 61.0 | 61 | 14 | 14.0 | 14 | 4.4 | 4.4 | 4.4 |
| 13132500 | Big Lost River nr Arco ID | 43.582 | -113.271 | 7 | 170 | 210 | 240 | 48 | 59.9 | 71 | 12 | 14.4 | 16 | 3.7 | 4.1 | 4.5 |
| 13132520 | Big Lost River Bl Ineel Div nr Arco ID | 43.516 | -113.082 | 3 | 110 | 136.7 | 180 | 33 | 40.1 | 52.4 | 7.1 | 9.2 | 12.5 | 4.2 | 4.4 | 4.6 |
| 13140800 | Big Wood River at Stanton Crossing nr Bellevue ID | 43.329 | -114.319 | 2 | 160 | 160 | 160 | 47.4 | 47.7 | 48 | 9.4 | 9.5 | 9.55 | 5.0 | 5.0 | 5.0 |
| 13141000 | Big Wood River nr Bellevue ID | 43.328 | -114.342 | 17 | 100 | 162.9 | 190 | 30 | 49.3 | 56 | 6.5 | 9.5 | 11 | 4.6 | 5.2 | 5.6 |
| 13141500 | Camas Creek nr Blaine ID | 43.333 | -114.541 | 4 | 59 | 77.8 | 94 | 18 | 23.8 | 29 | 3.3 | 4.4 | 5.3 | 5.3 | 5.4 | 5.6 |
| 13142500 | Big Wood River Bl Magic Dam nr Richfield ID | 43.248 | -114.356 | 1 | 110 | 110 | 110 | 33 | 33.0 | 33 | 6.8 | 6.8 | 6.8 | 4.9 | 4.9 | 4.9 |
| 13148500 | Little Wood River nr Carey ID | 43.389 | -114 | 4 | 77 | 106.8 | 150 | 21 | 28.8 | 41 | 6 | 8.2 | 11 | 3.3 | 3.5 | 3.7 |
| 13150430 | Silver Creek at Sportsman Access nr Picabo ID | 43.323 | -114.108 | 17 | 190 | 195.9 | 210 | 53 | 55.9 | 60 | 13 | 13.7 | 17 | 3.4 | 4.1 | 4.4 |
| 13152500 | Malad River nr Gooding ID | 42.887 | -114.802 | 44 | 71 | 163.5 | 210 | 20 | 42.8 | 60 | 5 | 13.8 | 18 | 2.2 | 3.2 | 4.3 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13152850 | Big Wood River at Upper Malad Dam nr Hagerman ID | 42.866 | -114.868 | 4 | 140 | 145 | 150 | 31 | 33.5 | 36 | 14 | 14.8 | 15 | 2.1 | 2.3 | 2.6 |
| 13152900 | Cove Creek nr Hagerman ID | 42.867 | -114.868 | 2 | 150 | 155 | 160 | 35 | 36.0 | 37 | 15 | 15.5 | 16 | 2.3 | 2.3 | 2.3 |
| 13154500 | Snake River at King Hill ID | 43.002 | -115.202 | 179 | 140 | 188.5 | 220 | 36 | 45.2 | 57 | 11 | 18.3 | 22 | 2.1 | 2.5 | 3.4 |
| 13168500 | Bruneau River nr Hot Spring ID | 42.771 | -115.719 | 15 | 25 | 42.2 | 59 | 8 | 13.5 | 19 | 1.2 | 2.0 | 2.9 | 5.6 | 6.6 | 7.5 |
| 13169500 | Big Jacks Creek nr Bruneau ID | 42.785 | -115.983 | 41 | 22 | 42.4 | 57 | 6.2 | 12.0 | 16 | 1.6 | 3.0 | 4.2 | 3.6 | 4.0 | 4.7 |
| 13172500 | Snake River nr Murphy ID | 43.292 | -116.42 | 13 | 170 | 193.1 | 210 | 41 | 45.4 | 49 | 16 | 19.4 | 21 | 2.2 | 2.3 | 2.7 |
| 13185000 | Boise River nr Twin Springs ID | 43.659 | -115.726 | 13 | 18 | 29.5 | 34 | 6.3 | 10.5 | 12 | 0.48 | 0.8 | 1 | 11.1 | 12.7 | 15.7 |
| 13186000 | Sf Boise River nr Featherville ID | 43.496 | -115.308 | 4 | 32 | 38.3 | 47 | 11 | 13.3 | 16 | 1 | 1.3 | 1.8 | 8.9 | 10.4 | 12.0 |
| 13200000 | Mores Creek ab Robie Creek nr Arrowrock Dam ID | 43.648 | -115.989 | 4 | 23 | 35.5 | 44 | 7.2 | 11.3 | 14 | 1.1 | 1.7 | 2.1 | 6.5 | 6.6 | 6.7 |
| 13202000 | Boise River nr Boise ID | 43.519 | -116.059 | 3 | 25 | 30.7 | 37 | 8.3 | 10.1 | 12 | 1 | 1.2 | 1.6 | 7.5 | 8.3 | 9.1 |
| 13203510 | Boise R Bl Diversion Dam nr Boise ID | 43.54 | -116.094 | 10 | 22 | 32.1 | 39 | 7.57 | 10.8 | 13 | 0.838 | 1.2 | 1.5 | 8.1 | 8.9 | 10.0 |
| 13203760 | Boise River at Eckert Rd nr Boise ID | 43.566 | -116.131 | 1 | 32 | 32 | 32 | 11 | 11.0 | 11 | 1.1 | 1.1 | 1.1 | 10.0 | 10.0 | 10.0 |
| 13204400 | 51n Storm Drain at Walnut Street at Boise ID | 43.601 | -116.187 | 3 | 91 | 103.7 | 120 | 29 | 32.3 | 37 | 4.5 | 5.6 | 6.7 | 5.4 | 5.8 | 6.4 |
| 13205300 | 44s Storm Drain @ Boise State U. at Boise ID | 43.605 | -116.203 | 3 | 16 | 51.3 | 100 | 5.2 | 16.7 | 32 | 0.75 | 2.4 | 5 | 6.4 | 7.5 | 9.3 |
| 13205505 | 39n Storm Drain at 9th Street at Boise ID | 43.611 | -116.208 | 3 | 34 | 76 | 130 | 10 | 24.7 | 42 | 2.1 | 3.1 | 5 | 4.8 | 7.7 | 10.0 |
| 13205518 | 43 St. Storm Drain at Garden City ID | 43.631 | -116.251 | 3 | 28 | 54.7 | 100 | 9.5 | 18.8 | 35 | 1 | 2.1 | 3.8 | 8.6 | 9.1 | 9.5 |
| 13205524 | 31n Storm Drain at Americana Blvd at Boise ID | 43.616 | -116.221 | 5 | 19 | 34 | 60 | 6.4 | 10.4 | 15 | 0.75 | 2.0 | 5.5 | 2.7 | 7.2 | 8.7 |
| 13205642 | Boise R at Veterans Memorial Parkway at Boise ID | 43.639 | -116.246 | 2 | 46 | 46 | 46 | 15 | 15.0 | 15 | 2 | 2.1 | 2.1 | 7.1 | 7.3 | 7.5 |
| 13206000 | Boise River at Glenwood Bridge nr Boise ID | 43.66 | -116.278 | 31 | 25 | 39.6 | 55 | 8.48 | 13.1 | 18 | 0.941 | 1.7 | 2.5 | 6.8 | 8.0 | 9.3 |
| 13206200 | Boise River nr Eagle ID | 43.675 | -116.317 | 2 | 55 | 55.5 | 56 | 18 | 18.0 | 18 | 2.5 | 2.6 | 2.6 | 6.9 | 7.1 | 7.2 |
| 13206305 | Boise River South Channel at Eagle ID | 43.675 | -116.354 | 2 | 69 | 74 | 79 | 22 | 23.5 | 25 | 3.5 | 3.8 | 4.1 | 6.1 | 6.2 | 6.3 |
| 13209500 | Boise River South Channel at Linder Rd nr Eagle ID | 43.674 | -116.411 | 3 | 88 | 91 | 95 | 27 | 28.0 | 29 | 4.9 | 5.1 | 5.4 | 5.4 | 5.5 | 5.7 |
| 13209800 | Boise R at Sundance Ranch nr Star ID | 43.683 | -116.461 | 2 | 66 | 78 | 90 | 21 | 24.5 | 28 | 3.2 | 4.1 | 4.9 | 5.7 | 6.1 | 6.6 |
| 13210050 | Boise River nr Middleton ID | 43.684 | -116.573 | 9 | 34 | 66.2 | 86 | 11 | 21.1 | 27 | 1.5 | 3.3 | 4.6 | 5.9 | 6.4 | 7.3 |
| 13213000 | Boise River nr Parma ID | 43.782 | -116.971 | 49 | 52 | 143.6 | 180 | 16 | 39.6 | 48 | 3 | 10.7 | 14 | 3.3 | 3.8 | 5.3 |
| 13213100 | Snake River at Nyssa Or | 43.877 | -116.984 | 11 | 110 | 171.8 | 210 | 29 | 41.3 | 50 | 10 | 16.7 | 20 | 2.2 | 2.5 | 3.0 |
| 13235000 | Sf Payette River at Lowman ID | 44.085 | -115.622 | 14 | 21 | 33.7 | 44 | 7.5 | 12.2 | 16 | 0.4 | 0.8 | 1.03 | 13.3 | 15.8 | 32.5 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13239000 | Nf Payette River at Mccall ID | 44.908 | -116.118 | 14 | 6 | 8.4 | 32 | 1.9 | 2.9 | 12 | 0.26 | 0.3 | 0.4 | 6.1 | 8.5 | 30.0 |
| 13240000 | Lake Fork Payette River ab Jumbo Cr nr Mccall ID | 44.914 | -115.996 | 1 | 5 | 5 | 5 | 1.8 | 1.8 | 1.8 | 0.02 | 0.0 | 0.02 | 90.0 | 90.0 | 90.0 |
| 13245000 | Nf Payette River at Cascade ID | 44.525 | -116.046 | 18 | 10 | 11.9 | 14 | 3 | 3.7 | 4.5 | 0.5 | 0.7 | 0.79 | 4.5 | 5.4 | 9.0 |
| 13250600 | Big Willow Creek nr Emmett ID | 44.074 | -116.485 | 1 | 47 | 47 | 47 | 11 | 11.0 | 11 | 4.6 | 4.6 | 4.6 | 2.4 | 2.4 | 2.4 |
| 13251000 | Payette River nr Payette ID | 44.043 | -116.924 | 18 | 19 | 48.1 | 81 | 6.2 | 14.2 | 23 | 0.95 | 3.1 | 5.7 | 3.9 | 5.0 | 6.5 |
| 13258500 | Weiser River nr Cambridge ID | 44.579 | -116.643 | 6 | 23 | 36.8 | 48 | 5.9 | 9.3 | 12 | 2 | 3.3 | 4.4 | 2.6 | 2.8 | 3.3 |
| 13266000 | Weiser River nr Weiser ID | 44.268 | -116.771 | 17 | 24 | 46.9 | 73 | 6 | 11.1 | 17 | 2.2 | 4.6 | 7.4 | 2.3 | 2.4 | 2.7 |
| 13269000 | Snake River at Weiser ID | 44.246 | -116.98 | 57 | 78 | 148.5 | 210 | 15 | 35.9 | 49 | 6.8 | 14.0 | 20 | 1.1 | 2.6 | 3.2 |
| 13293800 | Salmon River @ Hwy 93 Abv Redfish Cr nr Stanley ID | 44.164 | -114.886 | 10 | 39 | 54.7 | 68 | 14 | 19.4 | 24 | 1 | 1.5 | 1.9 | 11.7 | 12.7 | 14.0 |
| 13293900 | Redfish Lake Creek Bl Lake nr Stanley ID | 44.156 | -114.911 | 2 | 11 | 11.5 | 12 | 3.6 | 3.9 | 4.2 | 0.4 | 0.4 | 0.4 | 9.0 | 9.8 | 10.5 |
| 13296000 | Yankee Fork Salmon River nr Clayton ID | 44.288 | -114.72 | 2 | 22 | 27 | 32 | 7.4 | 9.2 | 11 | 0.9 | 1.0 | 1.1 | 8.2 | 9.1 | 10.0 |
| 13296500 | Salmon River Bl Yankee Fork nr Clayton ID | 44.268 | -114.733 | 1 | 23 | 23 | 23 | 7.5 | 7.5 | 7.5 | 1.1 | 1.1 | 1.1 | 6.8 | 6.8 | 6.8 |
| 13297450 | Little Boulder Creek nr Clayton ID | 44.099 | -114.447 | 1 | 23 | 23 | 23 | 8.2 | 8.2 | 8.2 | 0.57 | 0.6 | 0.57 | 14.4 | 14.4 | 14.4 |
| 13298000 | Ef Salmon River nr Clayton ID | 44.224 | -114.286 | 1 | 46 | 46 | 46 | 15 | 15.0 | 15 | 2 | 2.0 | 2 | 7.5 | 7.5 | 7.5 |
| 13301510 | Grouse Creek at Road Crossing nr May ID | 44.447 | -113.887 | 1 | 290 | 290 | 290 | 97 | 97.0 | 97 | 12 | 12.0 | 12 | 8.1 | 8.1 | 8.1 |
| 13301535 | Sulphur Creek at Road Xing nr May 14n 21e 13aac1 | 44.549 | -113.915 | 1 | 180 | 180 | 180 | 43 | 43.0 | 43 | 18 | 18.0 | 18 | 2.4 | 2.4 | 2.4 |
| 13302005 | Pahsimeroi River at Ellis ID | 44.525 | -114.047 | 10 | 160 | 178 | 200 | 42 | 46.1 | 51 | 14 | 15.2 | 17 | 2.9 | 3.0 | 3.3 |
| 13302500 | Salmon River at Salmon ID | 45.184 | -113.895 | 16 | 42 | 100.3 | 140 | 13 | 28.7 | 40 | 2.4 | 6.7 | 9.6 | 3.8 | 4.4 | 5.4 |
| 13305000 | Lemhi River nr Lemhi ID | 44.94 | -113.639 | 16 | 120 | 186.9 | 240 | 30 | 46.8 | 62 | 11 | 17.0 | 21 | 2.1 | 2.7 | 3.1 |
| 13307000 | Salmon River nr Shoup ID | 45.322 | -114.441 | 4 | 69 | 106.3 | 140 | 20 | 30.0 | 39 | 4.7 | 8.0 | 11 | 3.4 | 3.9 | 4.4 |
| 13309220 | Mf Salmon River at Mf Lodge nr Yellow Pine ID | 44.722 | -115.016 | 2 | 34 | 36 | 38 | 12 | 12.5 | 13 | 0.94 | 1.1 | 1.3 | 10.0 | 11.4 | 12.8 |
| 13310700 | Sf Salmon River nr Krassel Ranger Station ID | 44.987 | -115.724 | 5 | 7 | 10.2 | 13 | 2.4 | 3.8 | 4.9 | 0 | 0.2 | 0.34 | | | |
| 13313000 | Johnson Creek at Yellow Pine ID | 44.962 | -115.499 | 14 | 12 | 33.1 | 44 | 3.9 | 10.5 | 14 | 0.45 | 1.6 | 2.2 | 5.9 | 6.7 | 8.7 |
| 13316500 | Little Salmon River at Riggins ID | 45.413 | -116.325 | 13 | 18 | 46.9 | 74 | 5.6 | 14.3 | 23.1 | 0.9 | 2.7 | 3.95 | 4.0 | 5.3 | 6.3 |
| 13317000 | Salmon River at White Bird ID | 45.75 | -116.324 | 99 | 22 | 57.4 | 92 | 6.9 | 16.8 | 26 | 1.1 | 3.7 | 6.5 | 3.3 | 4.8 | 7.5 |
| 13317046 | Spring Abv Swartz Pond Near White Bird ID | 45.805 | -116.271 | 1 | 35 | 35 | 35 | 8.63 | 8.6 | 8.63 | 3.24 | 3.2 | 3.24 | 2.7 | 2.7 | 2.7 |
| 13317048 | White Bird Cr at Bridge Abv Price Cr nr White Bird | 45.778 | -116.279 | 2 | 19 | 99.5 | 180 | 4.46 | 22.5 | 40.6 | 1.79 | 10.2 | 18.7 | 2.2 | 2.3 | 2.5 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13336300 | Gedney Creek nr Selway Falls Guard Station ID | 46.058 | -115.314 | 7 | 6 | 7.7 | 12 | 1.8 | 2.5 | 4.1 | 0.2 | 0.4 | 0.48 | 5.2 | 6.8 | 10.0 |
| 13336500 | Selway River nr Lowell ID | 46.087 | -115.514 | 3 | 5 | 8 | 11 | 1.8 | 2.7 | 3.7 | 0.2 | 0.3 | 0.5 | 7.4 | 8.4 | 9.0 |
| 13337000 | Lochsa River nr Lowell ID | 46.151 | -115.587 | 3 | 7 | 10 | 13 | 2.5 | 3.3 | 4.2 | 0.3 | 0.5 | 0.7 | 5.3 | 6.6 | 8.3 |
| 13338500 | Sf Clearwater River at Stites ID | 46.086 | -115.977 | 15 | 9 | 19.4 | 33 | 2.7 | 5.3 | 8.5 | 0.6 | 1.5 | 2.8 | 3.0 | 3.6 | 4.5 |
| 13338650 | Unnamed Spring Blw Nikesa Creek nr East Kamiah ID | 46.21 | -116.004 | 2 | 92 | 94.5 | 97 | 22 | 22.5 | 23 | 8.94 | 9.3 | 9.62 | 2.4 | 2.4 | 2.5 |
| 13339500 | Lolo Creek nr Greer ID | 46.372 | -116.163 | 5 | 9 | 10 | 11 | 2.6 | 2.8 | 3 | 0.6 | 0.7 | 1 | 2.8 | 3.9 | 4.4 |
| 13341300 | Bloom Creek nr Bovill ID | 46.858 | -116.291 | 1 | 24 | 24 | 24 | 6.8 | 6.8 | 6.8 | 1.7 | 1.7 | 1.7 | 4.0 | 4.0 | 4.0 |
| 13341500 | Potlatch River at Kendrick ID | 46.612 | -116.658 | 1 | 35 | 35 | 35 | 10 | 10.0 | 10 | 2.3 | 2.3 | 2.3 | 4.3 | 4.3 | 4.3 |
| 13342450 | Lapwai Creek nr Lapwai ID | 46.427 | -116.804 | 18 | 54 | 100.9 | 150 | 14 | 25.7 | 38.9 | 4.5 | 9.1 | 13.3 | 2.6 | 2.9 | 3.1 |
| 13342490 | Lapwai Creek at Spalding ID | 46.448 | -116.816 | 3 | 49 | 99.7 | 130 | 0.02 | 18.7 | 31.1 | 0.002 | 6.9 | 11.6 | 2.7 | 2.7 | 2.9 |
| 13342500 | Clearwater River at Spalding ID | 46.449 | -116.826 | 98 | 7 | 14.3 | 29 | 2.2 | 4.2 | 7.8 | 0.3 | 0.9 | 2.2 | 3.4 | 4.7 | 9.3 |
| 13344800 | Deep Creek nr Potlatch ID | 46.961 | -116.934 | 1 | 49 | 49 | 49 | 14 | 14.0 | 14 | 3.4 | 3.4 | 3.4 | 4.1 | 4.1 | 4.1 |
| 13345000 | Palouse River nr Potlatch ID | 46.915 | -116.95 | 18 | 14 | 26.2 | 37 | 3.8 | 7.3 | 11 | 1.2 | 1.9 | 2.5 | 3.2 | 3.8 | 4.8 |
| 13346800 | Paradise Cr at University Of Idaho at Moscow ID | 46.732 | -117.023 | 1 | 160 | 160 | 160 | 44 | 44.0 | 44 | 11 | 11.0 | 11 | 4.0 | 4.0 | 4.0 |
| 124131265 | Ef Ninemile Creek Abv Success Mine nr Blackcloud | 47.53 | -115.874 | 1 | 24 | 24 | 24 | 7.66 | 7.7 | 7.66 | 1.21 | 1.2 | 1.21 | 6.3 | 6.3 | 6.3 |
| 124131267 | Ef Ninemile Creek nr Blackcloud, ID | 47.524 | -115.88 | 1 | 10 | 10 | 10 | 3.23 | 3.2 | 3.23 | 0.526 | 0.5 | 0.526 | 6.1 | 6.1 | 6.1 |
| 130626914 | Angus Creek Reservoir | 42.827 | -111.4 | 2 | 290 | 335 | 380 | 78.6 | 92.3 | 106 | 22.4 | 25.3 | 28.2 | 3.5 | 3.6 | 3.8 |
| 130626924 | Angus Creek 0.7 Miles Blw Angus Cr Res nr Henry ID | 42.835 | -111.407 | 1 | 250 | 250 | 250 | 70.3 | 70.3 | 70.3 | 18.4 | 18.4 | 18.4 | 3.8 | 3.8 | 3.8 |
| 133170462 | Outflow From Swartz Pond nr White Bird ID | 45.801 | -116.271 | 1 | 150 | 150 | 150 | 23.8 | 23.8 | 23.8 | 21.9 | 21.9 | 21.9 | 1.1 | 1.1 | 1.1 |
| 1313457010 | Bell Rapids Mutual Irr Co Pumping Plnt nr Hagerman | 42.83 | -114.937 | 1 | 200 | 200 | 200 | 48 | 48.0 | 48 | 20 | 20.0 | 20 | 2.4 | 2.4 | 2.4 |
| 422750114251201 | High Line Canal Near Twin Falls Airport | 42.464 | -114.42 | 11 | 150 | 158.2 | 170 | 35.4 | 41.2 | 44.5 | 11.2 | 13.1 | 14.6 | 2.5 | 3.2 | 3.6 |
| 431854114091200 | Silver Creek at The Nature Conservancy Preserve | 43.315 | -114.153 | 1 | 190 | 190 | 190 | 57.2 | 57.2 | 57.2 | 12.2 | 12.2 | 12.2 | 4.7 | 4.7 | 4.7 |
| 472721116480100 | Coeur D Alene Lake Btwn Harrison And Harlow Point | 47.456 | -116.8 | 4 | 17 | 28.3 | 38 | 4.35 | 7.1 | 9.49 | 1.48 | 2.6 | 3.53 | 2.5 | 2.8 | 2.9 |
| 472839115545001 | Canyon Creek Seepage Site No. A-7 | 47.478 | -115.914 | 2 | 47 | 48.5 | 50 | 13.2 | 13.5 | 13.8 | 3.42 | 3.5 | 3.65 | 3.8 | 3.8 | 3.9 |
| 472852115541401 | Canyon Creek Seepage Site No. A-6 | 47.481 | -115.904 | 2 | 47 | 48 | 49 | 13.3 | 13.5 | 13.7 | 3.48 | 3.5 | 3.54 | 3.8 | 3.8 | 3.9 |
| 472905115534301 | Canyon Creek Seepage Site No. A-4 | 47.485 | -115.895 | 1 | 43 | 43 | 43 | 12 | 12.0 | 12 | 3.08 | 3.1 | 3.08 | 3.9 | 3.9 | 3.9 |
| 472931115531501 | Canyon Creek Seepage Site No. A-1.2 | 47.492 | -115.888 | 1 | 43 | 43 | 43 | 12 | 12.0 | 12 | 3.06 | 3.1 | 3.06 | 3.9 | 3.9 | 3.9 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 472931115581201 | Sf Coeur D Alene River Seepage Site No. B-1 | 47.492 | -115.97 | 3 | 46 | 60.3 | 69 | 12.5 | 16.2 | 18.4 | 3.6 | 4.9 | 5.59 | 3.3 | 3.3 | 3.5 |
| 473005115593201 | Sf Coeur D Alene River Seepage Site No. B-1.1 | 47.501 | -115.992 | 1 | 67 | 67 | 67 | 17.8 | 17.8 | 17.8 | 5.44 | 5.4 | 5.44 | 3.3 | 3.3 | 3.3 |
| 473007115585601 | Sf Coeur D Alene River Inflow Pipe Ds Of B-1.1 | 47.502 | -115.982 | 1 | 170 | 170 | 170 | 33.5 | 33.5 | 33.5 | 20 | 20.0 | 20 | 1.7 | 1.7 | 1.7 |
| 473019115523501 | Canyon Creek Seepage Site No. A-1 | 47.505 | -115.876 | 2 | 41 | 42.5 | 44 | 11.6 | 11.9 | 12.2 | 2.98 | 3.1 | 3.17 | 3.8 | 3.9 | 3.9 |
| 473022115592001 | Sf Coeur D Alene River Seepage Site No. B-2 | 47.506 | -115.989 | 3 | 46 | 61 | 70 | 12.2 | 16.2 | 18.6 | 3.65 | 4.9 | 5.62 | 3.3 | 3.3 | 3.3 |
| 473037116004101 | Sf Coeur D Alene River Seepage Site No. B-5 | 47.51 | -116.011 | 3 | 48 | 62 | 70 | 12.9 | 16.5 | 18.6 | 3.84 | 5.0 | 5.67 | 3.2 | 3.3 | 3.4 |
| 473059116013901 | Sf Coeur D Alene River Seepage Site No. B-7 | 47.516 | -116.028 | 3 | 45 | 60 | 69 | 12 | 16.1 | 18.5 | 3.58 | 4.9 | 5.61 | 3.3 | 3.3 | 3.4 |
| 473107116020901 | Sf Coeur D Alene River Seepage Site, Rosebud Gulch | 47.519 | -116.036 | 1 | 74 | 74 | 74 | 18.7 | 18.7 | 18.7 | 6.55 | 6.6 | 6.55 | 2.9 | 2.9 | 2.9 |
| 473107116021301 | Sf Coeur D Alene River Seepage Site No.B-8 | 47.519 | -116.037 | 3 | 49 | 62.3 | 70 | 13.2 | 16.7 | 18.7 | 3.94 | 5.1 | 5.7 | 3.2 | 3.3 | 3.4 |
| 473208116064501 | Sf Coeur D Alene River Seepage Site No. C-1 | 47.536 | -116.113 | 2 | 48 | 59 | 70 | 13 | 15.7 | 18.3 | 3.85 | 4.8 | 5.8 | 3.2 | 3.3 | 3.4 |
| 473210116070601 | Sf Coeur D Alene River Seepage Site, Milo Creek | 47.536 | -116.118 | 2 | 35 | 35.5 | 36 | 9.59 | 9.9 | 10.3 | 2.22 | 2.6 | 2.88 | 3.3 | 4.0 | 4.6 |
| 473251116101701 | Sf Coeur D Alene River Seepage Inflow, Govt Gulch | 47.548 | -116.171 | 1 | 27 | 27 | 27 | 7.55 | 7.6 | 7.55 | 1.86 | 1.9 | 1.86 | 4.1 | 4.1 | 4.1 |
| 473252116095301 | Bunker Cr at Mouth Of Culvert at Kellogg, ID | 47.548 | -116.165 | 1 | 2100 | 2100 | 2100 | 594 | 594.0 | 594 | 145 | 145.0 | 145 | 4.1 | 4.1 | 4.1 |
| 473252116101101 | Sf Coeur D Alene River Seepage Site No. C-6 | 47.548 | -116.17 | 3 | 65 | 98.3 | 120 | 17.3 | 26.4 | 32.6 | 5.19 | 8.2 | 9.96 | 3.1 | 3.2 | 3.3 |
| 473253116094001 | Sf Coeur D Alene River Seepage Site No. C-5 | 47.548 | -116.161 | 2 | 54 | 66.5 | 79 | 14.3 | 17.5 | 20.7 | 4.44 | 5.6 | 6.7 | 3.1 | 3.2 | 3.2 |
| 473253116130901 | Sf Coeur D Alene River Seepage Site No. C-10 | 47.548 | -116.219 | 3 | 76 | 105.3 | 120 | 20.4 | 27.8 | 31.6 | 6.17 | 8.9 | 10.2 | 3.1 | 3.2 | 3.3 |
| 473259116122301 | Sf Coeur D Alene River Seepage Site No. C-9 | 47.55 | -116.206 | 1 | 75 | 75 | 75 | 20.1 | 20.1 | 20.1 | 6.03 | 6.0 | 6.03 | 3.3 | 3.3 | 3.3 |
| 473302116115901 | Sf Coeur D Alene River Seepage Site No. C-8 | 47.551 | -116.2 | 3 | 74 | 124.7 | 180 | 19.8 | 33.9 | 49.8 | 5.87 | 10.1 | 14.4 | 3.2 | 3.3 | 3.5 |
| 473328115545601 | Beaver Cr. ab Ferguson Cr nr Delta, ID | 47.558 | -115.916 | 16 | 13 | 27.6 | 50 | 3.68 | 8.3 | 15.2 | 0.72 | 1.7 | 3.07 | 4.3 | 5.1 | 7.6 |
| 473329115541800 | Dobson Creek | 47.558 | -115.905 | 1 | 22 | 22 | 22 | 6.9 | 6.9 | 6.9 | 1.18 | 1.2 | 1.18 | 5.8 | 5.8 | 5.8 |
| 473330115541500 | Bc12-Old | 47.558 | -115.904 | 2 | 33 | 34.5 | 36 | 10.5 | 11.4 | 12.2 | 1.39 | 1.5 | 1.52 | 6.9 | 7.8 | 8.8 |
| 473344115525600 | Cc4-Adit | 47.562 | -115.882 | 2 | 120 | 125 | 130 | 39.5 | 42.5 | 45.5 | 3.61 | 3.6 | 3.68 | 10.7 | 11.7 | 12.6 |
| 473344115531400 | Cc6-Mid | 47.562 | -115.887 | 2 | 16 | 24 | 32 | 5.3 | 8.0 | 10.7 | 0.73 | 0.9 | 1.15 | 7.3 | 8.3 | 9.3 |
| 473345115524500 | Cc2-Above | 47.563 | -115.879 | 2 | 10 | 13.5 | 17 | 3 | 4.1 | 5.1 | 0.63 | 0.9 | 1.07 | 4.8 | 4.8 | 4.8 |
| 473347115534600 | Bc10-Mid | 47.563 | -115.896 | 2 | 10 | 13.5 | 17 | 3.2 | 4.4 | 5.6 | 0.58 | 0.7 | 0.77 | 5.5 | 6.4 | 7.3 |
| 473348115533600 | Pioneer Creek | 47.563 | -115.893 | 1 | 12 | 12 | 12 | 3.7 | 3.7 | 3.7 | 0.72 | 0.7 | 0.72 | 5.1 | 5.1 | 5.1 |
| 473349115532201 | Carbon Cr ab Mouth nr Delta, ID | 47.564 | -115.889 | 7 | 16 | 55.6 | 82 | 5.2 | 18.2 | 27.1 | 0.72 | 2.5 | 3.52 | 7.1 | 7.6 | 9.4 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 473350115532201 | Beaver Cr ab Carbon Cr nr Delta, ID | 47.564 | -115.889 | 3 | 7 | 11 | 16 | 2.1 | 3.2 | 4.5 | 0.49 | 0.8 | 1.23 | 3.7 | 4.4 | 5.3 |
| 473356115515201 | Carbon Cr Bl Headwaters nr Delta, ID | 47.566 | -115.864 | 1 | 23 | 23 | 23 | 6.14 | 6.1 | 6.14 | 1.76 | 1.8 | 1.76 | 3.5 | 3.5 | 3.5 |
| 473404115554801 | Beaver Cr ab No Name Gulch nr Delta, ID | 47.568 | -115.93 | 1 | 52 | 52 | 52 | 14.7 | 14.7 | 14.7 | 3.77 | 3.8 | 3.77 | 3.9 | 3.9 | 3.9 |
| 473421115522200 | Ubc5-Mid | 47.573 | -115.873 | 2 | 7 | 8 | 9 | 2 | 2.4 | 2.7 | 0.46 | 0.5 | 0.57 | 4.3 | 4.5 | 4.7 |
| 473423115520300 | Ubc3-Above | 47.573 | -115.868 | 2 | 7 | 8 | 9 | 2 | 2.4 | 2.8 | 0.45 | 0.5 | 0.57 | 4.4 | 4.7 | 4.9 |
| 473505115555601 | Pony Gulch Cr ab Mouth nr Delta, ID | 47.585 | -115.932 | 1 | 50 | 50 | 50 | 11.3 | 11.3 | 11.3 | 5.36 | 5.4 | 5.36 | 2.1 | 2.1 | 2.1 |
| 473525115440301 | Prichard Cr ab Jo Gulch nr Murray, ID | 47.59 | -115.734 | 1 | 7 | 7 | 7 | 1.98 | 2.0 | 1.98 | 0.556 | 0.6 | 0.556 | 3.6 | 3.6 | 3.6 |
| 473532115475301 | Granite Gulch Cr Bl Moonshine Gulch nr Murray, ID | 47.592 | -115.798 | 1 | 12 | 12 | 12 | 3.45 | 3.5 | 3.45 | 0.77 | 0.8 | 0.77 | 4.5 | 4.5 | 4.5 |
| 473541115453201 | Prichard Cr ab Monarch Gulch nr Murray, ID | 47.595 | -115.759 | 1 | 7 | 7 | 7 | 2.01 | 2.0 | 2.01 | 0.484 | 0.5 | 0.484 | 4.2 | 4.2 | 4.2 |
| 473545115451201 | Paragon Gulch Creek Abv. Mouth nr Murray, ID | 47.596 | -115.753 | 2 | 13 | 13 | 13 | 3.45 | 3.5 | 3.53 | 0.952 | 1.0 | 0.979 | 3.6 | 3.6 | 3.6 |
| 473551115474201 | Granite Gulch Cr ab Mouth nr Murray, ID | 47.598 | -115.795 | 1 | 10 | 10 | 10 | 3.06 | 3.1 | 3.06 | 0.686 | 0.7 | 0.686 | 4.5 | 4.5 | 4.5 |
| 473553115473901 | Prichard Cr Abv Confluence Of Granite Cr nr Raven | 47.598 | -115.794 | 1 | 12 | 12 | 12 | 3.48 | 3.5 | 3.48 | 0.877 | 0.9 | 0.877 | 4.0 | 4.0 | 4.0 |
| 473554115473601 | Prichard Cr ab Granite Gulch Cr nr Delta, ID | 47.598 | -115.793 | 1 | 11 | 11 | 11 | 3.2 | 3.2 | 3.2 | 0.812 | 0.8 | 0.812 | 3.9 | 3.9 | 3.9 |
| 473555115561701 | Beaver Cr Bl Gleveland Gulch nr Delta, ID | 47.599 | -115.938 | 1 | 38 | 38 | 38 | 9.88 | 9.9 | 9.88 | 3.27 | 3.3 | 3.27 | 3.0 | 3.0 | 3.0 |
| 473605115475401 | Bear Gulch Cr ab Mouth nr Murray, ID | 47.601 | -115.798 | 1 | 7 | 7 | 7 | 1.95 | 2.0 | 1.95 | 0.506 | 0.5 | 0.506 | 3.9 | 3.9 | 3.9 |
| 473630115562901 | Trail Cr ab Mouth nr Delta, ID | 47.608 | -115.941 | 1 | 76 | 76 | 76 | 19 | 19.0 | 19 | 6.86 | 6.9 | 6.86 | 2.8 | 2.8 | 2.8 |
| 473641115492701 | Prichard Cr Bl ID Gulch nr Murray, ID | 47.611 | -115.824 | 1 | 10 | 10 | 10 | 2.74 | 2.7 | 2.74 | 0.715 | 0.7 | 0.715 | 3.8 | 3.8 | 3.8 |
| 473648115493501 | Butte Gulch Cr nr Muuray, ID | 47.613 | -115.826 | 1 | 15 | 15 | 15 | 4.13 | 4.1 | 4.13 | 1.05 | 1.1 | 1.05 | 3.9 | 3.9 | 3.9 |
| 473655115470201 | Bear Gulch Cr nr Round Top Mtn nr Murray, ID | 47.615 | -115.784 | 1 | 7 | 7 | 7 | 1.83 | 1.8 | 1.83 | 0.531 | 0.5 | 0.531 | 3.4 | 3.4 | 3.4 |
| 473702115572501 | Beaver Cr Bl Prospect Gulch nr Delta, ID | 47.617 | -115.957 | 1 | 44 | 44 | 44 | 10.8 | 10.8 | 10.8 | 4.02 | 4.0 | 4.02 | 2.7 | 2.7 | 2.7 |
| 473732115513001 | Prichard Cr at Murray, ID | 47.626 | -115.858 | 12 | 5 | 9.2 | 13 | 1.27 | 2.5 | 3.5 | 0.361 | 0.7 | 0.945 | 3.5 | 3.6 | 3.8 |
| 473840115551701 | Prichard Cr Abv Eagle Cr at Eagle, ID | 47.644 | -115.921 | 1 | 13 | 13 | 13 | 3.4 | 3.4 | 3.4 | 1.08 | 1.1 | 1.08 | 3.1 | 3.1 | 3.1 |
| 473841115551601 | Eagle Cr Abv Mouth at Eagle, ID | 47.645 | -115.921 | 1 | 15 | 15 | 15 | 4.04 | 4.0 | 4.04 | 1.29 | 1.3 | 1.29 | 3.1 | 3.1 | 3.1 |
| 473925115530200 | Ef Eagle Creek nr Mouth nr Prichard, ID | 47.657 | -115.884 | 1 | 12 | 12 | 12 | 2.92 | 2.9 | 2.92 | 1.04 | 1.0 | 1.04 | 2.8 | 2.8 | 2.8 |
| 473930115530101 | Ef Eagle Cr Abv Fancy Gulch nr Eagle, ID | 47.658 | -115.884 | 12 | 5 | 9.8 | 15 | 1.23 | 2.5 | 3.72 | 0.429 | 0.9 | 1.31 | 2.8 | 2.9 | 3.0 |
| 474011115450401 | Tributary Cr Bl Headwaters nr Jack Waite Forks, ID | 47.67 | -115.751 | 1 | 83 | 83 | 83 | 17.4 | 17.4 | 17.4 | 9.5 | 9.5 | 9.5 | 1.8 | 1.8 | 1.8 |
| 474017115530601 | Wf Eagle Cr Abv Nocelly Gulch nr Eagle, ID | 47.671 | -115.885 | 12 | 6 | 9.7 | 13 | 1.54 | 2.6 | 3.61 | 0.509 | 0.8 | 1.08 | 3.0 | 3.2 | 3.3 |

| Station ID | Descriptive Name | latdd | longdd | Count | Min. H | Ave. H | Max H | MinCa | AveCa | MaxCa | MinMg | AveMg | MaxMg | MinCa/Mg | AveCa/Mg | MaxCa/Mg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 474041115484401 | Ef Eagle Cr Bl Toboggan Cr nr Jack Waite Forks, ID | 47.678 | -115.812 | 1 | 14 | 14 | 14 | 3.36 | 3.4 | 3.36 | 1.34 | 1.3 | 1.34 | 2.5 | 2.5 | 2.5 |
| 474111115465201 | Upper Ef Eagle Cr Blw Trib Cr nr Jack Waite Fork | 47.686 | -115.781 | 1 | 22 | 22 | 22 | 5.27 | 5.3 | 5.27 | 2.13 | 2.1 | 2.13 | 2.5 | 2.5 | 2.5 |
| 474118115463101 | Prichard Creek Blw Paragon Creek nr Murray, ID | 47.688 | -115.775 | 2 | 6 | 6 | 6 | 1.8 | 1.8 | 1.8 | 0.406 | 0.4 | 0.407 | 4.4 | 4.4 | 4.4 |
| 474118115463201 | Tributary Cr ab Mouth at Jack Waite Forks, ID | 47.688 | -115.776 | 5 | 23 | 24.6 | 29 | 5.27 | 5.7 | 6.65 | 2.31 | 2.5 | 2.97 | 2.2 | 2.2 | 2.3 |
| 474212115513501 | West Fork Eagle Creek Abv Bobtail Cr nr Eagle, ID | 47.703 | -115.86 | 4 | 12 | 12.3 | 13 | 3.18 | 3.2 | 3.32 | 0.988 | 1.0 | 1.05 | 3.2 | 3.2 | 3.2 |
| | Total observations | | | 3594 | | | | | | | | | | | | |
| | | Minimum | | | 4 | 4.7 | | | | | | | | | | |
| | | 1st-percentile | | 4.23 | 4.7 | | | | | | | | | | | |
| | | 5th percentile | | 6.15 | 4.7 | | | | | | | | | | | |

Appendix B

## How to measure insignificance? Comparisons between NOECs, EC1s, and EC0s and the lower confidence limit of EC10s to estimate "insignificant effects"

### Summary

To assist in our analysis, NMFS considered what toxicity test statistic best approximated a "true" no-effect concentration for evaluating risks to ESA listed species, We made a comparison of "no-observed effect concentrations" (NOECs) versus regression or distribution based methods for estimating no- or very low effects concentrations. The alternative statistics were regression- or distribution based estimates of the EC1 or EC0 (i.e., concentrations causing adverse effects to 1% or 0% of a test population), and the lower 95$^{th}$ percentile confidence limit of the concentration affecting 10% of the test population (LCL- EC10), which is a statistic used in human health risk assessment for determining benchmark doses of materials that present low increased risk (EPA, 2000a), Our conclusion was that if the data sets had a gradient of effects that would allow calculation of an EC0, the EC0 would be the preferred, best estimate of no-effect value from a toxicity test. If data were insufficient to calculate an EC0, the NOEC may be the best appropriate statistic.

### The problem

In evaluations of the risks of chemicals to aquatic species listed as threatened or endangered, the statistical interpretation of toxicity testing has become an issue. Classically, the interpretation of chronic or sublethal tests has involved the use of statistical hypothesis testing, the results of which are commonly reported as "no-observed effect concentration" (NOEC) or "lowest-observed effect concentration" (LOEC). Definitions vary, but for this analysis the LOEC will be considered the lowest concentration for which there is a 95% probability that the biological response of interest (survival, growth, fecundity, etc.) is different from the control response. Similarly, the NOEC is considered the next lowest treatment. It has been assumed that somewhere between the NOEC and LOEC lies a maximum acceptable toxicant concentration (MATC) that represents a "true" but unknown threshold for unacceptable effects. In practice, the MATC concentration is estimated as a simple geometric mean between the NOEC and LOEC (Gelber *et al.*1995). This is the value usually used in EPA criteria documents to estimate "safe" concentrations from a chronic toxicity test, although the term "MATC" is avoided in the Guidelines and instead the statistic is called a "chronic value" for a test. MATCs in turn are averaged to obtain species mean chronic values, and ultimately to set chronic criteria values.

The EPA criteria approach seems to conflict with concepts for evaluating risk to listed species because the EPA approach of averaging NOECs and LOECs assumes that aquatic communities are resilient to, or can recover from, some low-level of adverse effects. In contrast, if a species was listed as threatened or endangered, it is assumed to have substantially less resiliency than general aquatic communities. Therefore, in interpreting toxicity test data, a statistic that by definition includes some uncertain but probably low level of adverse effect such as the EPA "chronic value" is inappropriate as a statistic of effects on listed species that are expected to be discountable or insignificant. In the ESA Consultation Handbook for evaluating effects of actions to listed species, states that*" ' insignificant effects' relate to the size of the impact and should never reach the scale where take occurs. Discountable effects are those extremely unlikely to occur. Based on best judgment, a person would not: (1) be able to meaningfully measure, detect, or evaluate insignificant effects; or (2) expect discountable effects to occur." (USFWS and NMFS 1998).* Thus a meaningful measurement of low-effects from a toxicity test such as an EC10 or EC5 is inherently in conflict with a definition that requires "insignificant" effects to be unmeasurable.

An obvious substitute for use in ESA consultations is the NOEC, and indeed that is the default statistic selected in EPA's methodology for conducting biological evaluations of aquatic life criteria (EPA 2003). However, in recent years the concept of the NOEC has been battered in the ecotoxicology literature. The three complaints relate to the common design of toxicity experiments which usually involve a series of about five treatment concentrations plus a control, each replicated about three times. Complaint #1 is that a NOEC has to be one of the concentrations tested, so its precision is dependent on the number and spacing of treatment concentrations. So for example, if the unknown "true" no-effect concentration is 1.8 µg/L a test series of 1.0, 1.2, 1.6, 2.0, …. will give a more precise NOEC estimate than a series of 1, 2, 4, 8, 16, ….(1.6 vs. 1.0 µg/L). Complaint #2 is that for the low levels of replication used (often 3), the minimum statistically detectable effect level can vary widely, easily from 5 to about 40% for endpoints with low or high variability (e.g. growth in fish (low) or fecundity in invertebrates (highly variable). The NOEC statistic by itself gives no insight into whether a "significant" effect is biologically trivial or whether an effect is biologically serious but too variable to be significant at the arbitrary limit that no more than a 5% risk of being wrong is acceptable (acceptable to the evaluators, not whether it is acceptable to the organism). Complaint #3 is related in that the NOEC-LOEC approach is solely focused on the "Type I" error, or the risk of declaring an adverse effect when the observed effects occurred solely by chance, with no or little regard for Type II error, the risk of failing to detect an adverse effect that was really present but the test had insufficient power to detect it. Type II error rates may be quite high in ecotoxicological studies that fail to detect effects as "significant" at the 5% Type I error rate (Stephan and Rogers, 1985; Laskowski, 1995; Moore and Caux, 1997; Crane and Newman, 2000; McGarvey, 2007; Newman, 2008; Brosi and Bilber, 2009).

An alternative often put forth to the NOEC-LOEC approach is regression or distribution based techniques that fit an effects curve to the observed data, and then any point along that curve can be used to estimate effects at a given concentration. This regression or distribution based approach is the most common technique for defining $LC_{50}$s in acute data but obviously other effect concentrations percentiles (ECp) besides the 50$^{th}$ percentile could be of interest. The catch in this approach is that it is up to the assessor to independently determine what level of effect is "important." Choices of what level of effect is "important" have either been made subjectively or by comparisons of ECp values back to NOECs and LOECs. For example, in the interpretation of EPA's chronic whole-effluent toxicity (WET) tests, NOECs are assumed to be equivalent to an EC25. The conclusion that a 25% adverse effect in a biologically important endpoint therefore represents a no-observed effect concentration was supported by a citation to an analysis of 23 pooled chronic WET test results for red algae, sheepshead minnows, sea urchins, *Ceriodaphnia*, and fathead minnows in which NOECs were more frequently similar to EC20s (EPA, 1991, p. 27). No reason was given why the EC25 was endorsed over the EC20, since the analysis supported the use of the EC20, but regardless the EC25 is often the trigger statistic in WET tests.

