U.S. FISH AND WILDLIFE SERVICE
BIOLOGICAL AND CONFERENCE OPINION
for
USEPA's Proposed Approval of Oregon Water Quality Criteria for Toxics

Action Agency: U.S. Environmental Protection Agency

Consultation Conducted by: U.S. Fish and Wildlife Service
Oregon Fish and Wildlife Office

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LIST OF ABBREVIATIONS

ACR – acute to chronic ratio
ANOVA – analysis of variance
AMD – acid mine drainage
BAF – bioaccumulation factor
BCF – bioconcentration factor
BE – Biological Evaluation
BMC – benchmark concentration
BMD – benchmark dose
BMR – benchmark response
BOD – biological oxygen demand
CCC – Criterion Continuous Concentration
CCR – Crescent Coastal Research
CFR – Code of Federal Regulations
CFWR – Coast Fork Willamette River
CH – critical habitat
CHU – Critical Habitat Unit
ChV – chronic value
CLI – Climate Leadership Initiative
CMC – Criterion Maximum Concentration
CNDDB – California Natural Diversity Database
CSA – Cambridge Scientific Abstracts
CTR – California Toxics Rule
CWA – Clean Water Act
DO – dissolved oxygen
DOGAMI's –Department of Geology and Mineral Industries
DPS – Distinct Population Segment
DSL – Department of State Lands
EC – effect concentration
ECOTOX – ECOTOXicology Database
ESA – Endangered Species Act
ESHB – Engrossed Substitute House Bill
GIS – geographic information system
HC – hazard concentration
HCP – Habitat Conservation Plan
IC – inhibitory concentration
ICE – Interspecies Correlation Estimate
IPCC – Intergovernmental Panel on Climate Change
ISAB – Independent Scientific Advisory Board
LAA – likely to adversely affect
LC – lethal concentration
LC50 – median lethal concentration
LOEC – lowest observed effect concentration
LOEL – lowest observed effect level

LSR – Late-Successional Reserve
MATC – maximum acceptable threshold concentration
MBTSG – Montana Bull Trout Scientific Group
MFWR – Middle Fork Willamette River
mg/L – milligram(s) per liter
MILO-2 – Mineral Information Layer for Oregon, Release 2
MSE – mean square error
NA – not applicable
NLAA – not likely to adversely affect
NMFS – National Marine Fisheries Service
NOAEL – no observed adverse effect level
NOEC – no observed effect concentration
NOEL – no observed effect level
NPDES – National Pollutant Discharge Elimination System
NRC – National Research Council
NRCS – Natural Resources Conservation Service
NWEA – Northwest Environmental Advocates
NWFP – Northwest Forest Plan
OAR – Oregon Administrative Rules
OCLC – Online Computer Library Center, Inc.
ODA – Oregon Department of Agriculture
ODEQ – Oregon Department of Environmental Quality
ODFW – Oregon Department of Fish and Wildlife
ODOT – Oregon Department of Transportation
ONRC – Oregon Natural Resources Council
OSN – olfactory sensory neuron
PCE – Primary Constituent Element
PCH – Proposed Critical Habitat
PCP – pentachlorophenol
POEA – polyethoxylated tallowamine
ppm – part(s) per million
RM – River Mile
SHA – Safe Harbor Agreement
SMAV – species mean acute value
SSD – species sensitivity distribution
STP – sewage treatment plant
TBT – tributyltin
TL – total length
TMDL – Total Maximum Daily Load
TNC – The Nature Conservancy
TSS – total suspended solids
µg/L – microgram(s) per liter
USBLM – Bureau Land Management
USBR – U.S. Bureau of Reclamation
USDA – U.S. Department of Agriculture

USDI – U.S. Department of the Interior
USEPA – U.S. Environmental Protection Agency
USFWS – U.S. Fish and Wildlife Service
USGS – U.S. Geological Survey
WADT – Washington State Department of Transportation
WCPI – Wetland Conservation Plan Inventory
WDFW – Washington Department of Fish and Wildlife
WWF – World Wildlife Fund
WWTP – wastewater treatment plant

