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).

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[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

[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.

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[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.

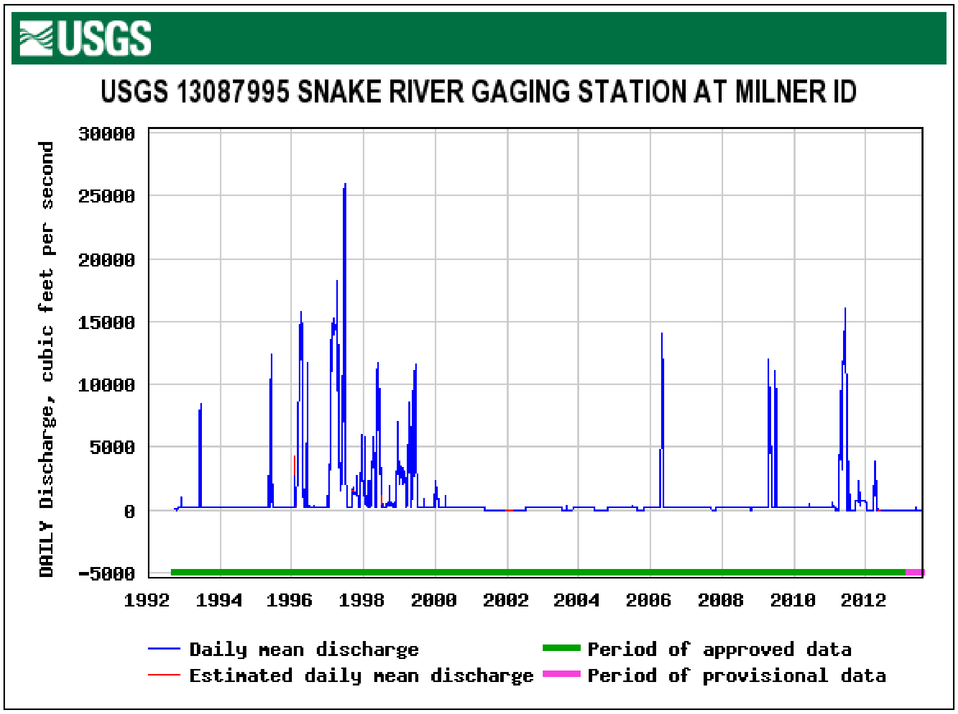

Figure 1. Snake River Flows at Milner Dam from 1993 (time of listing) through early 2013.

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

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[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 19th and early 20th 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 19th 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).

[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).