Subsequent analyses have also shown that NOECs are usually higher than point estimates of low toxic effects such as the EC10 (Moore and Caux, 1997; Crane and Newman, 2000). In an analysis limited to the effects of cadmium, NMFS found that the typical expected adverse effect associated with MATC was often about 20-30% with invertebrates and about 10-15% for fish (Mebane, 2006). However, using ECx values that correspond with a NOEC or MATC to select "x" as a suitable replacement for the unsuitable NOEC falls into circular reasoning. A counterpoint could be made that comparisons of ECps and NOECs to support an ECp value to replace NOECs is a tautology. Instead of matching statistics, biological arguments could be made for assuming different "acceptable" ECp values based upon patterns of variability of the same endpoints in natural populations, life history strategies, projecting effects in population models, and field studies relating year class survival to size differences. No comprehensive analysis along these lines is known to have been published.

Mebane and Arthaud (2010) gave an example of what effect-statistics could be related to population extinction risks or recovery trajectories for a headwaters threatened Chinook salmon population. In this population, Marsh Creek in the upper Salmon River, Idaho, survival of juvenile migrants is strongly related to the size of the fish. A size reduction of 4% as length, i.e., an EC04, was associated with survival reductions ranging from 12 – 38% for different migrant groups from a trap near the headwaters to the first dam encountered downstream. In the toxicity tests with Chinook salmon and rainbow trout that were analyzed for the study, a 4% reduction in length corresponded with about a 12% reduction in weight. When the survival reductions associated with a length EC04 were extrapolated through a population model to changes in extinction risk or recovery time, little difference in extinction risk was projected but an appreciable delay in recovery was projected. This indicates that at least for the length endpoint in chronic fish toxicity tests, the statistical threshold for important adverse effects may not be much higher than statistics such as an EC0 or EC01. Yet for the commonly used weight endpoint in chronic fish toxicity tests, the statistical threshold for important adverse effects would be higher, around the EC10. Presumably, if endpoints are more variable, such as the number of eggs produced per female (fecundity), then a higher ECp value (e.g. EC20) might be appropriate. While the relevance of this example to other species or even different populations of Chinook salmon is not known, it does at least serve as one example of a basis to judge the importance of an ECp value without relying on circular comparisons back to other statistics.

## Comparisons between statistics

For this exercise, NMFS evaluated data from a variety of available toxicity tests results that were available in the syntax required by the statistics models. While such data are not comprehensive or necessarily definitive, they are preferable to many journal articles because the latter are sometimes too summarized to make any subsequent analysis of. We selected the examples to illustrate a variety of response patterns ranging from classic, concentration-responses to test results that are difficult to interpret.

NMFS used either reported NOECs or those that could be estimated using Dunnett's test. ECp values were estimated for growth and reproduction using a distribution analysis for survival data (respondents are either alive or dead) or nonlinear regression for more or less continuous data (growth or fecundity measurements). For each type of analysis, a choice of underlying distributions of the populations must be assumed.

(1) Gaussian (Normal) Distribution: This is based on the familiar "bell curve" or gaussian distribution. This produces a sigmoidal toxicity relationship with infinite tails, and is equivalent to prohibit analysis.

(2) Triangular Distribution: This produces a sigmoidal toxicity relationship similar to the gaussian distribution, but with a finite threshold exposure below which responses are zero and a finite exposure above which all organisms are affected. It is also referred to as a "sigmoid threshold" (Erickson 2008).

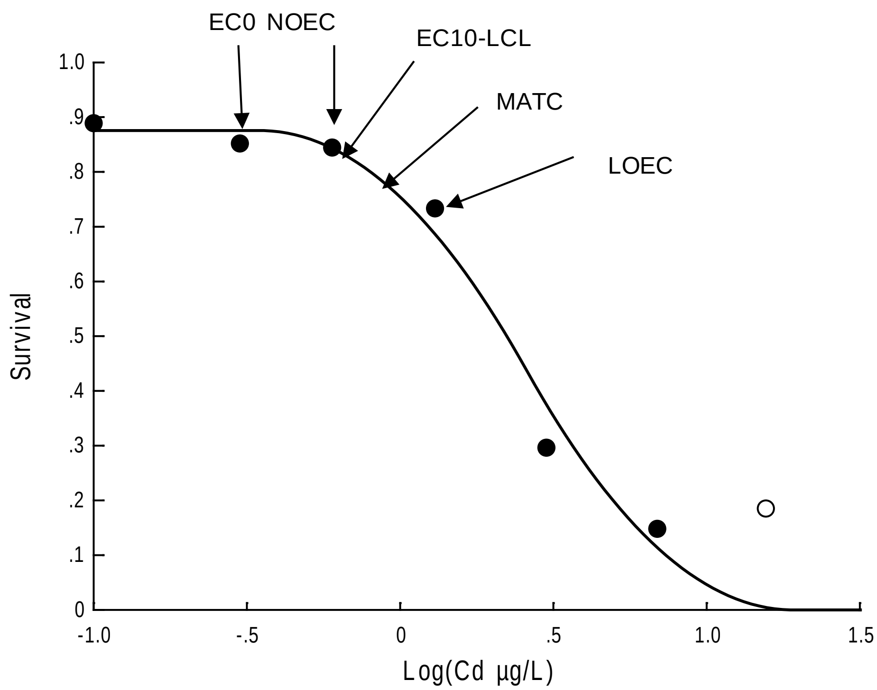
(3) Uniform (Rectangular) Distribution: This produces a piecewise-linear toxicity relationship, for which there is a finite lower and upper exposure limit like the triangular distribution, but for which the decline in response between these limits is linear rather than sigmoidal. Similar analyses have been called "jackknife distributions" in the literature because of its shape.

The assumed statistical distribution and behavior of the data in the tails of the distribution are usually of little consequence when one is trying to estimate the middle of the distribution ($LC_{50}$). However, when one is trying to estimate no-effects data, these estimates are at the extreme tails of the distribution, and the shape of the tails and the behavior of the models become more important. In the Gaussian, normal distribution, an EC0 can never be achieved because the tails are infinite; in other words some rare organisms are assumed to be infinitely resistant and some sensitive to infinitesimal exposures. Because that assumption is not plausible for ecotoxicology data, methods have been developed using discrete distributions with definite ends, i.e. no organism is infinitesimally sensitive, and an EC0 can be calculated.

NMFS calculations used a beta version of the Toxicity Response Analysis Program, under development EPA's National Health and Environmental Research Laboratory, Mid-Continent Ecological Division (Erickson 2008).

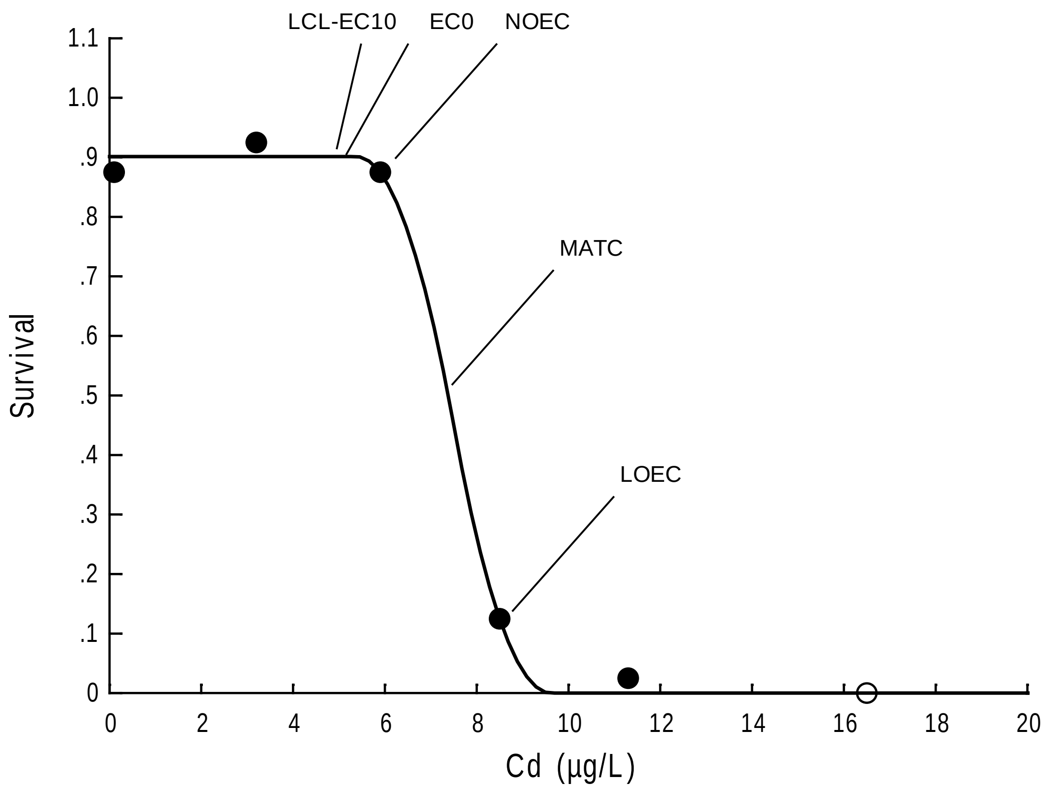
## Examples:

**Example 1.  Rainbow trout 53-day survival with cadmium, using the sigmoid threshold model based upon an assumed triangular distribution.  Open circles indicate data points that were excluded from the regression**




| ECp | ECp est | 95 LCL | 95% UCL |
|---|---|---|---|
| 10 | 0.85 | 0.62 | 1.17 |
| 0 | 0.35 | 0.21 | 0.59 |

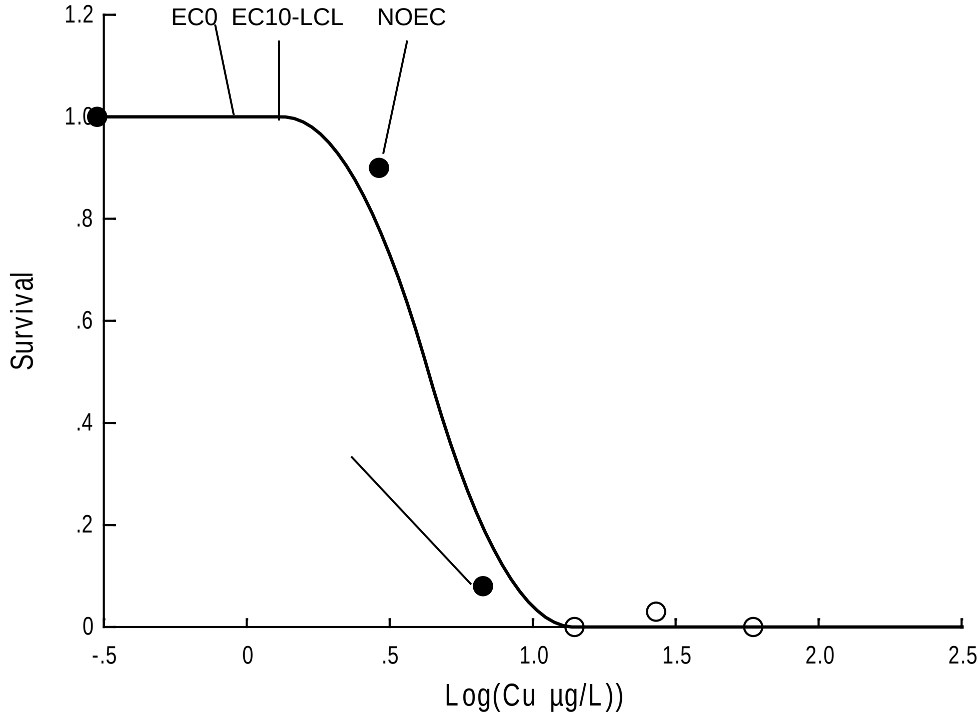
**Example 1 was selected to illustrate the classic ski jump curve shape, where the initial part of the curve from the control out to the 2nd treatment shows a slight decline, followed by a steep drop in the center region of the curve where intermediate effects occur, followed by a flattening out of the slope at the bottom as almost all animals are predicted to be killed (Mebane *et al.*, 2008).**




**Example 2.  Fountain darter, 7-day survival with Cd, sigmoid threshold, showing a very steep curve that results with (nearly) all-or-nothing responses.   In this case, all of the "nearly-no-effect" estimators give similar values.**

| ECp | ECp est | 95 LCL | 95% UCL |
|---|---|---|---|
| 10 | 6.33 | 5.06 | 7.59 |
| 0 | 5.38 | 2.84 | 7.92 |

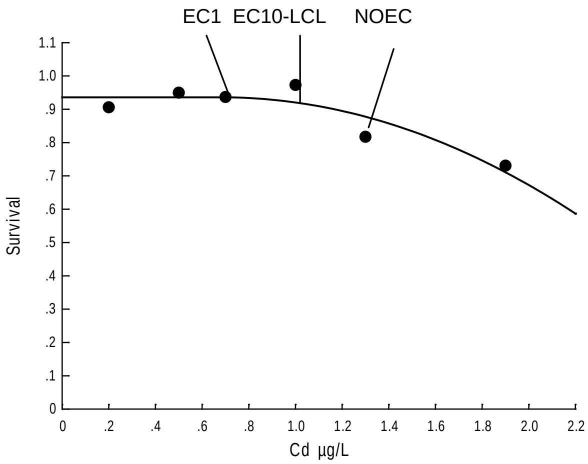
(Castillo and Longley, 2001)

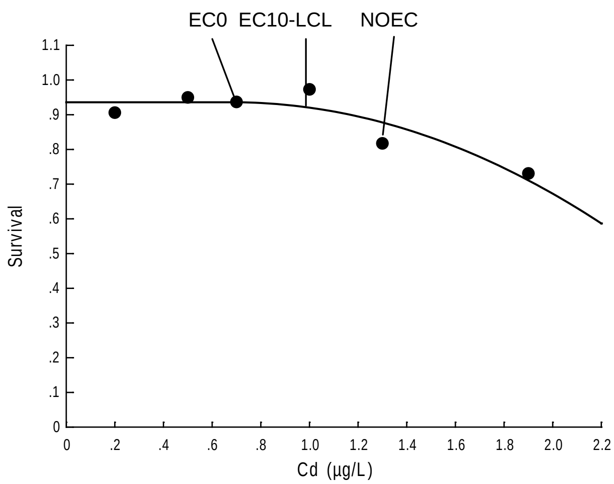


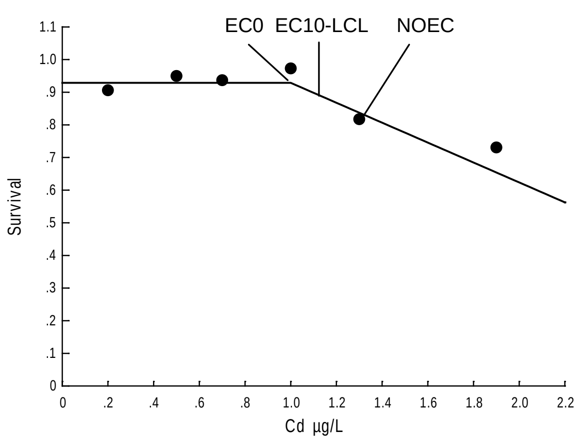

**Example 3.  Mottled sculpin, 14-day survival with copper (Besser and others, 2007).  As with example 2, these data had inadequate partial responses resulting in an uncertain fit between the control and treatment 1, the NOEC.  Even so, ECp estimates are reasonable and confidence limits are not large. These type of data are often encountered working with listed species or other poorly tested species for which investigators have little idea in advance what exposure change to test.**

| ECp | ECp est | 95 LCL | 95% UCL |
|---|---|---|---|
| 10 | 2.255 | 1.841 | 2.762 |
| 5 | 1.934 | 1.516 | 2.466 |
| 0 | 1.334 | 0.924 | 1.925 |

LOEC



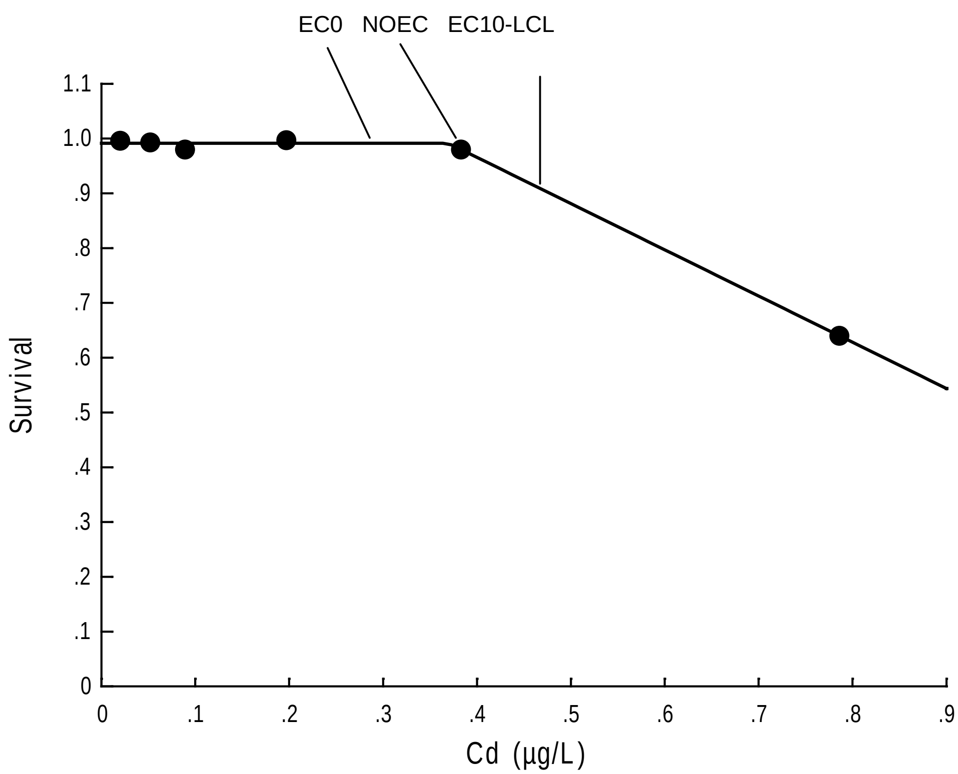



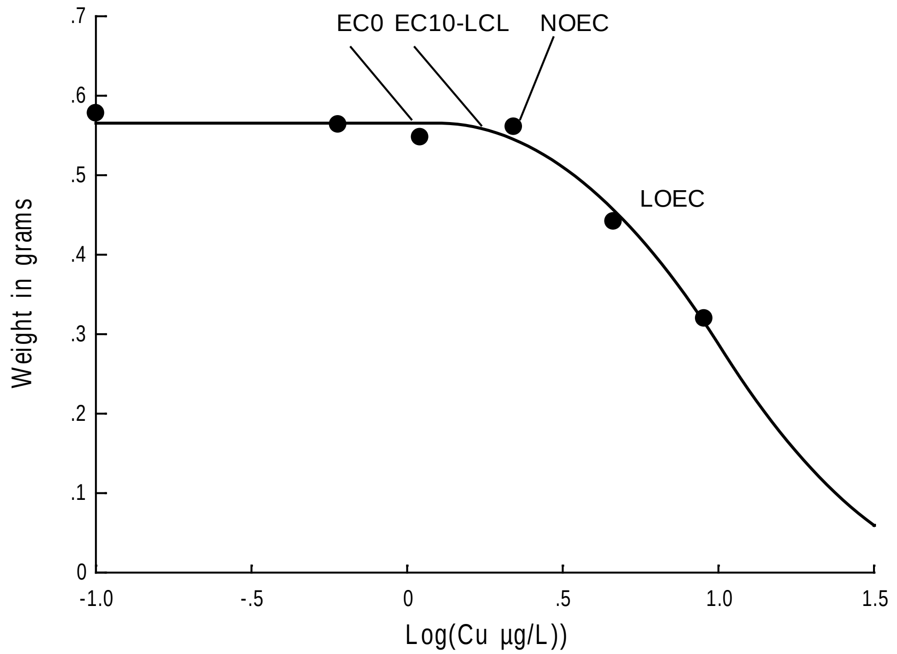

**Example 4. Chinook salmon 120 day survival with Cd (Chapman, 1982), illustrating differing ECp estimates resulting from different statistical models. Note that EC0s are conceptually impossible using the normal distribution, but the EC1 in the top figure is close to the EC0 in the middle figure using the triangular distribution. In this example, the linear model (bottom) does the best job of finding the no-effect estimate (visually, treatment 3, the 4$^{th}$ point from the left). Despite the very different underlying models, all ECp estimates were similar in this example.**

Chinook and Cd ECp values | Bull trout and Cd ECp values:

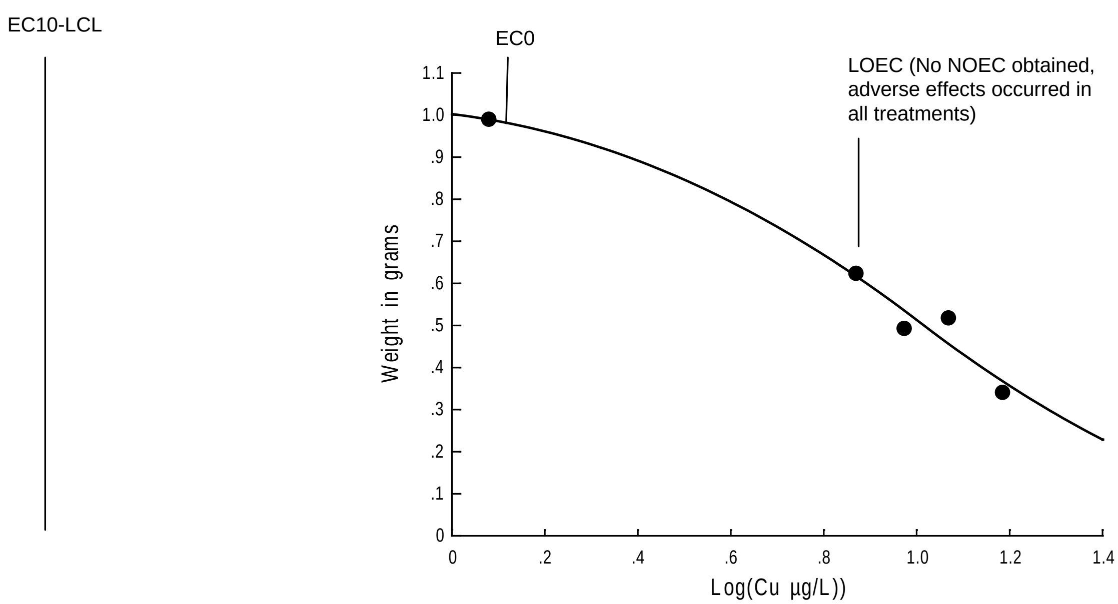
| Gaussian | ECp est | 95 LCL | 95% UCL | ECp | ECp est | 95 LCL | 95% UCL |
|---|---|---|---|---|---|---|---|
| 20 | 1.802 | 1.541 | 2.063 | | | | |
| 10 | 1.480 | 1.042 | 1.919 | | | | |
| 5 | 1.215 | 0.503 | 1.927 | | | | |
| 1 | 0.717 | -0.789 | 2.223 | | | | |
| Triangular | | | | | | | |
| 20 | 1.792 | 1.529 | 2.056 | | | | |
| 10 | 1.466 | 1.144 | 1.788 | 10 | 0.555 | 0.477 | 0.633 |
| 5 | 1.236 | 0.764 | 1.707 | 5 | 0.479 | 0.379 | 0.578 |
| 0 | 0.679 | -0.454 | 1.811 | 0 | 0.294 | 0.117 | 0.471 |
| Rectagular | | | | | | | |
| 20 | 1.609 | 1.366 | 1.852 | 20 | 0.60496 | 0.58075 | |
| 10 | 1.304 | 1.114 | 1.495 | 10 | 0.48715 | 0.46026 | |
| 5 | 1.152 | 0.953 | 1.352 | 5 | 0.42824 | 0.39815 | |
| 0 | 1.000 | 0.763 | 1.237 | 0 | 0.36933 | 0.33522 | |



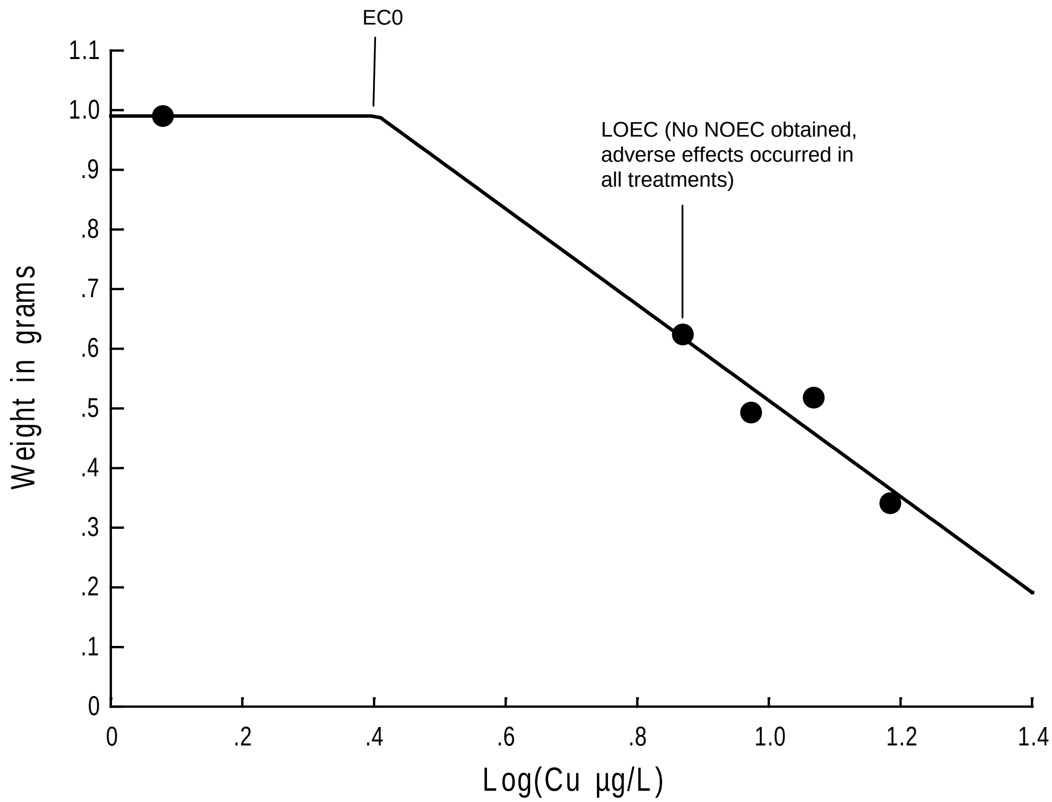

**Example 5. Bull trout, 55-day survival with Cd (Hansen and others, 2002c). Here the NOEC is lower than the LCL-EC10. Similar to the Chinook salmon and Cd example, these data would give an inadequate and highly unreliable response for an LC50. However, with chronic testing the interest is in the low-effect part of the curve.**




**Example 6. Growth of rainbow trout after 60-days Cu exposure (Marr and others, 1996). This data set is nicely balanced with 3 nearly no-effect treatments and 2 treatments above a clearly defined effects threshold.**




**Example 7. Growth of Chinook salmon after 120-days Cu exposure, sigmoid threshold model (Chapman, 1982). This data set presents uncertain EC0 values because adverse effects occurred in all tested treatments. The LCL-EC10 is less than zero which is clearly impossible and using the sigmoid model, the EC0 falls close to the control. There is no NOEC, although in some data compilations the "less than" for this treatment was lost in translation and the NOEC or chronic value has been treated as 7.4 µg/L rather than < 7.4 µg/L. This mistake results in a 40% reduction in growth being treated as a low- or no-effect.**



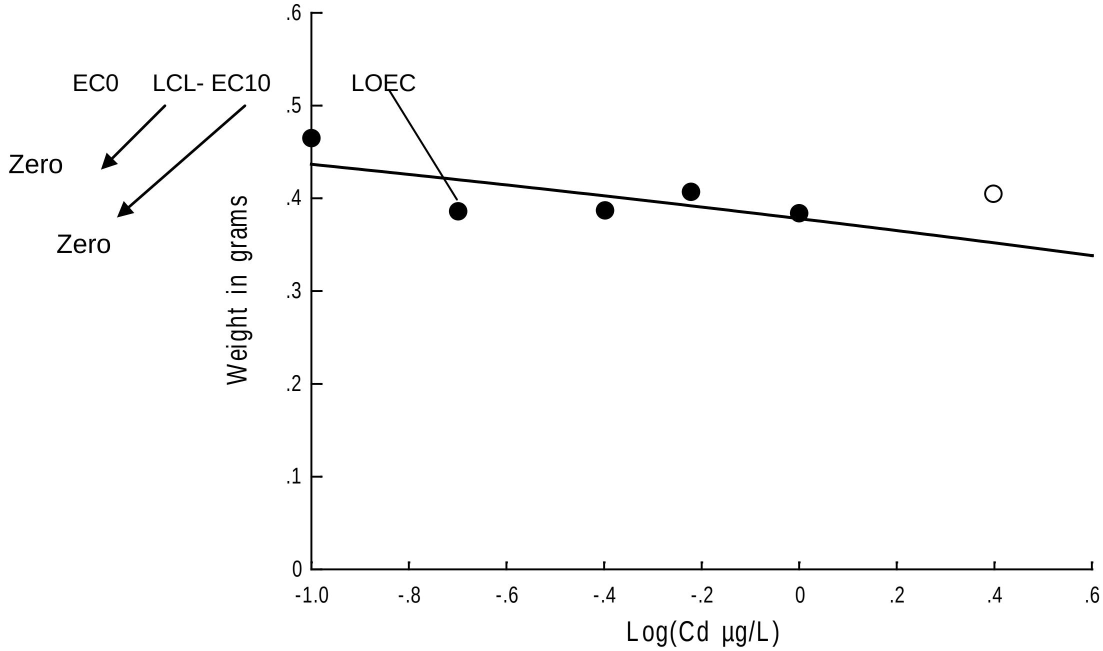

**Example 8. Growth of Chinook salmon after 120-days Cu exposure, piecewise linear response (Chapman, 1982). Curves do not always give better fits; here it is more plausible that the onset of adverse effects occurs at a higher copper concentration than the controls. However, in data sets such as this, the interpolation between the control and first treatment data set is so large that the shape of the curve and thus the response is less a statistical question than a professional judgment about what seems most plausible.**

Chinook growth (sigmoid threshold)

| ECp | ECp est | 95 LCL | 95% UCL |
|---|---|---|---|
| 10 | 2.215 | 0.026 | 185.270 |
| 1 | 0.954 | 0.000 | 12238.000 |
| 0 | 0.646 | 0.000 | 652500.000 |

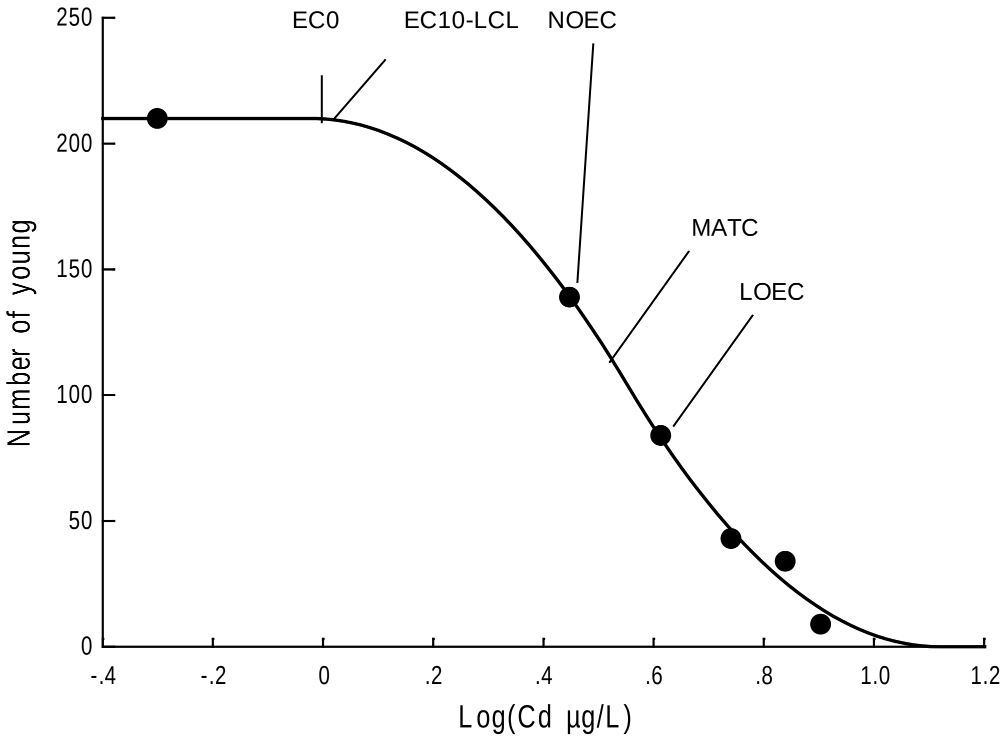
Chinook growth (piecewise linear)

| | | | |
|---|---|---|---|
| 10 | 3.386 | 0.699 | 16.399 |
| 1 | 2.623 | 0.396 | 17.354 |
| 0 | 2.550 | 0.372 | 17.468 |



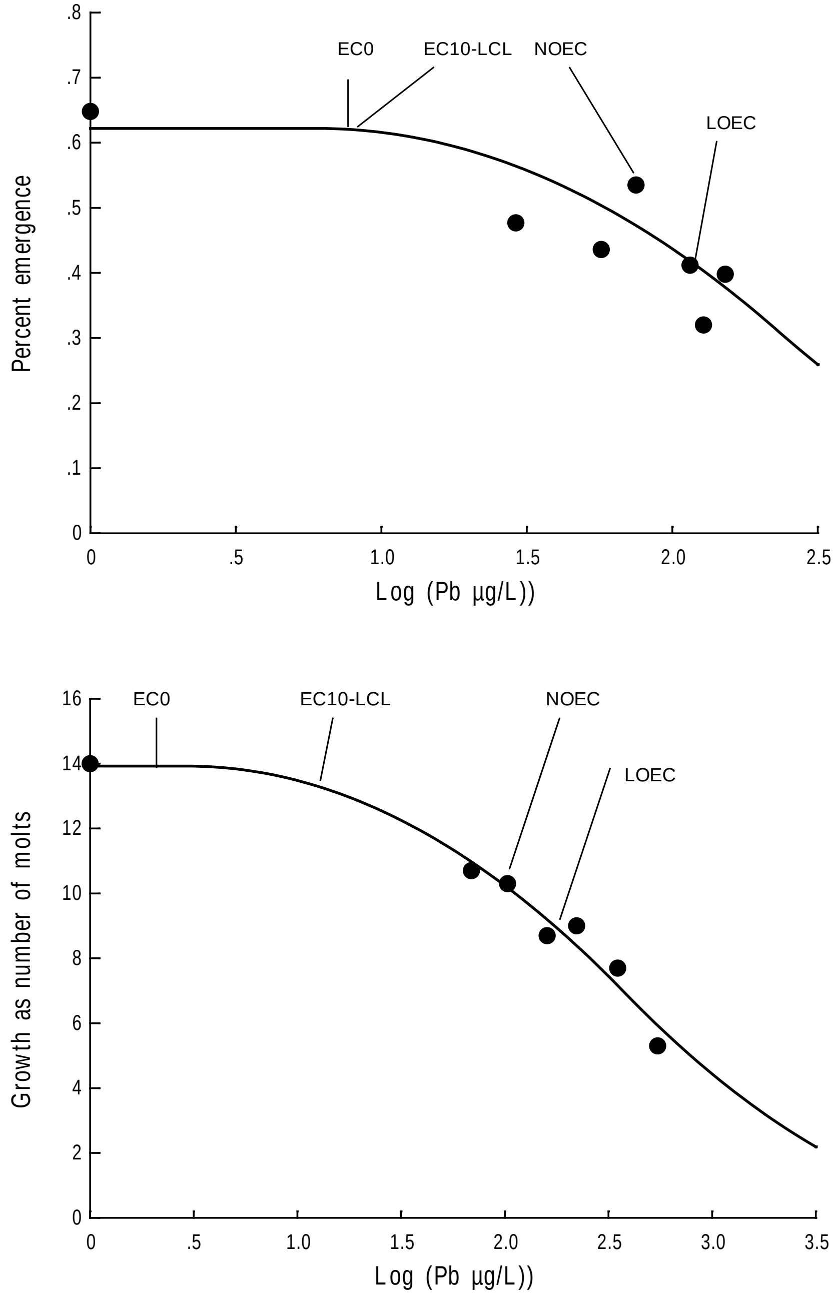

**Example 9. Rainbow trout growth after 62-d exposure to Cd (Mebane *et al.* 2008). This example is similar to the Chinook salmon and Cu example in that statistically significant effects were observed in all treatments and no NOEC could be obtained. Further, because no monotonically decreasing concentration response was observed, the curve was almost flat and ECp values are meaningless (numerous errors and warnings were overridden to create this example). In this example, statistics of any type offer little help in interpreting the data.**

| ECp | ECp est | 95 LCL | 95% UCL |
|---|---|---|---|
| 50 | 16.61600 | 0 | Infinity |
| 20 | 0.02234 | 0 | Infinity |
| 10 | 0.00080 | 0 | Infinity |
| 5 | 0.00008 | 0 | Infinity |
| 0 | 0.00000 | 0 | Infinity |

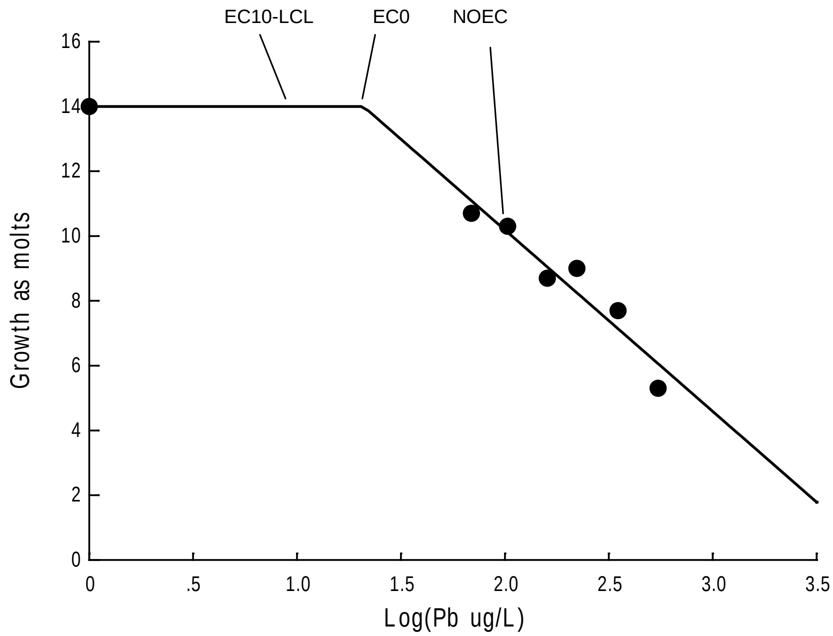



**Example 10.  Reproduction of *Ceriodaphnia dubia* after 7-d exposure to Cd (Castillo and Longley, 2001).  In this test, the NOEC reported by the authors corresponded to about a 35% reduction in reproduction, and greater than a 50% reduction for the MATC .**

| ECp | ECp est | 95 LCL | 95% UCL |
|---|---|---|---|
| Sigmoid | 2.800 | | |
| 25 | 2.414 | 1.970 | 2.957 |
| 10 | 1.716 | 1.239 | 2.375 |
| 1 | 1.148 | 0.577 | 2.282 |
| 0 | 0.953 | 0.482 | 1.887 |




**Example 11.  Emergence of midge (*Chironomus tentans*) larvae following 21-days exposure to Pb (Top); Mayfly (*Baetis tricaudatus*) molting during 10-days exposure to Pb  (Mebane *et al.* 2008). Examples of less than ideal datasets that can arise from testing of non-standard organisms or tests conducted in environmentally realistic but noisy experiments (these were streamside tests).  The shape of the curves in both datasets suggest an onset of effects below the lowest concentration tested. This suggests both that NOECs may not be conservative and that low ECp values are uncertain.**




**Example 12. (Continued) Same mayfly (*Baetis tricaudatus*) as above, but using a piecewise linear or jackknife distribution. As with the case of copper and Chinook salmon growth, assuming a curved distribution would cause the EC0 estimates to be near the control. If that were to be considered implausible, the jackknife "curve" provides a higher "no-effect" value that statistically is equally valid.**

| **C. tentans, Pb Emergence, logistic** | | | |
|---|---|---|---|
| ECp | ECp est | 95 LCL | 95% UCL |
| 10 | 30.697 | 5.793 | 162.670 |
| 5 | 19.039 | 1.962 | 184.720 |
| 0 | 6.009 | 0.067 | 540.460 |

| **Mayfly, Baetis tricaudatus - logistic** | | | | **Mayfly, Baetis tricaudatus – "Jackknife** | | |
|---|---|---|---|---|---|---|
| ECp | ECp est | 95 LCL | 95% UCL | ECp est | 95% UCL | 95% UCL |
| 20 | 63.159 | 25.394 | 157.090 | 65.972 | 28.888 | 150.66 |
| 10 | 25.713 | 6.721 | 98.375 | 37.103 | 13.112 | 104.99 |
| 5 | 13.620 | 2.514 | 73.806 | 27.825 | 8.7638 | 88.342 |
| 0 | 2.937 | 0.206 | 41.873 | 20.867 | 5.8379 | 74.585 |

## Conclusions

In most of these comparisons, the rank order of the "effects" concentrations were EC0< LCL-EC10 <NOEC. Of the statistics examined, the LCL-EC10s seems particularly suspect. Generally, LCL-EC10 estimates were close to EC0 or EC1 values, however, in all cases where reasonable LCL-EC10 estimates could be obtained, so could EC1 or EC0 values. Confidence intervals on very low effect estimates are large, but at least for EC1 or EC0 values, confidence intervals can be calculated. No confidence limits can be calculated on a confidence limits, and there is no logical reason why the LCL-EC10 is a better estimate of an EC1 or EC0 than would be the EC1 or EC0 themselves. In sum, no empirical or theoretical reason for using LCL-EC10 statistic could be envisioned.

In most instances, the differences between the NOECs, LCL- EC10s, and EC0s were small. This suggests that given the magnitude of uncertainty involved in other aspects of evaluating risks to listed species such as extrapolating effects between species, and extrapolating acute-to-chronic effects, the choice of which statistic used to estimate "no-effect" for a given test response may be of less importance. Some datasets were less than ideal for the statistical models. For most datasets, estimates of these extreme statistics seemed reasonable, based on the datasets from which they were derived. Confidence limits were very large, but the estimates themselves seemed reasonable. Some ECp analyses were uncertain, most commonly because of inadequate partial effects resulting in uncertainty in the shape of the response curve. In other tests, adverse effects resulted in all treatments, so no NOEC could be determined. Differences in results obtained using different assumed statistical distributions (normal, triangular, rectangular) were small.

The results of NMFS' analysis suggest that for initial screening of large databases for chronic effects concentrations to compare with criteria values, any of the NOEC, LCL-EC10, EC1, or EC0 statistics could be useful, and the choice of which statistic to use will probably depend on which is most available. However, in instances where the test is influential in the assessment, a more careful review of the original research might enable the assessor to make a more informed judgment of whether the test indicates reassurance of the lack of effects or indicates that adverse effects are likely. These judgments cannot always follow rote statistical analyses.

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# Appendix C

## An evaluation of the accuracy and protectiveness of EPA's 2007 biotic ligand model (BLM)-based copper criteria for copper

*Contents* 

**Summary**

In 2007, EPA revised their national freshwater ambient water quality criteria for copper. The 2007 criteria replaced the longstanding statistical hardness-toxicity site-specific modifiers of copper toxicity with the much more advanced and complex biotic ligand model (BLM). The BLM uses a more mechanistic approach, which combines a geochemical model of copper speciation and binding to dissolved organic carbon (DOC) in the water, and a model of competition between copper and major ions in water for binding sites on the gills of fish or other biological surfaces. The BLM predictions include the concentration of total dissolved copper in water that is predicted to accumulate on the gills of fish, or for small invertebrates that have less defined gill structures, other surface tissues, to non-specific critical accumulation levels that kill the organisms. The version of the model used in the 2007 criteria was supported by a large body of research (Di Toro *et al.* 2001; Santore *et al.* 2001; EPA 2003b; Niyogi and Wood 2004; EPA 2007a). Because the hardness-based copper criteria which date from the 1980s have not been demonstrated to be consistently protective of listed salmonids and their ecosystems, the application of the more recent and scientifically advanced BLM-based criteria is an obvious potential alternative.

However, there are fundamental questions about the BLM's performance and BLM-criteria's protectiveness. These include:

1. The BLM concept is intended to be capable of predicting copper toxicity to any aquatic animal but the performance of the BLM has been principally validated with toxicity data from fathead minnows and daphnids. Does the BLM reliably predict the toxicity or non-toxicity to other aquatic organisms including salmonids?

2. Earlier versions of the BLM were criticized for under predicting toxicity in low hardness waters and over predicting the mitigating effect of dissolved organic carbon (DOC). Is this still the case with the 2007 version? If so, are these concerns important enough to recommend against the use of the 2007 criteria?
3. The BLM that the 2007 criteria are based upon is an acute toxicity model for predicting short-term, lethal (acute) toxicity of copper. However, for the 2007 criteria it was extrapolated to predict for, and protect against chronic effects as well. No analyses of the efficacy of this extrapolation were included or referenced in the 2007 criteria document. Does this BLM also predict long-term, chronic effects?
4. Because the 2007 BLM-based criteria were developed solely from acute lethality data, no consideration of sublethal effects related to chemosensation and behavior such as impaired olfaction, predator avoidance, and prey capture were considered. These types of behaviors are considered fundamental for salmonids and other fish to complete their life cycles in the wild. Does the BLM reasonably predict and prevent against impairment of these types?
5. Laboratory experiments with single-species have an inherent artificiality to them. Field tests or tests in experimental ecosystems can be very different from those in laboratory experiments. Do the BLM-based criteria appear protective in more natural field settings or with experimental ecosystems?