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INTRODUCTION

The Endangered Species Act (ESA) of 1973 (16 U.S.C. 1531 *et seq.*), as amended, establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat on which they depend. Section 7(a)(2) of the ESA requires Federal agencies to insure, in consultation with the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS), as appropriate, that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitats. Section 7(a)(4) of the ESA requires Federal agencies to confer with USFWS and NMFS (the Services), as appropriate, in cases where the agency or the Services have determined that a proposed or ongoing Federal action is likely to jeopardize the continued existence of species proposed to be listed under section 4 of the ESA or result in the destruction or adverse modification of critical habitat proposed to be designated for such species. The USFWS also encourages Federal agencies to conference on actions that may affect a proposed species or proposed critical habitat. In such cases, conference concurrence determinations or conference opinions can be adopted as formal concurrences or biological opinions, respectively, after a proposed species is listed or the critical habitat is designated. Such an approach can avoid disruption of project implementation due to the need to initiate and complete formal consultation at the time of listing or designation. It also facilitates or promotes action agency consideration of the conservation needs of proposed species and the recovery function of proposed critical habitat. This Biological and Conference Opinion is the product of an interagency consultation and conference pursuant to sections 7(a)(2) and 7(a)(4) of the ESA and its implementing regulations found in the Code of Federal Regulations (CFR) at 50 Part 402.

This document transmits the USFWS's Biological and Conference Opinion based on our review of the U.S. Environmental Protection Agency's (USEPA) proposed action to approve revised Oregon water quality standards for toxics. USEPA's biological evaluation (BE) was received on January 16, 2008, with a request for formal section 7 consultation. This document is USFWS's Biological and Conference Opinion which addresses the effects of the proposed action on the threatened bull trout (*Salvelinus confluentus*) and its designated critical habitat, the threatened Oregon chub (*Oregonichthys crameri*) and its designated critical habitat, the endangered Lost River sucker (*Deltistes luxatus*) and its proposed critical habitat, the endangered shortnose sucker (*Chasmistes brevirostris*) and its proposed critical habitat, the threatened vernal pool fairy shrimp (*Branchinecta lynchi*) and its designated critical habitat, and the threatened marbled murrelet (*Brachyramphus marmoratus marmoratus*) and its designated critical habitat. This document also includes our formal concurrence on USEPA's determinations that the proposed action may affect, but is "not likely to adversely affect", the other listed species and their respective critical habitats (discussed in the Concurrences section below). The Conference Opinion addresses the effects of the proposed action on proposed critical habitat for the Lost River and shortnose suckers.

This Biological and Conference Opinion is based on information provided in USEPA's 2008 BE, published literature and other sources of information. A complete decision record for this consultation is on file at the USFWS's Oregon Fish and Wildlife Office in Portland, Oregon.

CONSULTATION HISTORY

On July 8, 2004, the Oregon Department of Environmental Quality (ODEQ) submitted to USEPA revised water quality standards for toxics pollutants. The updated criteria incorporated all of USEPA recommend criteria for toxic pollutants that were current at the time.

From June to September of 2004, meetings between USEPA, NMFS, USFWS, and ODEQ were conducted to discuss the scope of the consultation. It was agreed that the consultation would focus on aquatic life toxic criteria and not human health toxic criteria. Furthermore, the scope of the consultation was limited to only those aquatic life criteria that were changed from previous criteria.

On August 10, 2005, a meeting was held to discuss using the draft ESA consultation methodology (Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual, Draft 3/2/05) in the biological analysis of the effects to threatened and endangered species.

On April 7, 2006, Northwest Environmental Advocates (NWEA) filed suit against USEPA for failure to perform their non-discretionary duties under the Clean Water Act (CWA) to approve or disapprove Oregon's submitted water quality standards within the timelines set forth in the CWA.

On January 16, 2008, USFWS received a letter and BE from USEPA requesting formal consultation.

On April 21, 2008, USFWS sent a letter to USEPA acknowledging receipt of the BE and stating that additional information was needed to complete USFWS's evaluation of the BE for the purposes of initiating consultation. The USFWS listed three initial additional information needs.

On April 22, 2008, a meeting amongst USEPA, USFWS, and NMFS regarding the BE and proposed consent decree. USFWS presented concerns with the BE methodology and summarized a proposed approach to complete consultation. In addition, USFWS requested financial support from USEPA to assist with the analysis.

On April 22, 2008, USEPA entered into a consent decree with NWEA to complete their approval action. The consent decree outlined a schedule for USEPA, including consultation with the Services.

On May 29, 2008, the court entered the consent decree between USEPA and NWEA for taking final action on the USEPA approval/disapproval of the Oregon water quality standards for toxic chemicals, including actions dependent on the USFWS completing a biological opinion.

On September 16, 2008, a conference call occurred with USEPA and the Services to discuss the upcoming action of USEPA withdrawing their consent to allow continuation of ESA consultation beyond the date of December 30, 2008.

On September 29, 2008, USEPA sent a letter to USFWS (and NMFS) responding to the April 21, 2008, letter from USFWS requesting additional information for evaluating the BE. USEPA provided descriptions and dates for the additional information provided.