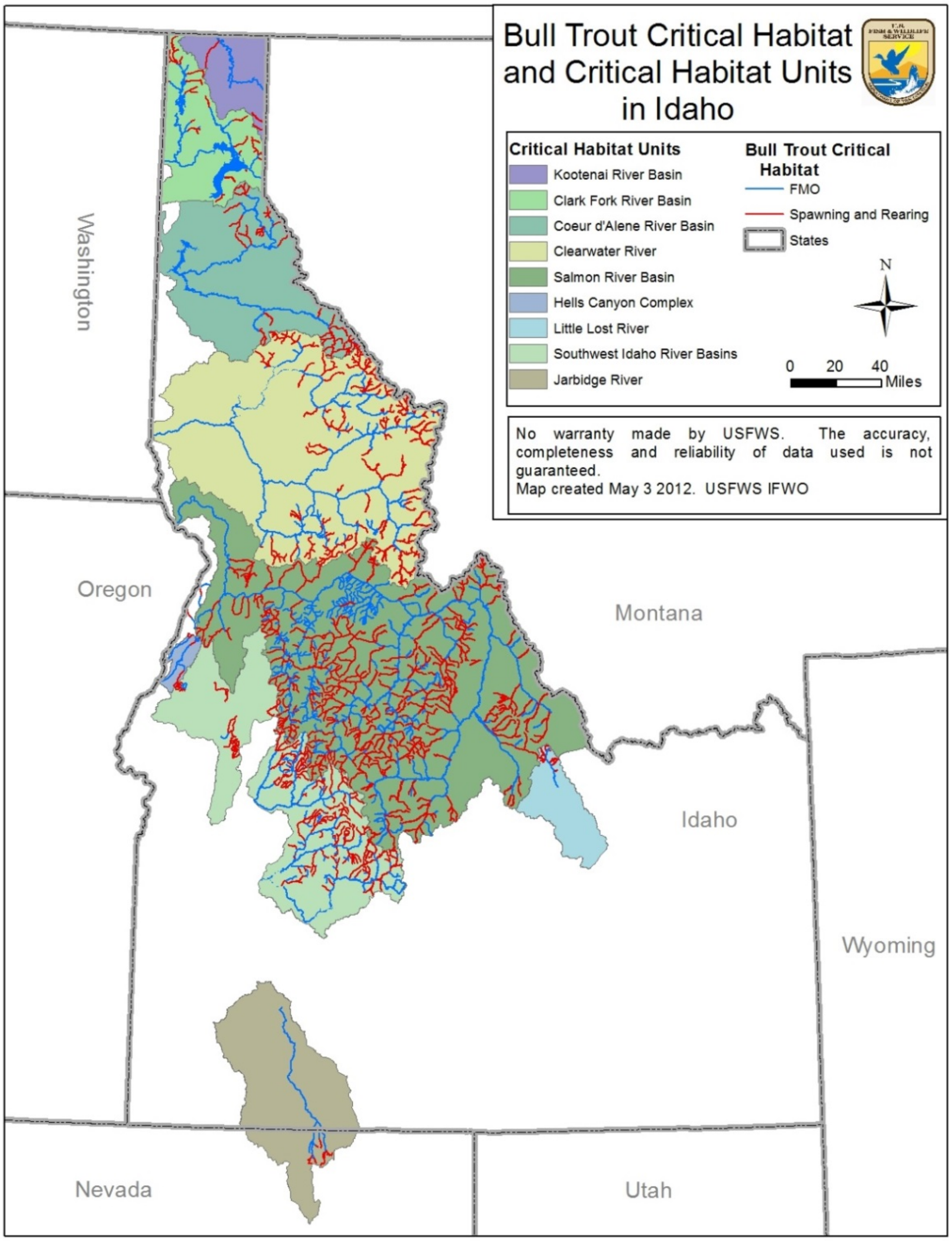

Figure 2. Bull trout critical habitat in Idaho, by Critical Habitat Unit and type of designation (i.e., spawning and early rearing or foraging, migrating, and overwintering.

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.

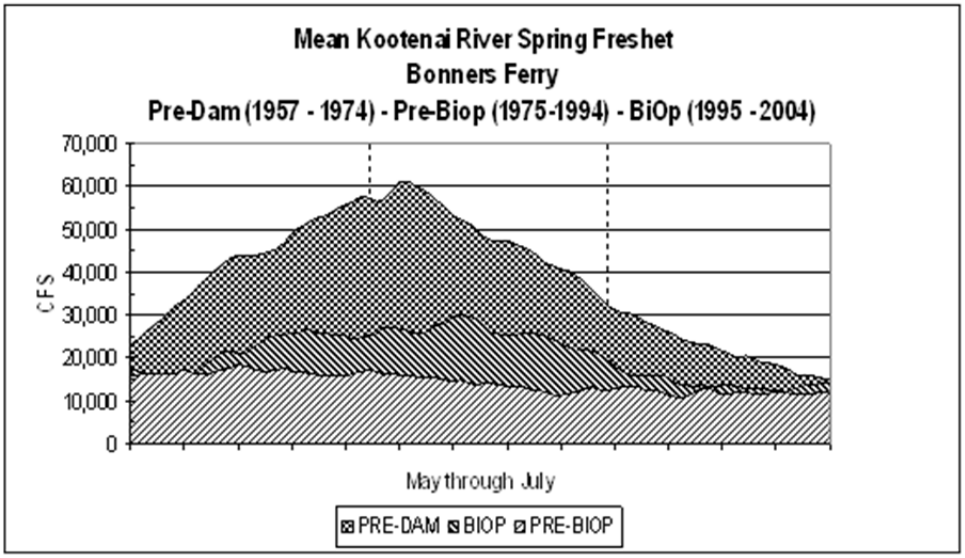

Figure 3. Mean seasonal (May through July) hydrograph (calculated; Bonners Ferry) for pre-dam (1957 – 1974), pre-biological opinion (BiOp) (1975-1994), and BiOp (1995-2004).

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 = $\frac{4\ \mu g/L}{4\mu g/L/TU} + \frac{0\ \mu g/L}{8\mu g/L/TU}$ =1 TU, (obviously, for a single substance), *or*

2 µg/L Cd + 4 µg/L Cu = $\frac{2\mu g/L}{4\mu g/L/TU} + \frac{4\mu g/L}{8\mu g/L/TU}$ =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.

Hardness

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 >≈ pH >Ca > Na ≈ alkalinity ≈ 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 (>≈ 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 ≥ 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.

[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.