This review addresses these questions through analyses of existing data sets using the BLM. The results of our analysis are mostly favorable toward the performance of the BLMs and also mostly favorable toward the protectiveness of the 2007 criteria values resulting from the BLM outputs. Our review suggested opportunities for refining the BLM. For example, the model appears to be overly sensitive to dissolved organic carbon (DOC) and under sensitive to calcium. That is, in the data sets reviewed, increasing DOC in water predicted a greater protection than appeared to be the case, and increasing calcium predicted a lower protection than appeared to be case. Regardless, in most cases, the 2007 criteria appeared protective from the adverse effects described in the studies. That is, for the water quality characteristics of a particular test, the copper criteria values produced from the BLM were mostly lower than the corresponding, measured adverse effect values. Although the data were thin, some of the field studies indicated risks that adverse shifts in invertebrate communities could occur at copper concentrations lower than those estimated for the 2007 criteria for the situations. However, in the field studies from which adverse effects were inferred, the 2007 copper criteria values were as low as or lower than corresponding hardness-based criteria equations. Appropriately designed and well executed field studies of the effects copper in the context of BLM predictions would be particularly valuable. Still, for the present, the 2007 BLM-based copper criteria appear sufficiently protective for listed salmonids and their ecosystems.

Reviews of seasonal time series data from a variety of streams that were considered representative of conditions within the action area indicate that "critical conditions," i.e., conditions when the BLM predicts that organisms would be most vulnerable to a given concentration of copper, are highly predictable. If the site-specific water chemistry information needed to directly calculate the BLM-based criteria were unavailable, table values are suggested for conservative but realistic critical conditions for waters across the range of anadromous salmon in Idaho that could be used to ensure protective conditions for listed salmon and steelhead.

## Introduction

The purpose of this review is to evaluate whether EPA's (2007a) aquatic life criteria for copper would be a protective alternative to apply in lieu of EPA's (1985) hardness-dependant aquatic life criteria for copper, which are the criteria adopted by Idaho and under review in this consultation. Whereas the 1985 copper criteria, along with most other metals criteria developed by EPA prior to 2007, were based upon statistical regressions between water hardness and toxicity, the 2007 copper criteria are based on a fundamentally different approach. The 2007 copper criteria are derived from the biotic ligand model (BLM) which predicts copper toxicity based on copper's expected bioavailability to aquatic organisms, as estimated using a geochemical model (HydroQual 2007). The BLM concept can be generalized to a variety of metals, and a variety of effects measurements such as short-term acute exposures that kill organisms outright, long-term chronic exposures that may not, predicting death or sublethal effects such as sensory impairment or reduced growth. However, for brevity, henceforth "BLM" refers to the version of the BLM for predicting acute toxicity of copper that was incorporated into EPA's 2007 copper criterion.

The EPA's Biological Assessment of for Idaho's toxics criteria proposed the use of the BLM as a "*strategy for reduction in uncertainty of water quality criteria for the protection of threatened and endangered species*." Further, EPA Region 10 committed to "*review the schedule and plan for updating the aquatic life criterion for copper*" and that "*the Services and EPA Region 10 will determine if the plan for updating the criteria will provide protection for salmonids.*" (EPA 2000, p. 24). Consistent with their commitments in the BA, EPA developed and published an updated, BLM-based copper criterion (EPA 2007a).

In 2005, the Idaho Department of Environmental Quality (IDEQ) updated all of its aquatic life criteria for metals, except for copper. The metals criteria in use in Idaho as of 2005 had been developed in the 1980s and were initially promulgated for use in Idaho through EPA's (1992) National Toxics Rule (NTR). EPA completed a series of updates to its metals criteria in 1995; these updates were subsequently published in a 2002 compendium of recommended water quality criteria, which in turn provided the technical basis for most of IDEQ's updates in 2005 (EPA 1996, 2002). IDEQ in 2005 adopted updates to all of their metals criteria except for copper. The explanation given in public meetings held by IDEQ was that although EPA's 2002 copper criteria were more protective than the NTR versions, EPA had also published the 2003 draft BLM update to the copper criteria, and that IDEQ would rather wait until the pending BLM-based criteria was published in final form, rather than revising their copper criteria twice. In February 2007, EPA published their final revised aquatic life criteria for copper (EPA 2007a). Nevertheless, there has been no indication that IDEQ is considering updating their 1992 NTR version of the copper aquatic life criteria.

The EPA's training materials for implementing the copper BLM suggest an incremental implementation as the most feasible and efficient means of implementing the updated criteria. EPA (2010) suggested that this incremental approach "s*hould result in more appropriate criteria more quickly for waters where the hardness-based copper criteria may be potentially <u>overprotective</u>, such as waters with high DOC, or potentially <u>under-protective,</u> such as waters with low pH.*" (emphasis added). Despite the apparent even-handed treatment of risks of either over- or under-protection of the hardness-based criteria in the previous sentence, all the examples given in EPA (2010) of site-specific application were for effluent influenced waters that were expected to provide considerably higher criteria. Further, the quoted sentence could be misleading, since waters with near neutral pH of 7.5, the hardness-based criteria may be underprotective by more than 6X in sites with moderately-hard waters and low DOC (Table 1).

This review focuses on whether the updated criteria as published will likely provide adequate protection for juvenile salmonids, their invertebrate prey, and other aquatic life. The EPA's (2007a) aquatic life criteria for copper represented a fundamental and ambitious change from earlier statistical regression-models in part because the BLM seeks to actually simulate some of the mechanisms of toxicity. Previous criteria just reflected overall statistical regression models of toxicity using water hardness, which is one of many factors that influence toxicity. In effect, the BLM is expected to be applicable and flexible enough across a variety of water quality conditions that it would produce a site-specific criterion for any specific location. While the 2007 BLM-based copper criteria are the most advanced and complex water quality criteria developed by EPA to date (Di Toro *et al*. 2001; Niyogi and Wood 2004), there are fundamental untested assumptions and unanswered questions relating to their protectiveness for aquatic ecosystems, especially those ecosystems inhabited by threatened or endangered species. These include:

1. Low hardness or soft water streams are common in Idaho. An earlier version of the acute copper BLM severely under predicted toxicity in very soft waters (Sciera *et al*. 2004; Van Genderen *et al*. 2005). Does the 2007 version predict toxicity accurately in soft waters? Regardless, would criteria values be lower than observed toxicity concentrations?

2. Some studies have suggested prediction bias in waters related to dissolved organic carbon (DOC). In tests with *Daphnia magna* and rainbow trout, the default BLM predictions tended to over predict the mitigating effect of elevated DOC on copper toxicity, and better model predictions were obtained by reducing the "metals reactive" portion of DOC by half in the model inputs (e.g., De Schamphelaere *et al*. 2004; Welsh *et al*. 2008). Is this borne out by other studies? If so, is this bias of a magnitude to undermine the protectiveness of the 2007 criteria?

3. The published validation of the reliability of the acute copper BLM for accurately predicting effects from short-term copper exposures was based only on three species, the fathead minnow, *Pimephales promelas*, and the cladocerans *Daphnia magna* and *Daphnia pulex* (Santore *et al*. 2001). Does the model produce reasonably accurate toxicity predictions for salmonids? What about other species that might be representative of prey or other co-occurring taxa?

4. The BLM was developed using short-term data, but has also been used to extrapolate against long-term effects (or the lack thereof) using acute to chronic ratios (ACRs). This approach has been criticized for its implicit assumption that acute and chronic effects are the results of similar internal mechanisms and as violating the mechanistic foundations of the BLM (Niyogi and Wood 2004). For example, the protection of factors such as DOC or calcium from copper accumulation and toxicity might be leaky. For instance, organic carbon as humic acid delayed the loss of sodium in longer-exposures of rainbow trout to copper, but did not ultimately prevent sodium losses (McGeer *et al*. 2002), also adding calcium to soft water protected against the

acute respiratory and osmoregulatory effects of exposure to a combined, relatively high Cd and Cu concentration on trout, but did not protect against the longer term ionoregulatory effects of the Cd and Cu mixture and the longer term accumulation of Cd and Cu by the fish (Richards and Playle 1999). Does the 2007 acute copper BLM also predict chronic toxicity? Does the ACR extrapolation result in criteria that are protective for fish or other aquatic organisms?

5. The sense of smell in fish is tied to critical behaviors including predator evasion, finding mates, and navigation but particularly so with migratory salmonids. Copper interferes with olfactory function in fish, and the olfactory bulb in a fish snout has very different structure and function than do the gills, for which the BLM was developed (Hansen *et al*. 1999a; Hansen *et al*. 1999b; McIntyre *et al*. 2008b). Do the 2007 BLM based criteria sufficiently protect against olfactory impairment?

6. As with almost all EPA's criteria, the 2007 BLM-based copper criteria predicts the absence or presence of adverse effects in the field from extrapolations of mostly short-term laboratory toxicity tests with "standard" laboratory test species that may not represent any real ecosystem. The 2007 criteria represent a fundamental change from previous criteria, and under some conditions can produce criteria values that can allow considerably higher copper concentrations than previous versions, and some field validation of the criteria protectiveness seems prudent. Are there field studies that indicate whether the BLM-based criteria are likely protective?

**Development of the BLM**

The BLM relies on the concept that metals in water are not toxic to fish and other aquatic organism per se, but rather, only when metals accumulate to critical concentrations in tissues does toxicity result. Thus it is not necessary to specify some metal species such as free $Cu^{2+}$ in water as being bioavailable and other metal species such as CuOH as being less bioavailable. Rather, the presence of the gill causes the chemical equilibria in water to change. Thus it is the degree to which the gill complexation sites are occupied by metal that determines whether toxicity will occur. A fundamental concept and assumption of the BLM is that the fraction of receptor binding sites occupied by a toxic metal would be the same for a given biological response, independent of water chemistry.

With fish, the gill is considered to be the target organism tissue used to predicts toxicity as function of three "C's": 1) complexation of copper with dissolved organic carbon (DOC) and carbonate in the water, 2) concentration of copper forms that can be toxic, which are assumed to be ionic free copper and copper hydroxides; and 3) competition between copper and other ions in water such as calcium, hydrogen, magnesium, and sodium for essential calcium and sodium channels on the surface of fish gills, or in the case of invertebrates that may not have distinct gill surfaces, on the surface of the "biotic ligand."

The development of the BLM can be traced back to the demonstration that the concentrations of cadmium accumulated on the gills of fish were a reliable predictor of cadmium caused deaths (Mount and Stephan 1967), that concentrations of metals accumulated on gills of fish can be predicted as a function of inorganic water chemistry (Pagenkopf 1983), and that organic carbon in water is an important modifier of metal accumulation (Playle *et al.* 1993b). These concepts were further refined and validated to develop what has become known as biotic ligand models for copper that could be manipulated to make predictions for any taxa (Di Toro *et al.* 2001; Santore *et al.* 2001). The EPA then extended the BLM concept from predicting toxicity for single species to predicting non-toxicity to 95% of the taxa represented in a species sensitivity distribution of available data (EPA 2003b, 2007a). A more detailed history of the BLM is given by Paquin *et al.* (2002) however Mount and Stephan's (1967) original insights do not appear to have previously been credited. More recently, an important practical aspect that has greatly popularized the BLM in recent years was the development of functional personal computer software that has made the ability to make BLM predictions for several taxa accessible to non-specialists.[9]

The data requirements to calculate copper criteria using the BLM are greater than that for the hardness based criteria (i.e., calcium and magnesium or direct titration). The BLM requires data on temperature, <u>pH</u>, <u>DOC</u>, <u>Ca</u>, Mg, <u>Na</u>, K, sulfate, chloride, and <u>alkalinity</u> and the underlined values appear more important, especially DOC and pH.

While the intended level of protection for 95% of the species-sensitivity distribution is unchanged when EPA updates a criterion, as a practical matter, the higher or lower criteria concentrations allowed for the same characteristics of a water body make different criteria more or less protective for species and more or less stringent for dischargers. Comparing the 1992 copper criterion under consultation with the updated 2002 or 2007 criteria values show that there is little difference between the 1992 and 2002 versions. In contrast, the 2007 BLM-based chronic criteria values are strikingly different from the Idaho values under consultation (Table 1). For the moderately hard "BLM-standard" water conditions used in the 2007 criteria derivation that were used to make data more comparable, the BLM based criteria are over 6X lower than the Idaho/NTR criteria . However, the BLM-based criteria are strongly influenced by the concentration of dissolved organic carbon (DOC) in the water, and when the DOC is increased to 8 mg/L but other water characteristics are kept the same, the BLM-based criteria are twice as high as the Idaho/NTR criteria. Therefore, a key question for reviewing the accuracy and protectiveness of the BLM-toxicity predictions is whether DOC is likely to control toxicity to the extent predicted by the BLM.

Table 1. Comparison of chronic copper criteria (CCC) from the 1992 hardness-based NTR, hardness-based 2002 update, and 2007 BLM-based updated criteria.

| Water-chemistry condition | NTR CCC µg/L | 2002 CCC µg/L | 2007 CCC µg/L |
|---|---|---|---|
| Hardness 85 mg/L, pH 7.5, DOC 0.5 mg/L (ASTM/EPA moderately-hard water) | 9.9 | 7.8 | 1.5 |
| Same except DOC of 1 mg/L | 9.9 | 7.8 | 2.8 |
| Same except DOC of 2 mg/L | 9.9 | 7.8 | 5.5 |
| Same except DOC of 4 mg/L | 9.9 | 7.8 | 11. |
| Same except DOC of 8 mg/L | 9.9 | 7.8 | 22. |
| Same except DOC of 12 mg/L | 9.9 | 7.8 | 33 |

The DOC range of 0.5 to 12 mg/L includes the vast majority of DOC measurements in Pacific Northwestern streams although higher values likely occur briefly during runoff or in waters with extensive riparian or littoral marshes).

[9] http://www.hydroqual.com/wr_blm.html

**Analyses of the accuracy of the copper-BLM and criteria for predicting toxic or non-toxic conditions**

To address the five questions from the introduction, copper effects data from many relevant studies were tracked down and reviewed to see they had sufficient data to analyses in the BLM-context. If so, the water characteristics corresponding with the empirical effects data where run through the HydroQual, Inc. The BLM software using the 2007 model parameter values from Table 2. To evaluate the ability of the model to predict the observed effects across different species, types of effects, and diverse waters, a critical or lethal accumulation value (CA or LA) was estimated at the biotic ligand associated with a given effect as the sum of predicted biotic ligand concentrations of $Cu^{+2}$ and $CuOH^{+}$ for each test value. For example, a copper concentration causing 50% mortality in a test, that is the LC50, would be used with the model to predict a LA50 for each test. When multiple CAs for the same endpoint and species were available, such as with rainbow trout for LC50 concentrations or concentrations causing a 10% growth reduction (EC10) from multiple tests for example, a geometric mean CA was calculated. This mean CA50 was in turn used to predict how well the model could predict LC50s or EC10s for that species across diverse water conditions.

The parameters used in the 2007 criteria were noted to be different differed from those that were described in previous technical evaluations supporting the technical basis of the BLM-based copper criteria. The differences are that binding affinity factors between the biotic ligand magnesium (Mg) and copper hydroxide ($CuOH^{+}$) are included in the computer parameter files but not in the model documentation (Santore *et al.* 2001; EPA 2003b). The bioavailability and toxicity of $CuOH^{+}$ were described in EPA (2003) and its omission from the summary parameter Table 2 was probably simply an oversight. In contrast, Mg was specifically excluded in earlier versions of the gill binding model, including the public review draft of the criteria update, because it did not mitigate toxicity as much as Ca (Santore *et al.* 2001; EPA 2003a). While not in the criteria documentation, discussions with the model and criteria developers indicated that the about face on including Mg in the criteria version of the model was not documented because it occurred shortly before publication. This was because despite earlier evidence of the lack of protectiveness of Mg for fish in "normal" waters (Erickson *et al*. 1996; Welsh *et al*. 2000; Santore *et al.* 2001; Naddy *et al.* 2002), at least in some extremely hardwater effluents in the arid west, Mg did have some protective effects against copper toxicity (Van Genderen *et al*. 2007). Because of the late and informal addition of Mg to the model, the protectiveness of the BLM-criteria in natural waters with differing Mg content is specifically considered here.

In addition to the inorganic species listed in Table 2, a fundamental part of the BLM is its procedure for estimating the amount of copper in the water column (i.e. before the copper ever gets to the gill) that is bound to dissolved organic carbon (DOC). This is implemented in the BLM through an implementation of the Windermere Humic Aqueous Model (WHAM V) originally developed by Tipping and Hurley (1992). As will be shown, the BLM is extremely sensitive to DOC, making the accurate measurement of DOC in the waters of interest highly important to the performance of the BLM.

Table 2. Parameters of the BLM versions used in the criteria technical support document that was prepared for peer review (EPA 2003b) and revised parameters used in the 2007 criteria (EPA 2007a)

| **log K conditional equilibrium stability constants of binding affinity of the biotic ligand (BL) with inorganic species** | **BLM Technical support document** (EPA 2003b) | **2007 BLM versions 2.2.1 (EPA 2007a)and 2.2.3**[10] |
|---|---|---|
| log $K_{BL-Cu}^{2+}$ | 7.4 | 7.4 |
| log $K_{BL-CuOH}^{+}$ | Not included | 6.22 |
| log $K_{BL-Ca}^{2+}$ | 3.6 | 3.6 |
| log $K_{BL-Mg}^{2+}$ | not used | 3.6 |
| log $K_{BL-Na}^{+}$ | 3.0 | 3.0 |
| log $K_{BL-H}^{+}$ | 5.4 | 5.4 |
| Fathead minnow critical gill lethal accumulation value (LA50, nmol/gill ww) predicted to cause 50% mortality, on the average | 6.2 | 2.97 |
| *Daphnia magna* " " " | Not included | 0.0483 |
| *Ceriodaphnia dubia* " " " | Not included | 0.052 |
| Rainbow trout " " | Not included | 0.4424 |
| Final acute value (FAV) " " " | Not included | 0.03395 |

Acute toxicity predictions for fish

The BLM is expected to be at its strongest for predicting the acute toxicity of copper to fish because the BLM was initially developed with acute toxicity data for fish, and the initial published calibration of and validation of model was with fish (Santore *et al.* 2001). In particular, Santore *et al.* (2001) used an extensive data set by Erickson *et al.* (1996) in which fathead minnows were tested with copper while manipulating natural water from Lake Superior with varying factors that could potentially control toxicity, such as DOC (humic acid), pH, Ca, Mg, Na, temperature, and suspended solids. While a very good fit was obtained for most data, subsequent analyses with fathead minnows suggested that the BLM for fathead minnows was biased high and underpredicted toxicity (that is, over predicted LC50s). Underpredictions were more pronounced in very soft water (Van Genderen *et al*. 2005). However, these analyses were based upon an earlier version of the BLM fathead minnow model than that derived by EPA (2007). The NMFS compiled and analyzed these and many other tests with fathead minnows using the 2007 model to predict toxicity. While the fathead minnow is not of direct interest in the present consultation, the fathead minnow is emphasized because it is a model organism that is extensively used by aquatic toxicologists worldwide to evaluate the relative potency of compounds and factors affecting toxicity (Ankley and Villeneuve 2006). Patterns developed with fathead minnows are thus presumed relevant to other fish species, even though in an absolute sense, fathead minnows are probably less sensitive to copper than are the salmonids that are the focus of the present analysis and consultation (EPA 2007a).

Before one can validate or refute BLM predictions by comparing them to empirical results from toxicity tests, one must evaluate the inherent precision and repeatability of empirical toxicity tests. Santore *et al.* (2001) found that after excluding outlying data, their model was able to predict the toxicity of copper to fatheads across a wide range of water chemistries by about a factor of 2. Santore *et al*. (2001) considered agreement within a factor of 2 for predicted and

[10] http://www.hydroqual.com/wr_blm.html last accessed 12 August 2010

measured LC50s to be "quite good" noting that replicate toxicity tests by Erickson *et al*. (1996) with copper and fatheads in un-manipulated Lake Superior water sometimes varied by up to a factor of 6. Because the inherent limits on the accuracy of the BLM model are a fundamental benchmark for evaluating the model and criteria, these comparisons of replicate variability were reproduced from Erickson's original dataset (Figure 1).

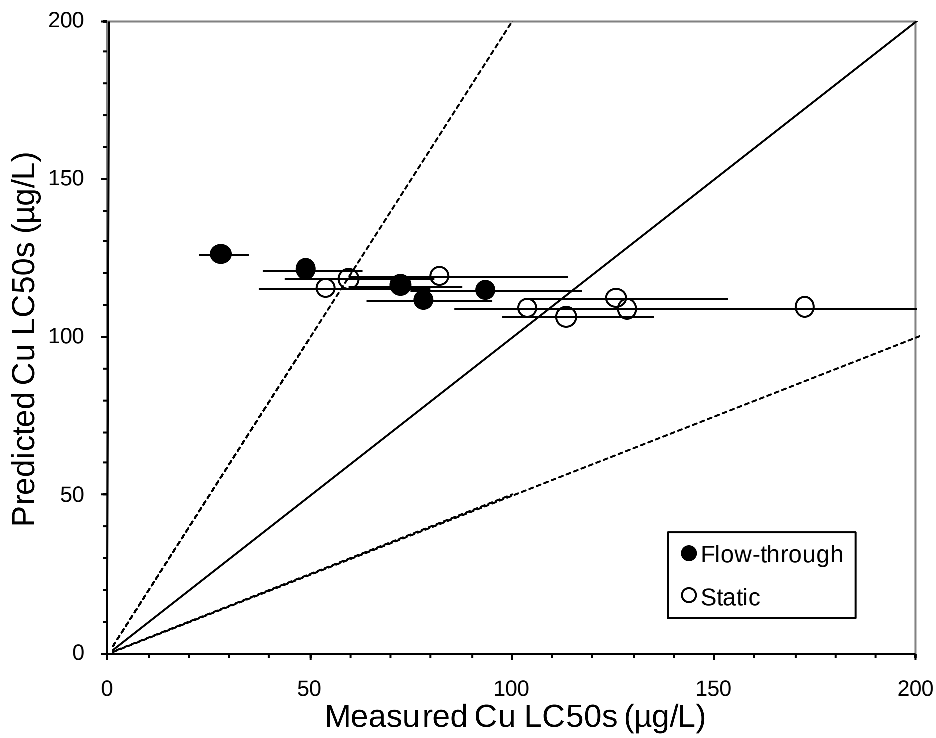



**Figure 1. Comparison of measured and BLM predicted copper LC50s for fathead minnows and copper using flow-through or static exposures with unmanipulated Lake Superior dilution water. Unmanipulated Lake Superior water was used as a reference condition as part of a larger study of the effects of water chemistry on the acute toxicity of copper (124 tests total, Erickson *et al.* 1987, 1996). Lake Superior water is commonly used as the dilution water for toxicity testing at the EPA's Duluth laboratory because of its stable characteristics and low background contaminant levels. Closed symbols denote flow-through test results and open symbols denote renewal test results, error bars show 95% confidence intervals on observed LC50s. The solid line indicates the 1:1 line of perfect agreement, dashed lines indicate 1:2 and 2:1 lines, i.e., bounds for predicted values being within 2X of observed values. The same convention is used in following figures.**

Lake Superior water may be nearly ideal as a standard reference water for comparing the performance of toxicity tests. The water chemistry near the EPA's Duluth, Minnesota Environmental Research Laboratory appears to be very stable based upon different analyses over time and the water has low background contaminant levels (McKim and Benoit 1971; Maier and Swain 1978; Erickson *et al.* 1996; Cotner *et al.* 2004). Average conditions for the tests in Figure 1 were DOC 1.35 mg/L, alkalinity 42 mg/L, hardness 45 mg/L, pH 7.9, although the differences in the predicted fathead minnow tests (vertical scale) result from variance from these average values. All tests were initiated with <24 hour old fish, so the confounding issue of size or age of fish should be minimized. The tests are grouped by whether they were conducted as "flow-through" tests where the test solutions are constantly being replaced with an average water residency in the test chambers of about 45 minutes, and as "static" tests where the fish were placed in test solutions at the start of the test and the solution was not refreshed during the 96hr test. Figure 1 clearly shows that copper was more toxic in the flow through tests than in the static tests. Among the flow-through tests, copper LC50s ranged over a factor of about 3.5X, from about 25 to 90 µg/L and among the static tests, copper LC50s ranged over a factor of about 2.5X, from about 50 to 125 µg/L, not including an outlying LC50 at about 170 µg/L (Figure 2).

This analysis illustrates why it is not reasonable to expect the BLM to predict toxicity much more precisely than by about a factor of ± 2, since replicate tests often vary by more than that. This supports the convention started by Santore *et al*. (2001) to use the "within a factor of 2" prediction factor as one guideline for evaluating model performance. In our review, when the predicted/empirical comparisons showed a pattern in their residual errors, we investigated the bias to see if it indicated systematic error, especially underprotection by the model and criteria.

NMFS compiled a large number of toxicity tests with fathead minnows, independent of the Erickson *et al*. (1996) data used to calibrate the model. Test water chemistries were used to predict the toxicity of fathead minnows through the BLM, and we compared these predictions to the empirical LC50s for each test. The 2007 version of the Hydroqual BLM (v. 2.2.3) includes a fathead minnow prediction using a critical accumulation of copper on the gill surface of 5.48 nmol/g gill wet wt (ww) (i.e., the "LA50"). However, the data sources of this fathead minnow prediction file were not described and using the 2007 Hydroqual BLM for fathead minnows using the standard water conditions used by EPA (2007) to normalize data with the BLM produces a fathead minnow LC50 that is 2X higher than the species mean acute value (SMAV) for fathead minnows derived by EPA (2007), 116 µg/L vs. 63 µg/L for the 2007 Hydroqual version 2.2.3 of the BLM and EPA SMAV respectively. Thus to evaluate the performance of the BLM as used in the 2007 criteria, it was necessary to reconstruct the LA50 for fathead minnow and other species in the same manner as done in EPA (2007). A "critical" species mean LA50 for fathead minnow value was estimated as the geometric mean of 141 "LA50" values which in turn had been estimated from LC50 values listed in Table E of EPA (2007). Table E has 150 tests with fathead minnows, but Table 1 says that "Underlined LC50s or EC50s not used to derive SMAV because considered extreme value." No tests in Table 1 were underlined, although in the 2003 draft report, 9 tests with fathead minnows were marked as excluded. Excluding those same tests reproduces the SMAV given in the document, whereas if all tests were used, a critical value of 2.37 would be obtained, which produces a lower SMAV than given in the 2007 document. Thus Table 1 in the March 2, 2007 version is apparently in error, and instead Table 1 of the draft 2003 version is actually the reference for tests used or excluded from the 2007 "final" document. The reconstructed LA50 for fathead minnows produced a SMAV of 69.34 µg/L which is nearly identical to the fathead SMAV of 63.69 µg/L given in Table 1 of EPA (2007). Critical LA50 estimates for *Ceriodaphnia dubia*, *Daphnia magna*, and rainbow trout were similarly reconstructed and also agreed well with their respective 2007 SMAVs. These reproductions of the EPA (2007) values confirmed both the EPA values as well as giving reassurance that the present analyses are indeed comparable.

For species and endpoints that are not based on EPA (2007) values such as critical LA50 values for acute LC50s other species or untested endpoints such as chronic growth reductions or olfactory impairment, we estimated "critical" values in the same manner as used in EPA (2007).

Effects values were determined (i.e., EC50s, EC20s, EC10s), chemistry compiled, and the BLM was run in geochemical speciation mode to predict the copper accumulation on the ligand as $Cu^{2+}$ and $CuOH^{+}$ , the sum of which was considered the "critical accumulation" for the test. Where multiple values were available, the geometric mean of test critical accumulation values for a particular endpoint and species was used as the species mean critical value for the endpoint. These critical values were in turn used to predict EC values for the same tests. The scatter or bias of these predictions was used to evaluate the performance of the BLM.

The BLM predictions are compared with empirical or so-called "observed" effects data by plotting scatterplots of the observed and predicted values along with the 1 to 1 line of perfect agreement, bracketed by lines illustrating the factor of 2 test of good agreement between the modeled and observed values. Also, linear regressions are shown, where a slope of 1.0 indicates perfect agreement, and $R^2$ coefficient of regression values indicate the proportion of variability explained by the regression. Optimal performance would be reflected by a tight scatter of points close to the 1:1 line of perfect agreement; poor model performance would be reflected by a random "shotgun" pattern or distinctly biased patterns that systematically over- or under-predicted toxicity.

The first of these examples is the original calibration data set with fathead minnows that was presented in EPA (2003) and in Santore *et al.* (2001). This modeling used a comprehensive set of toxicity tests with copper and fathead minnows in a variety of artificial and amended natural waters in which the effects of changes in hardness, calcium, magnesium, sodium, alkalinity, humic acid, temperature and other factors were tested (Erickson *et al.* 1987; 1996). (Erickson *et al*. 1987 and 1996 describe the same testing although some data details and tests were not included in the shorter 1996 published version.). As a benchmark, copper toxicity was also tested by Erickson *et al*. (1987; 1996) in un-amended Lake Superior water which varies little in consistency (Figure 1). The model performance for the Erickson dataset was remarkably good, with very little bias in the predictions or scatter, considering that the LC50s ranged from about 10 to 1000 µg/L. Both the highest and lowest LC50s fell very close to the 1:1 line of perfect agreement (Figure 2, top). Because these data and modeling were seminal for the BLM development and the technical support of the subsequent criteria, the methods and sources described in EPA (2003) were repeated here to see if the results were reproducible. For most of the datapoints, the results from Santore *et al.* (2001) and EPA (2003) were successfully reproduced, particularly for the less-toxic samples with measured LC50's >≈ 100 µg/L. While the Santore/EPA results were also very good for the more toxic samples, the reconstructed results underpredicted toxicity. The results of the discrepancy are not easy to reconcile, but might be related to uncertainty about whether or not biotic ligand-bound $CuOH^{+}$ was considered toxic in the Santore *et al.* (2001)/EPA (2003) modeling (Table 2).

In Erickson *et al*.'s (1996) data, copper tended to be more toxic in tests that used the flow-through exposure methods rather than static exposures (Figure 2, bottom). In flow through tests, the test solution is intermittently metered into the vessels, replacing the complete water volume several times each day. In static tests, the fish are introduced into the test vessel and maintained in the same volume of water throughout the duration of the tests (e,g., 96-hours). In the Erickson *et al.* (1996) test, the replacement rate resulted in a test residence time of about 45 minutes, or about 32 volume replacements per day. The "renewal" method is a compromise between the flow-through and static methods. Renewal tests are the same as static, except that the majority of the test solution is siphoned off and replaced midway through the tests (ASTM 1997).

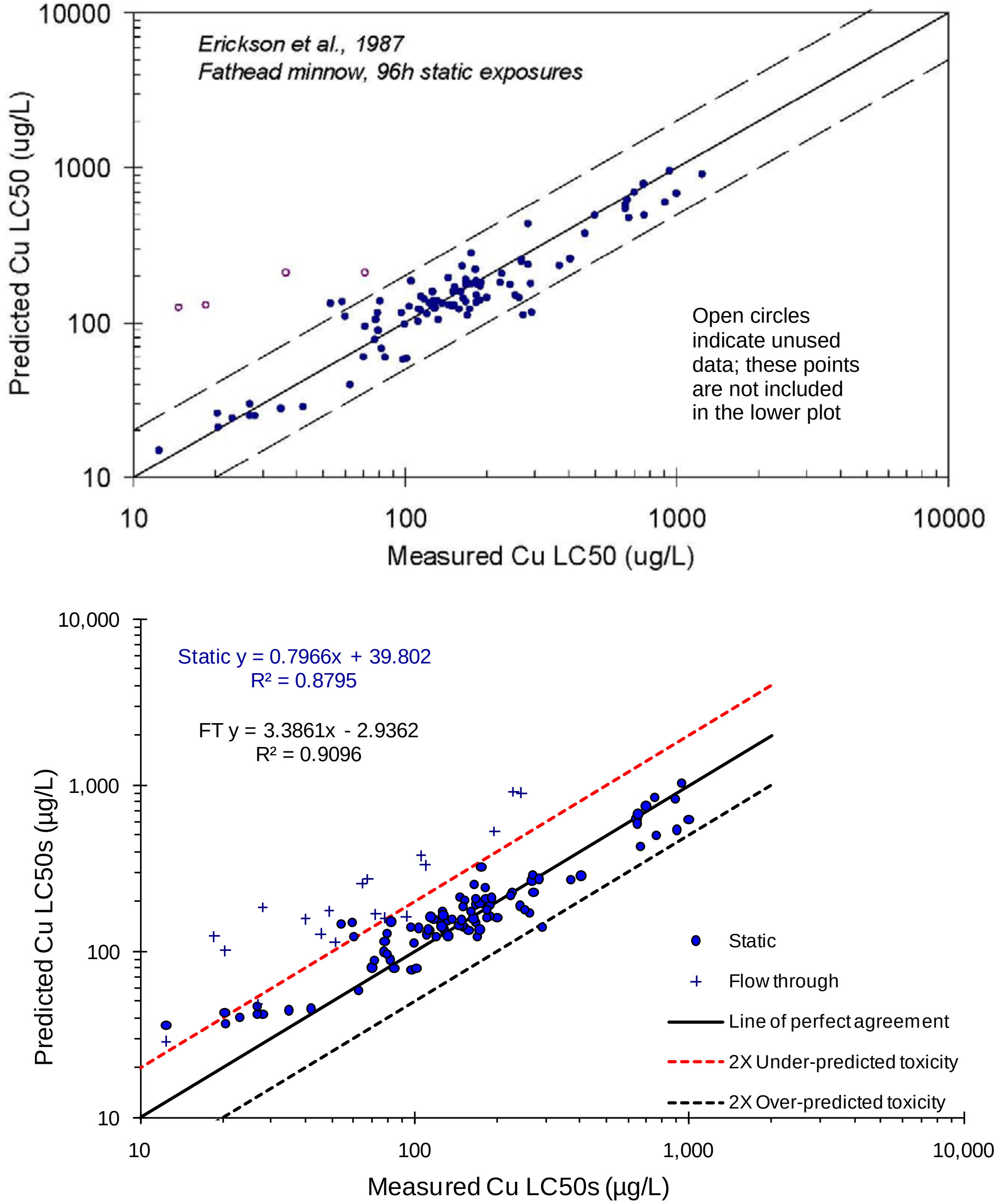



**Figure 2.** **Top - Biotic ligand model predicted versus observed LC50 values for fathead minnows in static toxicity exposures. Figure is from EPA (2003, their Figure 14) in which data from Erickson *et al.* (1987, 1996) were used with the BLM parameters and a fathead minnow LA50 of 6.32 listed in Table 1. The solid line is the 1:1 line of perfect agreement and the dotted lines show the 2:1 and 1:2 lines showing**

**values 2X more toxic than predicted or 2X less toxic than predicted. Bottom – Reconstruction of EPA's top figure using original data and BLM parameters from EPA (2003) as listed here in Table 2. The modeling very nearly reproduced most values as shown by the nearly identical patterns of points between the two plots. In EPA's 2003 modeling at top, the most toxic measured conditions at the bottom left of the plot with the lowest LC50s were predicted by the model quite accurately. However, in the reconstruction toxicity tended to be under-predicted. This discrepancy is unexplained.**

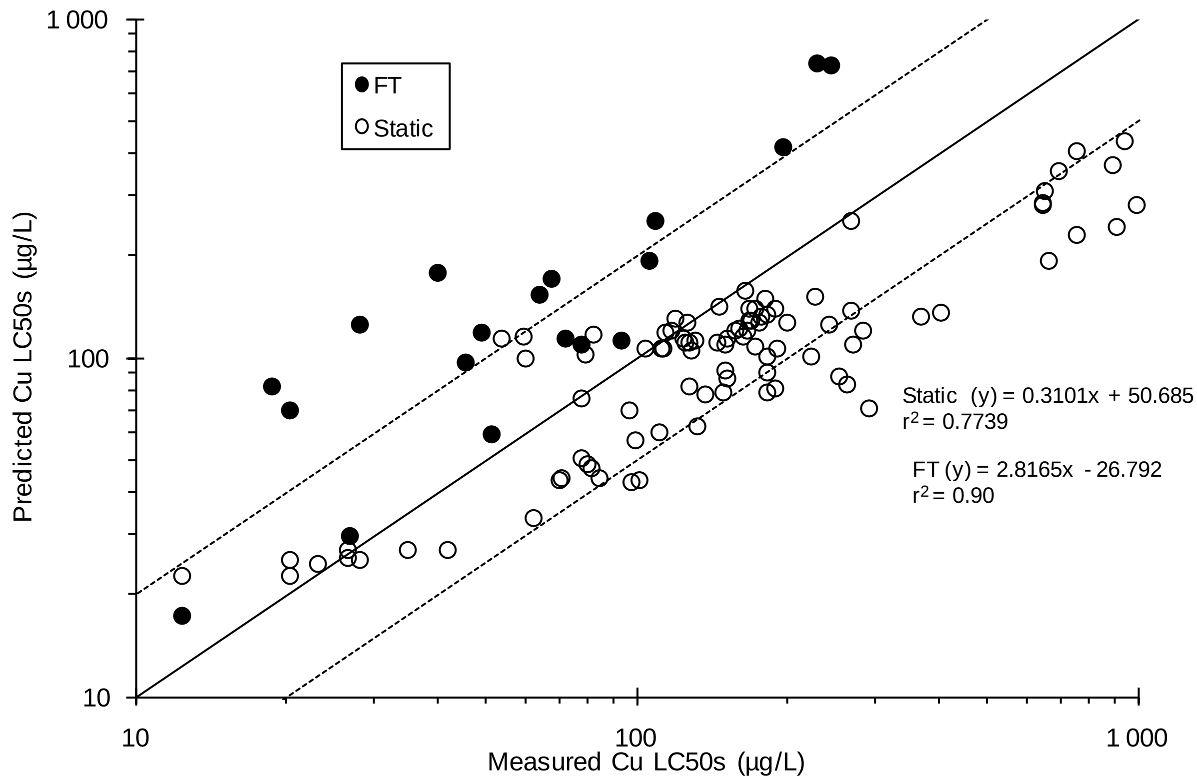



**Figure 3. Fathead minnow BLM predicted and measured Cu LC50s, from Erickson's 1996 tests, using EPA's 3-02-2007 LA50 and model parameters from Table 2. "FT" – flow through tests.**

Santore *et al*. (2001) interpreted the increased toxicity in the flow-through tests as an indication that the copper had not yet reached equilibrium with DOC and because the metal speciation and complexing equations in the BLM were based on the assumption of equilibrium, the static results were relied upon by EPA in EPA's (2003b) validation (Figure 2, top). In contrast, Welsh *et al.* (2008) also showed that DOC may build up in renewal tests and can explain lessened copper toxicity to rainbow trout. Their flow-through rates were lower than Erickson's (about 4 to 6 hours vs. 45 minutes per volume replacement), suggesting that copper and DOC had more of an opportunity to approach equilibrium than in the Erickson's tests with fathead minnows. The "non-equilibrium" and "increased DOC concentration" explanations for increased copper toxicity in flow-through tests relative to static or renewal tests are not mutually exclusive. Further, metals in streams may not be at chemical equilibria, which can influence toxicity of metals (Nimick *et al.* 2003; Meyer *et al.* 2007; Nimick *et al.* 2007). Thus, in this review, we examined results from both flow-through and static or renewal tests.

The different patterns of copper toxicity in flow-through or static tests are obvious in the plots of BLM predicted and measured values from Erickson's (1996) data. Initially, the model was calibrated using the static results only, following the belief that the flow-through results greatly exaggerated toxicity (Figure 2). However, EPA (2007a) compiled additional toxicity data on fathead minnow (and other species) toxicity that was analyzed in their (EPA 2003b) technical support document. When we used the 141 test values used by EPA (2007a) to establish the species mean acute value (SMAV) for fathead minnows to estimate a species mean LA50, we obtained considerably more sensitive estimates of copper toxicity to fathead minnows (Table 2). When we predicted the same Erickson *et al.* (1996) data using the BLM parameters used in EPA (2007a), a very different impression resulted. Instead of the static results looking "about right" and the flow-through results looking skewed, both the flow-through and static results roughly straddle the 1 to 1 line of perfect agreement (Figure 3). More interestingly, in the EPA 2003b version, the static data set had a measured to predicted regression slope that was at least 0.9 and a $R^2$ coefficient of variation that was probably at least 0.9 (described as "at least" because by eye the original 2003 plot had a better fit than the reconstructed plot in Figure 2). However, using the 2007 parameters, the Erickson static data take on a much shallower slope of 0.3 and a lower $R^2$ value (Figures 2 and 3).

Some previous efforts to validate the BLM-toxicity predictions noted that the model severely underpredicted toxicity in very soft water. Erickson *et al.'s* (1996) data covered a wide hardness range from ~19 to 250 mg/L, although most were conducted at hardnesses of ~40 mg/l and above. Curiously, in Figure 2 bottom, the cluster of solid points at the lower left corner with predictions that drift toward being less protective at lower LC50 values correspond with the lowest hardness levels tested. In tests in very-soft natural waters from the South Carolina plain, the BLM so underpredicted the toxicity of copper to fathead minnows that the LA50 that had been derived primarily from the Erickson data had to be empirically lowered by a factor of 36 to fit the model (Van Genderen *et al.* 2005). Similar results were attained by Sciera *et al*. (2004). These results lead to the question, 'does that the BLM may systematically under predict toxicity in softwater'? This is a significant concern for application in the Pacific Northwest or other areas where softwater is common. Thus NMFS compiled these additional datasets and other softwater toxicity data and compared them to the 2007 version of the BLM. We also analyzed two additional datasets from the Canadian Shield area of central Ontario (Welsh *et al.* 1993; Welsh *et al.* 1996). The Canadian Shield is characterized by thin soils over crystalline bedrock which leads to very low calcium contents in the waters and pH values less than 7 units. DOC may range from as low as 0.5 to over 20 mg/L. While few streams in headwaters regions of the Salmon or Clearwater Rivers in Idaho or most other mountainous regions of the Pacific Northwest have DOC values as high, Canadian Shield waters otherwise appear to have many aquatic chemistry characteristics as waters draining the granitic watersheds in the Idaho Batholith region of central Idaho or the Precambrian metamorphic rocks found further north in much of the Clearwater River watersheds. A fourth important dataset is one in which fathead minnows were tested with copper under uniform hardwater conditions, but with various concentrations of DOC from natural organic material that had been isolated from Nordic reservoirs (Ryan *et al.* 2004).

Comparison of the BLM-predicted and measured EC50s show a systematic bias where the BLM tends to under predict the toxicity of copper in the softwater settings (tests with higher toxicity/lower LC50s) that plot near the bottom left corner (Figure 4.) Across the different datasets, the under prediction bias is diminished as the tests waters become less toxic (higher LC50s) which corresponds with increasing hardness. The tests by Ryan *et al.* (2004) in hardwater with various DOC show no obvious bias. The model performance in softwater was at least an improvement over the magnitude of under prediction in the version used by Van Genderen *et al.* (2005).

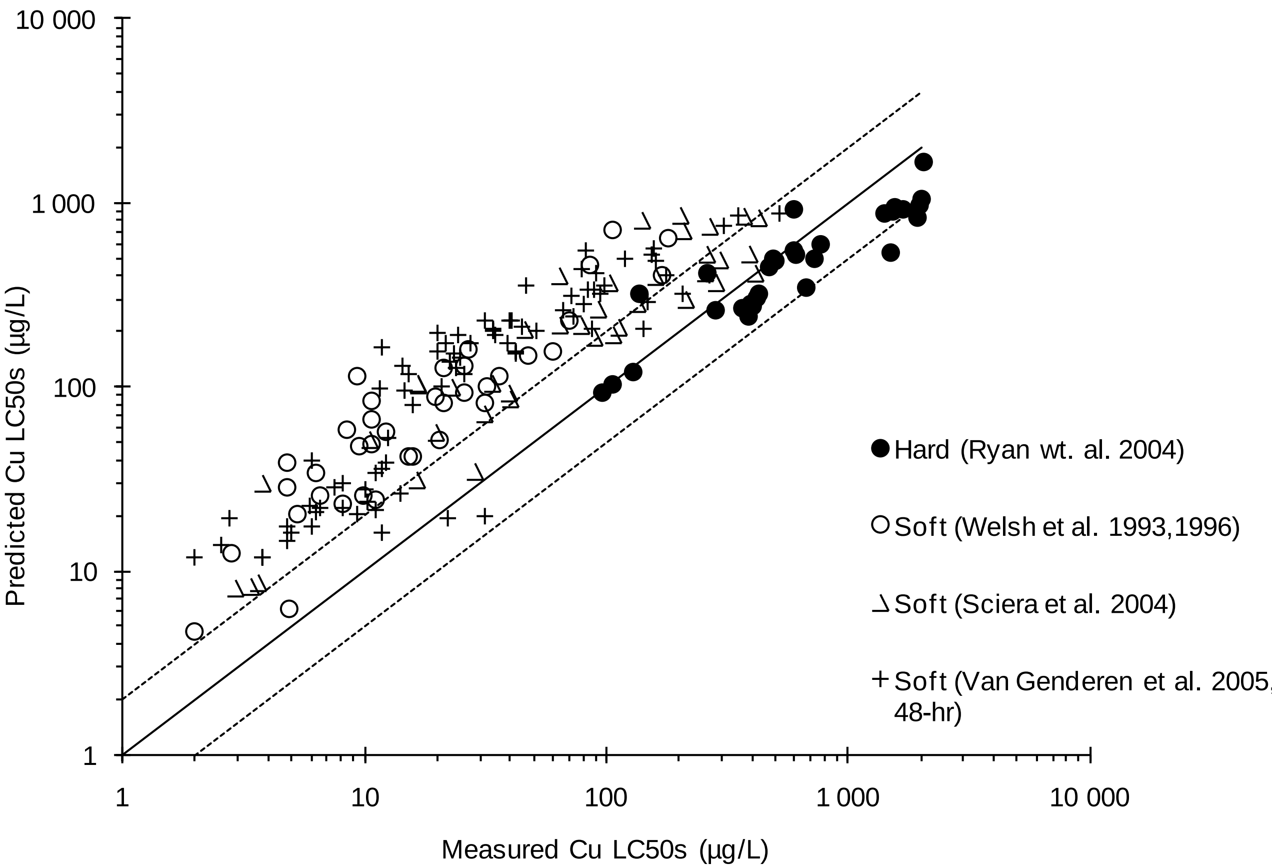



**Figure 4.** **Fathead minnows, BLM predicted and measured Cu LC50s labeled by hard or soft dilution waters, using BLM v 2.2.3 and EPA's 2007 LA50. The comparison shows the BLM generally underpredicted toxicity in the more toxic samples with low measured LC50s.**

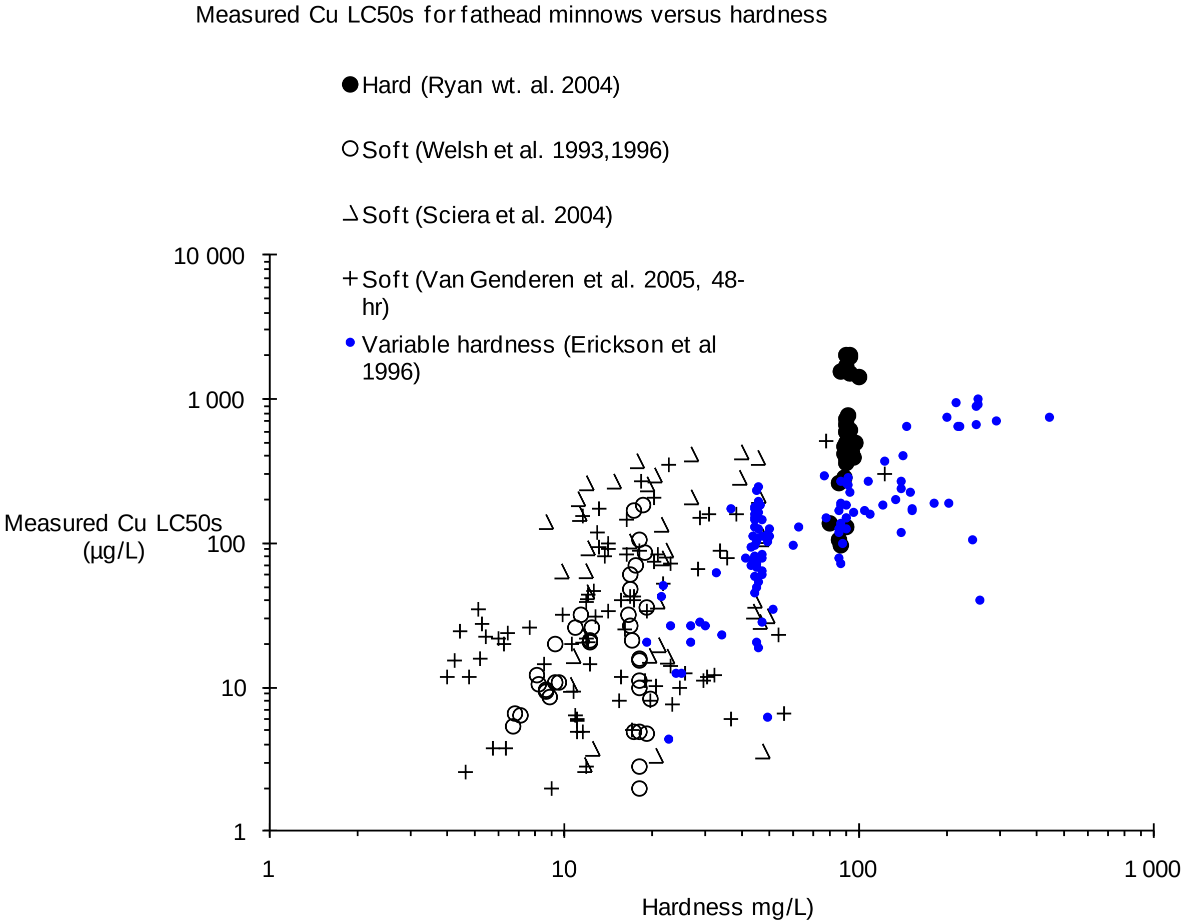



**Figure 5. Hardness as a predictor of copper toxicity to fathead minnows: at a hardness of 20 mg/L, fathead minnow LC50s could range from about 2 to 300 µg/L, and at a hardness of about 90 mg/l, LC50s could range from about 100 to over 2000 µg/L.**

Because there appeared to be a prediction bias in the BLM that was associated with hardness in these softwater datasets, NMFS compared the LC50s with hardness to see if hardness may be a better predictor of toxicity than the BLM (Figure 5). While there is clearly a pattern of increasing LC50s with increasing hardness (i.e., decreasing toxicity), the variability is so severe as to render a hardness-toxicity relationship dubious for water quality management. For instance, at a hardness of 20 mg/L, fathead minnow LC50s could be anywhere from 2 to over 300 µg/L (factor of 150), and at a uniform hardness of about 90 mg/L, LC50s range over a factor of about 20 (~100 to >2000 µg/L). In contrast, although the BLM-predictions were severely skewed, the predictions seldom varied by more than a factor of five and most of the data varied by much less. This suggests that with additional calibrations, it would be feasible to better tune the model performance in soft waters. In fact, encouraging results with this problem have been recently published (Ryan *et al.* 2009).

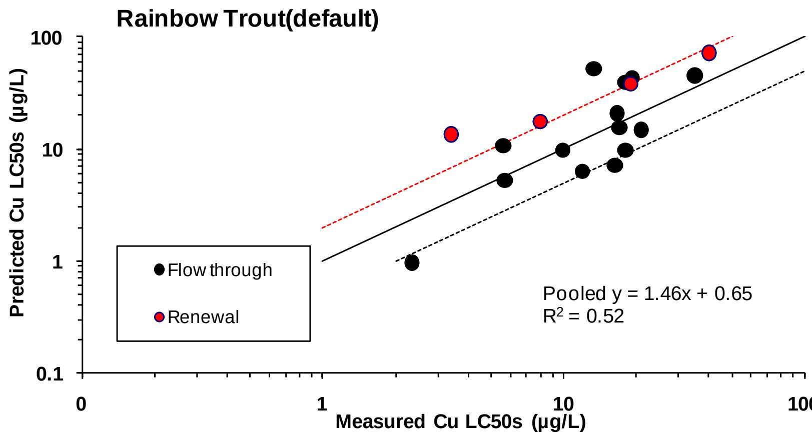
The accuracy of BLM predictions and the protectiveness of BLM-based criteria are related but not identical issues. A goal of criteria development is to be able to make useful predictions whether a specific addition of a toxic agent such as copper to a particular aquatic ecosystem will cause any unacceptable effect on that ecosystem (Stephan 1986). However, from the perspective of protecting listed species, where it is better to err on the side of the species, if exceeding a criterion fails to predict adverse effects, that is not a problem for the species. Rather, what is essential is that the criterion is protective of the listed species and their ecosystems. Thus the suboptimal performance of the BLM in predicting copper toxicity in softwaters, indicates that the BLM-criteria would provide less protection in these waters than intended. Yet it does not necessarily demonstrate that the criteria would be unprotective for the fish tested (fathead minnows). When we calculated the 2007 FAV for each individual test condition, we found that only 4 of 187 or 2% of the FAVs were greater than the empirical LC50s for the same waters. The reason that the FAV was sufficiently protective even though it was biased in softwater is likely because the fathead minnow is sufficiently less sensitive to copper than were the more sensitive Daphnid and mollusc species that defined the FAV.

Next, we consider the performance of the BLM with salmonids. While the foundational work to develop the copper BLM used experiments with rainbow trout (Playle *et al*. 1992; Playle *et al.* 1993b, a; MacRae *et al.* 1999), relatively little has been published since then regarding the performance of the copper BLM criteria with rainbow trout our other salmonids (but see Welsh *et al.* 2008). Despite this, we located several very relevant datasets that were well suited for evaluating the protectiveness of the copper BLM for salmonids.

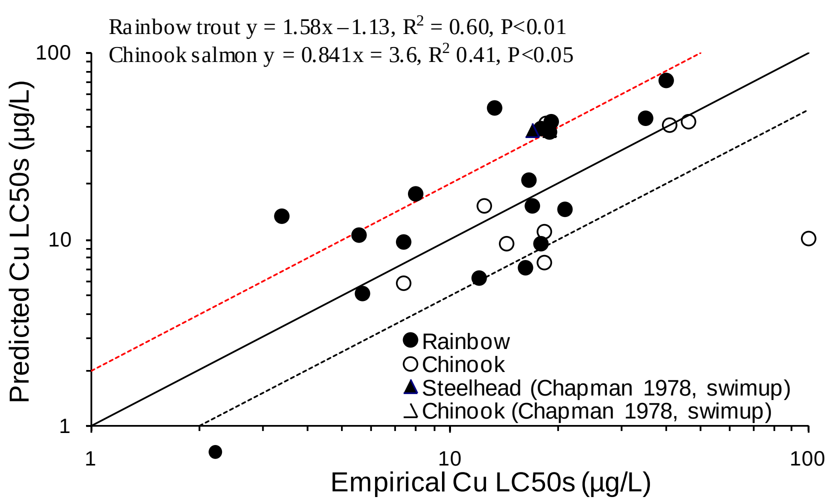
The first dataset with salmonids was from a comprehensive study that tested the comparative sensitivity of rainbow trout and Chinook salmon to copper in natural waters of the upper Sacramento River in northern California. Tests were also conducted in laboratory waters in which calcium, magnesium, and pH were manipulated (Stratus 1996, 1998). All necessary water chemistry parameters were measured and experimental controls were exceptionally tight and well described. With rainbow trout, tests were conducted under both flow-through and renewal designs, but Chinook salmon were only tested with a flow through design. The natural river waters used tended to have soft water, low DOC, and pH in the ranges that are typical of other salmon and steelhead waters in Idaho and the Pacific Northwest. Although the ranges of water chemistry data are fairly narrow, these data are otherwise nearly ideal for the evaluation of the BLM performance under environmentally realistic conditions. The study reports contain a wealth of data and are well supported by data quality control and quality assurance information, and some of the tests were available for incorporation into EPA's (2007) criteria dataset; however, these data have never been further published and have mostly been unavailable to the scientific community.

The results of our review of this dataset were reasonably favourable to the BLM's performance, with the regression slopes not greatly different from 1.0. When comparing rainbow trout predictions by whether they used a flow-through or renewal design, the plots do suggest that renewal tests tended to higher LC50s (lower toxicity) for given predicted values than did the flow through tests (Figure 6, top). However, the apparent disparities were not nearly as pronounced as those with fathead minnows discussed earlier (Figures 1 through 3). Thus, it seems that considering renewal or flow-through tests as being more or less appropriate for testing copper toxicity or to use with the BLM is probably not warranted. At least this appears the case for tests with salmonids that were conducted in aquaria with slower water replacement times (longer water residence times) than was used with the fathead minnow mini-diluter study design.

The Chinook salmon predicted and observed toxicity values fell among the rainbow trout values, indicating that at least for the tested stocks, the sensitivity of the two species to copper is very similar (Figure 6, middle). In fact, the predicted toxicity values were produced using LA50 estimates developed for rainbow trout without any obvious sensitivity bias between the species.

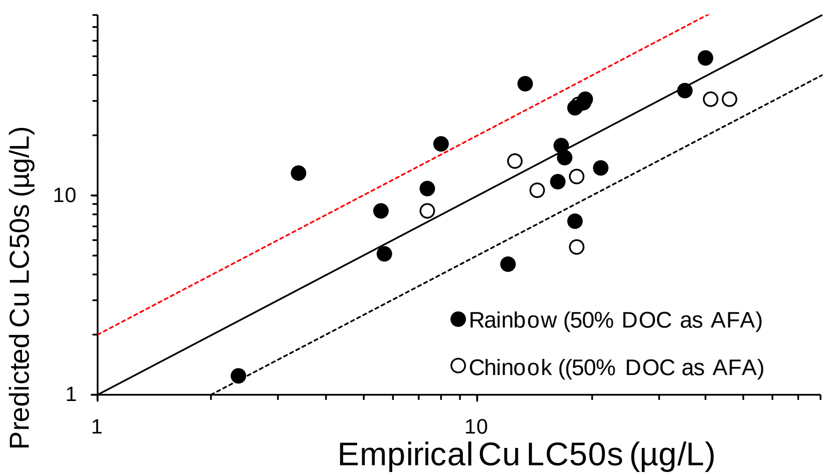



A.




B.

Rainbow trout y = 1.46x +0.65, $R^2$ = 0.52,
Chinook salmon y = 0.96x + 0.92, $R^2$ 0.61,




C.

Rainbow trout y = 0.94x +4.6, $R^2$ = 0.52,
Chinook salmon y = 0.56x + 0.92, $R^2$ 0.61,

**Figure 6. Rainbow trout and Chinook salmon: empirical and BLM predicted toxicity in 96-hour tests using natural Sacramento River water and lab waters, DOC <0.11 to 2.0 mg/L, pH manipulated from 6-**

**8, hardness 19-60 mg/L (Stratus 1996, 1998). A. Flow-through vs. renewal tests with rainbow trout; B. rainbow vs. Chinook; and C. rainbow vs. Chinook after reducing DOC availability by 50%.**

An issue that has been unresolved in the scientific literature on BLM development is whether some empirical adjustment to the copper reactivity of different DOC sources is beneficial. The argument is that if only 50-65% of DOC in natural waters is reactive with copper (De Schamphelaere *et al.* 2004; Schwartz and Vigneault 2007; Welsh *et al.* 2008), then if 100% of DOC were treated as copper reactive in the BLM it could bias toxicity predictions high when DOC is abundant, and conversely bias predictions low when DOC is scarce. The approximation of metal-binding by a large, complex, and variable group of organic acids making up natural DOC in waters is an extremely difficult problem, and some studies have found that the WHAM model used in the BLM may markedly over-predict organic carbon complexation of copper, resulting in measured free-ion concentrations exceeding predicted values (Boeckman and Bidwell 2006).

The 2007 BLM-based criteria treat 100% of DOC as copper reactive. We evaluated this issue in several of the datasets including the Stratus Sacramento River data with rainbow trout and Chinook salmon by reducing the input DOC by 50% as fulvic acid and generating a new LA50 and predicting toxicity. Welsh *et al.* (2008) give more details on the "50% active fulvic acid (AFA)" adjustment. Curiously, this adjustment slightly improved results with rainbow trout, but slightly worsened predictions with Chinook salmon (Figure 6, bottom). With rainbow trout, the 50% AFA adjustments brought the slope of the empirical vs. predicted line to nearly 1.0, reduced the standard error and reduced the average prediction error slightly from 2.0 to 1.7 (i.e. with a prediction error or "prediction factor" of 2.0, on the average predicted values were within a factor of 2 of the empirical values). However, with Chinook, the prediction factors were little changed with 1.55 to 1.60 for the default and 50% AFA approaches. Thus the 50% AFA "improvement" was not important for the Chinook data.

A second large, comprehensive, and similarly unpublished dataset with rainbow trout and copper is from a "water-effect ratio" (WER) study from the Clark Fork River, Montana (ENSR 1996). The WER approach involves toxicity testing in tandem in dilution waters collected from the site water of interest and in a standard reconstituted laboratory water. The WER is the ratio of the test LC50 in site water divided by the LC50 in laboratory water; the ratio is then multiplied by the aquatic life criteria to obtain a WER-adjusted site-specific criteria. The WER approach is considered here to be a fundamentally limited concept because it is unrealistic to expect any laboratory water to represent the variety of natural and synthetic waters used in testing laboratories. However, in instances such as the Clark Fork testing, WER studies may produce important datasets that are very useful for evaluating BLM performance, because tests are well matched, often conducted across a wide range of DOC and inorganic chemistries, and the better studies measure detailed water characteristics that may influence copper toxicity. In the case of the Clark Fork testing, rainbow trout were tested in laboratory and in natural waters from tributary and river sites during different seasonal "rounds" of testing with measurements of all BLM chemical parameters. This resulted in values ranging from very soft to very hard waters and DOC from less than 1 mg/L to 11 mg/L. Because the "Round 1" and "Round 4" data were collected from the same places in September 1994 and September 1995, but DOC values were much higher in Round 1 and higher than USGS data for similar locations, we considered the DOC data from Round 1 unreliable and excluded it from our evaluation. This still left a very robust censored data set of 73 tests conducted in diverse waters (Figure 7).

Using the censored data, the BLM predictions followed the empirical LC50s reasonably well, with an average prediction error of 1.65 and the worst prediction error of 5.0. When we tried the 50% AFA adjustment as described earlier, the $R^2$ coefficient of determination value was noticeably improved and the average prediction error was lessened to 1.46 with the worst prediction error lowered to 3.5. The "50% AFA" approach improved the prediction errors in 46 pairs and worsened the errors in 27 pairs. Thus the 50% AFA "improvement" seemed real with this dataset.

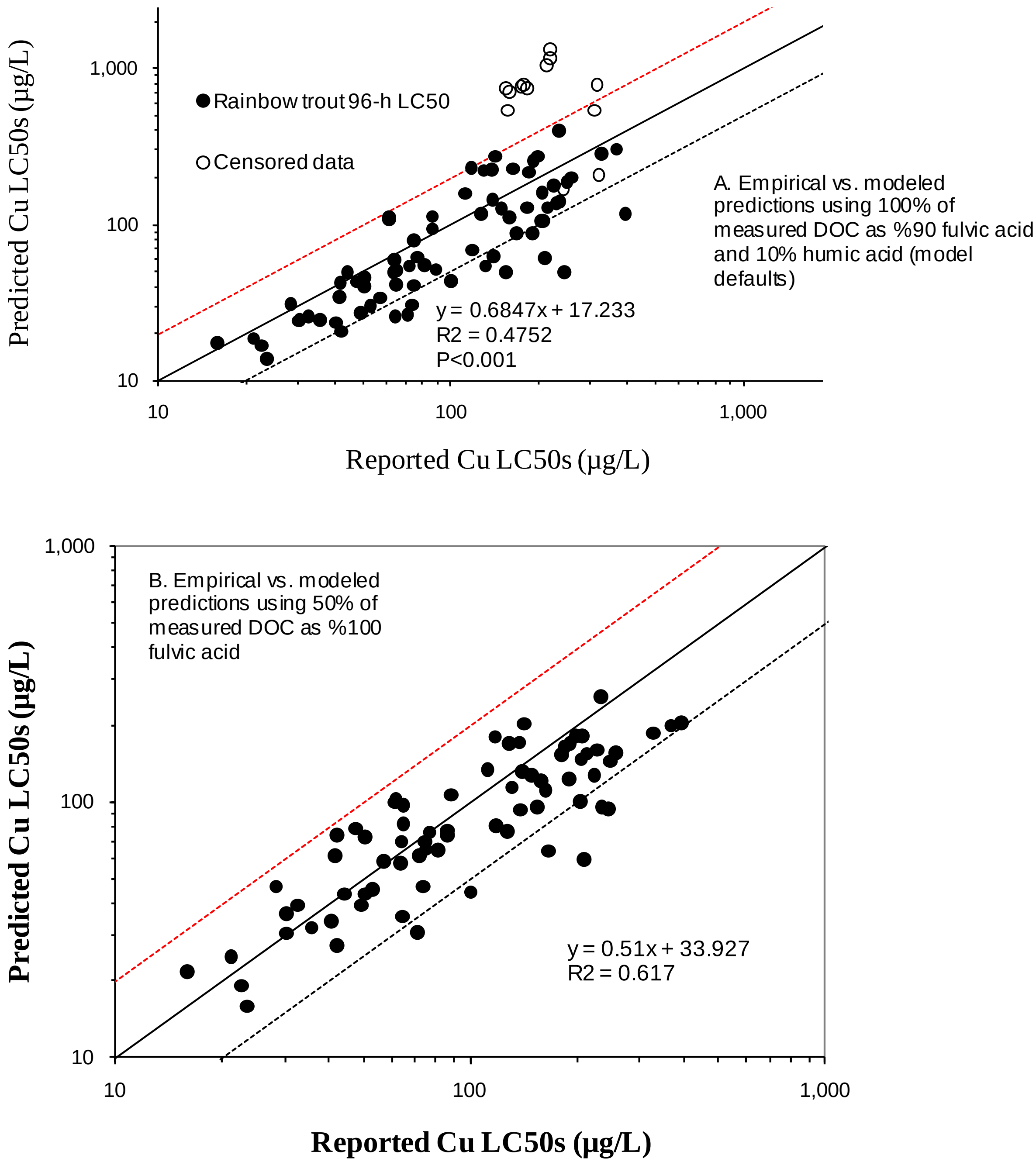



**Figure 7. Rainbow trout: predicted vs. empirical toxicity, using the 2007 BLM, in 96-hr renewal tests using lab and site waters, hardness 23-308 mg/L, DOC from <1 to 11 mg/L. Data from ENSR (1996).**

In contrast to the studies we described here that evaluated BLM performance in natural waters where DOC and pH were probably the most important factors, the following evaluations consider inorganic factors that affect copper toxicity, such as calcium, magnesium, and alkalinity. These comparisons allow better evaluation of performance of BLM parameters than with natural waters, because in natural waters inorganic chemical factors tend to be correlated with each other. If the model performs well in replicating observed toxicity, then evaluations with natural waters are persuasive. However, if the model performs poorly, if the factors all rise and fall together, there is no way to tease out which factors need adjusting in the model. Thus even though the chemical combinations in such "factors testing" may be contrived in ways that would seldom ever occur in nature, together with testing in natural waters these "factors tests" may provide a thorough examination of model performance.

Welsh *et al.* (2000; 2001) and Naddy *et al.* (2002) tested the sensitivity of rainbow trout to copper in waters in which they tested the relative importance of Ca or Mg in mitigating toxicity by concocting waters which had similar hardnesses, but different Ca and Mg ratios. Both studies found that Mg conferred little protection from copper toxicity to fish, although Naddy *et al.* (2002) found Mg did reduce copper toxicity to Daphnia. The 2003 version of the BLM for copper did not include Mg. However, Mg was included on an equal basis to Ca in the 2007 version (Table 2). The 2007 BLM performed poorly in our review, with the BLM predicting toxicity to decrease with increasing Mg contribution to hardness, when little or no reduction occurred (Figure 8). For example, the Naddy *et al.* (2002) tests were all predicted to have LC50s of about 50 µg/L, when in fact they varied from about 15-70 µg/L. In a pair of tests with different Na content, the BLM accurately predicted the observed pattern.

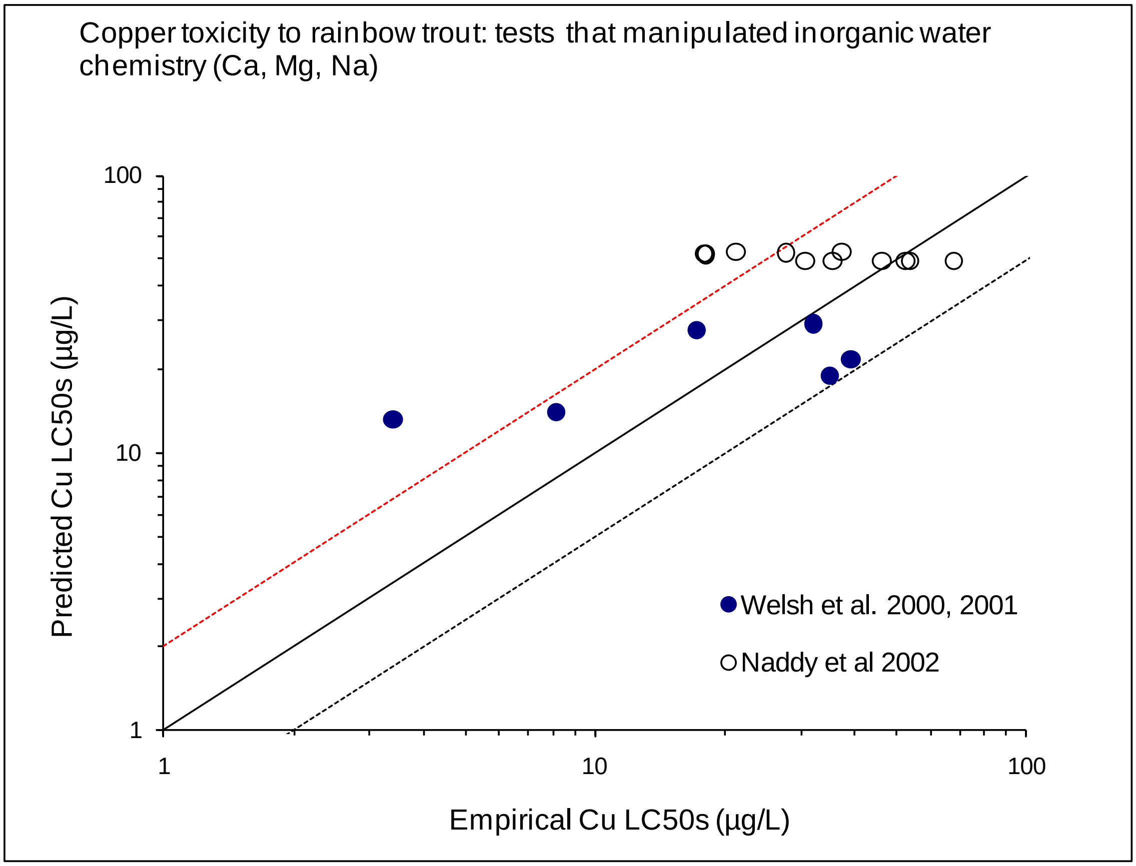



**Figure 8. Rainbow trout copper 96h LC50s, with varying Ca and Mg while keeping hardness and alkalinity about the same and with uniform low DOC. Data from Welsh *et al.* (2000; 2001) and Naddy *et al.* (2002), using the 2007 BLM.**

Naddy *et al*. (2002) also attempted one test in which magnesium hardness made up all of the total hardness, that is no calcium was added. However, all of the fish died within 48-hours even in the controls with no added copper. This reinforces the critical role of calcium in stream water, that very low calcium waters are stressful independent of metals, and that metals can be exceptionally toxic in low hardness water.

For NMFS' final evaluation of the BLM performance with inorganic factors, we evaluated a series of 9 tests with cutthroat trout and copper that alternately held alkalinity constant and varied hardness or vice versa (Chakoumakos *et al.* 1979). Because the alkalinity manipulations involved different proportions of spring water and amended distilled water, and the spring water contained higher DOC, we grouped the tests by alkalinity, for which DOC was probably about uniform across the tests.

In all, these results suggest that Ca and Mg should not be treated as equally important in the BLM (Table 2), but that Ca should be given a higher binding affinity log K value. While these results suggest that the 2007 BLM modifications tend to lessen the BLM's conceptual improvement over the hardness-equations, because most natural waters tend to have more Ca than Mg (Appendix A), the poor model performance in these datasets probably should not be given more importance than performance with diverse natural waters.

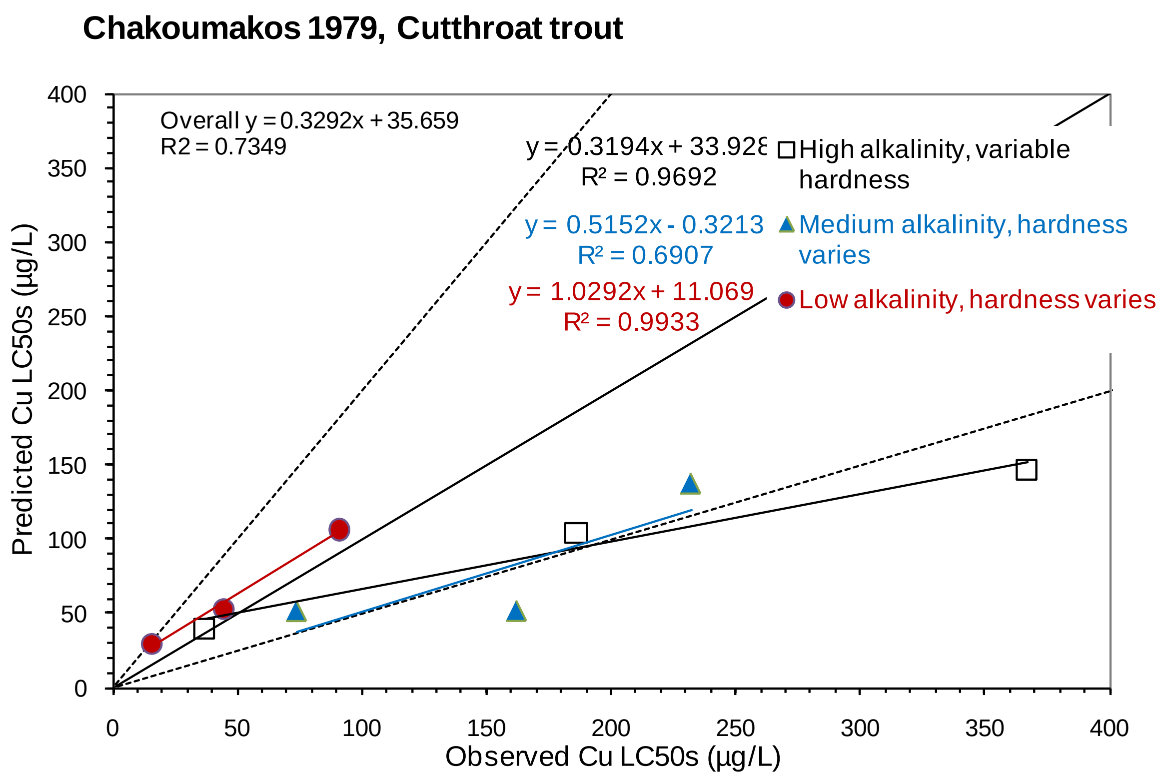



**Figure 9.** **Cutthroat trout modeled and predicted responses to copper under various combinations of low, medium, and high hardness and low, medium, and high alkalinity, in waters with DOC ranging from ~0.9mg/L in low alkalinity water to 3.3 mg/L in their high alkalinity spring water, 2.7 to 9.7g fish (Chakoumakos *et al*. 1979). Because DOC was nearly uniform within alkalinity treatments, data are grouped by alkalinity groups.**

The BLM performance seemed mixed in these comparisons (Figure 9). For the low alkalinity series, the BLM and empirical LC50 estimates were nearly perfect with a slope of 1.0 for predicted:empirical best fit line. Yet, for the tests at higher hardness, while the predictions were correlated with the empirical results, the slopes were progressively lower with copper being less toxic than predicted. This pattern is hard to interpret with just these data, but seems to support the idea that the log K value for Ca in the model could be higher.

In summary, this portion of NMFS' review evaluated the ability of the 2007 BLM to accurately predict acute copper toxicity by evaluating hundreds of separate tests with fathead minnows, rainbow trout, Chinook salmon, and cutthroat trout in diverse natural and artificial waters. With one exception, the BLM performed substantially better than did the hardness-toxicity derived criteria (i.e., the NTR and Idaho criteria). In the exception, Naddy *et al.*'s Ca and Mg manipulations shown in Figure 8, the BLM and hardness models were similarly poor; the BLM under predicted toxicity in very soft waters. The DOC influence in the BLM has been suggested to be too strong and a source of bias. This idea was generally supported in our evaluations of BLM performance in natural waters with fathead minnows and rainbow trout, but not of the (much smaller) Chinook salmon dataset. However, the magnitude of this apparent bias was not great. These analyses suggest areas of potential refinement and possible further improvement in the BLM, but do not necessarily indicate that the 2007 BLM is inappropriate to use as published. However, of the various analyses completed thus far, the evaluations of the overall BLM performance in natural waters is considered more important than "factors testing." In the great majority of tests, the BLM correctly predicted the direction of relations (i.e., more or less toxic) and most predictions of specific LC50 concentrations were reasonably close to empirical estimates.

Acute and chronic toxicity predictions with invertebrates

While our analyses so far evaluated the performance of the BLM with acute copper toxicity in fish, the criteria generated by the BLM apply to all aquatic animals, even invertebrates with gills that are too small or diffuse to directly test. This assumption that the BLM criteria are protective of all aquatic species must largely be true for the BLM to be a valid basis for protecting aquatic communities and, for example, avoiding adverse effects to food chains for ESA listed fish species. Previous versions of the model had good performance predicting toxicity to the zooplankter *Daphnia pulex* (Santore *et al.* 2001). Here, NMFS attempts to validate the performance of the 2007 BLM with other invertebrates. This is easier said than done, for invertebrates are woefully underrepresented in toxicity testing datasets compared to their relative diversity in the wild. For instance, there are at least 10X more aquatic insect species in North America than fish species, but few insects are represented, especially for long-term toxicity datasets (Mount *et al.* 2003; Mebane 2006). A more sensitive and practical approach is to test aquatic insect communities in experimental stream mesocosms, although these sturdies are complicated to interpret. (We discuss this in a later section "Accuracy of copper BLM toxicity predictions in field and experimental ecosystem studies.") We located and evaluated useable acute toxicity datasets with copper for invertebrate taxa: two additional zooplankters, *Daphnia magna* and *Ceriodaphnia dubia*, the freshwater benthic crustacean *Hyalella azteca*, and for two freshwater mussels, fatmucket, *Lampsilis siliquoidea* and rainbow mussel (*Villosa iris*).

We also located sufficient data to evaluate the ability of the BLM to predict chronic toxicity to rainbow mussel and *C. dubia*. This is of particular import because although the 2007 BLM is used to derive chronic criteria, it was developed as an acute model and is not known to have been previously validated for chronic predictions.

The first dataset we evaluated was an acute study with different combinations of pH and DOC tested with acute *Daphnia magna* (Meador 1991). Strengths of this data set include that the test conditions were well controlled and that natural DOC was concentrated from algae exudates, as opposed to some studies that evaluated the role of DOC by adding Aldrich humic acid that is commercially prepared for sale as a gardening soil amendment. The BLM predictions were reasonably favorable in comparison with the empirical results with a slope slightly less than 1.0. The BLM explained a little less than half of the variability in the data, although most values fell within the "factor of 2" rule of thumb for adequate model performance (Figure 10).

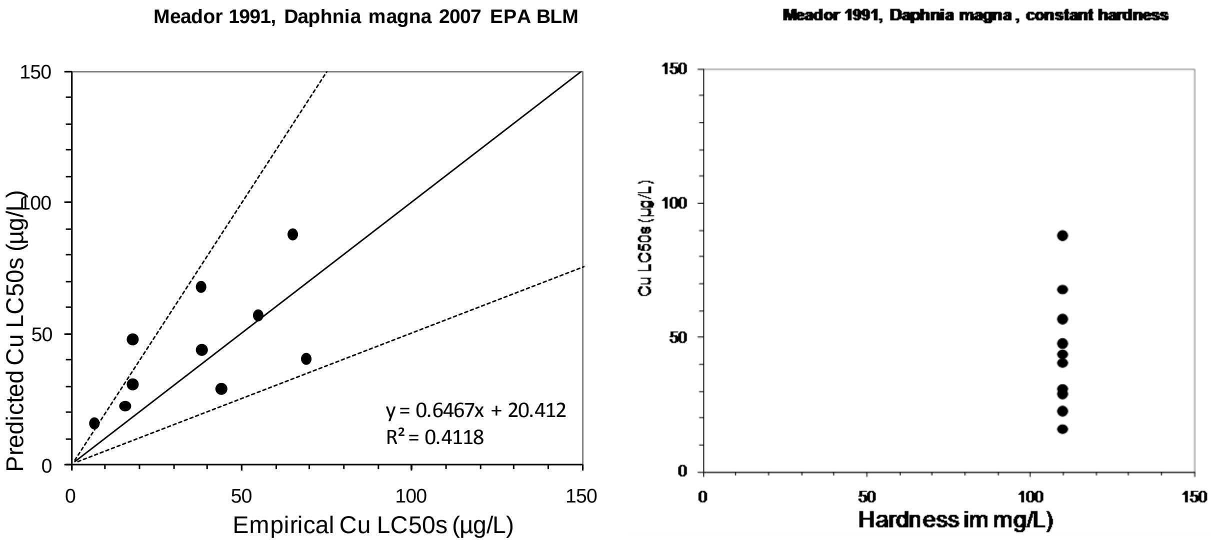



**Figure 10. (Left) Daphnia magna, BLM predicted and empirical copper toxicity, pH 6.9 to 7.9, DOC 2.4 – 6 mg/L, and (right) hardness as a predictor of toxicity (Meador 1991).**

We located two useful datasets with the amphipod *Hyalella azteca* in which the animals were tested across a gradient of water chemistry conditions within each study (Welsh 1996; Collyard 2002). Thus although one study compared *Hyalella* responses in 96-hour exposures and one in 48-hour exposures, the responses can be compared within the datasets for each study.

We reviewed but excluded two other studies with *Hyalella* and copper for our use comparison. The first was a large study on the effects of major ions (Ca, Mg, Na, and K) and pH on Cu toxicity to *Hyalella azteca* (Borgmann *et al*. 2005). These tests were not as useful for testing the BLM-copper toxicity predictions as the studies shown here because DOC concentrations were variable and uncertain. The data were from static, non-renewal, 1-week exposures in which the animals were fed twice during the tests. The DOC in the artificial media used as a dilution water rose from between <0.1 to 0.2 mg/L before introduction of animals or food to a range of 0.4 to 2.8 mg/L (average 1.72 mg/L) at the end of the test. Modeling the predicted LC50s using either the initial or average end of test DOC values showed that this uncertainty in DOC values alone was carried through to an average additional prediction error of 3X. Using 0.2 mg/L DOC in the model inputs resulted in an average 7-day LC50 prediction of 16 µg/L copper compared to 50 ug/L for the end of test conditions. For tests with low alkalinities, this uncertainty resulted in prediction differences greater than a factor 10. Thus, this dataset was not used to evaluate the BLM performance. The second, a study on the effects of pH on metals toxicity, did not include sufficient water chemistry to re-analyze their data through the BLM (Schubauer-Berigan *et al.* 1993). The EPA (2007a, Appendix E) had estimated ion content for the base dilution waters based on the recipe for very-hard reconstituted water. However, the base water was amended with hydrochloric acid (HCl) to experimentally lower the pH, which would have also lowered the alkalinity and raised chloride content relative to the base water. Differences in chloride can influence *Hyalella* growth and reproduction (Dave Mount, EPA, Duluth, MN, personal communication) so perhaps it is not too great a logical stretch to assume chloride might influence acute survival as well. Regardless, BLM predictions for copper are sensitive to the alkalinity of the waters, and alkalinity was unmeasured and assumed constant in the amended waters. Thus, the uncertainties regarding this dataset seemed such that they could invalidate validation attempts.

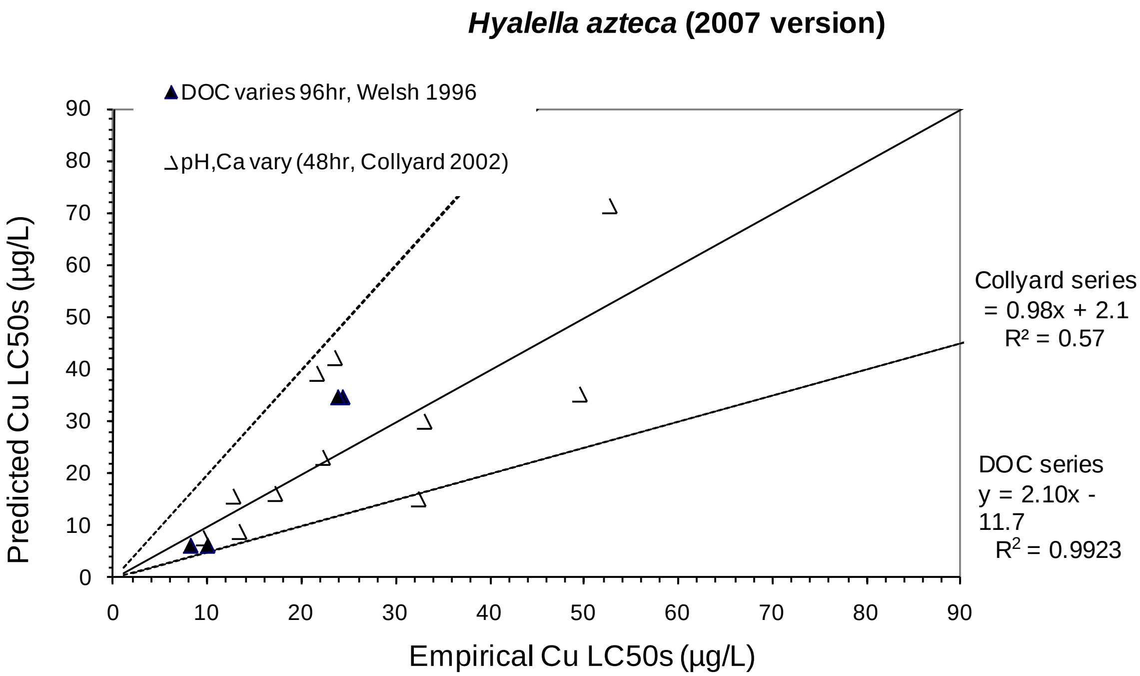



**Figure 11. Amphipod *Hyalella azteca*: Predicted and empirical copper toxicity to Hyalella azteca under conditions of varying natural organic matter (NOM), pH, and calcium.**

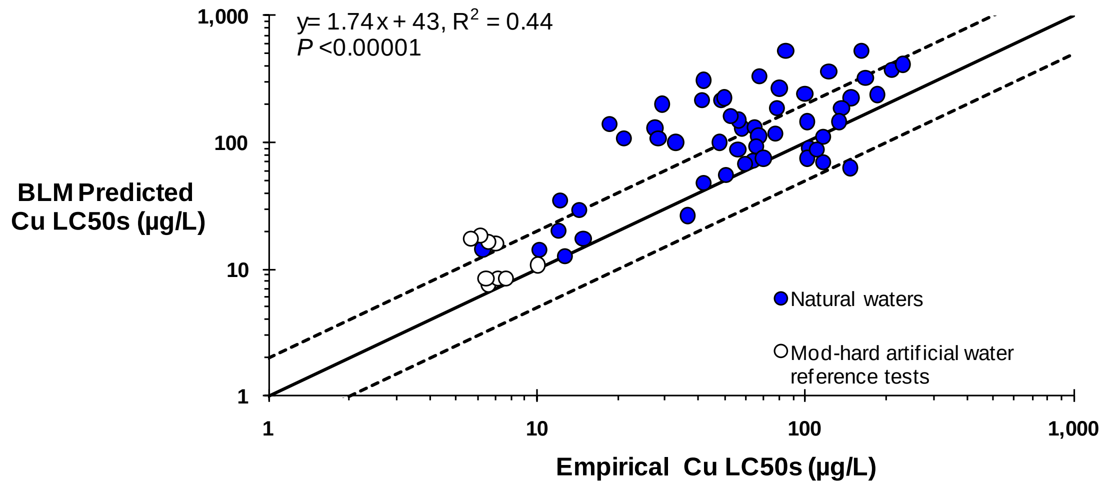
Of the *Hyalella* and copper datasets retained, the results of the BLM predictions were favorable. For the series of four tests using natural lake waters with different DOC (i.e., naturally occurring organic matter or NOM) concentrations, the predictions were highly correlated with observations although the slope of the predicted: empirical toxicity line was about 2, indicating that a stronger DOC effect was predicted than observed for these data (Figure 11). The results of the series with variable calcium and pH in the absence of DOC showed reasonable agreement, with the slope of the empirical to predicted toxicity regression approximately 1.0 and with the model explaining about half the variability in the data ($R^2$ of 0.56).

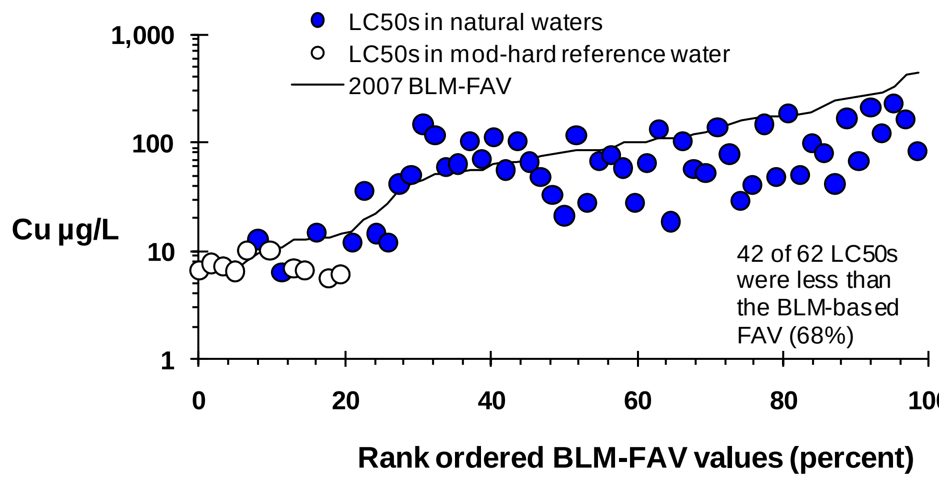
We then considered a comprehensive study of relative copper toxicity in natural waters collected from forested streams in Michigan's Upper Peninsula, with a wide range of hardnesses and DOC. The Great Lakes Environmental Center ("GLEC," a private environmental consulting firm) tested the toxicity to the cladoceran zooplankter *Ceriodaphnia dubia* in about 25 natural waters and in a moderately-hard artificial reference water. The hardnesses of the stream and lake waters ranged from about 17-185 mg/L $CaCO_3$ and 0.8 to 30 mg/L DOC. The toxicity in the artificial reference water was repeatedly tested as a benchmark of the inherent variability of (GLEC 2006).

The results showed that the *Ceriodaphnia* copper LC50s were correlated with the BLM predictions, with the regression explaining 44% of the variability in toxicity ($R^2$ value of 0.44). This is lower than some other datasets examined, and the slope of 1.7 is steeper than optimal (Figure 12, top). Still, the results were reasonably favorable, especially when compared to hardness-toxicity plot, where hardness explained less than 1% of the variability in the data, and the slope of the best fit line actually went the wrong way (decreasing toxicity with increasing hardness, Figure 12, bottom).

Comparing the *Ceriodaphnia* LC50 values with the BLM-based FAV shows that in general *Ceriodaphnia* would not be fully protected by the BLM copper criteria (Figure 12, middle). This is a two edged observation. First, *Ceriodaphnia* arguably never were intended to be protected by the 2007 copper criteria because they fall below the 5$^{th}$ percentile of the species sensitivity distribution (SSD) used to define the criteria and presumably *Ceriodaphnia* provide redundant ecosystem functions and are not "important" species that warrant a downward adjustment of the criteria to afford them protection. This emphasizes how fundamental the assumptions are that protecting 95% of representatives in a dataset is sufficient to protect freshwater ecosystems. With copper, these assumptions could be questionable since of the 27 genera included in the 2007 copper FAV, the lowest two genus mean acute values and lowest three species mean acute values (GMAVs and SMAVs) are for cladocerans. While individually, a cladoceran species may not necessarily be "important" because of the assumed functional redundancy provided by other cladoceran zooplankters, cladocerans as a group probably have keystone ecological functions by their intermediate role in lake food webs between algae (phytoplankton) and planktivorus fish species such as sockeye salmon. In theory, cladocerans might not be fully protected by the acute copper criterion, since of the four species included in the acute toxicity dataset, three (*Ceriodaphnia dubia*, *Daphnia magna* and *Daphnia pulicaria*) have SMAVs near or below the FAV, and the fourth should not have been included because it does not meet data quality guidelines for criteria derivation. This fourth GMAV, *Scapholeberis sp.*, listed 5$^{th}$ most sensitive of 27 genera, was from a single test with an adult (EPA 2007a). Under EPA's Guidelines, tests with daphnids and other cladocerans should be started with organisms that are less than 24 hours old. Tests with older animals could be considered as "other data" (Stephan *et al.* 1985, p. 33). Because the three daphnid species in the criteria dataset cannot be assumed to be the most sensitive of the widely distributed North American daphnids (Koivisto *et al*. 1992; Harmon *et al.* 2003; Shaw *et al*. 2006), this undermines conceptual support for the 5$^{th}$ percentile SSD criteria approach in cases where a major taxonomic group might go under protected together.