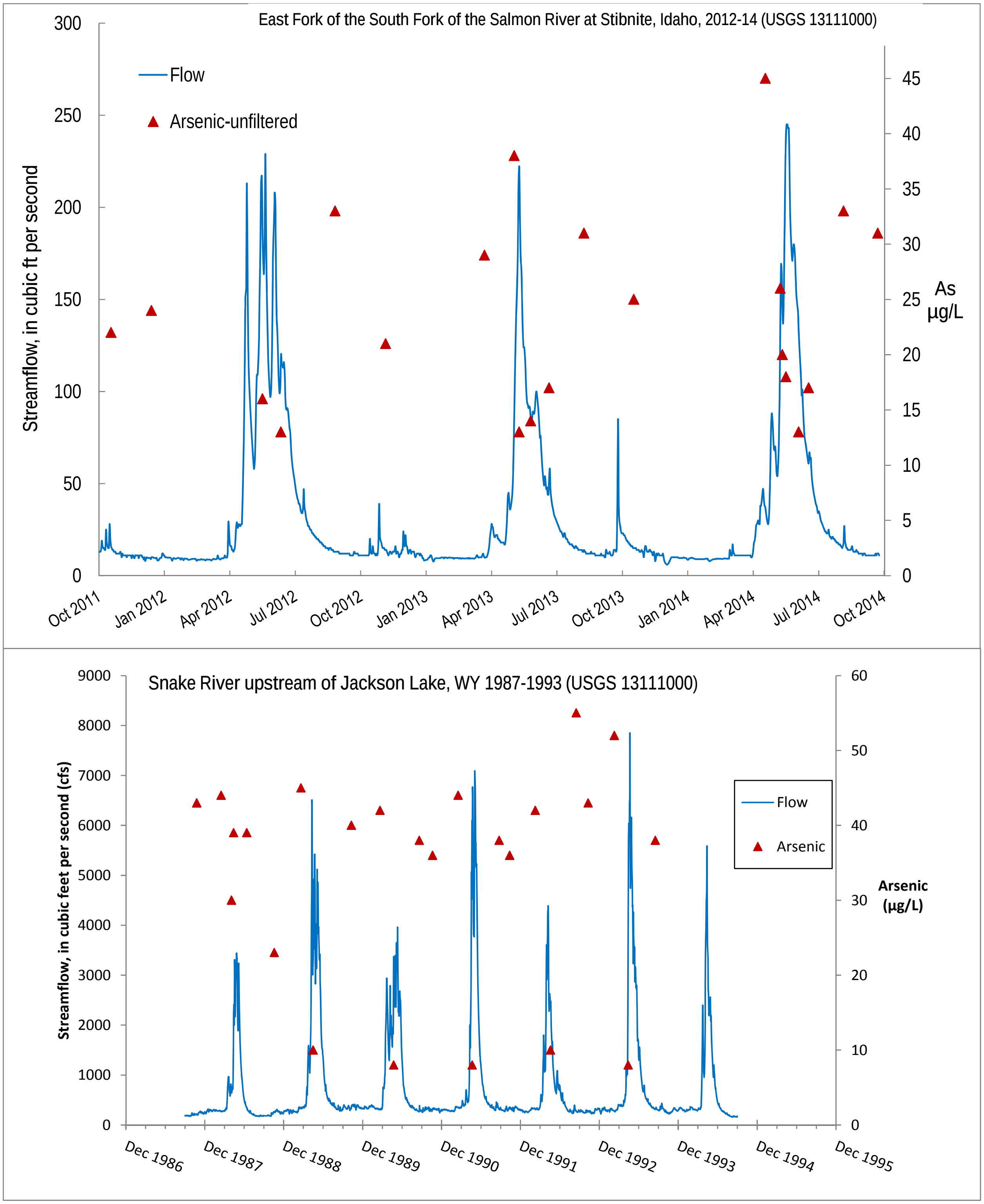

Figure 4. Arsenic versus streamflow time series from two river settings, in which arsenic sources are from groundwater, and close to 100 percent of the total arsenic is present in a dissolved form.

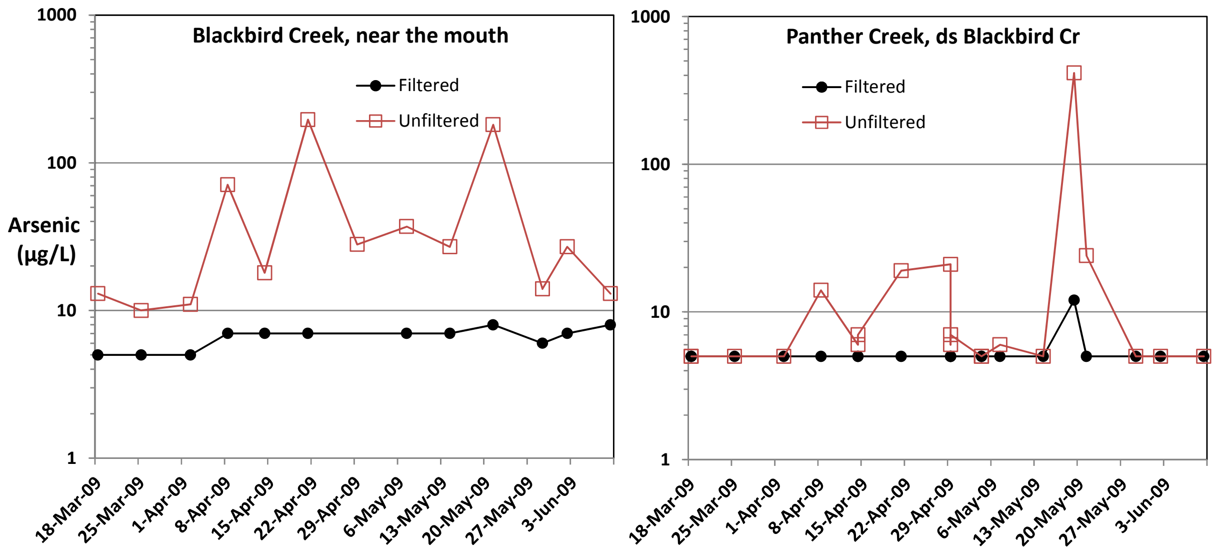

Figure 5. Unfiltered ("total") and filtered ("dissolved") arsenic concentrations from stream settings in which arsenic sources are mostly derived from snowmelt runoff remobilization of arsenic-laden particles from the streambed and stream banks on Panther Creek and its tributary, Blackbird Creek, in 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 (5th and 9th 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.

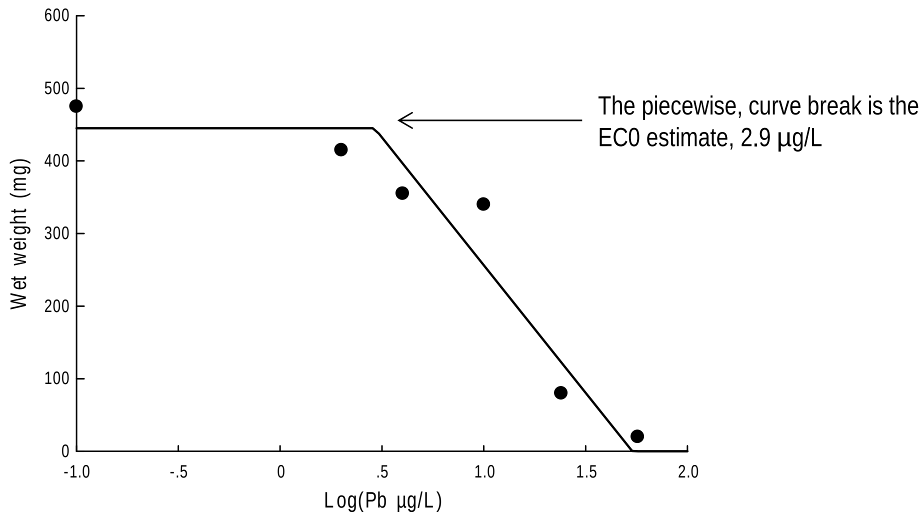

Figure 6. *Lymnaea stagnalis* growth, under different lead (Pb) exposures. Data taken from Brix et al (2012) curve fitting was done using the nonlinear regression, piecewise regression function in the Toxicity Response Analysis Program (TRAP) (Erickson 2010).

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.