*Ceriodaphnia dubia*, **EPA 2007 parameters**, Upper Penisula, MI (GLEC 2006)



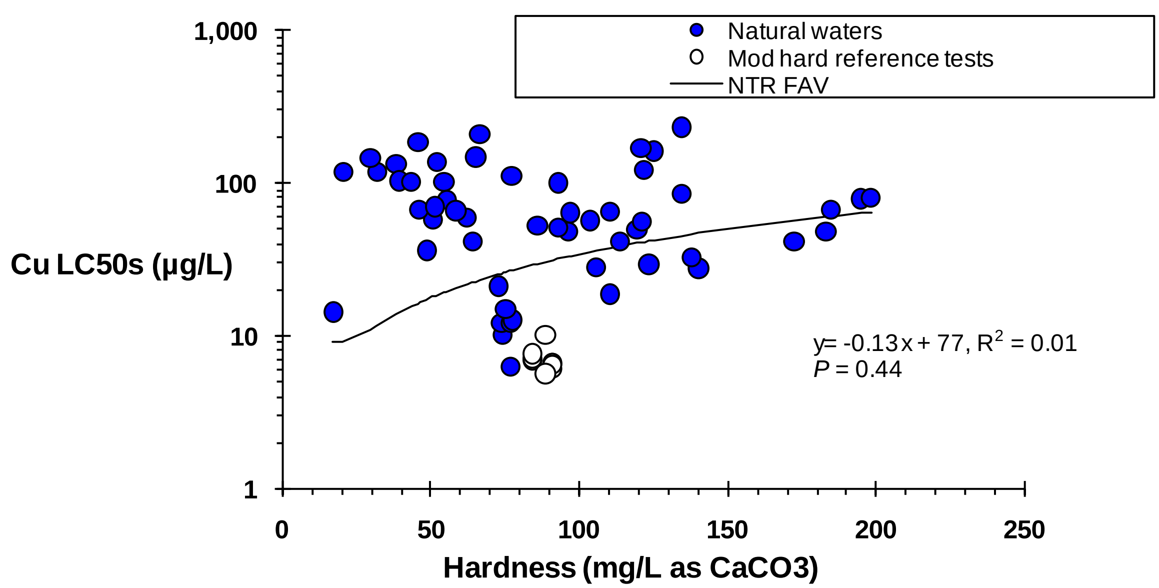



**Figure 12. Cladoceran *Ceriodaphnia dubia:* correspondence between BLM predicted and empirical copper LC50s for in natural and reference waters (top); comparison of copper FAV and LC50s (middle); and lack of correspondence between hardness and Ceriodaphnia LC50s (bottom). Data from (GLEC 2006).**

There is a flip side to the concerns that *Ceriodaphnia* and its relatives may not be adequately protected by the 2007 BLM based criteria or the hardness-based criteria. That is, because the *Ceriodaphnia dubia* three brood test is one of the two routinely required for whole effluent toxicity (WET) tests for effluent discharges, this shows that, for copper the *Ceriodaphnia* test is a sensitive tool for evaluating instream toxicity from discharges. Because there are sometimes similarities in relative sensitivity rankings of organisms with different substances, these patterns suggest the *Ceriodaphnia* WET test could be sensitive to other metals and other substances as well.

*Chronic toxicity predictions with invertebrates*

The mostly favorable evaluations of the performance of the BLM and the protectiveness of the BLM-based criteria for invertebrates thus far considered only acute test data. The 2007 BLM criteria are based upon an acute model of copper toxicity, and the acute BLM predictions are extrapolated to derive chronic criteria through a fixed acute-to-chronic ratio (ACR). This approach has been criticized as counter to knowledge on different mechanisms of acute and chronic toxicity of metals (Niyogi and Wood 2004) and by practical arguments that an acute copper biotic ligand model (BLM) for *D. magna* could not serve as a reliable basis for predicting chronic copper toxicity (De Schamphelaere and Janssen 2004a).

Considering these criticisms, we compiled four well-characterized datasets of chronic copper toxicity to invertebrates in order to evaluate the performance of the acute-BLM to predict chronic toxicity. The datasets evaluated were:

1. cladoceran (*Ceriodaphnia dubia*) in tests with DOC varying from <1 mg/L to 10 mg/L (Wang *et al*. 2011);

2. parallel tests with copper in 28-day exposures with the rainbow mussel (*Villosa iris*) (Wang *et al*. 2011);

3. three tests with Daphnia magna growth, survival, and reproduction in 21-day exposures in water with low DOC and a range of water hardnesses (Chapman *et al*. 1980); and

4. 35 tests with *Daphnia magna* growth, survival, and reproduction in 21-day exposures in water with mostly high DOC (range 2 – 22 mg/L) and a range of water hardnesses and pH (De Schamphelaere and Janssen 2004b).

Our review of the mussel and *Ceriodaphnia* study showed that the BLM performed remarkably well, explaining over 90% over the variability observed with the mussel survival and growth endpoints and the *Ceriodaphnia* acute and chronic survival endpoints. The empirical estimates of *Ceriodaphnia* reproduction endpoint did not vary as much as did the predicted estimates, producing weaker relations between predictions and empirical results with the model explaining only about 33% of the observed variability (Figure 13).

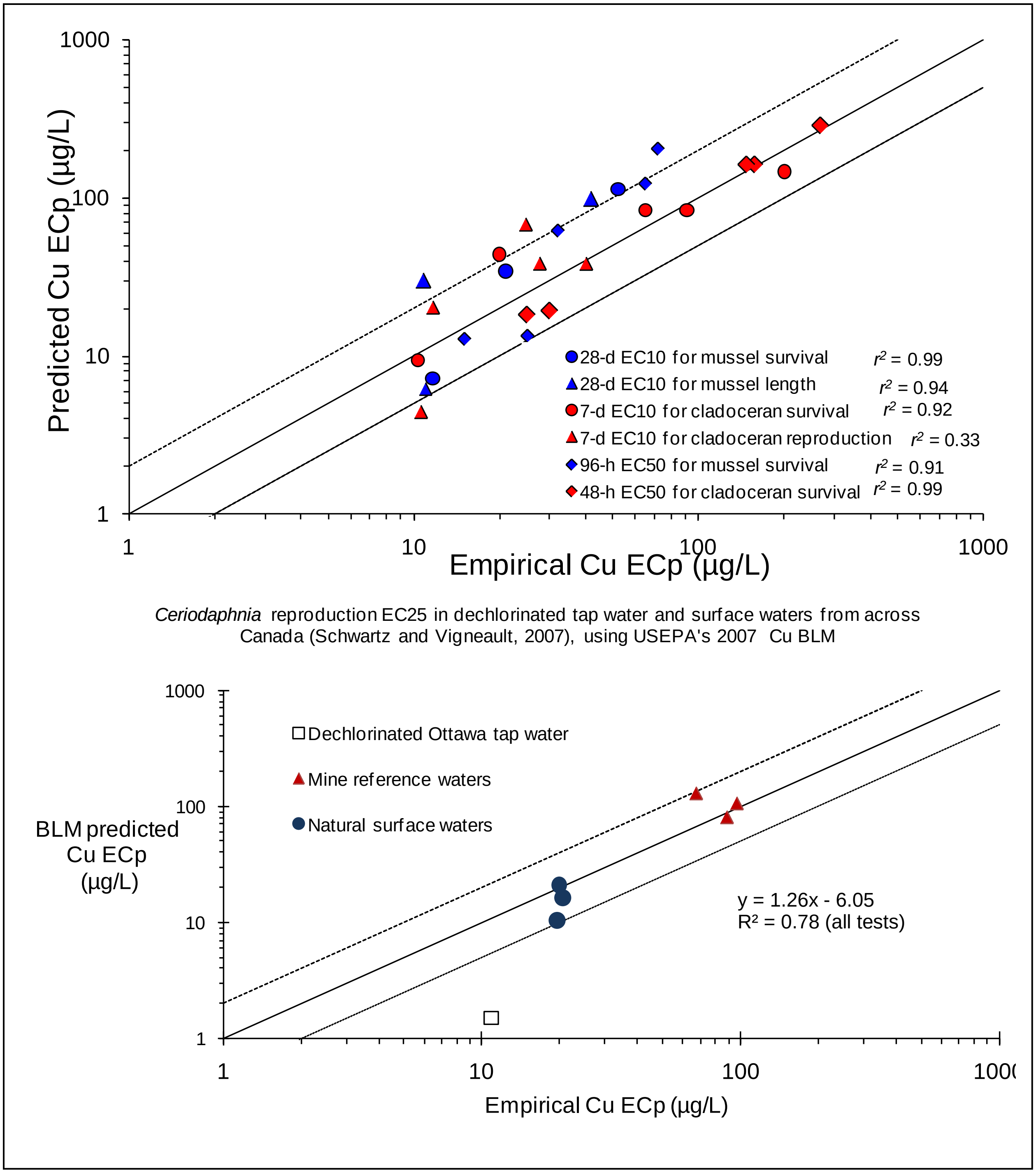



**Figure 13.** **Chronic toxicity of copper to rainbow mussel (*Villosa iris*) and the cladoceran (*Ceriodaphnia dubia*): A. BLM-predicted and empirical acute and chronic toxicity of copper to in tests with DOC varying from <1 mg/L to 10 mg/L (Wang *et al.* 2011), B. reproductive impairment of *C. dubia* in waters with DOC ranging from 0.4 to 33 mg/L and water hardness of 23 to 170 (Schwartz and Vigneault 2007).**

The difference in comparisons between endpoints may reflect the inherent variability in biological testing, and reproductive endpoints are generally more variable than growth or mortality endpoints. Using the mean critical accumulation value estimated for the BLM from

Wang *et al*'s (2011) *Ceridaphnia dubia* reproductive EC20s, *Ceriodaphnia dubia* reproductive EC25s determined with diverse surface waters from across Canada by Schwartz and Vigneault (2007) were predicted remarkably well by the BLM (Figure 13, bottom)

The performance of the BLM-based copper criteria with freshwater mussels in chronic exposures is important because many freshwater mussels are in decline in the United States, some species are among the most sensitive taxa reported to date with copper, and traditional hardness-based copper criteria are under-protective of mussels (March *et al*. 2007). Yet with acute and chronic exposures in artificial lab waters with very low DOC and with acute exposures in natural waters with a range of DOC concentrations from different sources, the BLM-based copper criteria appear mostly protective (Wang *et al*. 2007a; Wang *et al*. 2007b; Wang *et al*. 2009). The most recent study with a freshwater mussel in waters with natural DOC added indicates that the BLM-based chronic copper criteria is protective for at least the species tested, and presumably also for closely related, untested taxa.

De Schamphelaere and Janssen (2004b) tested the effects of pH (5.3–8.7), water hardness ($CaCO_3$ at 25–500 mg/L), DOC concentration (1.6–18.4 mg/L), and DOC source on the chronic toxicity of copper to *Daphnia magna* as 21-day survival or reproductive impairment. Chapman *et al*. (1980) similarly tested the chronic toxicity of copper to *Daphnia magna* as 21-day survival or reproductive impairment, in waters with hardness varying from about 50 to 200 mg/L. Reproduction was the most sensitive endpoint in both tests, and the statistical no-observed effect concentrations (NOECs) are used here for comparability between the studies.

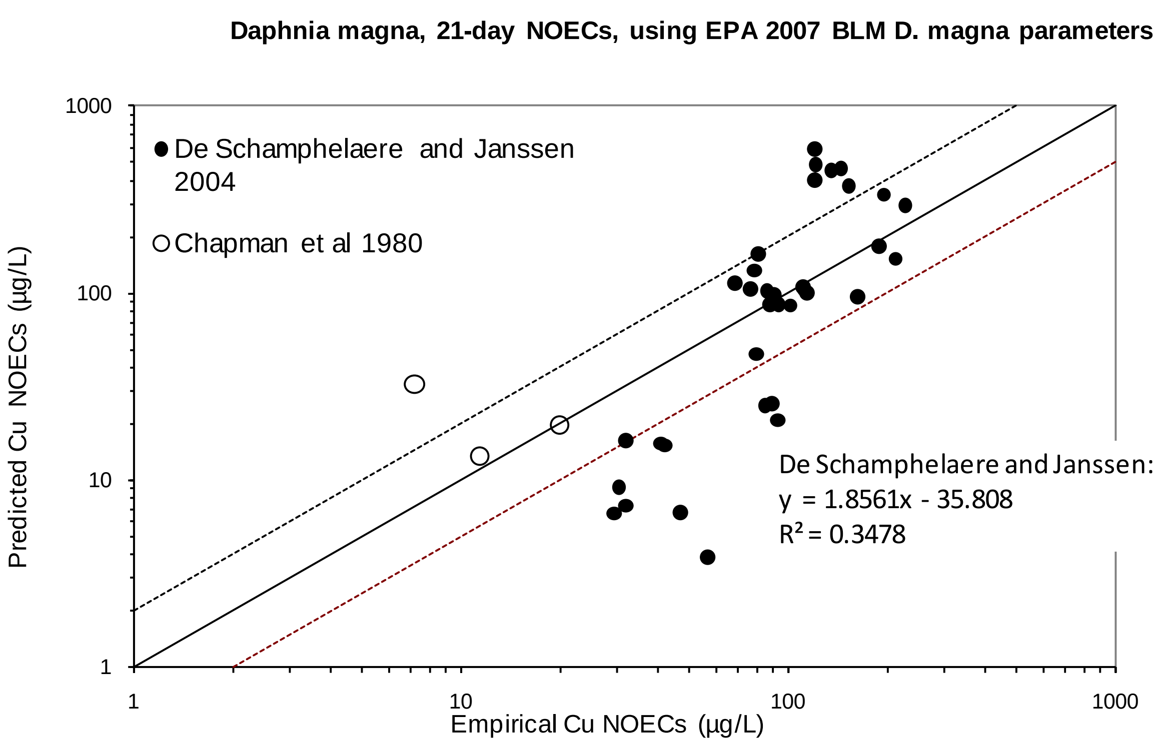



**Figure 14. With chronic *Daphnia magna* in the De Schamphelaere and Janssen dataset, the tests with higher DOC tended to underpredict toxicity (NOECs too high) in tests with relatively high DOC and overpredict toxicity in tests with low DOC. In the Chapman dataset, the two points falling on the 1 to 1**

**line of agreement are with hardness 50 and 100 mg/L. Their test at hardness 200 mg/L was more sensitive to copper than the tests at lower hardness.**

With these data, the agreement between the BLM predictions was considerably worse than in most other datasets (Figure 14). This might in part be related to working with the NOEC statistic. The NOEC is not the best statistic for comparing effects between tests because the NOEC has to be one of the concentrations tested, and the actual magnitude of this "no-observed effect" might in fact range from a 0% to 30% or more reproductive impairment, whereas with EC10 or EC50 values, a common magnitude of effect, e.g., 10% or 50%, is estimated from nonlinear curve fits (Appendix C). However, by inspecting the underlying values, this seems unlikely to explain these "noisy" relations. Instead, De Schamphelaere and Janssen (2004b) suggested a bias associated with the BLM predicting more mitigation of copper effects from DOC than indicated in the empirical data and pH having a stronger influence than accounted for in the model. The Chapman data are particularly puzzling because while toxicity was reduced exactly as predicted from a hardness of about 50 to 100 mg/L, in the test at a hardness of 200 mg/L, copper was more toxic than at either lower hardness condition. Review of the raw data from the tests indicated no test performance data quality problems, and this result remains unexplained.

We estimated critical accumulation values for Daphnia magna 21-day reproduction from the De Schamphelaere and Janssen (2004b) study with the BLM in order to make the comparisons between predicted and empirical effects. Curiously, we found, it was nearly identical to the species mean critical accumulation value ("LA50") for 48-hour lethality to Daphnia magna estimated from the EPA (2007) SMAV for Daphnia magna. Generally, 48-hour LC50 values for *Daphnia magna* are expected to be considerably higher than 21-day reproductive no-apparent values for *Daphnia magna*. Such was the case with the Chapman *et al*. (1980) study. The reason for the relative insensitivity of the De Schamphelaere and Janssen (2004b) chronic data is unexplained, although they suggest that their treatment of only considering about 50% of the DOC as having a role in reducing copper toxicity, compared to the EPA (2007) approach of treating 100% of the DOC as having a role in reducing copper toxicity as one factor (De Schamphelaere and Janssen 2004a).

<u>Chronic toxicity predictions with fish</u>

No previous evaluations of the 2007 BLM, developed as an acute model of copper toxicity, are known of for predicting chronic copper toxicity to fish, especially listed salmonids or their surrogates. As with the case of predictions of chronic effects to invertebrates, the extrapolation of the acute model to chronic predictions through a fixed acute-to-chronic ratio (ACR) has been strongly criticized as speculative and counter to knowledge on different mechanisms of acute and chronic toxicity of metals (Niyogi and Wood 2004). However, this is the approach used by EPA (2007). Here the question considered is not whether the ACR extrapolation compromises the mechanistic basis of the model (it does), but as a pragmatic issue, can the 2007 acute BLM produce reasonable estimates of chronic toxicity to fish? If not, are the criteria still protective?

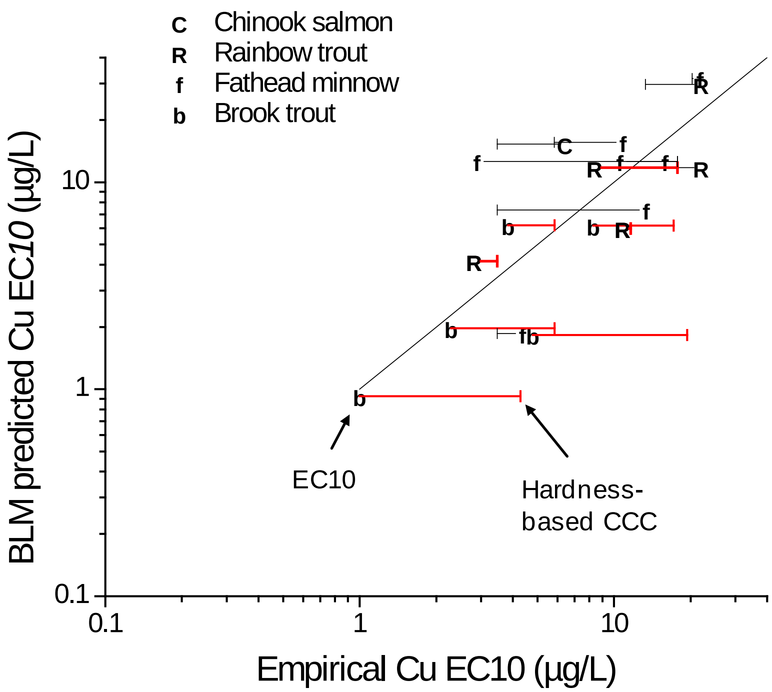
Because of the expense and complexity of chronic toxicity testing, chronic data are much rarer than acute data. Also, with acute data, usually only the 50% mortality endpoint is considered, but with chronic data a variety of endpoints may be tested (e.g., growth as length or weight, fecundity, survival), and statistical endpoints that approach the threshold for the onset of adverse effects are of more interest than an extreme 50% effect. Test durations of "chronic" tests generally range from 30 days to over 2 years. For instance, with growth reductions, fish will often die before growth reductions on the order of 50% are ever realized. We located several datasets, although the conditions tested were much more limited than with acute data. Few data tested different water chemistry conditions within a study. Exceptions were a pair of 30-day tests with fathead minnows in natural waters with different DOC and pH by Welsh 1996; and a study with nine tests of different hardness and pH, although in that case some assumptions about missing water chemistry had to be made (Waiwood and Beamish 1978).

With salmonids, we located interpretable datasets with rainbow/steelhead trout (Waiwood and Beamish 1978; Seim *et al*. 1984; Marr *et al*. 1996; Hansen *et al*. 2002a; Besser *et al*. 2005), Chinook salmon (Chapman 1982), and brook trout (McKim and Benoit 1971; McKim and Benoit 1974; Sauter *et al.* 1976; Besser *et al.* 2001). With fathead minnows, interpretable data sets were located that used natural waters (Lind *et al*. 1978; Welsh 1996) and laboratory waters (Mount 1968; Besser *et al*. 2005).

Where needed, we estimated necessary major ion chemistry and DOC inputs to the BLM from other studies or regional values, or from data collected at different times from the same water source. For example, the Sauter *et al.* (1976) tests used as a water supply a 400 ft deep well into bedrock that is still in use. Because the water chemistry in the well appears stable (Mark Cafarella, Springborn Smithers Laboratories, personal communication), and hardness, alkalinity, and pH from the well were similar in 2006 and 1976, major ion and DOC data from 2006 were assumed similar as well. Waiwood and Beamish (1978) said their water chemistry was similar to their (1978) study, which still didn't have all the needed information but was assumed to be similar to that reported by Dixon and Sprague (1981) for the same laboratory. No information on DOC in the University of Guelph toxicity laboratory water supply or in the City of Guelph drinking water reports[11] could be found. Because the water source was dechlorinated City of Guelph tap water that originated as well water, a fixed DOC value of 1 mg/L, since that has been considered a reasonable estimate for similar waters (EPA 2007a, Appendix C).

As an estimate of thresholds of adverse effects, EC10s were estimated for the most sensitive endpoint of each test by nonlinear regression using EPA's Toxicity Response Analysis Program (TRAP) (Erickson 2008). In the Waiwood and Beamish (1978) dataset, no raw data were reported, so it was necessary to use their EC25 values for reduced growth. Swimming performance (10% reduction) was proportional to 25% growth reductions over the same pH and hardness combinations, but growth was more sensitive (Waiwood and Beamish 1978). Thus only growth was evaluated here.

[11] e.g., http://guelph.ca/uploads/ET_Group/waterworks/2003_Waterworks_Summary_Report.pdf



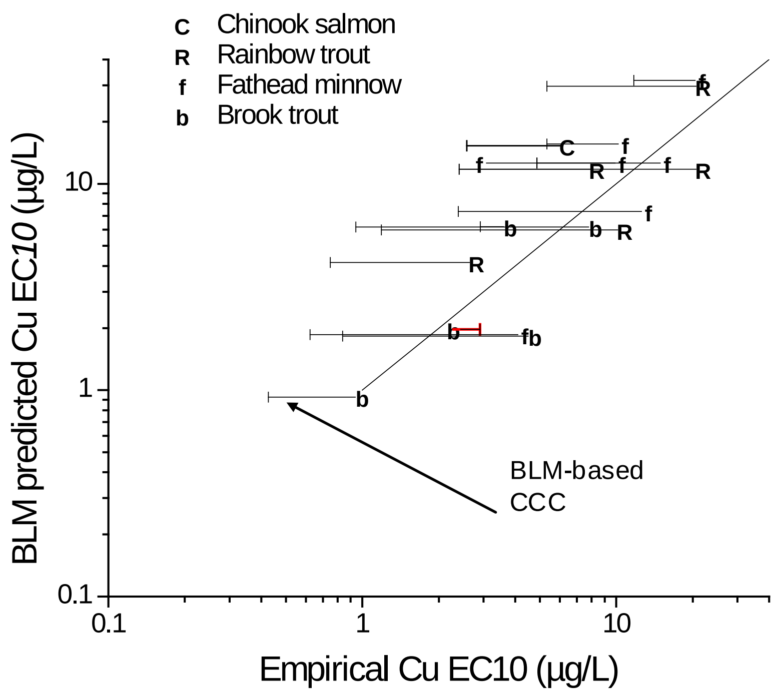
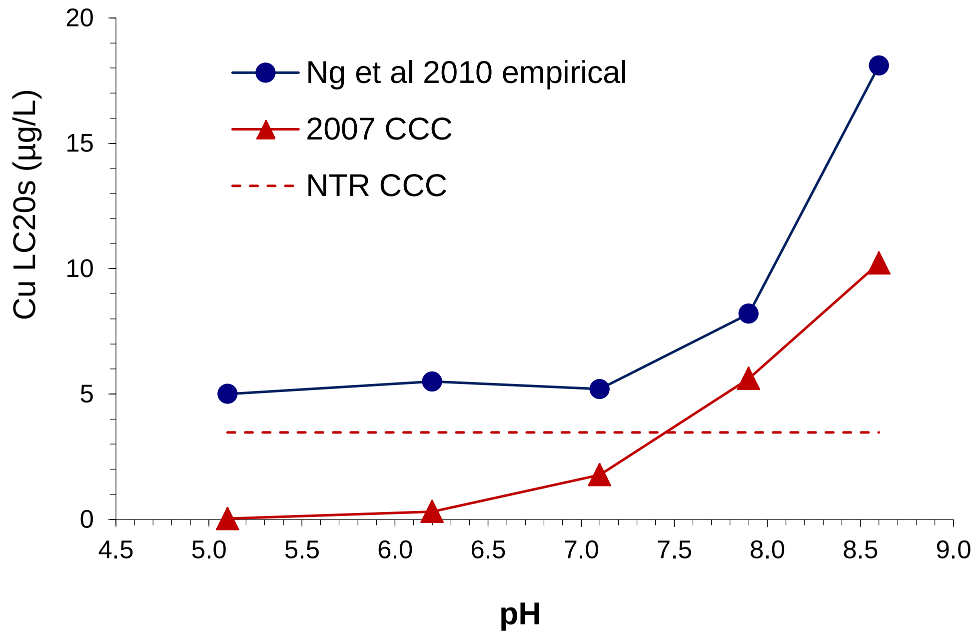




**Figure 15. Protectiveness or non-protectiveness of both hardness-based (top) or BLM-based Cu criteria (bottom) and observed vs. BLM-predicted chronic EC10 values for Chinook salmon, rainbow trout, brook trout and fathead minnows. Horizontal error bars leading to the left of the symbols indicate the difference between the EC10 and the CCC or "safety margin" for the occurrence of low adverse effects and the criterion; horizontal error bars, emphasized in red, leading to the right of the symbols indicate**

**the degree of non-protectiveness of the criterion for that test value. Solid diagonal line is the 1:1 line of perfect agreement. (BLM version 2.2.3, default parameters)**

Despite the disparate test methods used across studies and the many estimates needed to come up with the inputs for the BLM, the results were surprising good, as good or better than some of the acute comparisons. The BLM was able to account for 54% to 71% of the variability in the chronic datasets (Figure 15).



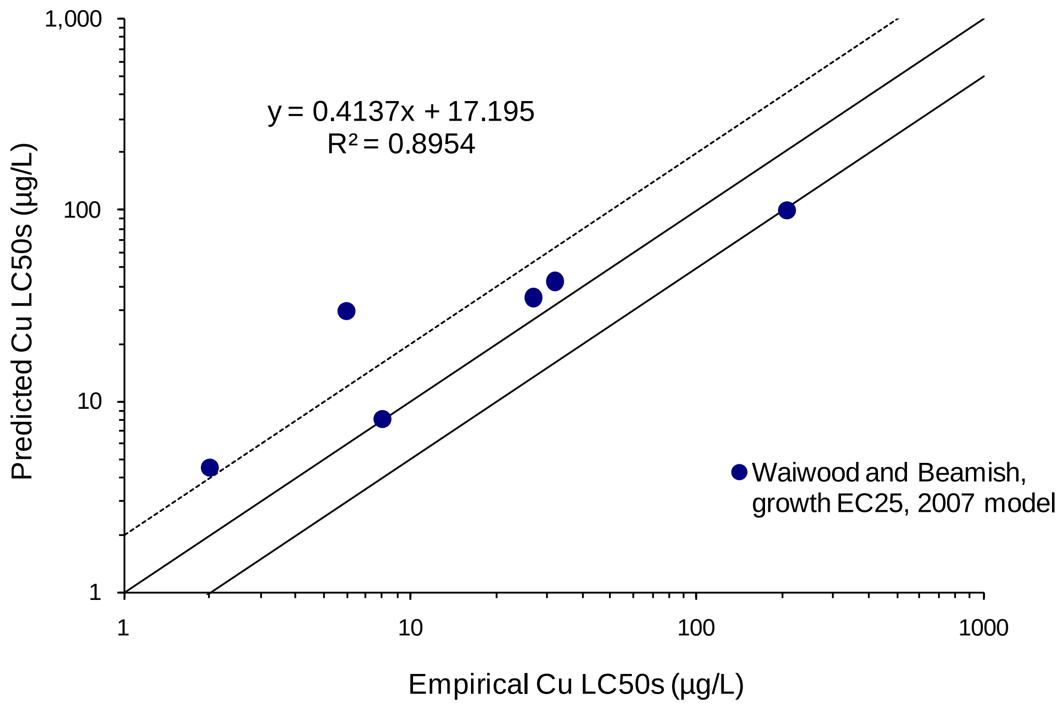




**Figure 16.** **Comparisons of empirical results of juvenile rainbow trout 30 day exposures with BLM predictions. (Top) Rainbow trout copper 30-day LC20s across pH gradient in low hardness (11-22 mg/L), water with DOC ~1.5 mg/L (Ng *et al.* 2010). Both the BLM-based and hardness-based chronic Cu criteria were protective, however, only the BLM-based criteria mimicked the direction of responses. The BLM-based criterion values were very low compared to the empirical results at pH <6.5. (Bottom) BLM compared with empirical EC25 estimates of growth (weight) reductions from 30-day growth tests that manipulated inorganic water chemistry (pH and hardness), using the 2007 BLM parameters (Waiwood and Beamish 1978).**

For these chronic tests estimating the thresholds where adverse effects just begin to occur (EC10, discussed in Appendix B), rather than the "factor of 2" prediction bands, the protectiveness or lack thereof of the chronic criterion is shown as a horizontal line extending to the left or right of each EC10 estimate. The ends of the horizontal lines show the chronic criteria for the water chemistry conditions of each test. Thus, a horizontal line extending left from each point is favorable, indicating that the chronic criterion was lower than the test EC10. For 17 of 18 tests analyzed in this way, the chronic criterion was protective for that test. The exception was one of three replicate 30-day fathead minnow tests conducted by Besser *et al.* (2005); this point is located parallel with the legend entry "Fathead minnow") in Figure 15.

The comparisons of the BLM predictions and empirical estimates with Waiwood and Beamish's tests in which they manipulated pH, hardness, and alkalinity were also surprisingly strong, considering the number of estimates needed for the water chemistry model inputs. The empirical effects estimates ranged from 2 to 206 µg/L, and the BLM predictions ranged from about 6 to 100 µg/L, with the BLM accounting for almost 90% of the observed variability (Figure 16).

Chemosensory or behavioral effects

Another type of adverse effect caused by copper is neurotoxicity, which can impair the ability of fish to complete normal migrations and prevent salmonids from migrating downstream or from homing on their natal stream (Hecht *et al*. 2007; McIntyre *et al.* 2008a; Green *et al*. 2010). Copper is neurotoxic to fish and interferes with the function of the peripheral olfactory nervous system, as well as the function of the mechanical sensory cells located on the lateral lines of fish that keep fish oriented to currents, schooling behaviors, and flight responses among other functions. However, the structure of the olfactory epithelium, lateral line epithelium, and gill epithelium all differ, leading to debates whether the BLM which was developed for the gill epithelium, functions adequately for olfactory or lateral line toxicity (Linbo *et al.* 2006; McIntyre *et al*. 2008a; Meyer and Adams 2010).

To evaluate this issue, we re-interpreted three studies: (1) destruction of lateral line hair cells on larval zebrafish following copper exposures under differing inorganic chemistry and with natural organic matter additions (Linbo *et al.* 2009); (2) olfactory inhibition in coho salmon following short-term (30 minute) exposures to copper under differing pH, hardness, alkalinity and DOC conditions (McIntyre *et al.* 2008b, a); and (3) reduced survival of olfactory inhibited copper-exposed coho salmon in staged encounters with a wild predator, cutthroat trout (McIntyre *et al.* 2012).

Our analysis of the lateral line study showed very strong correlations between predicted and empirical EC50 values for mechanical-sensory hair cell destruction. However, the patterns were very different for different water quality parameters (Figure 17). Increasing alkalinity by adding sodium bicarbonate had little effect on empirical EC50s, although predicted EC50s rose steeply with increasing alkalinity and sodium. Predictions also nearly perfected correlated with empirical results from increasing DOC, although predicted values increased more than empirical values, with a slope of 2.7. Calcium, Mg, Ca+Mg, and Na additions resulted in very high $R^2$ values, with slopes less than 1. No hair cell EC50s were lower than the BLM-based FAV. As with olfaction, these results support the value of further investigations, but suggest that the 2007

BLM acute criterion is probably protective of lateral line damage from copper, assuming that the function and physiology of lateral lines is similar across fish species (e.g. Linbo *et al*. 2009).

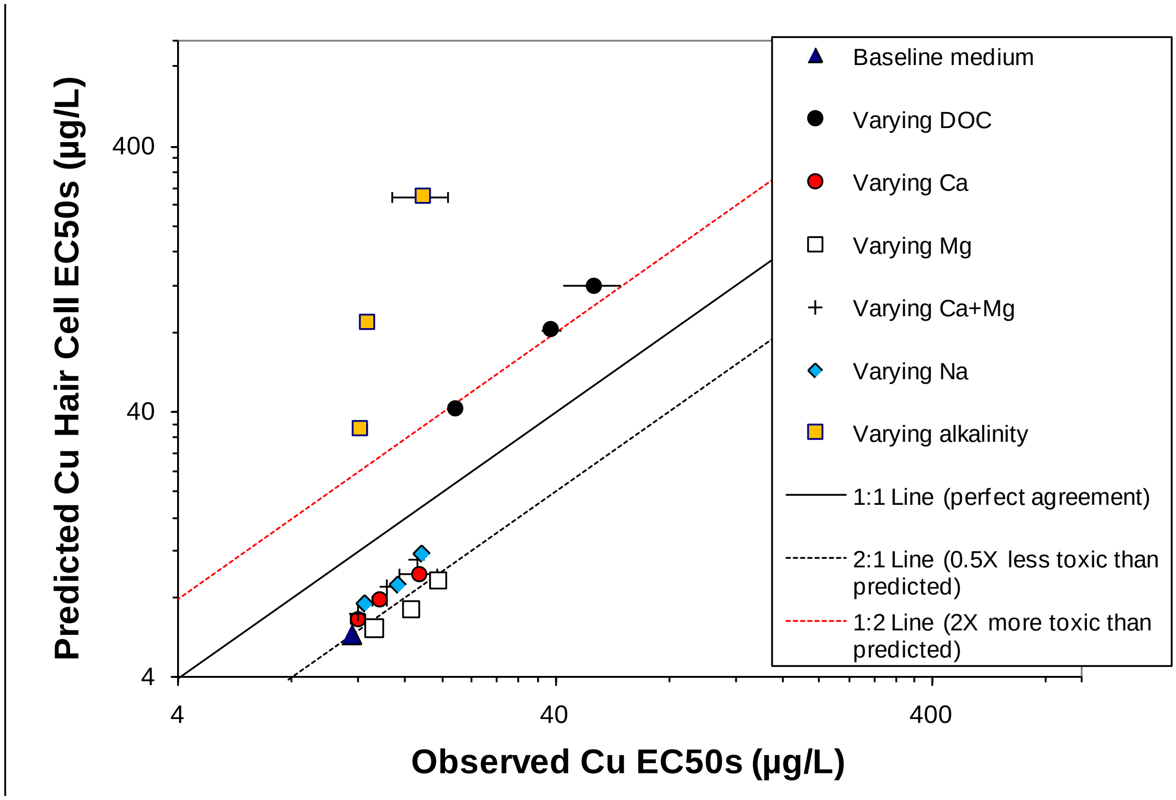
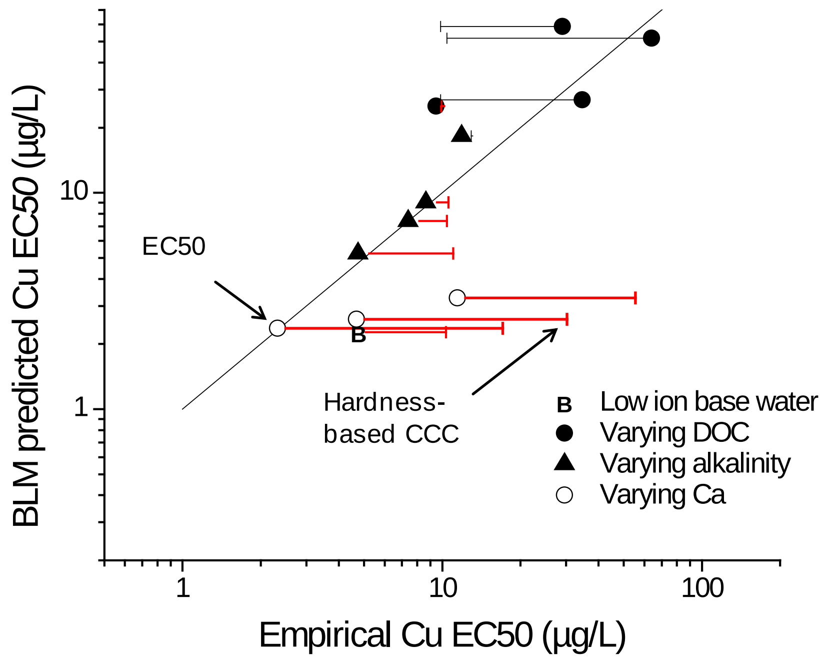



**Figure 17. Destruction of lateral line hair cells in zebra fish, empirical and BLM predicted values (data from Linbo *et al.* 2009).**

With the inhibition of olfaction in coho salmon from copper (McIntyre *et al*. 2008b, a), the results showed reasonable agreement between the BLM predicted copper concentrations causing 50% olfactory inhibition in coho salmon using the "factor of 2" rule-of-thumb for evaluating model predictions (Figure 18). However, there were differences in how well the BLM handled the different water chemistries. With varying alkalinity, predictions were nearly perfectly correlated with empirical estimates, although the slope was steeper than 1.0. With varying calcium, correlations were similarly very strong, but the slope was much shallower than 1.0. With added DOC, correlations were more scattered and weaker. These patterns suggest further evaluations would be appropriate, yet the overall pattern of model predictions seemed favorable. Further, none of the empirical results were lower than the BLM based FAV, indicating that the acute BLM-based criteria would likely be protective from olfactory inhibition in coho salmon caused by copper.

Other tests from the same laboratory suggest that other salmon species have similar responses to copper, and thus these results with coho salmon are considered to be relevant to other salmonid species including Chinook salmon and steelhead (Baldwin *et al.* 2010).



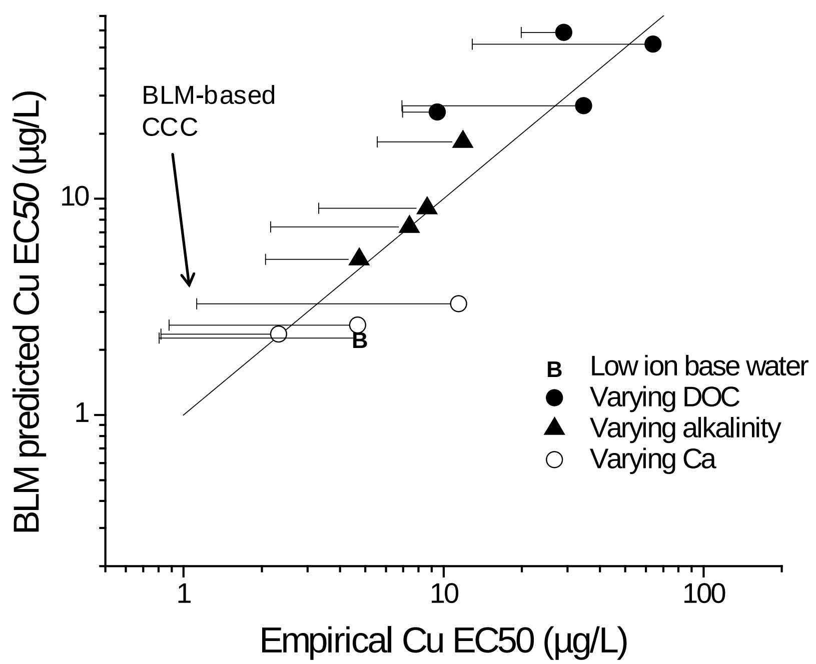




**Figure 18. Inhibition of olfaction in coho salmon from copper following 30 minute exposures, observed and BLM predicted values (data from McIntyre *et al.* 2008a,b). Horizontal error bars leading to the left of the symbols indicate the margin of safety between the EC50s and the Final Acute Value (FAV); error bars, emphasized in red, leading to the right of the symbols indicate the degree of non-protectiveness of**

**the FAV for that test value. Both the EPA (1992) hardness-based criteria (top) and the EPA (2007) BLM-based criteria (bottom) are shown. Solid diagonal line is the 1:1 line of perfect agreement.**

Prey fishes have a behavioral alarm response to olfactory predation cues that provides a survival benefit when under attack (Mirza and Chivers 2001, 2003). It logically follows that that survival benefit could be compromised if a pollutant such as copper disrupts the behavioral alarm response (Scott and Sloman 2004; Sandahl *et al.* 2007). This presumed reduction in survival of copper-exposed prey fish in predator-prey encounters was demonstrated by McIntyre *et al*. (McIntyre *et al*. 2012). They found that copper exposure altered prey (juvenile coho salmon) response to olfactory predation cues in the presence of predators (adult cutthroat trout), and that this "info-disruption" reduced prey fish survival. Copper exposure made prey easier for predators to detect and capture. The primary impact of copper on predator-prey dynamics in her study was faster prey detection, shown as faster time to attack and time to capture. Copper-exposed prey were more active than control prey during predation trials. For visual predators of juvenile fishes (e.g. salmonids, birds, otters), prey activity is a critical determinant of detection by predators (McIntyre *et al*. 2012).

McIntyre *et al.* (2012) conducted the predator-prey interactions in relation to copper exposures in two trials. The first trial tested encounters between copper-exposed prey and non-copper exposed predators. The result was a graded decline in prey survival times over copper concentrations ranging from about 0.2 µg/L in controls to 20 µg/L, in a freshwell water with very low organic carbon concentrations (≤ 0.25 mg/L DOC). No threshold of response was found; reduced survival times were observed at the lowest concentration tested, 5 µg/L (Figure 19). In McIntyre *et al*.'s second trial, both the prey and predators were exposed to copper in a subset of the trials, which did not markedly improve the ability of copper-exposed prey to evade the copper-exposed predator (Figure 19). This result was attributed to the fact that cutthroat trout are visual predators and copper exposure is not expected to affect their eyesight.

For the water chemistry conditions of the exposure waters from McIntyre *et al*.'s (2012) study, the 2007 BLM-based acute copper criterion is only slightly higher than the average copper concentrations measured in the control waters. In contrast, the acute hardness-based criterion (the Idaho and NTR criterion, EPA 1992) values for a hardness of 56 mg/L $CaCO_3$ is about 10 µg/L, well into the range of decreased prey survival. In this study, no minimum threshold below which copper-exposures have no or little effect on predator-prey interactions was obtained, so no strong conclusions about "safe" copper concentrations could be made. However, adverse effects at the BLM-based criterion concentration would seem unlikely since it was close to the control concentration.

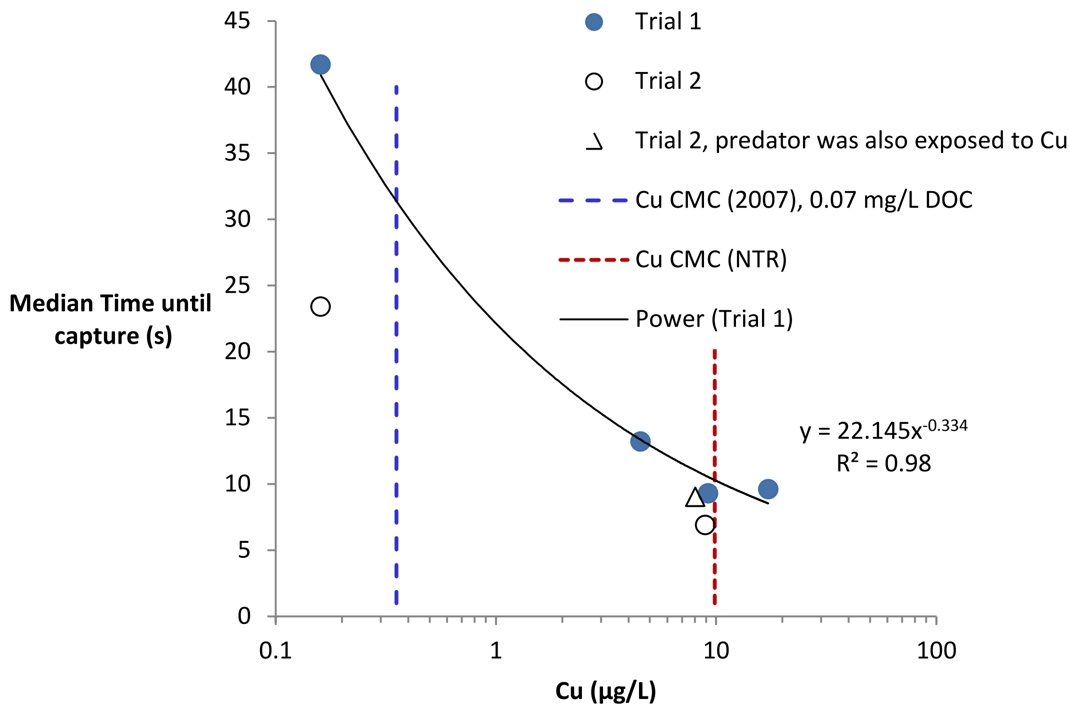



**Figure 19. Survival times of juvenile coho salmon prey before being eaten by an adult cutthroat trout predator, as a function of copper exposure concentrations (3-hr durations) prior to the predation experiments. Unlike natural environments, in the circular tank there was no place to hide or escape, so all prey were eventually eaten regardless of copper exposure. In a natural environment with hiding places and escape routes, the prey that evaded capture longest in would presumably have a better chance of ultimate escape. Data from McIntyre *et al*. (2012).**

Chinook salmon and rainbow trout have also been shown to be very sensitive to avoidance, and the loss of olfactory responses at copper concentrations less than the hardness-based copper criteria (Hansen *et al*. 1999a). Hansen *et al.* (1999a) found that behavioral avoidance of copper in soft water differed greatly between rainbow trout and chinook salmon. Chinook salmon avoided at least 0.7 µg Cu/L, whereas rainbow trout avoided at least 1.6 µg Cu/L in water with low DOC and a hardness of 25 mg/L in 20-minute exposures. These lowest observed effect concentrations (LOECs) were considerably lower than the hardness-based acute criterion for the test waters, 4.6 µg/L. In contrast, the estimated BLM-based allowable acute criterion concentrations for the test conditions were below or close to these LOECs for behavioral avoidance, about 0.4 to 1.1 µg/L. The BLM-based criterion concentrations are estimated from other studies because Hansen *et al.* (1999a) did not measure all the necessary BLM parameters. The major ion data were taken from Marr *et al.* (1996), a nearly contemporaneous study at the same lab, with the same targeted blend of well and reverse osmosis (RO) deionized water, and most of the same investigators. Hansen *et al*. (2002b) reported DOC near 0.1 mg/L in further tests that used similar softwater blends in the same lab with about the same proportions of well and RO water as did the Hansen *et al.* (1999a) study. This yielded a BLM-based acute criterion concentration of about of 0.4 µg/L for Hansen *et al.*'s (1999a) behavioral tests.

Hansen *et al.*'s (1999a) behavioral avoidance results were also reinterpreted by Meyer and Adams (2010) in the context of whether the BLM-based or hardness-based copper aquatic life criteria would be protective. Meyer and Adams' (2010) reinterpretation differed from that used here in that instead of comparing LOECs, they developed regression based estimates of 20% and 50% increases in behavioral avoidance. This avoided a limitation of using statistics like LOECs when comparing effects across studies. The LOECs simply reflect the lowest concentration with a response that with 95% confidence was statistically different from the controls, but tell nothing about the size of the effect that was different, for instance whether a 5% or 50% response was "different." Using Meyer and Adams' (2010) estimate of 20% avoidance effect as a threshold of appreciable avoidance (EC20) of about 0.84 and 0.91 µg/L would also indicate that the BLM-based acute criterion was close to or below the concentration causing olfactory-related impairment. Meyer and Adams' (2010) argued that while less than 20% olfactory-impairment might be considered important for some species of concern, the variability associated with behavioral testing would make a smaller effect percentile of questionable meaning. As with the LOECs, the EC20s and even EC50s were lower than the hardness-based acute criterion.

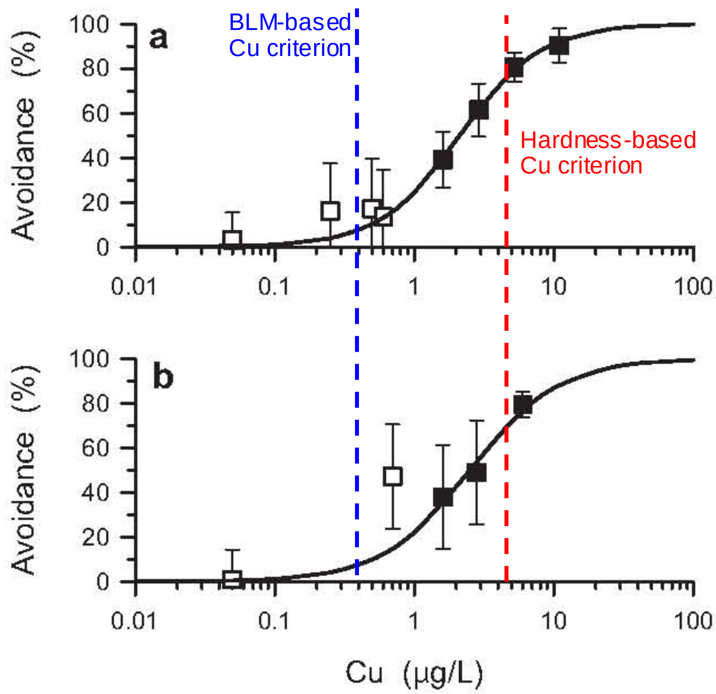



**Figure 20. Avoidance of copper by rainbow trout (a) and Chinook salmon (b) in softwater with low DOC in relation to 2007 BLM-based (blue dashed line) or 1992 hardness-based acute copper criteria (red dashed line). Open or closed symbols indicate values lower or above the copper detection limit used. Base figure was taken from Meyer and Adams (2010) using original data from Hansen *et al.* (1999a).**

In summary, the available information indicates that the BLM-based copper acute criterion would likely be protective against neurological damage or behavioral impairment resulting from short-term (<< 1 day) copper exposures. The older hardness-based copper acute criterion (the Idaho criterion under consultation) would be considerably underprotective for chemoreception, behavioral avoidance, predator avoidance, and survival from predators.

Field and experimental ecosystem studies

Our reviews up to this point have relied on carefully controlled laboratory studies. For, our final evaluation of the protectiveness of the BLM-based chronic criterion we consider how the BLM is likely to perform under more realistic field conditions. Field validation of laboratory or mathematical models through field surveys or ecosystem manipulations may represent some ideal for ecotoxicology, but it is an elusive ideal to achieve. This is in part due to the scale of effort needed to conduct a rigorous study and ethical constraints on manipulating natural ecosystems, but also because field studies tend to be specific to the locale, season, etc. studied and may be difficult to extrapolate to other ecosystems. Yet some ambitious experimental manipulations of whole streams by adding copper have been completed that are very relevant, as well as small scale stream tests constructed streamside. NMFS located and re-interpreted three high quality field experiments and a small scale microcosm test relevant to our evaluation of the BLM-based copper criteria.

The most ambitious study we reviewed was an intensive, multi-year study of Shayler Run, an Ohio stream. The study took place prior to and during 33 months of copper additions, and during recovery from the copper additions. The Shayler Run drainage basins is underlain by limestone and other carbonate rocks and received sewage input from a small town upstream of the study area. Thus the hardness, alkalinity, pH, and DOC were all fairly high in Shayler Run. Measured direct effects on fish were death, avoidance, and restricted spawning. Chronic tests were done on-site at Shayler Run with stream species and fathead minnow(Geckler *et al*. 1976). The stream and test waters were well characterized chemically, and all necessary BLM data except for sulfate could be pieced together reasonably well from the study report. Sulfate was well estimated by regression from chloride ($R^2$ 0.94) from USGS data for station 03247400, Shayler Run near Perintown OH.

While well conducted field studies such as the Shalyer Run study may identify adverse effects thresholds with some precision, a difficult question is what stream chemistry conditions should be attributed to the observed effects? The effects probably resulted from long-term exposures to copper, but the stream characteristics such as pH, DOC, and thus the water quality criteria vary over the course of the experiment in a manipulated system like Shayler Run. Presumably adverse effects have resulted from some critical condition where for example, DOC was low and thus copper more toxic, but this is an educated guess that cannot be demonstrated or falsified from the available data.

For this review, we considered the approximate range of apparently "safe" copper concentrations for the stream ecosystem to be about 29-47 µg/L, using ACRs they determined with streamside acute and chronic tests (Geckler *et al*. 1976, p. 170). This upper range could be optimistic, since they noticed that the chronic tests underestimated the instream toxicity by about two times because only the effects of copper on survival, growth, and reproduction were measured; avoidance was not measured, and it was a significant effect in the stream. For instance, bluntnose minnows only spawned where total copper concentrations ranged from 35-77 µg/L (minimum they could access), but still a seven-fold reduction in fry occurred. With these considerations, for an effect benchmark for the overall study, we estimated an approximate

NOEC for the streamside chronic tests, of 29 µg/L. This value is about two times lower than the lower range of clearly adverse effects and might account for the unmeasured avoidance effects on minnow populations.

Figure 21 shows this benchmark in comparison to a monthly time series of BLM-chronic criteria values during the study. Since copper was held nearly constant in the streamside tests, but characteristics affecting toxicity varied, the conditions when copper would have been most toxic are shown as the dips in Figure 21, which occurred when DOC and pH were relatively low. These conditions seem more important than the peaks in graph when conditions were least toxic or some average condition. At these times the BLM-chronic criterion was lower than the selected benchmark. This suggests that the BLM-CCC probably would be protective for the Shayler Run situations.

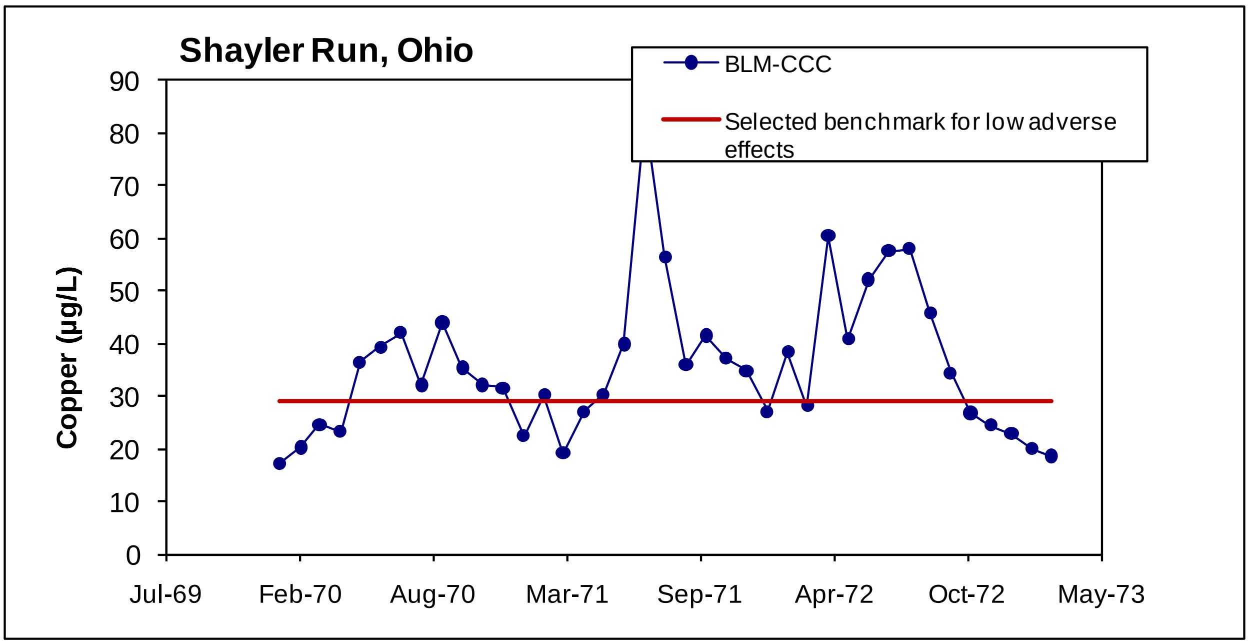



**Figure 21. Seasonal patterns in BLM-chronic criterion in Shalyer Run field study, in comparison to an estimated benchmark of adverse effects based on streamside tests. Average hardnesses 180 (126-220), DOC 6 (4-12 mg/L), pH 8,1 (7.75-8.3). Original data from Geckler *et al.* (1976).**

The second experimental stream study we examined was smaller in scale but was a western montane stream in the Sierra Nevada with soft water in a granitic drainage basin and thus particularly relevant to Idaho mountain streams. Convict Creek was dosed with copper for one year in different reaches with average copper concentrations of about 2.5, 5, 10, and 15 µg/L. Measured effects included stream ecosystem structural measures such macroinvertebrate community diversity and stream ecosystem functional measures such as stream metabolism energy production. (Kuwabara *et al.* 1984; Leland and Carter 1984, 1985; Leland *et al*. 1989). All needed BLM inputs except DOC were reported. We estimated a range of plausible DOC values for Convict Creek from a study of DOC in high lakes in the Sierra Nevada that were likely similar to that expected for the Convict Creek drainage basin, with a mean (range) of 1.9

(0.9 – 2.5) mg/L (Brooks *et al.* 2005; Daniel Dawson, Sierra Nevada Aquatic Research Laboratory, personal communication).

We interpret these data as follows. The minimum value from the Sierra high lakes seems prudent to use as the baseflow value that lasts most of the year. The maximum measured DOC value would be expected to occur during runoff high flow, which during the study water year appeared to have occurred in early July when calcium dropped. This assumption follows from our review of stream chemistry seasonal patterns discussed in the following sections.

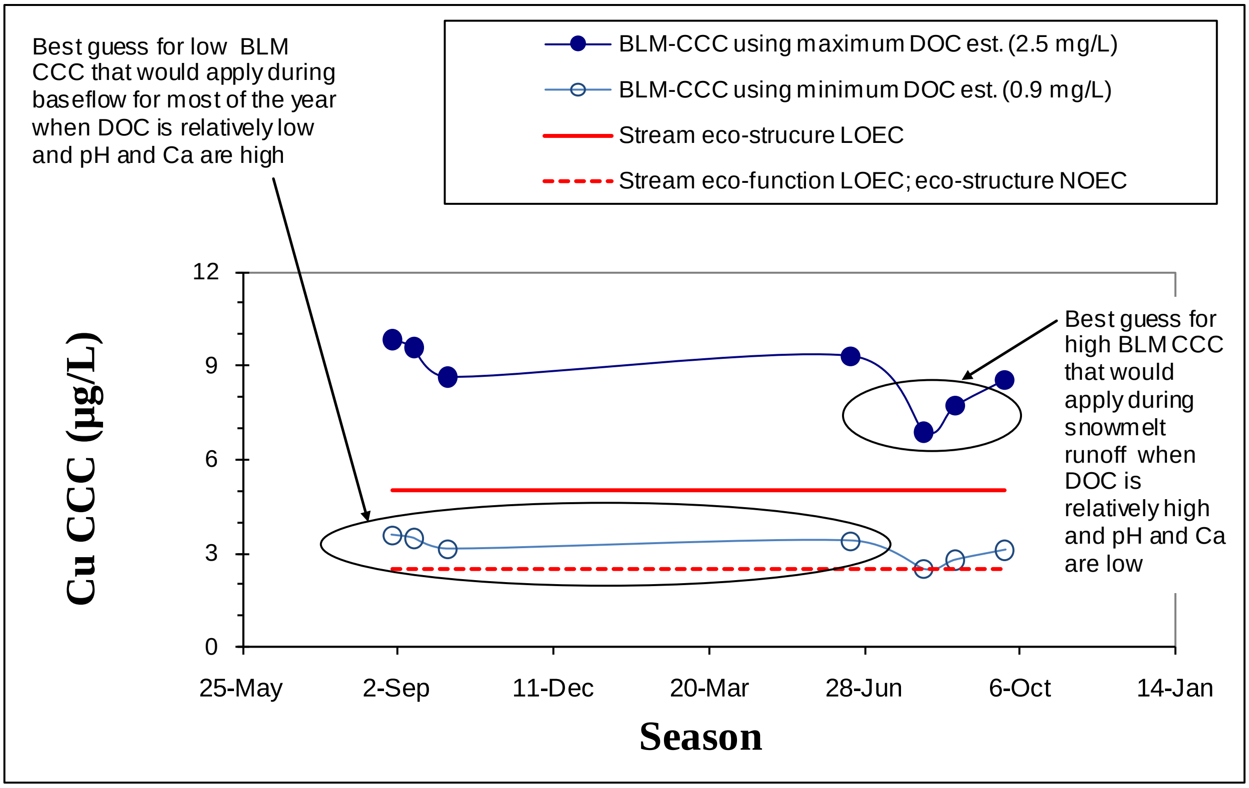



**Figure 22. Stream ecosystem alteration following copper additions in Convict Creek (Sierra Nevada), California vs. estimated BLM-based chronic copper criteria. Original data from Kuwabara *et al.* 1984; Leland *et al*. (1985,1989). LOEC for adverse effects to ecological function (energy production) LOEC for adverse effects ecological structure (invertebrate diversity).**

Measureable shifts in ecosystem function occurred in even the lowest copper treatment (decreased gross primary productivity and decreased respiration) of 2.5 µg/L. No effects of copper to the macroinvertebrate community were detected at 2.5 µg/L, but declines in population density of species representing all major orders (Ephemeroptera, Plecoptera, Coleoptera, Trichoptera, and Diptera) occurred at 5 µg/L copper and higher (Leland and Carter 1985; Leland *et al.* 1989). The BLM-CCC for the test conditions would probably be low enough to be below the adverse 5 µg/L treatment most of the time, and would be close to the 2.5 µg/L treatment that caused no apparent adverse effects to the macroinvertebrate community. In contrast, the Idaho/NTR-hardness based chronic criterion for these conditions would be between 6 and 8 µg/L. The BLM-CCC would not have been low enough to prevent stream metabolism depression (Figure 22). However, the depressed primary production via algae was not obviously reflected in secondary energy production in the macroinvertebrate community. Thus, the BLM-

CCC does not appear to be low enough to prevent measureable effects, but so long as secondary production from the macroinvertebrate community remained intact, the reduction in primary productivity is unlikely to carry over to salmonids.

Next we considered a rigorous, ecologically relevant streamside study of copper effects on macroinvertebrates. Natural assemblages of aquatic macroinvertebrates were established on substrate-filled trays which were then transferred to outdoor stream mesocosms adjacent to the New River, Virginia. Exposure of these communities to low levels of copper and zinc (target concentration = 12 µg/L) significantly reduced the number of taxa, number of individuals, and abundance of most dominant taxa within 4 days (Clements *et al*. 1988; Clements *et al*. 1989). Zinc concentrations on the order of 12 µg/L were unlikely sufficient to contribute to the reductions (Clements *et al*. 2000). A 42-day exposure almost completely extirpated mayfly communities as well as the sensitive *Tanytarsini* midges. In the second experiment, after 10 days, 6 µg/L copper was sufficient to eliminate 50% of the total macroinvertebrates (i.e, community EC50 of 6 µg/L) and even 2-3 µg/L copper were sufficient to cause a significant decline in macroinvertebrate communities.

As is commonly the case, only hardness, alkalinity, pH, conductivity, and temperature were measured as part of the study, which required NMFS to make estimates of other water chemistry parameters needed to run the BLM. The study was conducted close to a USGS monitoring station with sufficient data to make reasonable estimates of likely BLM-input for the experimental conditions (USGS station 0317500, New River at Glen Lyn, Virginia). During 1987 to 1988 when the biological tests were conducted major ion, conductivity, and pH data were collected by the USGS at the site, but not DOC. DOC data were collected during 1997. The inorganic parameters at the location were similar during the summers of 1987, 1988, and 1997. DOC concentrations during 1997 were not highly variable (1.1 to 2.4 mg/L). Assuming that DOC concentrations in the New River at Glen Lyn in the summer of 1997 were representative of DOC concentrations in the summers of 1987 and 1988, estimated BLM criteria for the macroinvertebrate experiments can be compared to the adverse effects values (Figure 23).

The results of the comparison shows that macroinvertebrate communities are very sensitive to copper with declines in abundance occurring as low as 2 -3 µg/L, which is about the same as the minimum BLM CCC estimate during summer at the study site. Severe effects occurred by 6 µg/L, and a 12 µg/L treatment for 42-days almost completely extirpated mayflies. In 1997, the BLM-CCC rose in late summer because of a half unit pH rise to about 8.3, which was similar to the fairly high initial pH values in the mesocosms. Still, pH variations of half unit or more over the course of a day are not uncommon in streams, even streams that are fairly oligotrophic (e.g., Nimick *et al*. 2011; Balistrieri *et al*. 2012), so perhaps the high BLM CCC values resulting from the high pH values should not get undue emphasis, and the BLM CCC calculated for the lower pH values should also be considered representative of the experimental conditions (Sep. 1996 to June 1997 values in Figure 23). These values are generally below the copper concentrations causing severe adverse effects in the New River experiments.

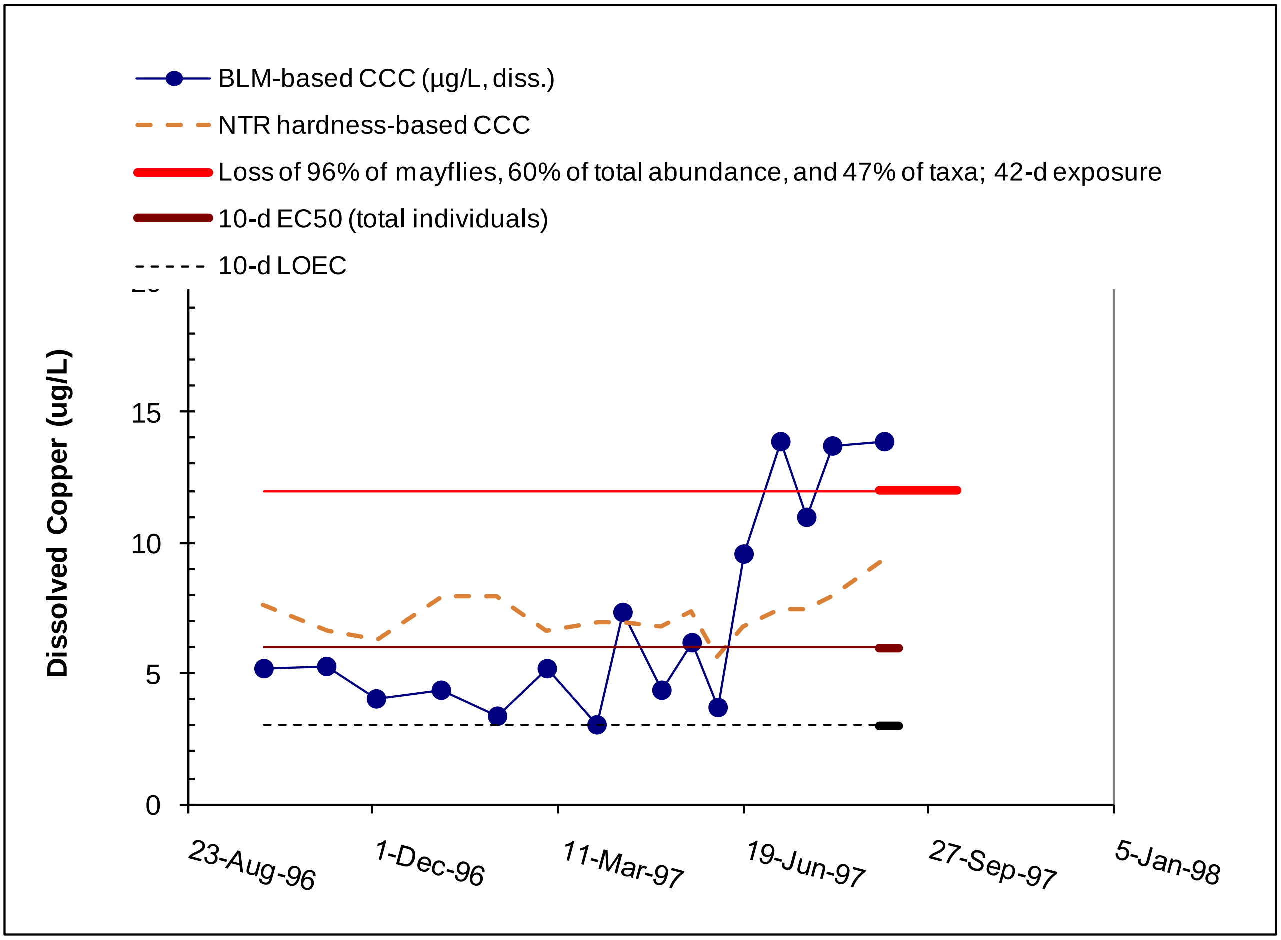



**Figure 23. Macroinvertebrate community effects concentrations and the chronic BLM-based copper criteria: New River at Glen Lyn, VA. Effects concentrations from Clements *et al.* (1988, 1989); BLM inputs estimated from USGS station 0317500 (New River at Glyn Lyn, Virginia). Hardness was 48-75 mg/L during the tests and DOC estimated in the ranges of 1.1 to 2.4 mg/L. Horizontal lines indicate different effects concentrations from the tests; the thick "handles" on the right ends of the lines correspond to the time of year that the tests were actually carried out.**

The best interpretation of the analyses summarized in Figure 23 may be that macroinvertebrate communities are very sensitive to copper, and that community richness may decline at concentrations lower than the BLM-based CCC. At best, there is a narrow range between allowable chronic criteria concentrations and pronounced adverse effects. This conclusion seems consistent with a recent field study relating mixtures of copper, cadmium, and zinc in Colorado streams to benthic community alterations using a modification of the BLM (Schmidt *et al*. 2010). Declines in diversity and abundance of the Cu-sensitive Heptageniid criteria co-occurred with Cu concentrations less than the BLM-based criteria.

The final experimental ecosystem study considered here used pond microcosms that were dosed with copper ranging from 4 to 420 µg/L (Hedtke 1984). In contrast to the previous ecosystem studies, these microcosms were much smaller, which might make them less realistic but allowed a more experimental control and replication of treatments. Most required BLM inputs were measured and reported in the article; we estimated pH and K from the water recipe.

Natural pond sediments were collected and allowed to develop in the microcosms for 30 days before the microcosms were dosed with copper for 32 weeks. A variety of ecosystem functional and structural endpoints were measured. The most sensitive results were the loss of most snails and most cladocerans in the 8.8 µg/L LOEC treatment. Additionally, at the 8.8 µg/L LOEC at 30 weeks, gross primary production, DOC production, and the filamentous green alga *Vaucheria* were significantly impacted. More severe effects developed at higher copper treatments.

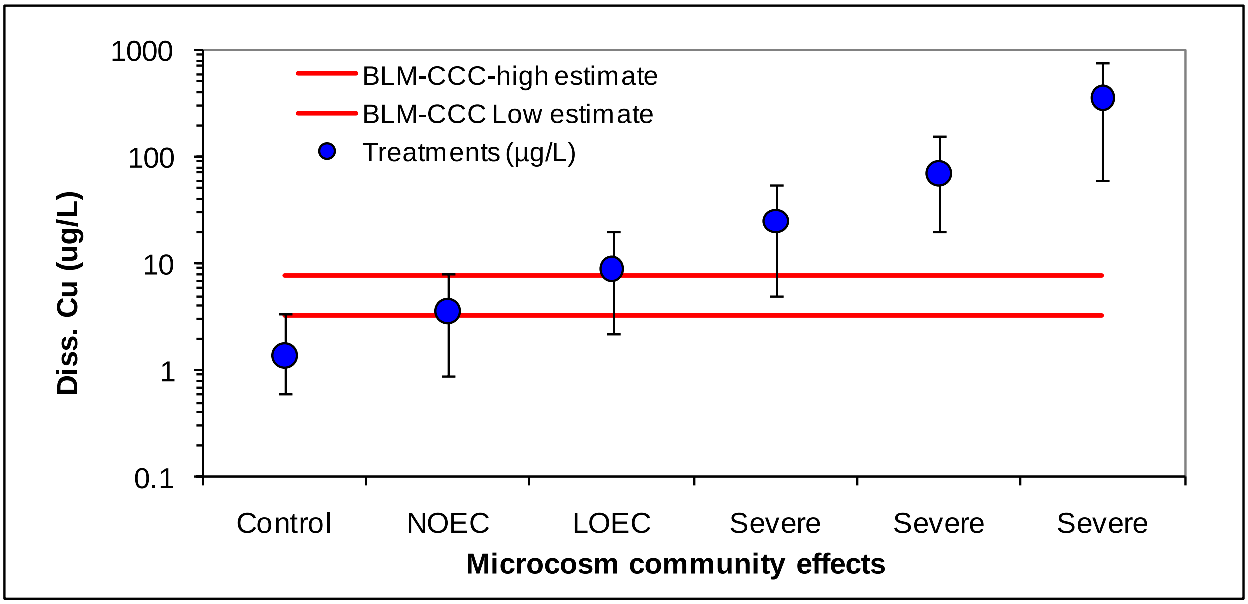



**Figure 24. Effects of copper in pond microcosm tests compared with BLM CCC. The values overlapped the no- and low observed effects concentrations, giving equivocal support for the protectiveness of the CCC. Original data from Hedtke (1984).**

The comparisons between the no- and lowest-effect concentrations and the BLM-CCC overlapped but leaned toward being favorable to the BLM CCC. The CCC was determined over the DOC range of 0.7 to 1.8 mg/L in the microcosm inflows, which is reflected in the BLM-CCC high and low estimates in Figure 24. The higher estimate of the BLM-CCC approached the LOEC although the lower estimate was below the LOEC and happened to match the NOEC. Presuming that adverse effects likely developed at times when copper was most bioavailable, i.e., when DOC was lowest, then the lower BLM estimate would be given more emphasis. However, as with the Convict Creek and New River experiments, these results emphasize the fine line between probable protectiveness of the BLM-CCC and water conditions in which copper causes much more severe effects to aquatic insects and other benthic macroinvertebrates. Still, the BLM-based 2007 chronic criterion is clearly more appropriate and protective than the Idaho/NTR hardness-based chronic criterion, which was 19 µg/L for the test conditions. In the microcosm treatment that was the closest match to the Idaho/NTR chronic criterion (25 µg/L) all measured ecosystem components except large oligochaetes were significantly impacted.

## Conclusions and Recommendations

Overall, NMFS' analyses of the performance of the 2007 BLM-based copper criteria tended to be favorable. With many independent data sets that tested a diverse assortment of aquatic organisms and endpoints across a wide variety of natural and laboratory waters, the 2007 copper BLM toxicity predictions were invariably at least correlated with empirical toxicity observations, which is considerably better than can be said for the Idaho/NTR hardness-based copper criteria. The analyses were most equivocal for the field experiments with aquatic insect and other benthic macroinvertebrates, yet even for these analyses the 2007 BLM-based criteria performance was clearly superior to that of the hardness-based criteria. Because listed juvenile steelhead and salmon are feeding generalists, so long as the overall benthic community remains diverse and abundant, steelhead or salmon populations could likely withstand minor losses of benthic diversity, which was evaluated under the heading "Salmonid Prey Items" in section 2.4.1.12 of the main body of this Opinion. Thus these adverse effects would not rise to the level of jeopardy or adverse modification of critical habitats.

Implementation Considerations

Our mostly favorable evaluation of the 2007 BLM-based criteria leads to the following logical problem: When compared to the old hardness-based criteria, the data requirements of the BLM-based criteria are novel and extensive. Could the 2007 BLM-based criteria be reasonably implemented as an alternative to the Idaho/NTR hardness-based criteria? Can the BLM-based criteria be safely estimated for different water body types even if measurements of all the BLM inputs are not available?

Some efforts have been made to develop regional estimates of BLM input parameters that could be used when measured data are unavailable. For instance, Carleton (2008) describes a proof-of-concept approach where "*one possible way to deal with such missing information is to develop conservative (realistic but protective) default values for these various model inputs… Given that ambient surface water chemistry reflects, among other things, the influences of local soil types and land uses, it may make sense that any such defaults be developed on some kind of regional or local basis*." The EPA (2012) gives further detail of such an approach, and gives potential interim values that could be used on an ecoregional geographic basis. Within the range of anadromous fishes in Idaho, waters and their drainage basins can be grouped according to their expected water chemistry characteristics. We compiled datasets of BLM input parameters for representative waters and examined for seasonal patterns of "critical conditions" which are the annual worst case conditions for that water body (i.e., when copper would be most toxic). Because of the paucity of streams with sufficient high quality chemistry data, some of the waters we used are located outside the range of anadromous fish, but we judged them likely to have characteristics similar to waters occupied by salmonids. Most data were obtained from the USGS National Water Information System database,[12] and were limited to data collected from 1994 or later. The 1994 cutoff was selected because older DOC data were consistently higher than more recent data from the same sites, suggesting there may systematic sample contamination from bottles, filters, or other sources (e.g., Yoro *et al.* 1999).

The first stream we considered was Panther Creek, one of the major tributaries to the Salmon River, Idaho. Since the early 1990s because of copper contamination from mining

[12] http://nwis.waterdata.usgs.gov

activities that led to the loss of Chinook salmon and steelhead populations, Panther Creek has been the focus of many studies, litigation, and restoration efforts (e.g., Mebane 1994, 2002; EcoMetrix 2007; EPA 2008). Water chemistry was measured in detail in Panther Creek during 1993-1994; more recent data are unfortunately inadequate for the BLM.

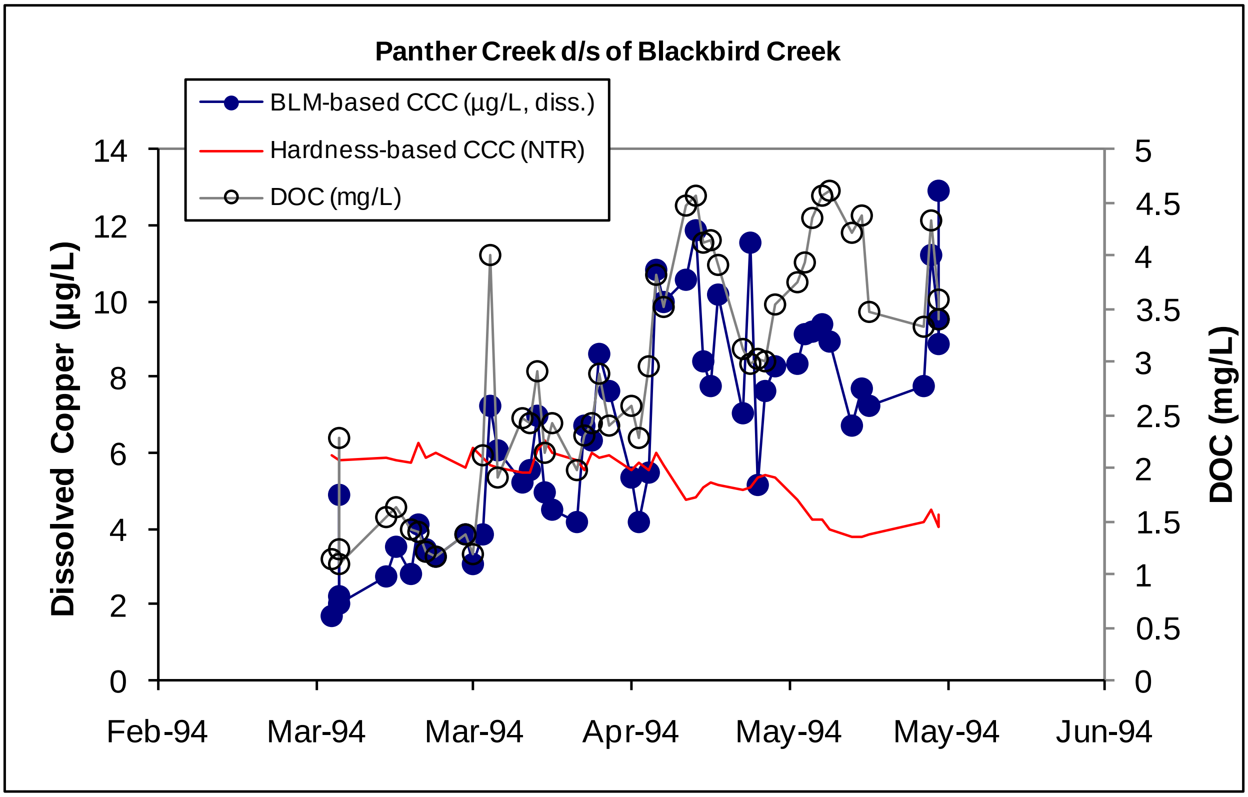



**Figure 25. The BLM-CCC in relation to DOC and the hardness-based CCC during the spring snowmelt and runoff in Panther Creek, Idaho. When sampling began in late winter baseflow conditions before the snowmelt began, DOC and the BLM-CCC were at their minimums and steadily rose as the runoff progressed. The hardness-based CCC shifted in the opposite direction. Thus the BLM- and hardness-based criteria give opposite indications of when copper would be most bioavailable and at risk. Original data from (Maest *et al.* 1995).**

The BLM-criteria were clearly driven by the DOC concentration of the water, since the two data series shift in nearly perfect unison (Figure 25). In contrast, hardness had little influence on the BLM criteria, with the hardness-based criteria dropping as the BLM-criteria increased. Thus the hardness-based criteria are telling us that the critical conditions for copper toxicity are at the end of the sampling period in late May, and that the lowest risk occurs at base flows. The BLM-criteria tell us just the opposite. Based on our previous review, it appears that the hardness-based criteria are giving misleading information and are completely wrong in their indications of relative risk to aquatic life from elevated copper concentrations.

Very similar patterns of the BLM- and the hardness-based CCC were apparent across datasets that we considered representative of drainage basins occurring within the range of listed salmon and steelhead in Idaho. The Teton River, Idaho (Figure 26, top) has geomorphic and water chemistry similarities to the Lemhi and Pahsimeroi Rivers, tributaries to the Salmon River, Idaho. In the Teton River, DOC clearly drove the BLM-criteria values, since the two data series track so closely together. (This tendency occurred in all of the data sets analyzed here as well as many other USGS river datasets reviewed but not presented, however the plots get cluttered and so DOC is omitted from some.) Opposite from the misleading information provided by the hardness-criterion, the critical conditions during which the BLM-criteria were near their annual minima again occurred during base flow conditions in fall or winter, and lowest risks for any given copper concentration occurred during spring snowmelt (Figure 26, top).

The Clark Fork River, Montana, has water chemistries that are probably roughly similar to those of the middle Salmon River between the confluences of the Lemhi River in the town of Salmon at river mile 260 and the Middle Fork near river mile 200. The Clark Fork has been the subject of much research and ecological risk assessment regarding copper risks to aquatic life, and so it is surprising that only one water year of BLM-quality water chemistry data was available (Figure 26, middle). Here again, the plots of BLM- and hardness-based criteria look like mirrored opposites, with the BLM indicating that copper would be most toxic during base flow in winter and least toxic in April to June (Figure 26, middle).

The Snake River, as it flows out of Yellowstone National Park in Wyoming near the Idaho border, has moderately low hardness and low DOC. These conditions probably make this location on the Snake River a reasonable surrogate for the upper Salmon River, upstream of the Pahsimeroi confluence near Salmon River mile 305 (Minshall *et al*. 1992). The Snake River at this location has one of the richest water quality datasets in the region, with comprehensive monthly sampling from 1993 until the USGS discontinued monitoring the site in 2004. Through the seasons, the BLM- and hardness-based criteria vary in nearly regular cycles that look almost like two sine waves that are out of phase. Again, the peaks in the BLM-criteria when copper is at its lowest risk correspond to the dips in the hardness-criteria, with their misleading risk indications (Figure 26, bottom).

The next panel of plots shows a very different situation of risk patterns for very soft waters and low organic content or low pH (Figure 25). The North Fork of the Coeur d'Alene River (NFCDA) is a comparatively well monitored stream in northern Idaho with very dilute water chemistries. The NFCDA has some similar characteristics to the upper Clearwater River basin streams and probably some of the Salmon River basin streams that are located in granitic geology with very dilute waters. In the NFCDA, the BLM-criteria are consistently much lower than the hardness-based criteria (Figure 27, top). The hardness-based criteria do not vary in these low hardness waters because the criteria-equations require that when the actual water hardness is less than 25 mg/L, the criteria shall be calculated using a hardness of 25 mg/L rather than the actual water hardness (EPA 1992).

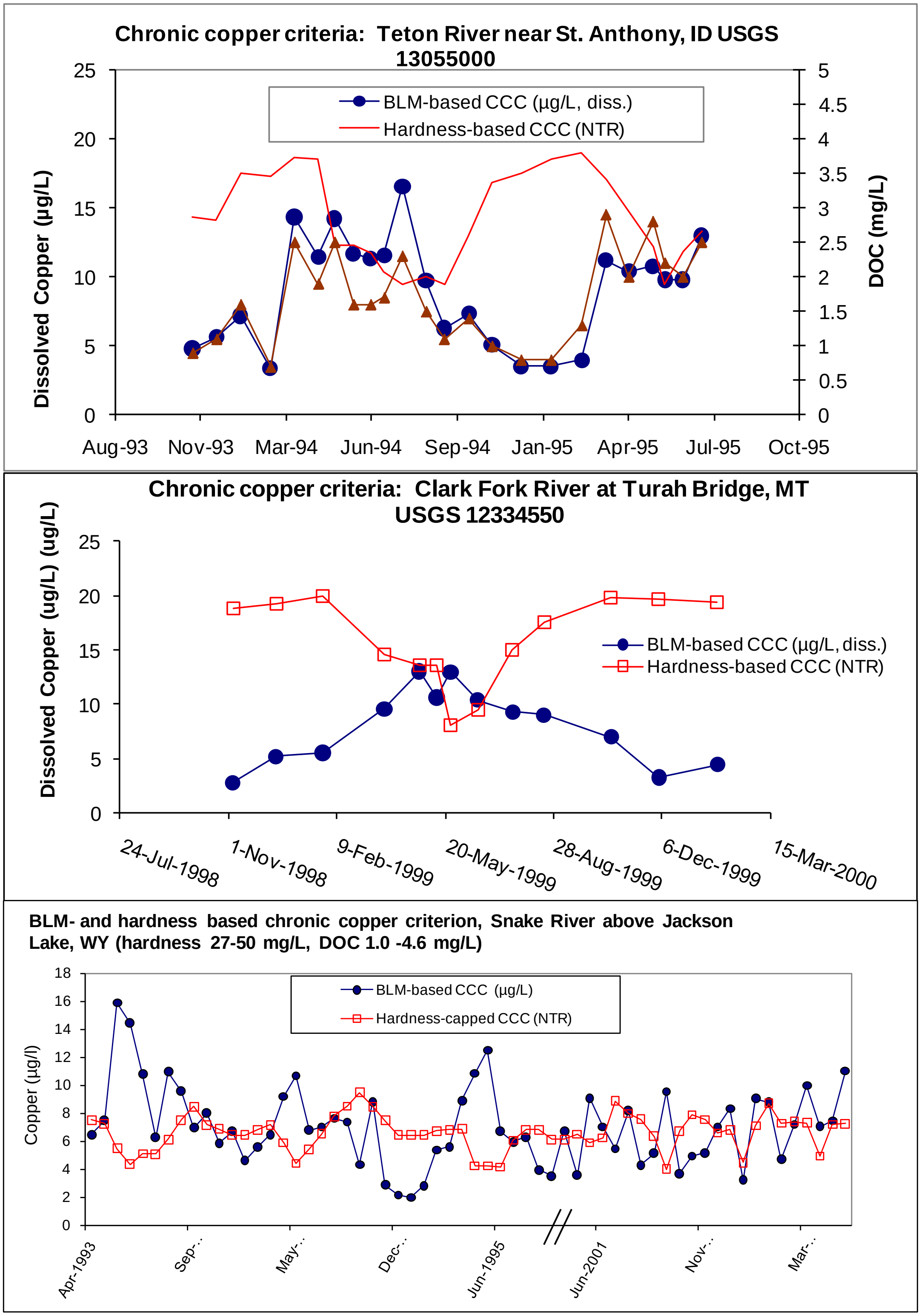



**Figure 26.** **Seasonal patterns in BLM-CCC and hardness-based CCC in streams with a strong spring snowmelt influence, and moderately-hard to softwater chemistry: the Teton River, ID; the Clark Fork River, MT; and the Snake River as it leaves Yellowstone National Park, WY. A consistent asynchronous pattern is apparent where the BLM and hardness-criteria shift in opposition to one another.**

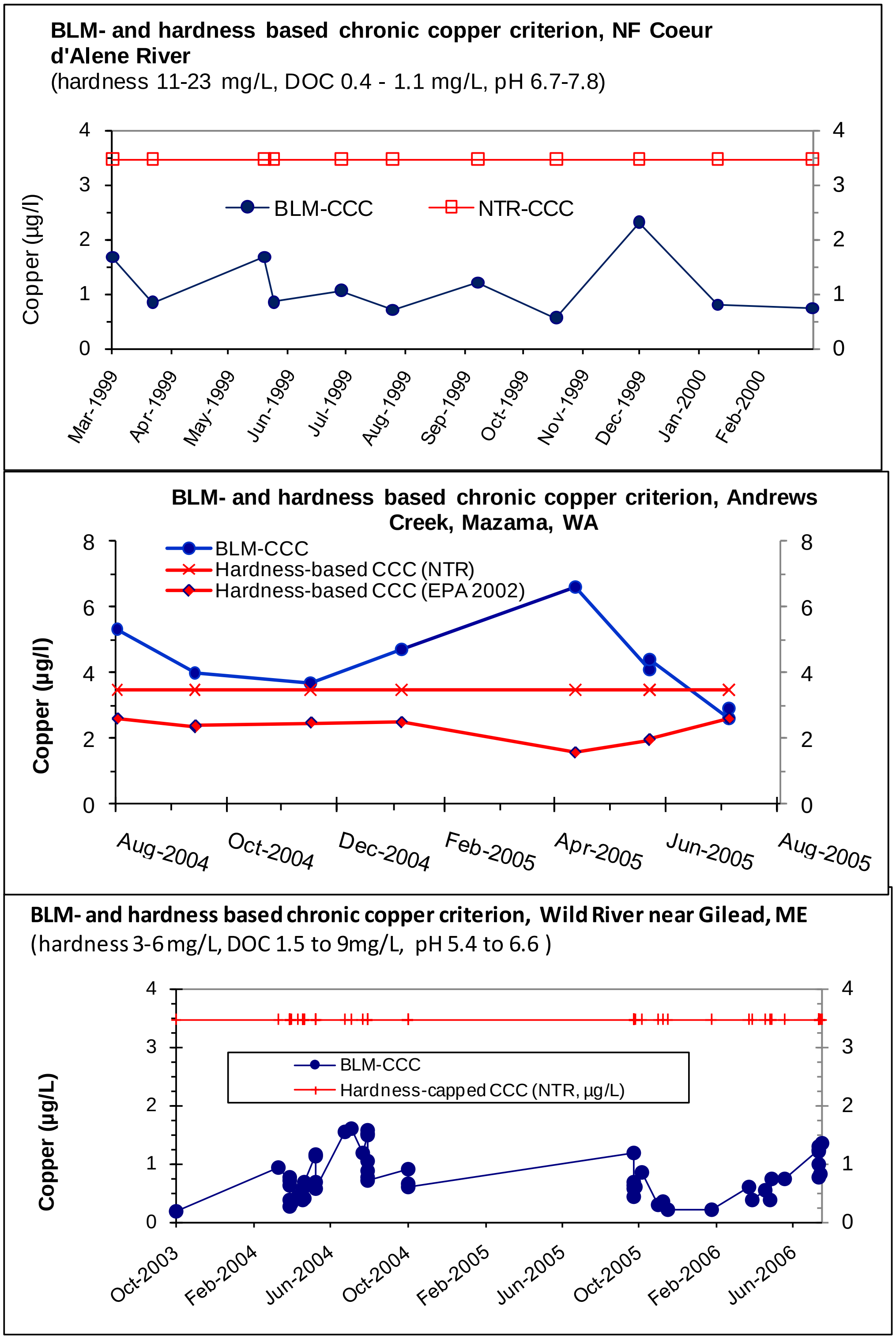
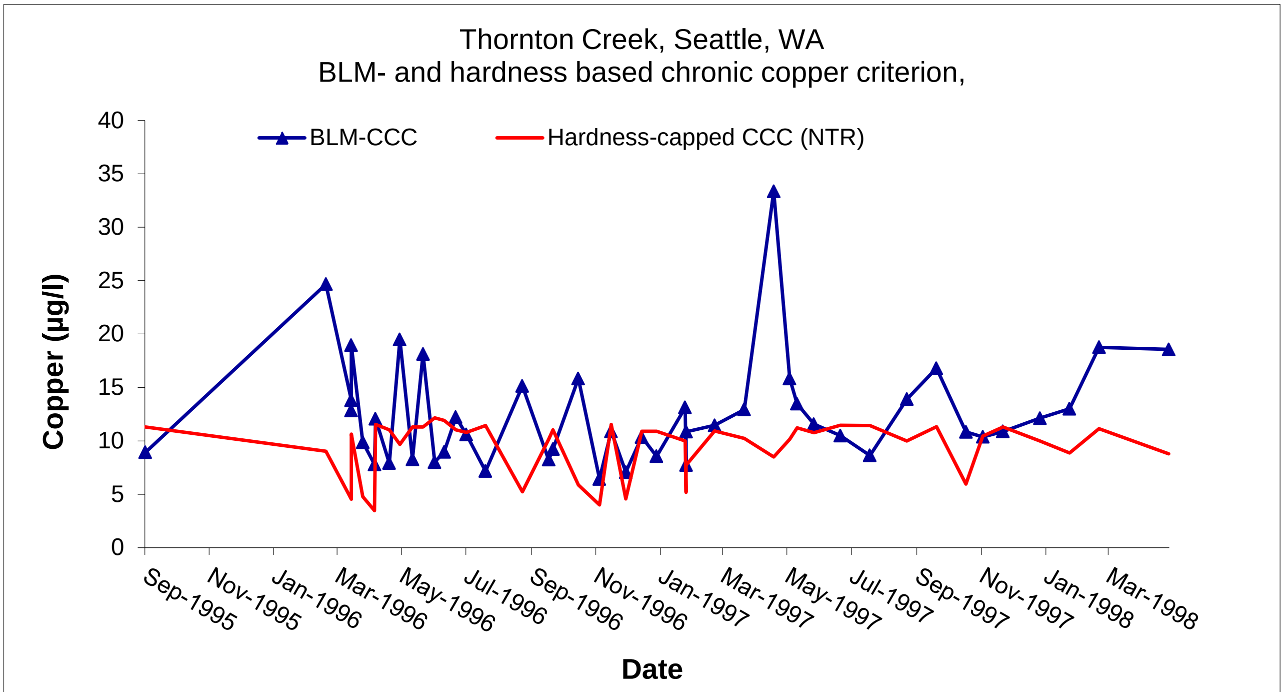



**Figure 27.** **Seasonal patterns in BLM- and hardness-based CCC values for three streams distinguished by snowmelt springflows, softwater, variable DOC, and in the bottom, low pH. Because in Idaho hardness-based criteria equations are "capped" at 25 mg/L, and the hardnesses of these streams never exceeded 25 mg/L, the hardness-based criteria are a constant 3.6 µg/L. In the NF Coeur d'Alene River,**

**Idaho, DOC is low year round resulting in low CCC values year round. In Andrews Creek, the "uncapped" CCC again mirrors the BLM-CCC with opposite trends.**

Andrews Creek is located on the eastside of the North Cascade Mountains near Mazama, Washington. Similar to Convict Creek in the Sierra Nevadas and to streams in the Idaho Batholith geology that underlies a large portion of the Salmon River drainage in Idaho, the Andrews Creek watershed is granitic with thin soils. Andrews Creek has very soft water and low organic matter, although not as extremely low as some waters in northern Idaho such as the NFCDA, Lochsa, or Selway River drainages. If the hardness-floor were ignored, the BLM- and hardness-based criteria again would show the now familiar opposite patterns, with the BLM-based low values occurring mostly in fall and winter (Figure 27, middle).