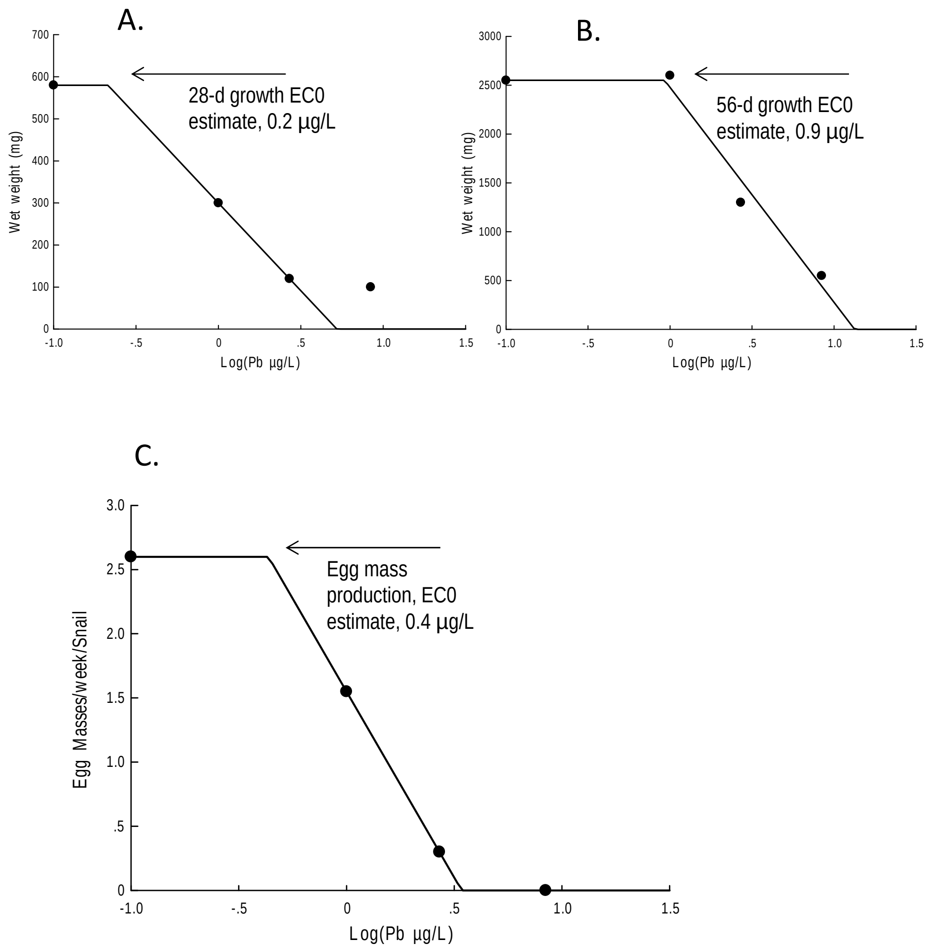

Figure 7. ***Lymnaea stagnalis*** **growth, under different lead (Pb) exposures. Data taken from Munley et al. (2013), curve fitting was done using the nonlinear regression, piecewise regression function in the Toxicity Response Analysis Program (TRAP) (Erickson 2010).**

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.

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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:

Acute zinc criterion (µg/L) = $e^{(0.8473[\ln(\text{hardness})]+0.884)}*0.978$

Chronic zinc criterion (µg/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

[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.

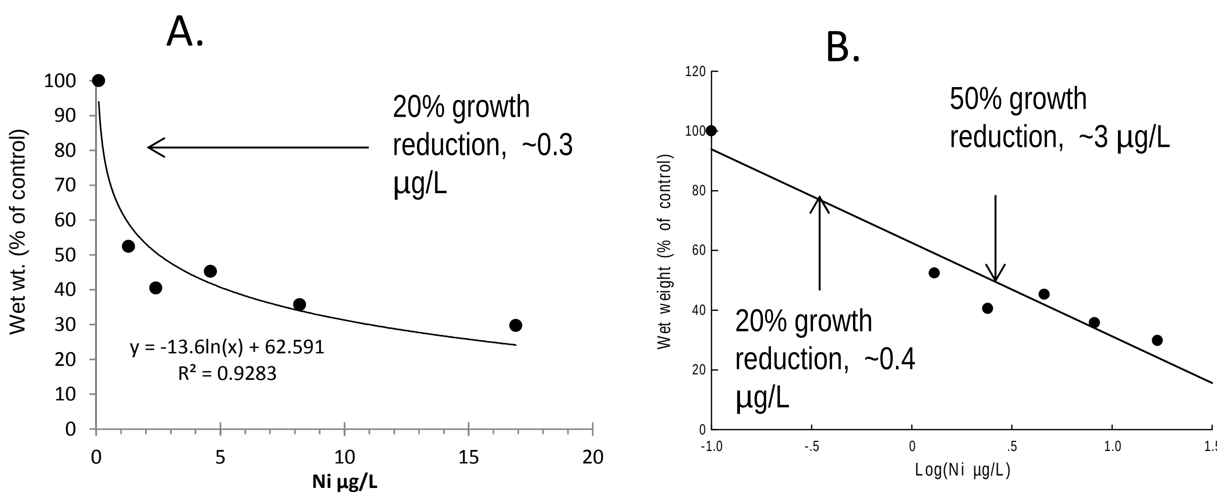

Figure 8. Lymnaea stagnalis growth, under different nickel (Ni) 21-day exposures. Data taken from Niyogi et al (2014). The data were not amenable to curve fitting to estimate EC values.

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.

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[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

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[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.

[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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