The Wild River near Gilead, Maine, is included to illustrate conditions that have produced some of the lowest BLM-based criteria time series values for natural waters we located (Figure 27, bottom). While not physically close to Idaho, the Wild River drainage has other geographic similarities to the action area. The Wild River drainage is underlain by erosion resistant bedrock with poorly buffered thin soils which results in extremely soft water. The pH of the Wild River is lower than that in any of the other "BLM-quality" datasets compiled for this review and is probably lower than is typical in softwater areas of Idaho. Still, in streams draining basins with granitic geology from Idaho Batholith or Precambrian metamorphic rocks, pH values are commonly in the low 6s and sometimes less than 6. While these BLM-based CCC values are very low (0.2 µg/L to <2 µg/L), 96-hour fathead minnow LC50s as low as 2 µg/L have been obtained in similarly mildly acidic, low calcium waters (Figure 4), and presumably had effects been obtained from longer exposures, sublethal endpoints, or more sensitive species would have been lower.



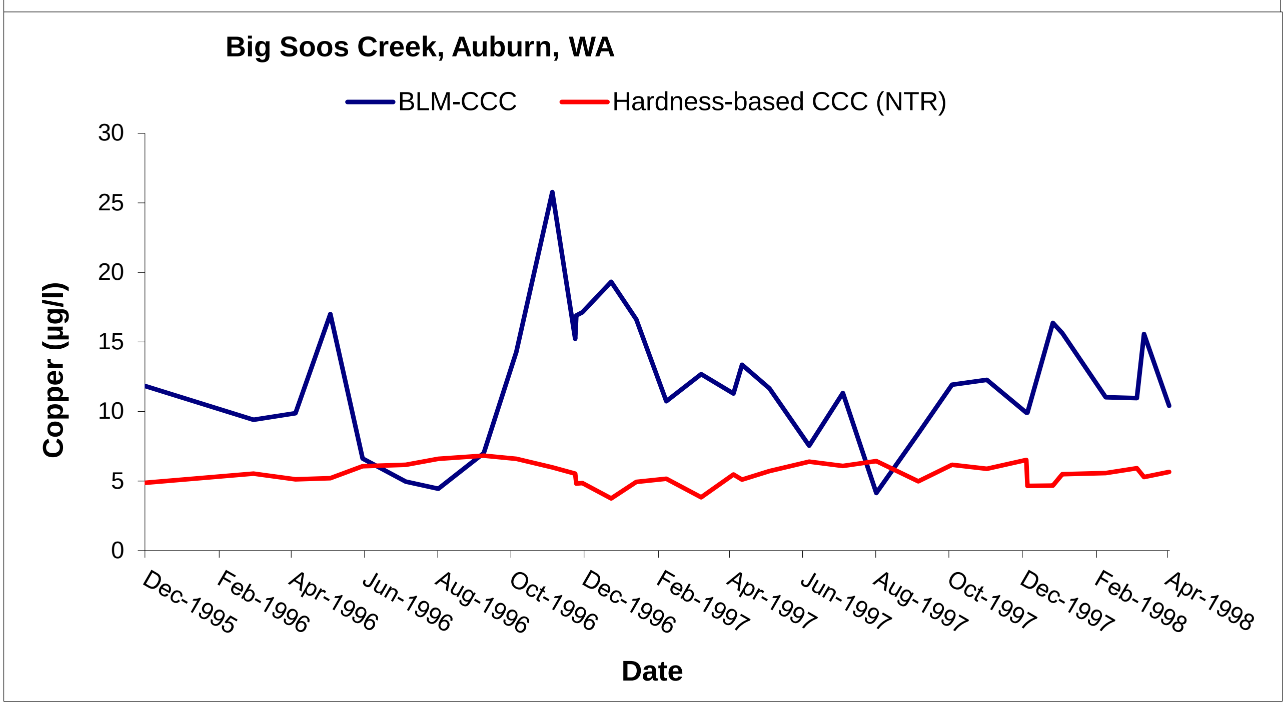




**Figure 28.** **Seasonal patterns in BLM- and hardness-based CCC values for two streams distinguished by high flows from winter rainfall instead of snowmelt. Thornton Creek is an urban stream with moderately hard water and higher DOC than most streams examined; Big Soos Creek is a softwater stream in a mostly rural area with periodic high DOC during the winter rainy season.**

DOC concentrations occurring across several streams sampled systematically from April through September 2007, show considerable variability in the timing of peak DOC, but except for the South Fork Coeur d'Alene River (SFCDA), for each stream the lowest DOC concentrations were measured in the late September samples (Figure 28). For the SFCDA, DOC was low and nearly uniform throughout the period of record. This particular sampling effort did not collect the major ion data needed to calculate BLM time series, but the DOC patterns give further support that critical conditions for vulnerability to copper toxicity are predictable and probably will occur in fall during base flow conditions.

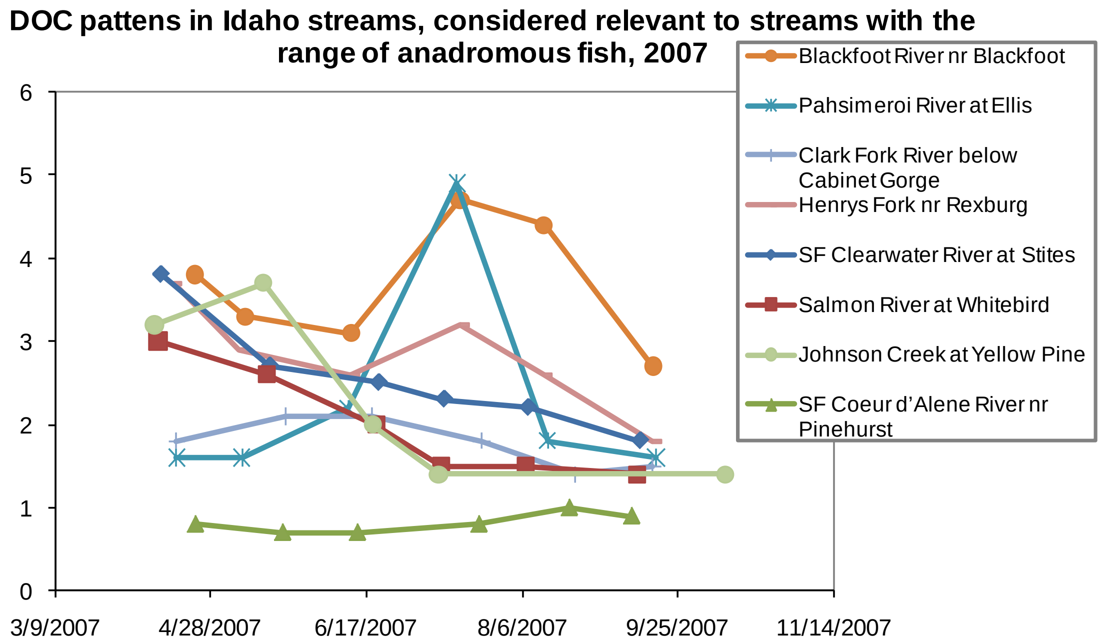



**Figure 29. Seasonal DOC patterns in Idaho streams considered relevant to listed salmon and steelhead waters, including four within their critical habitats. In all cases by late September, DOC was at or near its lowest measured value.**

Returning to the question posed at the beginning of this section on implementation considerations, are the regional and seasonal water chemistry patterns sufficiently predictable that conservative (realistic but protective) default BLM-criterion table values can be defined? For the annual critical conditions when copper would be at its most toxic, the answer appears to be "Yes." The most critical conditions almost invariably occur in the fall, and over the range of waters with listed anadromous fish in Idaho, data relevant to these conditions were either directly available or could be estimated from watersheds with similar characteristics (Table 3). Conservative high estimates of annual maxima could also be made. For example, if for the upper Salmon River, the Snake River above Jackson Lake is used as a surrogate, the lowest measured dip in the BLM-based criteria plots would be about 2 µg/L and the lowest annual peak would be about 6 µg/L (Figure 26).

The handling of discrete pH data is an important detail note in the BLM calculations to estimate late-summer copper benchmark concentrations in Table 3. None of the pH data in the USGS data for the streams in Table 3 were collected in the early morning hours near dawn when pH would be expected to be at the daily minimum. Some data were collected in the late afternoon, which is when pH would be expected to be near its maximum. In the copper BLM, pH is an important variable, and copper toxicity is predicted to markedly decrease as pH increases. Daily pH variations in excess of 0.5 units over the course of a day are not uncommon in streams, even streams that are fairly oligotrophic; and in streams with high primary productivity, pH can swing by at least 2 units (e.g., Nimick *et al*. 2011; Balistrieri *et al*. 2012). Accumulations of metals on gills can be rapid, with sufficient accumulation occurring over time periods of 45 minutes to 3 hours to predict later toxicity (Balistrieri and Mebane 2014). Until the importance of time varying pH for metals speciation, accumulation, and toxicity are better investigated, it seems prudent to use daily minimum pH values in BLM calculations. Thus, in the BLM calculations to estimate late-summer copper benchmark concentrations in Table 3, for those sites with pH >7.5, pH was lowered by 0.6 units to adjust for high bias from mid-day water samples.

*Table 3. Ranges of chronic copper criterion concentrations estimated for critical late summer/fall baseflow conditions in subbasins within the range of anadromous salmonids in the Snake River basin, Idaho.*

| Subbasin | Common subbasin geologic characteristics | Critical late-summer Cu benchmark concentration (µg/L) | Based upon EPA's 2007 Cu chronic criterion (CCC) using data collected or estimated using: |
|---|---|---|---|
| Selway, Lochsa, MF Clearwater R | Granitic or intrusive rocks from Idaho Batholith or Precambrian metamorphic rocks | 0.6 | St Joe River at Red Ives, 9/14/2007; SF Coeur d'Alene R at Pinehurst, 9/10/2007; NFCDA Fig 25 |
| SF Clearwater River | Idaho Batholith | 1 | SF Clearwater at Stites |
| MF and SF Salmon and tributaries | Idaho Batholith | 1 | Extrapolated using low conductivity measured in undisturbed streams in the Salmon R basin (Ott and Maret 2003), ~30 µs/cm, pH 6.9, using DOC of 1 mg/L and then estimating major ions with regression equations from streams in Coeur d'Alene R with similarly low conductivity |
| Upper Salmon R | Idaho Batholith and Challis volcanics | 3 | Snake River (Fig. 24); Johnson Creek at Yellow Pine, 10/10/2007 |
| Upper Salmon R tributaries | Challis volcanics | 3 | Assumed similar to Panther Creek |
| Panther Creek | Challis volcanics and Idaho Batholith | 3 | Minimum BLM=CCC calculated for low-flow, low DOC conditions from a 1994 dataset (Maest *et al*. 1995) |
| Lemhi and Pahsimeroi Rivers | Tertiary sediments from ancient lake bottoms | 6 | Pahsimeroi at Ellis, 9/18/2007 |
| Lower Salmon (downstream of SF Salmon) | Diverse | 3 | Salmon River at White Bird, 9/27/2007 |
| Snake River | Diverse | 6 | Minimum BLM calculated for Snake River at mouth (Burbank, WA) |

Data collected in 2007 were for a single data collection. It seemed reasonable to assume that late summer baseflow conditions were probably close the critical condition (i.e., annual minimum) CCC calculated using the BLM-based Cu criteria. However, because the BLM-based criteria is sensitive to pH and these mid-day collected samples probably represented close to the daily high for pH, pH was lowered by 0.6 units for those sites with high pH (>7.5) because pH can vary up to 2 units per day (Balistrieri *et al*. 2012), although in oligotrophic, coldwater streams in summer, pH swings on the order of 0.6 units over the day seemed more likely, with maximum pH occurring near midday. SpC = specific conductivity

This approach would also be consistent with the concept that greater conservatism in environmental management is appropriate when information is uncertain and this conservatism may be relaxed when uncertainties are reduced through better information. Assume for example that a facility manager was concerned that this approach of using conservative estimates of BLM-based criteria for regulating copper in effluents during base flow and that provided no relief from unnecessarily conservative hardness-based copper criteria during spring runoff when hardnesses were low, could result in costly discharge restrictions that might be of little environmental benefit. In such a case, since the BLM parameters probably only add a modest increase in sampling cost, compared to the labor costs of getting samples in the first place, it would be cost-effective for the facility manager to arrange to include the BLM parameters in their ambient monitoring program. The major dischargers operating within the range of anadromous fish in Idaho and that have metals limits are all mining facilities. These operations tend to collect ambient water quality data from their receiving waters four times a year, with one sampling event during low, base flows and three during the more variable April to June conditions. From the patterns observed from the 16 datasets shown here (Figures 23-27), such a seasonal sampling would be sufficient in at least streams with snowmelt dominated high flows, and over time could probably be backed off to one spring and one base flow sampling effort. A compromise seasonal table-value approach might be useful on a watershed or river reach approach in lieu of ongoing monitoring if risks of exceedences were low (Figure 30).

A more scientifically defensible and efficient approach would be to develop surrogate measures to predict the major ions and DOC in natural waters. In natural waters the inorganic parameters used in the BLM tend to be correlated with each other and with conductivity and water hardness. Similarly, DOC tends to be correlated with water color and with specific absorption (Schwartz *et al*. 2004; Dittman *et al.* 2009; Gheorghiu *et al*. 2010). It should be feasible to develop surrogate models to estimate BLM parameters with sufficient accuracy across diverse waters that would simply require a conductivity meter and a field spectrophotometer. These could be deployed in-situ and set to transmit in real-time, which offers promise as low-cost and data rich surrogate measures for DOC.

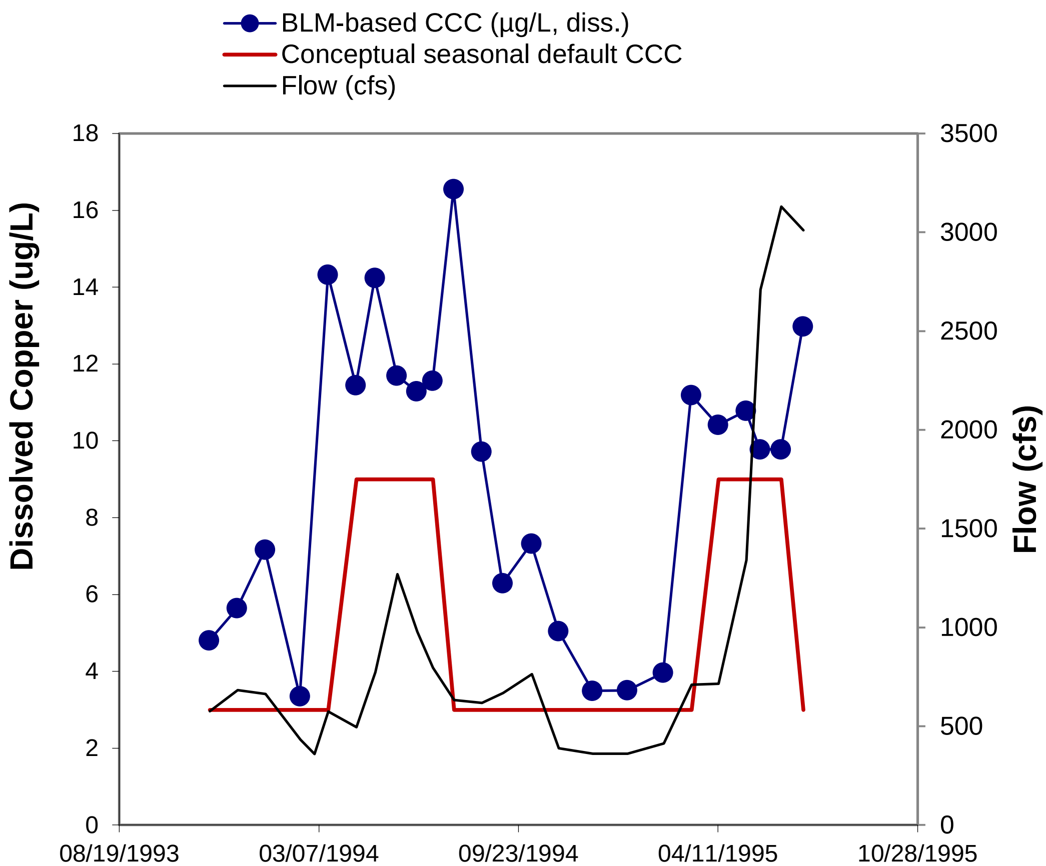



**Figure 30. Conceptual example of a simplified, default table-value approach to defining BLM-based copper criteria in lieu of routine monitoring of BLM data requirements. The sample data used are from the Teton River, ID (Figure 26).**

From a practical point of view for planning sampling for BLM inputs, targeting critical conditions that persist for several months during the low flow, dry season is considerably easier than trying to plan for sampling near the peak of runoff when hardnesses are at their annual minima, conditions that may develop quickly if an unexpected spring thaw occurs and may only last for a few days, and when access to collect samples may be hampered by rotten snow, high water, and mud.

In summary, NMFS' review has shown that Idaho's hardness-based copper criteria would likely result in instream copper concentrations above levels protective of listed salmonids and their critical habitats. Calcium, the main determinant of water hardness, is one factor affecting the toxicity of copper, but in natural waters it is generally less important than DOC or pH. Overall, EPA's 2007 version of the copper BLM did a credible job of predicting acute and chronic toxicity to taxonomically diverse organisms over a wide variety of waters, and had some predictive power with chemosensory functions in fish. Whether the BLM-based criteria would be fully protective of benthic macroinvertebrate communities is equivocal, but would be more protective than the alternative hardness-based criteria. While not optimal, minor losses of benthic diversity could likely be withstood by listed steelhead or salmon populations because juvenile steelhead and salmon are feeding generalists. Thus these likely adverse effects would not be expected to rise to the level of jeopardy or adverse modification of critical habitats.

The performances of the 2007 BLM based criteria were not ideal, and refinements would be worthwhile to pursue. For example, the BLM performance in very soft waters could be re-evaluated in the light of developments subsequent to the 2007 version (e.g., Ryan *et al.* 2009; Paquin *et al.* 2011). Regardless of this prediction bias, in practice the BLM-based criteria still produced quite low values in natural soft waters relative to toxicity values (Figure 5, Figure 25). So while our analyses suggest areas where the 2007 version copper BLM could be refined (e.g., treatment of DOC, competitive conditional stability constants), its mostly robust performance with a diverse array of organisms with sublethal and lethal endpoints in diverse waters validate earlier testing of the BLM performance (e.g., Santore *et al.* 2001; EPA 2003b). As is, the 2007 criteria represent a huge improvement over the NTR copper criteria and generally represent a major advance in the science of water quality criteria. Its application appears to be protective of listed salmon, steelhead, and their ecosystems.

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# Appendix D

## Conservative assumptions to be used in implementing criteria through effluent limits

The EPA's approach to implementing water quality based effluent limits in Idaho generally includes several conservative assumptions (EPA 2010:pp. 67-69). These conservative assumptions are designed to limit the discharge of pollutants in effluent such that pollutants would seldom be allowed to reach their "face value" criteria concentrations in waters receiving permitted discharges, downstream of mixing zones. Pursuant to Reasonable and Prudent Measure 2, Term and Condition 3.a., EPA will consistently apply three of these conservative assumptions in calculating effluent limits for discharges composed of any of the pollutants subject to this consultation for all NPDES permits in Idaho. The NMFS expects that application of these assumptions will reduce and minimize the take of the listed species subject to this Opinion.

The three conservative assumptions that EPA will implement pursuant to Term and Condition 3.a. are: (1) Assume that Only a Portion of the Low Stream Flow is Available for Mixing to Control Chronic Toxicity (mixing zone allowances); (2) Assume Receiving Stream Flows are Very Low; and (3) Assume the Maximum Permitted Discharge Volume. The NMFS evaluates the expected efficacy of these measures below

To evaluate the likely effectiveness of the three required conservative assumptions quantitatively, we selected the NPDES limits for the Thompson Creek Mine (TCM) as a relevant case study. This facility's permit was chosen because this facility had the necessary information most readily and transparently available to us. This information included:

1. Long-term flow records for the receiving waters were readily available via the internet, with a 37-year period of record (Figure D-2)[13];
2. A written description of the mixing zone allowances was available online (IDEQ 2000); and
3. The effluent limitations were available online and the calculations were described in a transparent and reproducible manner (EPA 2000).

The TCM facility has five permitted outfalls that discharge into three very different stream types:

1. Thompson Creek, a small stream with moderately-hard water (5$^{th}$ percentile hardness of 85 to 93 mg/L calcium carbonate) and very little dilution capacity during low flows with a 7Q10 flow of only 2.1 cfs (the 7Q10 is explained later);

[13] http://waterdata.usgs.gov/nwis.

2. Squaw Creek, a larger, hard water stream (5th percentile hardness of 290 mg/L calcium carbonate) with about double the flows of Thompson Creek and a 7Q10 of about 4.6 cfs; and

3. The upper Salmon River, a much larger, soft water stream (5th percentile hardness of 27 mg/L calcium carbonate) with a 7Q10 of about 323 cfs.

The characteristics of these discharges to these three water bodies are reasonably representative examples of the other facilities in the action area for which less information was readily available online.

## D.1 Conservatism of assuming that only a portion of the low stream flow is available for mixing (mixing zone allowances);

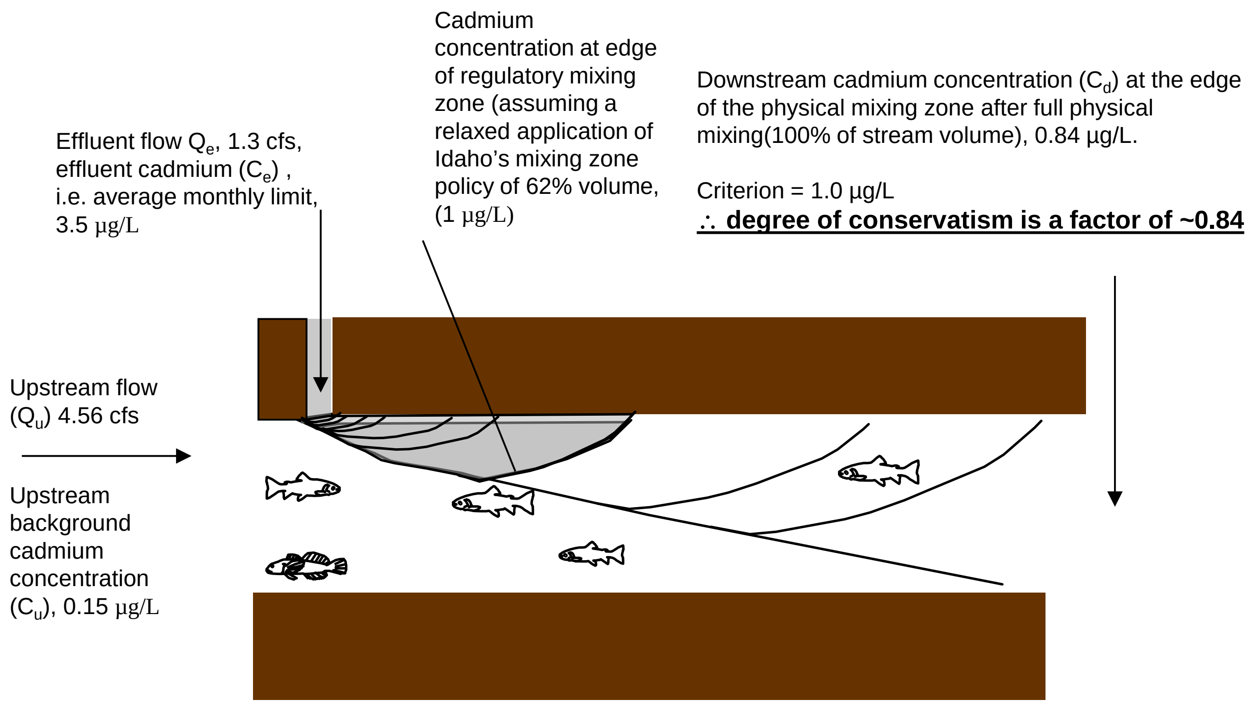
Under the first conservative assumption, EPA uses only a portion of the low receiving waterbody flow for dilution when calculating chronic limits. This is done in order to theoretically allow space in streams for passage of fish and other mobile aquatic species without having to pass through the mixing zone. This procedure further reduces the volume of the receiving stream which is used for permitting purposes, and therefore provides additional protection to aquatic species from chronic effects. The portion of the flow allowed for dilution is presumed to be 25% based on Idaho's Water Quality Standards, but based upon site-specific analyses of physical, biological, and chemical conditions, other fractions may be allowed. This discretion to relax or tighten the mixing zone percentage means that the actual conservative factor resulting from this policy may differ from the presumed limitation to 25% of the low stream flow. The State of Idaho is publishing more rigorous guidance on their mixing zone policies and it is now unlikely that mixing zone determinations would be proposed that would permit greater than 25% of receiving water flows to be used to dilute effluents without supporting technical analyses.[14]

For the TCM facility, some flexibility for both the listed species and the discharger was demonstrated by the state and EPA, with 0% mixing zone allowed for copper under certain flows and up to 62% of the stream volume allowed for cadmium. With cadmium, the allowable portions of receiving waters allowed for mixing range from 5% to 62% of actual stream flow for different streams and flow conditions (IDEQ 2000). The rationales for setting mixing zone fractions included avoiding concentrations likely to cause behavioral avoidance in salmonids, retaining sufficient zone of passage with suitable water velocities and depths for juvenile and adult salmonids, load allocations between outfalls, and limiting the travel time for drifting organism through the "acutely toxic" portion of effluent plumes to 1 minute or less, based upon the calculated instream concentrations and modeled time and distance for plume dilutions (IDEQ 2000, table 21).

Using the calculation methods of EPA (2000), NMFS evaluated the degree of conservatism resulting from various mixing zone limitations. A pessimistic (i.e., least-

[14] http://www.deq.idaho.gov/water/data_reports/surface_water/monitoring/mixing_zones.cfm accessed 01Oct2010.

conservative) example in which 50% of the portion of the receiving water flow was allowed for mixing of effluents is shown in Figure D-1. There, the degree of conservatism resulting from the limitation that only a portion of the receiving water stream flow could be used was a factor of 0.84. Other permitted conditions at the TCM facility were calculated as conservative factors ranging from a minimum of 0.22 for the most restrictive 5% mixing zone authorization; to 0.39 for the quasi-default mixing zone of 25% portion of flow; and to 0.84 for the mixing zone allowing 62% of the stream flow to be used.




$$C_d = \frac{C_u Q_u + C_e Q_e}{Q_e + Q_u} = C_{(downstream)} = \frac{(0.15\ \mu g/L \cdot 4.56\ cfs)+(3.5\ \mu g/L \cdot 0.87\ cfs)}{(0.87+4.56)\ cfs} = 0.84\ \mu g/L$$

**Figure D-1. Conservatism resulting from a liberal application of Idaho's mixing zone policy which allowed 62% of the stream flow to be used for diluting effluents.**

## D.2 Conservatism of assuming receiving stream flows are very low

The second conservative assumption measure is to assume that receiving stream flows are very low, based on EPA's concept of design flows for effluent discharges. Stream flows are variable and a target of effluent limitations is to approximate provisions in the aquatic life criteria that limit the tolerable frequency of excursions above water quality criteria. In the IWQS, for chronic criteria this is defined as the 7-day, once in 10-year low flow or 7Q10 (EPA 1991; IDEQ 2007).

In the Thompson Creek example, the concept of a 7Q10 was interpreted by EPA more liberally than a “7-day, once in 10-year low flow.” Rather, EPA defined “seasonal 7Q10s” where there is a conventional 7Q10, and then defined effluents set for a higher flow tier that occurs during spring snowmelt. By effectively having two 7Q10s for the same time period, the allowable frequency of excursions is greater than if a conventional 7Q10 were used. The higher flow tiers during spring runoff were considered appropriate by EPA (2000b) because of the extreme variability in effluent and receiving water flows. To keep comparable levels of protection during the high flow tiers when more effluents could be discharged, EPA (2000) required minimum dilution ratios as part of the permit.

We evaluated the degree to which the assumption that receiving stream flows are very low acts as a conservative measure (as stated in EPA (2010a)) by comparing the assumed low flows to the actual flows in Thompson and Squaw Creeks (Figure D-2). To avoid an optimistic review, we used water year 2007 because it was a year with lower than average flows. Flows in late summer and fall of 2007 (blue line) were considerably lower than the long-term average (brown line). Thompson Creek was in its higher flow tier for about 4 months of the year from March through July. The minimum measured flow in Thompson Creek in 2007 was effectively equal to the 7Q10 flow used in the effluent calculations, 2.1 vs. 2.05 CFS respectively (Figure D-2).

To determine to what extent the actual flows provided a “conservative factor,” we compared to the “low flow 7Q10” of 2.05 CFS and the “high flow 7Q10” of 7 CFS and divided the low or high “7Q10” by the actual flow for each day during calendar year 2007, and then calculated summary statistics for the year. The same thing was done with mean daily values for the 37 year period of record (i.e., the mean daily flow for October 1 for all 37 years, October 2 for all 37 years, and so on). These results are summarized in Table D-1.

For the four scenarios we analyzed, 95% of the time, the low-flow assumption resulted in a “conservative factor” of at least 0.84 (range 0.66-0.84). On the average, the “conservative factors” were about 0.4 (Table D-1).

When calculated in this manner, lower proportions are more conservative, and a value of one indicates no conservatism context. It would be equivalent to express the “conservative factors” as reciprocals so that bigger numbers correspond with increasing conservatism. Thus, it would be equivalent to say that 95% of the time, the low-flow assumption resulted in a “conservative factor” of at least 1.2 (range 1.2 to 1.6), and on the average the “conservative factors” were about 2.5.

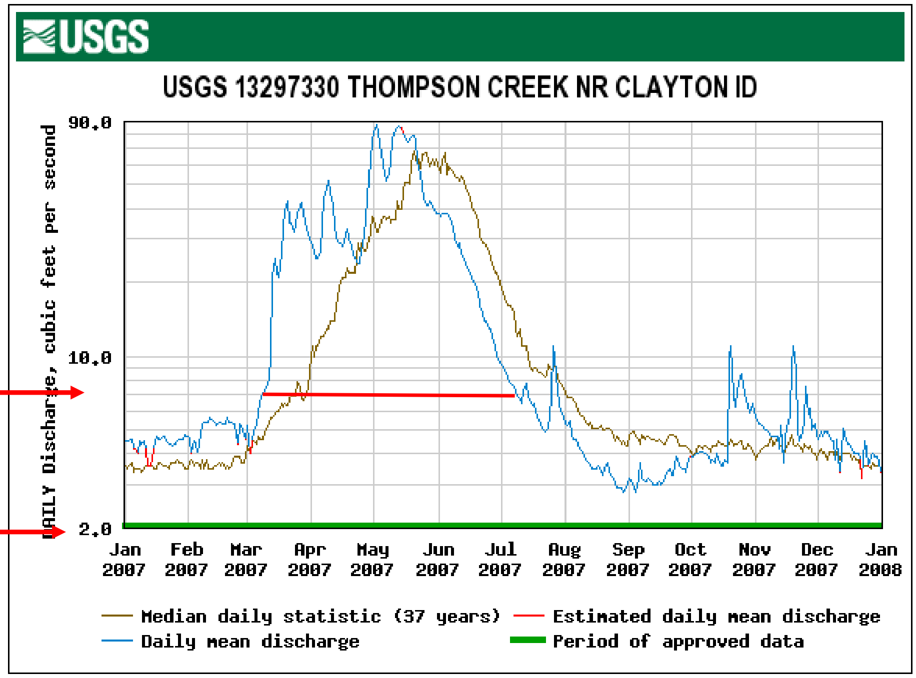
**Table D-1.** "Conservative factors" resulting from assumed low flows in two streams receiving mining effluents. Lower factors are more protective and a factor of 1.0 provides no additional conservatism.

| Conservative Factor | Thompson Creek 2007 | Squaw Cr 2007 | Thompson Cr – 37 year average | Squaw Cr – 37 year average |
|---|---|---|---|---|
| Median | 0.44 | 0.38 | 0.43 | 0.41 |
| Average | 0.45 | 0.40 | 0.41 | 0.40 |
| 90th percentile | 0.70 | 0.56 | 0.58 | 0.50 |
| 95th percentile | 0.84 | 0.75 | 0.76 | 0.66 |
| Least conservative | 1.00 | 1.00 | 1.00 | 0.98 |

Thus, a moderately pessimistic estimate of how much protection the "conservative factors" actually provided by limiting a portion of the low stream flow allowed for mixing is a factor of 0.84 and for assuming low receiving water flows coincidentally is also about 0.84 (i.e., 95% of the time it is more protective). Since these two measures are combined jointly, their product is 0.70.

Assumed instream flow used to calculate effluent limits to meet criterion during annual high-flow seasons (7 cfs)

Assumed instream flow used to calculate effluent limits to meet criterion during annual stable flow seasons (2.05 cfs)




Assumed instream flow used to calculate effluent limits to meet criterion during annual high-flow seasons (50 cfs)

Assumed instream flow used to calculate effluent limits to meet criterion during annual stable-flow seasons was 4.56 cfs, which is below the chart scale

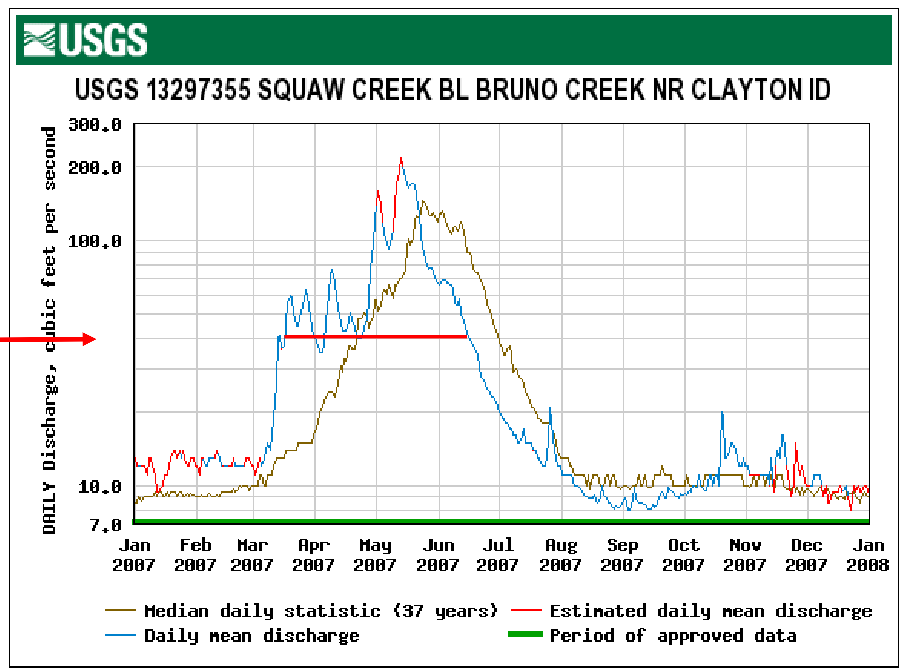



**Figure D-2. Examples of actual stream flows versus stream flows that were assumed to calculate seasonally variable wastewater discharge limits for a facility. Actual flows were estimated to be lower than the seasonally adjusted assumed flows about 98% of the time (IDEQ 2000; EPA 2000).**

## D.3 Conservatism of assuming the maximum permitted discharge volumes

The EPA's (2010a) final conservative measure is to assume the Maximum Permitted Discharge Volume is closely related to the analysis of receiving water stream flows. This assumption is overstated slightly in that EPA assumes a higher than average permitted discharge volume, not the absolute maximum. For example, at Thompson Creek outfall #2, the maximum effluent volume contributed 14% of the flow of Thompson Creek. The NPDES permit assumed that the effluent would contribute about 8% of the flow, which was close to the $99^{th}$ percentile of flow percentages. The $95^{th}$ percentile effluent volume contributed about 5% of upstream flows (IDEQ 2000; EPA 2000). This means that for this outfall, about 95% of the time, effluent volumes were less than or equal to about 5/8 of those permitted providing another "conservative factor" of 0.7. The likely compounded conservatism of this aspect of effluent limitations would be 0.7 X 0.84 X 0.84 for at least $0.95^2$ of the time which equals 0.5 for at least 90% of the time. This can be restated as follows.

The overall conservatism of the three conservative assumptions evaluated here can be summarized and were estimated as:

> Assumption 1: Limiting the portion of stream flow allowed for mixing of effluents. The conservatism factor for this measure was estimated at about 0.84 or less (from Figure D-1), where the conservatism factor is expressed as a proportion and smaller values are more conservative;
>
> Assumption 2: Assuming receiving stream flows are very low. About 95% of the time, the conservatism factor for this measure was also estimated as about 0.84 or less (from D-1); and
>
> Assumption 3: Assuming unusually high permitted discharge volumes. About 95% of the time, the conservatism factor for this measure was estimated at about 0.7 or less (from text following Figure D-2).

The overall conservatism of these factors can be estimated as their product, i.e., $1 \times 2 \times 3 = 0.84 \times 0.84 \times 0.7 \approx 0.5$. The protectiveness of assumptions 2 and 3 vary over time, thus the proportions of time need to be combined. If stream flows and effluent volumes vary independently, then the time proportions should be multiplied together, i.e., 0.95 x 0.95 = 0.9. This can be restated that at least 90% of the time, the overall conservatism factor of measures 1, 2, and 3 is a factor of 0.5 or less.

If the effluent and receiving water assumptions made for Thompson Creek are further assumed to not be much more stringent or lenient than is typical, then it could be assumed that these three conservative assumptions will reduce the allowed chemical concentrations from point source discharges to about 50% of the criterion values for the great majority of the time. This provides a significant reduction in exposure to pollutants from NPDES permit discharges and will minimize take of listed salmon and steelhead.

## References for Appendix D

D.4 References

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# Appendix E

## Biomonitoring of Effects

When Biomonitoring is necessary to implement RPAs or RPMs the following protocols are to be used.

1. At a minimum, samples will be collected upstream (reference) and downstream of the discharge location(s).

2. At a minimum, benthic macroinvertebrates are to be evaluated to make sure effects are not greater that those described in the effects section. Fish communities shall also be monitored, to the extent such monitoring is not otherwise prohibited by regulation or policy. At a minimum, monitoring shall be conducted annually during late summer or fall base flows; annual monitoring is recommended. Because of the need to minimize confounding variability other than the discharge constituents:

   a Reference and comparison sites need to be similar, except for the discharge; e.g. size, gradient, channel type, temperature, substrate, other variables that structure communities;

   b Because some biological variables can be confounded by natural upstream-downstream changes (e.g., temperature, habitat size), paired watershed, or other out-of-watershed reference sites are recommended in addition to within-watershed upstream reference sites;

   c Artificial substrates (rock baskets) may be needed for macroinvertebrate monitoring if comparable habitats cannot be located (e.g., similar sized gravels and cobbles, velocities, depths, and shading).

3. Taxonomic enumeration of macroinvertebrate samples will be sufficient to be comparable with that used in the IDEQ stream ecological assessment program (Grafe 2002). Generally this means that invertebrates must be identified to the lowest practical level, which for insects in the Ephemeroptera, Plecoptera, and Trichoptera orders (EPT, mayflies, stoneflies, and caddisflies) means to the species level; and for non-EPT insects other than chironomids, crustaceans and molluscs usually means at least to the genus level. Other non-insect invertebrates except annelid worms can usually be identified to family, annelids are often only identified to order.

4. Sampling for tissue residues of concern. With arsenic, the focus is evaluating residues in salmonid invertebrate prey. This is because adverse effects of arsenic at environmentally relevant concentrations have been demonstrated from feeding studies with trout (Cockell *et al.* 1991; Hansen *et al.* 2004; Erickson *et al.* 2010). Laboratory analyses of arsenic in invertebrate tissues should include both inorganic and total arsenic, because inorganic forms of arsenic appear to be more toxic to fish than organic forms such as arsenobetaine

and di- or monomethyl arsenic (Erickson *et al.* 2011). Monitoring should target representative, composite invertebrate samples for analysis. It seems reasonable to assume that benthic invertebrates that are vulnerable to capture with disturbance techniques (kick nets, rock scrubs) roughly represent those invertebrates that are likewise vulnerable to being eaten by juvenile salmonids.

With selenium, the focus is evaluating if tissue residues are accumulating to harmful concentrations in the fish themselves. Juvenile fish are recommended for sampling because adverse effects to juveniles are more likely to occur in the first place, or affect population dynamics more than adverse effects to adult fish (Lemly and Skorupa 2007; Van Kirk and Hill 2007). Sculpin may be a useful surrogate species to target in tissue monitoring because they are often abundant in streams, have significant dietary overlap with juvenile salmonids, have a sedentary life style that makes them more likely to have experienced and integrated the exposures at the place they are collected from, and permits to capture and kill sculpin are less likely to be obstructed by regulators than listed salmonids. Further, Rhea *et al.* (2013) found that sculpin were good indicators of selenium exposure and sublethal effects in the Yankee Fork, Idaho. However, based on anecdotes of sculpin being abundant in selenium enriched streams in southeastern Idaho, sculpin are probably not so sensitive that they would eliminated from streams with elevated selenium which would make them a poor choice of a monitoring species for tissue residues.

5. Adverse effects will be gaged in comparison to deviation from upstream or other reference sites using at least the following metrics or indexes (Table E-1).

    a Deviation from reference may be assessed based upon values compared to effects differences listed in table Table E-1 without the need for statistical testing. This is because sufficient replication necessary for statistical hypothesis testing approaches to be sensitive may be precluded by concerns about effects of monitoring or by costs. Further, the magnitudes of difference from expected reference conditions are probably more biologically meaningful than whether a reduction is statistically significant at a given probability. If statistical approaches are used, the following issues are to be considered.

    b Many valid approaches to statistical interpretation of monitoring data have been developed, and the following approaches are not intended to preclude other supportable approaches. However, the appropriateness of alternate approaches must be described.

    c Hypothesis tests, which aim to minimize type I errors (false positive results), are standard procedures in scientific research, but they are often inappropriate in ESA reviews, where the primary objective is to prevent type II errors (false negative results). Recognizing this disparity is particularly important when the best data available are sparse and therefore lack statistical power, because hypothesis tests that use data sets with low statistical power are likely to commit type II errors, thereby denying necessary protection to threatened and endangered species (Johnson 1999; McGarvey 2007).

d Hypothesis tests, if used, to test for statistical difference between sites for metrics that are expected to be sensitive to pollutants, should be interpreted with balanced power for type I and type II errors (Dayton 1998; Di Stefano 2003; Denton *et al.* 2011). That is, for macroinvertebrate data, if retrospective power analysis indicates an 80% probability of detecting a specified effect size ($\beta$ at 0.8) then the corollary test whether the effect was "statistically significant" is 20% ($\alpha$ of 0.2 or $p<0.2$). No fixed value for $\alpha$ (the probability of making a type I error, for example to incorrectly concluding impairment exists when in fact the apparent effect was only due to chance) is specified. While traditionally 'adequate' power has been settled by adherence to the 'five eighty' convention in which statistical significance (type I error rate, a) is fixed at 5% and statistical power considered adequate if it reaches 80% (type II error rate, b, of 20%) this places the 'burden of proof' disproportionately on those concerned about avoiding type II errors (Field *et al.* 2007). If statistical power analyses are used, the specified effect sizes are given in Table E-1.

Except for tissue residues, for which the bases for the table values are given in the respective sections of this Opinion, the references given in Table E-1 explain the methods and rationales for measuring the different effect metrics, but do not necessarily specify the effect values listed in Table E-1. Rather the magnitude of "critical effect" sizes were selected values are based upon subjective, professional judgments that were, in turn, influenced by two recent reviews (Munkittrick *et al.* 2009; Janz *et al*. 2010). The effect sizes listed in listed in Table E-1 probably are optimistic compared to the minimum differences detectable from statistical hypothesis testing using common replication efforts. This subjectivity and likely conservatism to the detectable differences seems both appropriate and unavoidable based on the information reviewed. For instance Janz *et al*. (2010) suggest that "*the stipulation of an effect size threshold is a judgment about biology, not simply a statistical or procedural decision, and relies on many underlying explicit or implicit judgments about the biological importance of an effect of a nominated magnitude.*"

**Table E-1. Biomonitoring metrics to evaluate for effects of toxic discharges**

| | Effects difference for comparison to reference or table value | Reference/notes |
|---|---|---|
| ***Macroinvertebrates in streams and rivers:*** | | |
| Idaho Stream Macroinvertebrate Index (SMI) | 10% | (Jessup and Gerritsen 2002) |
| SMI component metrics (9 metrics related to taxa richness, dominance and tolerance) | 10% for richness and dominance metrics; 20% for other metrics | (Carlisle and Clements 1999; Jessup and Gerritsen 2002) |
| Total macroinvertebrate biomass | 20% | |
| Abundance of invertebrates considered vulnerable to predation by juvenile salmonids | 20% | (Suttle et al. 2004) |
| Biomass of invertebrates considered vulnerable to predation by juvenile salmonids | 20% | (Suttle et al. 2004) |
| Similarity between reference and assessment stations (Jaccard similarity or comparable index, e.g. observed/expected (O/E) comparison) | 10% | Effects difference assumed to be similar to taxa richness measures |
| ***Fish*** | | |
| Community surveys (IBI) | 10% | (Mebane 2002b; Mebane et al. 2003) |
| Sentinel species (e.g. sculpin abundance or age classes) | 20% for abundance; no difference for age classes | (Janz et al. 2010) |
| Relative abundance (catch per unit effort, CPUE, or snorkel counts) | 20% | |
| Length-frequency analysis or numbers of age classes of salmonids or sculpins | 10% difference for median lengths; no difference in age classes | |
| Mean condition factor of salmonid species | | |
| Jaccard similarity | 10% | Minimum detectable difference assumed to be similar to taxa richness measures |
| ***Tissue Residues*** | | |
| Arsenic in benthic invertebrate prey organisms (as a representative composite community sample) | < 20 mg/kg dry weight | This review |
| Selenium in juvenile salmonids (whole-body) | < 7.6 mg/kg dry weight | This review |
| Selenium in adult sculpins (whole-body) | < 7.6 mg/kg dry weight | This review |

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# Appendix F

## Salmonid Zone of Passage Considerations

A zone of passage must be maintained around a mixing zone, sufficient to allow unimpeded passage of adult and juvenile salmonids. Determining what is “sufficient” may require site specific analysis. There is a long established precedent of using published expert opinion or expert consensus if no more than 25% of the cross sectional area was impinged upon, that would be sufficient for a zone of passage (FWPCA 1968; EPA 1994).

Recent examples have used different passage criteria. In a site-specific analysis, Mebane (2000) concluded that if the mixing zone of effluents into a small trout stream did not exceed 50% of the volume and width, then the unaffected portion of the channel was likely be sufficient for unimpeded passage around the mixing zone. That conclusion considered species and life stage requirements for appropriate depths, velocities, and habitat features including instream cover from predation in the unaffected portion of the channel. Other important considerations include situating mixing zones to avoid affecting or creating attractive habitat features in the effluent-exposed portions of the stream channel that could lead to fish congregating in mixing zones and risk disproportionate exposure to effluents. These habitat features to avoid influencing or creating might include locally important pool habitats, spawning areas or thermal refuges (e.g., Harper *et al.* 2009) in the mixing zones. The concept of avoiding spawning areas is necessarily subjective and cannot be defined in absolute terms. This is because fish can spawn in a variety of habitats, including those that experienced fisheries biologists might consider suboptimal, and the absence of spawning can rarely be proven. The intent is to avoid local concentrations of spawning habitat, not to preclude discharges into marginal habitats where spawning could potentially occur.

Instream flow studies for trout and salmon are another source of information for passage criteria. For example, the minimum depth criterion for adult fish passage must be present in greater than 25% of the total stream width in representative transects to allow passage (Maret *et al.* 2006).

The concept of mixing zone limitations are illustrated in Figures 2.9.1 and 2.9.2. Figure F-2 gives illustrates the results of effluent limit calculations with copper for a water body subject to the restriction that the volume of the receiving waters that is used for determining dilution and preserving a zone of passage for migrating fish and other aquatic life is limited to 25% of the stream volume. This example was calculated following the recent practices used by EPA Region 10 staff for determining effluent limits[15] and EPA’s technical support manual for water quality based effluent limits (EPA 1991).

In the calculations presented for copper, when the effluents are limited in this manner, the increase in copper concentrations allowed after complete mixing is less than 0.6 µg/L, a concentration increase likely to contribute to impairment of olfaction and predator avoidance in

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[15] http://yosemite.epa.gov/r10/WATER.NSF/NPDES+Permits/Permits+Homepage

juvenile salmon (Figure F-2). So long as the approaches described herein are followed, it seems likely that criteria for copper would adequately minimize adverse effects to listed salmonids. While an infinite variety of effluent and receiving water geometries, concentrations and flow conditions could be envisioned, the approach illustrated in Figure F-2 would result in similar results when the same decisions rules are applied in other configurations. Other cationic metals and other substances can cause chemosensory or avoidance behavior, but none were obviously more severe than copper (considered in the individual chemical sections). Thus, this approach would presumably be appropriate and as protective for other chemicals evaluated. A mixing zone demonstration in ESA waters should be rigorous enough to satisfy the information needs listed in Table F-1.

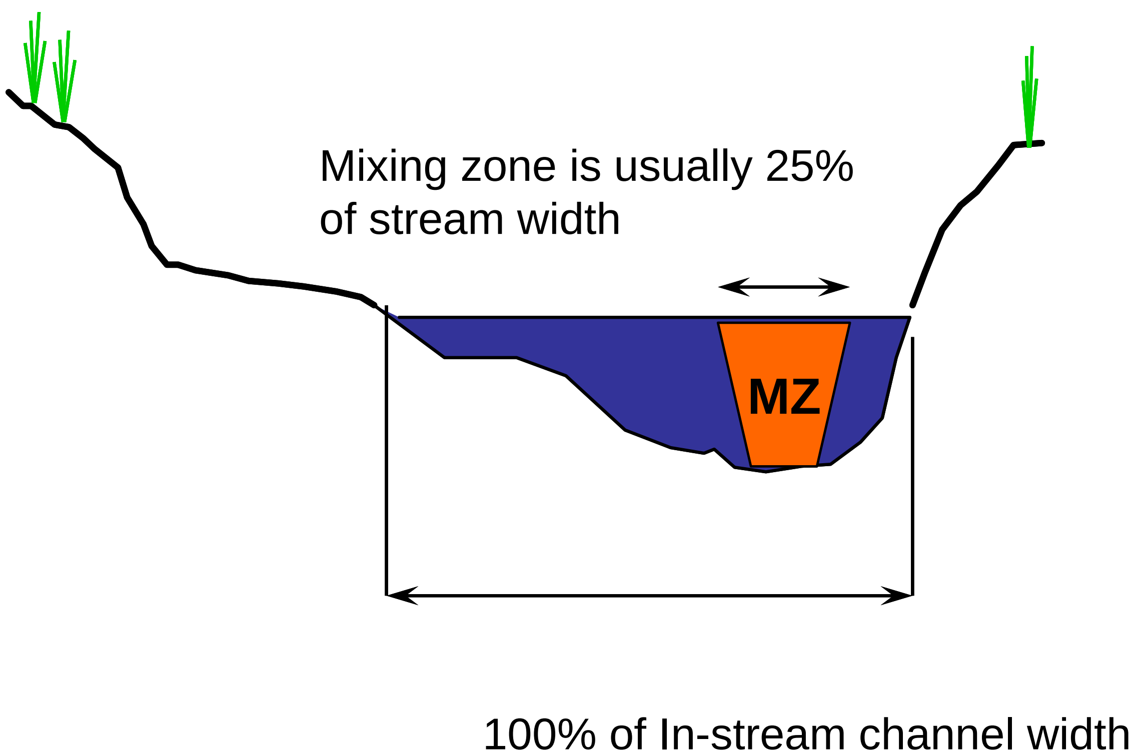
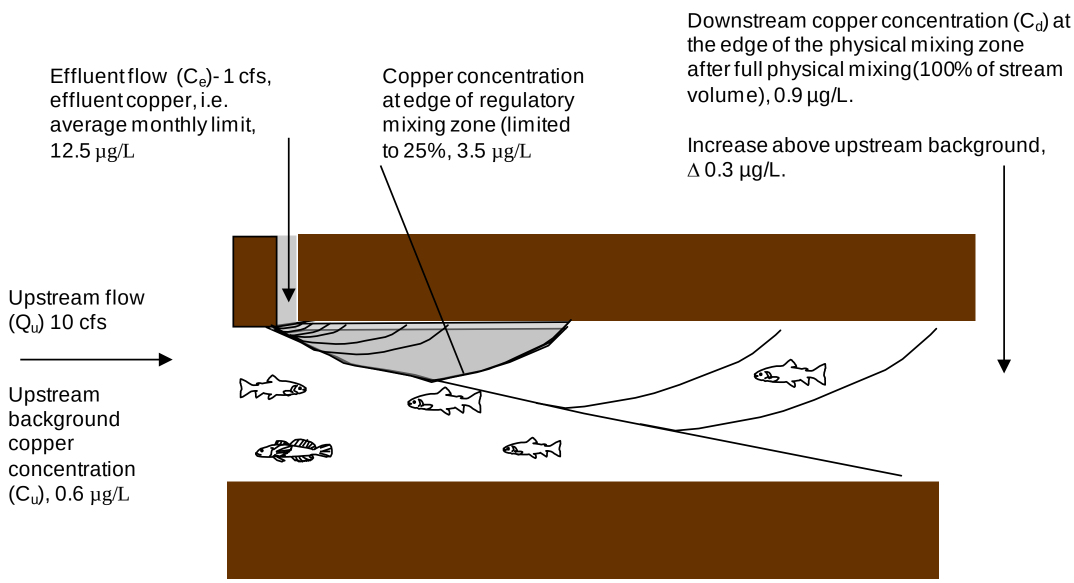



**Figure F-1. Illustration of an effluent mixing zone cross section, illustrating how an effluent plume containing copper or other chemicals (trapezoid) would be limited to a fraction of the actual receiving stream width.**




Calculated using simple mass balance equation

$$C_d = \frac{C_u Q_u + C_e Q}{Q_e Q_u}$$ where

$C_d$ is the concentration downstream of the effluent discharge, $C_u$ is the upstream concentration, $Q_u$ is the upstream flow, $C_e$ is the effluent concentration and $Q_e$ is the effluent flow.

**Figure F-2. Illustration of effluent limit calculations and resulting copper concentrations for a springtime, low-hardness scenario where the volume of the receiving water allowed to be used in calculated the effluent limits was limited to 25% of the actual stream volume.**

**Table F-1. Mixing zone demonstration in ESA waters which exceed either 25% volume or cross sectional area of a stream would require consideration of following elements:**

Definition of location, width, downstream extent (where should compliance be monitored). In open-water (reservoirs, lakes) describe where discharge-induced mixing ends;

Describe stream channel characteristics, including depth and velocity profiles at high and low flows. Present an interpretation of available suitable habitat for juvenile and adult salmon and steelhead either using simple fixed criteria comparisons (e.g., Bjornn and Reiser. 1991; Mebane 2000) or with habitat suitability curves(e.g., Maret et al. 2006).

Map habitat features within the mixing zone, including geomorphic channel units (pools, runs, riffles), presence of fish cover from predation, including overhanging vegetation, instream vegetation, woody debris or boulders. Describe habitat features in the affected reach context of the overall stream segment and any likely limiting habitat features for the area.

Describe measured or projected discharge and receiving water temperatures in the context of whether the effluents would represent an “attractive nuisance” by providing a thermal refuge and leading to disproportionately greater exposure of fish to effluents than would be expected based on spatial proportions. A difference of 3°C warmer in winter or 3°C cooler in summer between the effluent and receiving water respectively is considered sufficient to create a potentially harmful thermal attractant (Poole et al. 2001).

Show that the mixing zone is unlikely interfere with or block passage of fish or aquatic life. If mixing zone impinges on a large fraction of the zone of passage, e.g., more than 50% of the channel cross sectional area, then rigorous demonstration of passage adequacy by techniques such as telemetry may be needed. For copper, zone of passage is sufficient if at least 50% of channel cross sections (under critical conditions) have relative dissolved copper concentrations of $<0.6$ µg/L greater than background concentrations. Additionally, in at least 50% of channel cross sections, absolute dissolved copper should be no greater than that allowed by EPA’s (2007) biotic ligand model-based criteria.

Does not otherwise interfere with aquatic ecosystems (protect uses), as demonstrated through biomonitoring and WET testing.

Describe background, show that adjacent mixing zones do not overlap, evaluate whether the organisms would be attracted to the MZ.

Evaluate the size of mixing zone in relation to the availability of critical habitat for a species. Describe the extent (i.e., physical and temporal extents, including fraction of total).

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**Enclosure 1: National Marine Fisheries Service Comments on Environmental Protection Agency's Draft Aquatic Life Ambient Water Quality Criteria for Cadmium**

January 26, 2016

Enclosed are comments for the National Marine Fisheries Service (NMFS) on the Environmental Protection Agency's (EPA) Draft Aquatic Life Ambient Water Quality Criteria for Cadmium for species listed as threatened or endangered under the Endangered Species Act (ESA). Our comments take into account that we apply a threshold for insignificant effect under §9(a)(1) of the act, which states: *"...no one, public or private, can "take" an endangered species of fish or wildlife."* This is necessary because effects to individuals of an ESA-listed species can have significant implications for the persistence of the population they belong to. Thus the threshold that must be applied when evaluating effects to ESA-listed species is that "take" of an individual should not occur. Regardless of whether an ESA-listed species is more or less sensitive to a toxicant than common laboratory species in standards tests, the critical consideration for the protection of an ESA-listed species is the implication of losing the reproductive contribution of an individual for the persistence of the population.

## PRIOR ESA SECTION 7 CONSULTATIONS

Our West Coast Region Office have previously issues ESA Section 7 consultations with EPA on cadmium criteria proposed by the states of Oregon and Idaho. We refer EPA to these prior consultations and reference portions of these signed decisional documents with these comments. In particular, we include sections from both consultation documents that evaluate the suitability of EPA's national criteria methodology in arriving at guidelines that are protective of ESA-listed species (see NMFS Attachments 1 and 2).

### NMFS evaluation of Oregon's proposed water quality criteria for cadmium

At 2.1micrograms/liter (µg/L), EPA's freshwater acute guideline is slightly above Oregon's proposed criterion of 2.0 µg/L. NMFS determined that the Oregon criterion would jeopardize the continued existence of ESA-listed species occurring in that state. We understand that EPA Region 10 is currently working on a response to the reasonable and prudent alternative (RPA) proposed by NMFS to address this issue. The RPA requires EPA to disapprove the State of Oregon's acute criterion and recommend the state adopt an acute criterion derived using a more suitable approach, promulgating that criterion if necessary (see NMFS Attachment 3).

Specifically, several of the 96 hour LC50 data for ESA-listed species used in the derivation of the Oregon standard are below the criterion, so NMFS used these data to evaluate the implications on population growth rates. These analyses identified cases where population growth rates would be significantly altered on exposure to 2.0 µg/L cadmium (see NMFS Attachment 4).

EPA's chronic freshwater guideline for cadmium is also higher than the chronic criterion proposed by Oregon (0.73 µg/L vs 0.25 µg/L). NMFS analyses indicated exposure to 0.25 µg/L cadmium would result in sublethal effects, but the effects did not rise to the level of jeopardy. EPA's 0.73µg/L guideline is nearly three-fold Oregon's criterion and would be expected to result in more severe effects. However, the degree of severity cannot be inferred from the analyses in this consultation.

EPA's salt water acute and chronic cadmium guidelines are lower than those proposed by Oregon, and NMFS determined the Oregon-proposed values to not be a concern for Oregon's ESA-listed fish under NMFS jurisdiction. The NMFS determined the criteria were not likely to adversely affect ESA-listed sea turtles or large whales because their occurrence in waters affected by the criteria would be rare, infrequent, and transitory in nature and they would be unlikely to accumulate a significant amount of persistent pollutants such as cadmium because they primarily consume lower trophic-level prey.

**NMFS evaluation of Idaho's proposed water quality criteria for cadmium**

In 2006 Idaho proposed acute and chronic freshwater criteria of 1.3 and 0.6 μg/L, respectively. NMFS performed an independent analysis and, based on this analysis, concurred with EPA's assessment that these criteria were not likely to adversely affect Idaho's ESA-listed salmonids under NMFS jurisdiction (see NMFS Attachment 5). The application of the "take of an individual should not occur" threshold for effects in NMFS' independent analyses suggests that the Idaho criteria are protective of ESA-listed salmonids in other states. However, application of the criteria elsewhere would still require an analysis incorporating location-specific considerations.

## SPECIES NOT EVALUATED IN PRIOR CONSULTATIONS

**Sea Turtles**

The Oregon consultation concluded that ESA-listed sea turtles would be unlikely to accumulate a significant amount of cadmium specifically from state waters. However EPA's cadmium guidelines apply to all waters of the US, so exposures would occur throughout the US portion of sea turtle ranges. Further, cadmium accumulates in tissue with age, and sea turtles are understood to be very long lived species. For example, green turtles reach sexual maturity between 20 and 50 years of age. For such long lived species we would need to consider whether cadmium accumulation from US waters over a lifespan would reach tissue concentrations directly resulting in or contributing to adverse effects. Dietary exposure of the more omnivorous sea turtle species (i.e., leatherback, loggerhead) was a particular concern voiced by staff at the NMFS Southeast Regional Office

**Sturgeon and Smalltooth Sawfish**

Data on the effects of cadmium on smalltooth sawfish and Atlantic, Gulf, or shortnose sturgeon species are not available. In the absence of species-specific data, data for surrogate laboratory species are typically applied. For example, rainbow trout are commonly used in laboratory toxicity tests and, because they are cold water fish that are taxonomically closely related to ESA-listed salmonid species, they are considered suitable surrogate species for ESA-listed salmonids. Similarly, the fathead minnow is considered a suitable surrogate species for warm water fish. While there are some data for aquatic exposures of white surgeon to cadmium, an evaluation of ambient aquatic exposures alone would be inadequate to assess effects to ESA-listed sturgeon and smalltooth sawfish under NMFS jurisdiction. Like sea turtles, these species are long lived and dietary accumulation is likely a significant exposure pathway. For example, the lifespan of Atlantic sturgeon is 60 years and the lifespans of smalltooth sawfish and other sturgeon averages between 20-30 years, with Gulf and shortnose sturgeon maximum reported lifespans at 60 and 67 years, respectively. Further, sturgeon species use US fresh and marine waters exclusively and are known to ingest sediment (which may include particulate-bound cadmium originating from the water column) with their benthic prey.

**Corals**

Data on the effects of cadmium on ESA-listed coral species were not applied in the derivation of the cadmium water quality guidelines. A fertilization success study reported that success rate declined to 52% at cadmium concentrations of 5000 µg /L over a control rate of 78% success.[1] In another study, 14 day long cadmium exposures of coral resulted to increased antioxidant response at 5 µg/L after 4 days and mortality at 50 µg /L after 2-3 days.[2] A study evaluating the zooxanthellate sea anemone, *Aiptasia pulchellaas,* as a surrogate test species for corals reported 6 hour EC50s and 96 hour LC50s effects thresholds ranging from 249 to 2250 µg /L cadmium.[3] While these data suggest that the EPA guidelines for cadmium in marine waters are protective of coral species, this body of evidence is severely limited by the absence of data on colonization and recruitment, wound recovery, and predation activity.

## Conclusion

In light of the substantial data gaps and the concerns expressed in prior consultations regarding EPA's guideline development methodology, EPA needs to work with NMFS to conduct a more thoughtful evaluation of the implications of their guidelines for ESA-listed species and apply a more suitable analysis in guideline derivation, taking existing assessments of state-proposed criteria into consideration. New data are needed, but its generation needs to strategically target issues identified in prior consultations and those stated above.

**Remarks on EPA's approach to addressing its obligations under the ESA**

NMFS understands that EPA considers its development of water quality guidelines to not be subject to ESA consultation. EPA's reliance on ESA section 7 consultation only when the agency approves state-proposed water quality criteria results in a piecemeal approach when considering implications of such guidelines for broadly ranging species. The segmentation of an action under ESA section 7 leads to an incomplete consideration of the effects of the action that is legally vulnerable. Both agencies need to agree on and implement an assessment strategy that takes into account the aggregate effects of EPA's authorizations of state-proposed water quality criteria such that EPA can ensure that these authorizations, taken together, do not jeopardize the continued existence of ESA-listed species or adversely modify designated critical habitat. Given the scope of the guidelines, the conclusions of such an assessment and any associated implementation guidance would need to have the same authority/regulatory implications of a section 7 consultation.

[1] Reichelt-Brushett and Harrison, Coral Reefs (2005) 24: 524–534
[2] Mitchelmore et al., Aquatic Toxicology 85 (2007) 48–56
[3] Howe et al, Marine and Freshwater Research (2014) 65, 551–561

***Submitted Via State Public Comment Portal***

May 7, 2024

Casey Sixkiller
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**Re: Comments on Washington's Proposed Updates to Aquatic Life Toxics Criteria, WAC 173-201A-240 (CR-102)**

Dear Ms. Koberstein and Regional Administrator Sixkiller,

Please accepted the following public comments submitted on behalf of the Center for Biological Diversity (Center) and its 1.7 million members and supporters to the Washington Department of Ecology's (Ecology) proposal to revise Washington's aquatic life toxics criteria, WAC 173-201A-240.

The Center is concerned that the proposed criteria provide insufficient protections for federally listed endangered and threatened species and, in consideration of prior national, Oregon, and Idaho Section 7 consultation findings, likely violates the Endangered Species Act's prohibition on the take of listed species. The Center, therefore, urges Ecology to revisit its proposed criteria for the benefit of endangered and threatened species and revise downward those criteria to levels that meet the obligations of the Clean Water Act to support the most sensitive aquatic life uses[1] and the Endangered Species Act's requirement that "endangered species [] be afforded the highest of priorities." *Tennessee Valley Authority v. Hill*, 437 U.S. 153, 174 (1978).

## I. The Methodologies Used by Ecology and EPA for Deriving Water Quality Criteria Are Legally Deficient and Under-Protective of Endangered Species and Critical Habitats

The presence of toxic pollutants in waterways has a significant impact on aquatic and aquatic-dependent species' survival. According to the National Marine Fisheries Service (NMFS), "degraded water quality has been one of the contributing factors for the decline of almost all of

[1] *See* 40 C.F.R. § 131.11(a) (criteria must support the most sensitive use).

the anadromous fish species NMFS has listed since the mid-1980s."[2] Cyanide, cadmium, and mercury are three toxic pollutants that present significant threats to endangered and threatened aquatic species and their critical habitats.[3]

Over the last two decades, a series of lawsuits and consultations regarding EPA's national criteria and its approval of state standards and criteria for various pollutants—including cyanide, cadmium, and mercury—have raised profound concerns regarding the overall approaches that EPA utilizes in reviewing and approving water quality criteria; these cases also raise concerns about the inadequate and antiquated methodologies EPA used to establish national water quality criteria. *See, e.g., Center for Biological Diversity v. EPA*, Case No. 22-138, 2023 U.S. Dist. LEXIS 145674 (D. Ariz. Aug. 18, 2023) (finding that EPA acted unlawfully when it failed to engage in Endangered Species Act Section 7 consultation prior to issuing nationwide water quality criteria for cadmium and vacating EPA's 2016 chronic freshwater cadmium criterion); *Northwest Environmental Advocates v. National Marine Fisheries Service et al.*, Case No. 10-907-BR (2010) (dealing with the Oregon's Endangered Species Act consultation history and failures); *Northwest Environmental Advocates v. The National Marine Fisheries Service et al*., Case No. 13-00263-DCN (2013) (dealing with the Idaho's Endangered Species Act consultation history and failures).

The Center hereby attaches and incorporates into these comments past biological opinions and draft biological opinions and request they be made part of the record for this rulemaking as well as incorporated into EPA's review of Ecology's ultimate submission. The biological opinions describe severe methodological flaws and inadequate approaches that have inevitably yielded legally insufficient and under protective criteria. Each document included provides information that can guide Ecology's development of its criteria. More recent science, however, suggests the need for even more protective standards to fully comply with the Endangered Species Act.

Even further, because Washington is downstream of a number of states with known aquatic toxic pollution issues, including Idaho, Oregon, and even small portions of Wyoming and Montana, some of its waters are already receiving significant pollutants from upstream states, which raises concerns about cumulative impacts, and suggests even more stringent criteria are required to address pollution in a legally sufficient manner.[4] While in theory, Clean Water Act section 303(d)

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[2] NATIONAL MARINE FISHERIES SERVICE, DRAFT ENDANGERED SPECIES ACT SECTION 7 CONSULTATION BIOLOGICAL OPINION & CONFERENCE OPINION ON THE U.S. ENVIRONMENTAL PROTECTION AGENCY'S APPROVAL OF STATE OR TRIBAL, OR FEDERAL NUMERIC WATER QUALITY STANDARDS FOR CYANIDE BASED ON EPA'S RECOMMENDED 304(A) AQUATIC LIFE CRITERIA, 270 (2010) [hereinafter NMFS National Cyanide Draft BiOp].

[3] While these comments focus on the cyanide, cadmium, and mercury pollution and Washington's associated criteria, several additional pollutants are of concern to the Center. We request that Washington finalize toxics criteria across the board that are adequately protective of endangered and threatened species and their critical habitats.

[4] *See* EPA, Downstream Protection Guidance, Goal: Illustrate Considerations and Procedures Associated with Incorporating Downstream Protection into Development of Numeric Criteria, at 7 (2014) (describing that to develop downstream protections, the state should "establish numeric criteria in the receiving waterbody and build upstream")*; see also* 40 C.F.R. § 131.10(b) (a state "shall ensure that its water quality standards provide for the attainment and maintenance of water quality standards of downstream waters").

total maximum daily loads (TMDLs) are the mechanism to address total pollutant loading, Washington's TMDL program is largely moribund, it issues very few TMDLs for toxic pollutants, and its TMDLs do not take into consideration the cumulative effects of multiple toxic pollutants. For these reasons, Washington's water quality criteria for toxic pollutants must address the need to provide full protection of these downstream waters.

While the Center is generally supportive of Ecology's proposal to establish more stringent criteria, the proposed criteria still raise concerns regarding their effects on Washington's threatened and endangered species, including salmonids, southern resident orcas, and amphibians. Illustratively, for example, Washington's proposed chronic cyanide criteria is significantly higher than the level recommended in Fish and Wildlife Service's (FWS) biological opinion on EPA's national 304(a) cyanide criteria for bull trout. The proposal also does not appear to account for or address amphibian sensitivity to these toxics—another issue identified in FWS's biological opinion on EPA's national 304(a) criteria for cyanide.

## II. Washington's Proposed Cyanide Water Quality Criteria are Not Adequately Protective of Listed Species or Critical Habitats

| **Cyanide, Freshwater** | **Proposed Acute (μg/L)** | **Proposed Chronic (μg/L)** | **ESA Consultation History, if Applicable** |
|---|---|---|---|
| **Idaho** | 22 | 5.2 | Both received a jeopardy determination[5] |
| **EPA** | 22 | 5.2 | Both received a draft jeopardy determination[6] |
| **FWS Draft BiOp** | 13.77 | 0.68 | Recommended level for bull trout[7] |
| **NMFS Draft BiOp** | None Provided | None Provided | |
| **WA Ecology** | 12 | 2.7 | Yet to be fulfilled. |

a. Salmonids

Past consultations by FWS and NMFS on toxics criteria nationally and standards in several Pacific Northwest states indicate that the presence of cyanide threatens a number of federally listed salmonids species found in Washington, including bull trout, Chinook salmon, chum salmon, coho salmon, sockeye salmon, and steelhead.[8]

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[5] NATIONAL MARINE FISHERIES SERVICE, ENDANGERED SPECIES ACT SECTION 7(A)(2) BIOLOGICAL OPINION AND MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT ESSENTIAL FISH HABITAT (EFH) CONSULTATION, 299 (2014) [hereinafter NMFS Idaho Toxics BiOp].

[6] FISH AND WILDLIFE SERVICE, DRAFT BIOLOGICAL OPINION ON EPA'S PROPOSED PROGRAM OF CONTINUING APPROVAL OR PROMULGATION OF NEW CYANIDE CRITERIA IN STATE AND TRIBAL WATER QUALITY STANDARDS, 298 (2010) [hereinafter FWS National Cyanide Draft BiOp].

[7] *Id.* at 304.

[8] NMFS National Cyanide Draft BiOp at 270.

On the basis of these past actions, the bull trout appears to be the most sensitive of Washington's federally endangered and threatened species that is threatened by presence of cyanide. As detailed in the above chart, Ecology's proposed criteria for cyanide are higher than levels established through past biological opinions as necessary to adequately protect bull trout as required by the Endangered Species Act.[9]

Cyanide has been shown to cause reduced growth rates, reproductive performance, and survival in bull trout.[10] High chronic levels of cyanide can reduce the number of eggs spawned by females, reduce the number of eggs that hatch, and drastically reduce the survivorship of young fish. In the biological opinion for EPA's national 304(a) cyanide criteria, FWS found that exposure to bull trout at the chronic criterion proposed by EPA would likely "substantially reduce their reproduction" and that exposure at the proposed acute criterion would likely cause "substantial reductions in survival."[11] Based on this "magnitude of adverse effects," FWS found that the species was likely to be extirpated from the waters where they are exposed to cyanide toxicity at either criterion amount and suggested a chronic freshwater criterion of 0.68 µg/L—significantly lower than the chronic freshwater criterion of 2.7µg/L for cyanide the Ecology proposes here.

Washington should, therefore, revisit its proposed criteria and revise downward to a proposed chronic freshwater criterion for cyanide of no more than 0.68 µg/L, more so if updated science shows that a more stringent standard is necessary to protect bull trout and other salmonid populations; the Center does not take immediate issue with Washington's proposed acute freshwater criteria but request that it be revised as necessary subject to the outcome of further Washington-specific Endangered Species Act consultation activities.

b. Oregon Spotted Frog

In its 2010 consultation with EPA regarding national 304(a) water quality criteria for cyanide, FWS noted a lack of data for effects of cyanide on amphibian species but concluded that because amphibians are among the most sensitive species for a significant number of the pollutants examined, it is likely that amphibian species are highly sensitive to cyanide.[12] There, FWS used data for relative sensitivity of amphibians to rainbow trout, since rainbow trout is a species often used for criteria development.[13] Based on this analysis, FWS concluded that amphibian species are estimated to be as or more sensitive to cyanide than rainbow trout and thus likely to be adversely affected by exposure to cyanide at EPA's suggested chronic criterion of 5.2µg/L.

Since that consultation was completed, the Oregon spotted frog was listed as a threatened species in 2014 and has two critically imperiled populations in Washington.[14] The Oregon spotted frog is considered "the most aquatic native frog species in the Pacific Northwest (PNW)."[15] In making

[9] FWS National Cyanide Draft BiOp at 304.
[10] FWS National Cyanide Draft BiOp at 221.
[11] *Id.*
[12] *Id.* at 250.
[13] *Id.*
[14] 79 Fed. Reg. 51,658 (Aug. 29, 2014).
[15] *Id*. at 51,661.

its listing determination, the FWS determined that toxic chemicals pose a hazard to the Oregon spotted frog.[16] Yet, Ecology does not even appear to have included the Oregon spotted frog on its list of relevant Endangered Species Act listed species.[17] Cyanide criteria must therefore be adjusted accordingly following Endangered Species Act consultation.

c. <u>Orcas</u>

Southern Resident Orcas could also be indirectly affected by Ecology's proposed cyanide criteria due to the possible reduction in salmonid populations.[18] Salmon, particularly Chinook salmon, are a key food source for the southern resident orcas and if proposed criteria harm salmonids, it is likely that the orcas will suffer as well. In NMFS consultation for EPA's national 304(a) cyanide criteria, the agency found that EPA's criteria would "reduce freshwater production of all listed salmon species, as well as non-listed salmon species where cyanide concentrations are allowed to reach EPA's recommended aquatic life criteria concentrations."[19]

## III. Washington's Cadmium Water Quality Criteria are Not Adequately Protective of Listed Species and Critical Habitats

Cadmium is one of the most toxic metals to fish and can have various effects on aquatic organisms, including spinal deformities, inhibited respiration, immobility, and population alterations.[20] It can also cause neurotoxic effects in fish, manifesting as altered behavior, reduced growth, reproductive failure, and death.[21] Salmonids are particularly sensitive to cadmium pollution.[22] The principal acute effect of cadmium is gill toxicity, which causes an inability to breathe in aquatic organisms. Cadmium toxicity increases with water temperature.[23]

a. <u>Freshwater Cadmium</u>

| Cadmium, Freshwater | Proposed Acute (µg/L) | Proposed Chronic (µg/L) | ESA Consultation History, if Applicable |
|---|---|---|---|
| **Oregon** | 2.0 | 0.25 | Acute standard received jeopardy determination.[24] Both standards likely to adversely affect listed species. |

[16] *Id*. at 51,689-90.
[17] *See* Washington Dep't. of Ecology, Proposed Updates to Aquatic Life Toxics Criteria, WAC 173-201A-240 Technical Support Document, 31-32 (2024) [hereinafter Ecology Technical Support Doc].
[18] NMFS National Cyanide Draft BiOp at 271.
[19] *Id.* at 256.
[20] NATIONAL MARINE FISHERIES SERVICE, JEOPARDY AND DESTRUCTION OR ADVERSE MODIFICATION OF CRITICAL HABITAT ENDANGERED SPECIES ACT BIOLOGICAL OPINION FOR ENVIRONMENTAL PROTECTION AGENCY'S PROPOSED APPROVAL OF CERTAIN OREGON ADMINISTRATIVE RULES RELATED TO REVISED WATER QUALITY CRITERIA FOR TOXIC POLLUTANTS, 270 (2012) [hereinafter NMFS OR Toxics BiOp].
[21] *Id.* at 271.
[22] *Id.* at 270.
[23] *Id.* at 271.
[24] *Id.* at 547

| | | | |
|---|---|---|---|
| **Idaho** | 1.3 | 0.6 | NMFS independent analysis: standards not likely to adversely affect ESA listed Chinook salmon, sockeye salmon, or steelhead in the state, but noted that determination was location specific[25] |
| **EPA 2016** | 1.8 | [0.72] | No consultation.[26] Chronic criterion vacated to 2001 value; acute criterion levels remain in place but have been remanded back to EPA by court order[27] |
| **EPA 2001** | [2.0] | 0.25 | No consultation. |
| **WA Ecology** | 1.3 | 0.41 | Yet to be fulfilled. |

For cadmium, Ecology proposes a freshwater acute criterion of 1.3µg/L and a chronic freshwater criterion of 0.41 µg/L. Since EPA's nationwide 304(a) freshwater cadmium criterion was vacated by court order, the maximum concentration reverted back to the 2001 criterion of 0.25 µg/L; at a minimum, Washington must do the same.

However, based on the outcome of Endangered Species Act consultation, these criteria must be set at a level that is protective of federally listed species in Washington. Comparatively, the FWS biological opinion for Oregon toxics stated that "chronic exposure to cadmium at the proposed chronic level [of 0.25µg/L] is considered to have adverse effects to all bull trout potentially exposed by reducing their fitness through a reduction in growth."[28] The NMFS biological opinion for Oregon similarly found that "listed species exposed to waters equal to the acute or chronic [cadmium] criteria concentrations will suffer acute and chronic toxic effects."[29]

a. Saltwater Cadmium

| Cadmium, Saltwater | Proposed Acute (µg/L) | Proposed Chronic (µg/L) | ESA Consultation History, if Applicable |
|---|---|---|---|
| **Oregon** | 40 | 8.8 | Listed species will suffer acute or chronic toxic effects including mortality (moderate intensity) and sublethal effects (moderate intensity)[30] |
| **EPA 2016[31]** | 33 | 7.9 | |
| **WA Ecology 2024** | 33 | 7.9 | Yet to be fulfilled. |

[25] National Marine Fisheries Service, Comments on Environmental Protection Agency's Draft Aquatic Life Ambient Water Quality Criteria for Cadmium, 2 (Jan. 26, 2016).
[26] Center for Biological Diversity, EPA Approves Dangerous Water Quality Standards for Cadmium (April 1, 2016), https://www.biologicaldiversity.org/news/press_releases/2016/cadmium-04-01-2016.html.
[27] *Ctr. For Biological Diversity v. United States Env't Prot. Admin*, No. CV-22-00138-TUC-JCH, 2023 U.S. Dist. LEXIS 145674, at *44 (D. Ariz. Aug. 18, 2023).
[28] NMFS Oregon Toxics BiOp at 193.
[29] *Id.* at 270.
[30] *Id.* at 367.
[31] ENVIRONMENTAL PROTECTION AGENCY, AQUATIC LIFE AMBIENT WATER QUALITY CRITERIA CADMIUM – 2016, XV (2016).

Ecology's proposed change to saltwater cadmium criteria is also likely to put threatened and endangered species at risk. Ecology proposes to set saltwater cadmium criteria at EPA's 304(a) chronic criterion of 33µg/L and acute criterion of 7.9µg/L. During the peer review of EPA's 304(a) criteria, it was pointed out that the development of these criteria was based on insufficient toxicity data for effects on anadromous salmon and that "only one study evaluated Cd toxicity in coho salmon smolts in saltwater conditions, and this was at nearly full seawater strength."[32] This was a concern because anadromous salmonids encounter cadmium at lower salinities. It is important to better understand the impact of varying levels of salinity on cadmium toxicity of anadromous fish species and incorporate those findings into Washington's criteria.

The same peer review also noted that sea level rise associated with climate change is likely to cause saltwater intrusion into salmonid spawning habitat making it particularly important to understand how salinity affects cadmium toxicity.[33] Comparatively, in NMFS's biological opinion for Oregon's cadmium criteria, the agency pointed out various issues with EPA's criteria derivation methods, including for saltwater cadmium.[34] Therefore, relying on the EPA's 304(a) will not necessarily result in adequate protection for threatened and endangered species and their critical habitats in Washington waters.

## IV. Washington's Existing Mercury Water Quality Criteria are Not Adequately Protective of Listed Species or Critical Habitats and Must be Updated

Washington should learn from Idaho's mistakes and move forward with updating its water quality criteria for mercury.[35] In Idaho, which Ecology cites as a reason for not proceeding with amended mercury criteria at this time, EPA recently issued a proposed rule providing for both tissue and water column criteria for mercury.[36] The proposed chronic total mercury criteria are 0.225 µg/kg wet weight for muscle fish tissue, 0.162 µg/kg wet weight for whole body fish tissue, and 0.0021 µg/L for water column values.[37] In so doing, EPA asserted that these results were consistent with reasonable and prudent alternatives in the Services' biological opinions, and explained that it is important to include both a tissue and water column value in mercury and methylmercury criteria.[38]

In contrast, Washington is not only proposing to neglect updating its mercury criteria through this rulemaking but, in doing so, it is continuing to rely on an outdated freshwater chronic criterion which measures the proposed water column value at 0.012 µg/L. That is insufficient. First, "[b]ecause tissue measurements provide a more direct measure of toxicity for bioaccumulative pollutants such as mercury, . . . it appropriate to establish tissue criteria for these pollutants. However, criteria expressed as organism tissue concentrations can prove challenging

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[32] ENVIRONMENTAL PROTECTION AGENCY, EPA RESPONSE TO EXTERNAL PEER REVIEW COMMENTS ON THE DRAFT AQUATIC LIFE AMBIENT WATER QUALITY CRITERIA FOR CADMIUM, 39 (2015).
[33] *Id.*
[34] NMFS OR Toxics BiOp at 366-367.
[35] *See, e.g., Northwest Environmental Advocates et al. v. United States Environmental Protection Agency*, Case No. 13-00263-DCN (Memorandum Decision and Order, ECF No. 103, July 19, 2021).
[36] *See* EPA, *Mercury Criterion to Protect Aquatic Life in Idaho*, 89 Fed. Reg. 24,758 (April 9, 2024).
[37] *Id*. at 24,774.
[38] *Id*. at 24,762, 24,768.

to implement in CWA programs such as NPDES permitting and Total Maximum Daily Loads (TMDLs) because these programs typically demonstrate that water quality standards are met by using a water column concentration to calculate a load-based effluent limit or daily load, respectively."[39] Both are needed.

Second, per Idaho's earlier FWS biological opinion, which Ecology quotes in its TSD at 82, "[b]ased on the above information, implementation of the proposed chronic criterion for mercury is likely to adversely affect growth, reproduction, and behavior in the bull trout throughout its distribution in Idaho." Idaho's proposed freshwater chronic criterion was 0.012 µg/L or the same as Washington's current criterion. This means that Washingtons mercury criteria are, a minimum, likely not to be sufficiently protective of bull trout.

## V. EPA Methodologies for Derivation of Water Quality Criteria Do Not Prevent Adverse Effects to Listed Species and Critical Habitats

To the extent that Ecology based its proposed criteria on EPA's methodology, its analysis will suffer from the same issues as EPA's methodology—issues that are detailed in the NMFS biological opinions for EPA's national 304(a) cyanide criteria and Oregon's toxics criteria. The Center appreciates Ecology's attempts to account for some shortcomings in EPA's methodology by utilizing alternative derivation methods for some toxics and by using the 1st percentile of the genus toxicity data distribution rather than the 5th percentile. However, considering the extensive flaws underlying the toxicity data developed by EPA, using the 1st percentile of that data is not sufficient to protect endangered and threatened species.

For the freshwater acute cadmium criterion, for example, Ecology appears to be using the same derivation methods as EPA's recommendation;[40] for its chronic cadmium criterion, it used an EPA dataset and the 1st percentile of the toxicity distribution.[41] Although using the 1st percentile is more protective of species than the 5th, it is possible that issues in the underlying data still would not allow for a sufficiently protective calculation. Additionally, as discussed above, the proposed chronic cadmium criterion is in excess of the EPA criteria of 0.25µg/L, which is the current nationwide criteria following vacatur of EPA's 2016 criteria.

For cyanide, Ecology used new science in developing its proposed acute criterion, and an "acute to chronic" (ACR) ratio to develop its proposed chronic criterion because it lacked the toxicity data needed to calculate a chronic criterion using other methods.[42] The ACR is the ratio of the mean $LC_{50}$ (concentration causing 50% lethality following acute exposure) for the species to the concentration following chronic exposure that causes a level of adverse effect that is the threshold of unacceptability.[43] Since the ACR was calculated by EPA and is based on underlying values that could suffer from the flaws in EPA's methodology highlighted by NMFS in its national 304(a) cyanide and Oregon toxics biological opinions, it is possible that the values proposed by Ecology reflect some of those issues as well.

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[39] *Id*. at 24,762.
[40] Ecology Technical Support Doc. at 60.
[41] *Id.* at 62.
[42] *Id.* at 127–128.
[43] NMFS National Cyanide Draft BiOp at 245.

Importantly, EPA's methodology for calculating toxicity values at which adverse effects occur *does not* adequately account for compounding stressors such as temperature, dissolved oxygen, and others on the responses of aquatic life to toxics.[44] In its biological opinion for Idaho's toxics standards, FWS recommended that any new standards be calculated "using a temperature/toxicity correlation"[45] to account for the inverse relationship between cyanide toxicity and temperature.[46] Dissolved oxygen is also important to account for because in environments with less than optimal dissolved oxygen, fish compensate by increasing gill movement and ventilation volume to maintain adequate oxygen volumes. Since cyanide is a powerful asphyxiant, additional cyanide in waters with low dissolved oxygen further stresses fish and reduces the lethal concentration at which survival is expected.[47] In the NMFS biological opinion for the national 304(a) cyanide criteria, the agency pointed out that EPA's attempts to "avoid confounding factors" in their analysis that prevents them from replicating realistic conditions in the wild.[48]

It is not clear whether or to what extent Ecology accounted for the increased toxicity of cyanide at low temperatures. This is an important consideration, particularly for salmonids that spawn in cold waters and could face serious consequences from increased toxicity of cyanide at these low temperatures. It is also unclear whether the proposed criteria accounted for the impact of low dissolved oxygen or concurrent exposures with other contaminants and stressors.

## VI. Conclusion

Cyanide, cadmium, and mercury pollution threatens Washington's many endangered and threatened aquatic species. The Center urges Ecology to propose criteria that are sufficiently protective of Washington's federally protected endangered and threatened species, including by taking into consideration toxic pollution from upstream states and accounting for EPA's methodological limitations.

Please contact Hannah Connor at hconnor@biologicaldiversity.org with any questions.

Sincerely,

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[44] *Id.* at 266.
[45] FISH AND WILDLIFE SERVICE, BIOLOGICAL OPINION FOR THE WATER QUALITY STANDARDS FOR NUMERIC WATER QUALITY CRITERIA FOR TOXIC POLLUTANTS (2015) at 277 [hereinafter FWS Idaho Toxics BiOp].
[46] *Id*. at 143.
[47] NMFS National Cyanide Draft BiOp at 221.
[48] *Id.* at 266.

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