On December 16, 2008, the Services joined USEPA in a conference call with NWEA to discuss the reasons for delay in preparation of our biological opinions.

On April 22, 2009, USFWS provided a presentation to NWEA regarding our process for completing the consultation.

On May 26, 2009, NWEA filed a Notice of Intent to sue for unreasonable delay in completing a biological opinion for USEPA action on Oregon water quality standards for toxic pollutants.

On August 31, 2009, USFWS submitted to USEPA a *Draft Method for Analysis of the U.S. Fish and Wildlife Service's Biological Opinion for Oregon's Water Quality Criteria for Toxic Chemicals.*

In October 2009, USFWS officially entered into an interagency agreement with USEPA for funding to support a U.S. Geological Survey (USGS) scientist to assist with development of a copper and ammonia prototype analysis for the Biological Opinion.

On February 17, 2010, USFWS and USEPA discussed Region 10 comments on the *Draft Method for Analysis* document described above.

On March 26, 2010, USFWS received a letter from USEPA Headquarters with comments on USFWS *Draft Method for Analysis* document described above.

On August 18, 2010, USFWS entered into a settlement agreement with NWEA to complete the Biological Opinion by June 2012.

On June 29, 2011, USFWS provided USEPA with a draft *Method for Analysis of the U.S. Fish and Wildlife Service's Biological Opinion for Oregon's Water Quality Criteria for Toxic Chemicals and Evaluation of Copper and Ammonia.*

On August 23, 2011, USEPA Headquarters provided USFWS with a response to the copper and ammonia *Method for Analysis* document.

On August 31, 2011, USFWS provided USEPA with a final *Method for Analysis of the U.S. Fish and Wildlife Service's Biological Opinion for Oregon's Water Quality Criteria for Toxic Chemicals and Evaluation of Copper and Ammonia.*

On March 21, 2012, the Court granted the motion to modify the completion date for the USFWS Biological Opinion to July 30, 2012.

CONCURRENCES

In the BE (USEPA 2008), USEPA made initial effects determinations for all aquatic listed species and critical habitats by chemical. Based on our preliminary analysis of exposure potential (see Section 4.1), we disagreed with some of USEPA's determinations; see Table 1 for USEPA's original effect determinations and those with which we disagreed. Following further analysis and subsequent correspondence between our agencies, USEPA determined their proposed action was not likely to adversely affect (NLAA) many of the listed species and critical habitat designations under the jurisdiction of USFWS in Oregon. These NLAA determinations were based on: (1) lack of exposure to the chemicals considered in this opinion, or (2) where exposure could occur, it was the likely that the effects would be insignificant or discountable. See Table 2 for a summary of USEPA's revised effects determinations and the USFWS's findings relative to these determinations for each species and critical habitat designation addressed in this opinion.

Table 1. Summary of effects determinations for aquatic listed species by chemical made by USEPA for Oregon's water quality criteria for toxics (USEPA 2008); where USFWS disagreed with USEPA's determination, our recommended determination is shown in parentheses below USEPA's original determination. (NLAA = not likely to adversely affect; LAA = likely to adversely affect).

Chemical	Bull trout	Lahontan cutthroat	Oregon chub	Lost River sucker	Shortnose sucker	Modoc sucker	Warner sucker	Hutton tui chub	Foskett speckled dace	Borax Lake chub	Vernal pool fairy shrimp	Short-tailed albatross	Marbled murrelet	Western snowy plover
Arsenic														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Cadmium														
Freshwater	NLAA (LAA)	NLAA	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chromium III														
Freshwater	LAA	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chromium VI														
Freshwater	NLAA	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA (LAA)	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Copper														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Lead														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA (LAA)	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Nickel														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Chemical	Bull trout	Lahontan cutthroat	Oregon chub	Lost River sucker	Shortnose sucker	Modoc sucker	Warner sucker	Hutton tui chub	Foskett speckled dace	Borax Lake chub	Vernal pool fairy shrimp	Short-tailed albatross	Marbled murrelet	Western snowy plover
Selenium														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA (LAA)	NLAA
Silver														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	LAA	LAA	NLAA	LAA (NLAA)	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Zinc														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Pentachlorophenol														
Freshwater	NLAA	NLAA	NLAA	NLAA (LAA)	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA (LAA)	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Lindane														
Freshwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Dieldrin														
Freshwater	LAA (NLAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Endosulfan alpha														
Freshwater	LAA (NLAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Endosulfan beta														
Freshwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Endrin														
Freshwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Heptachlor Epoxide														
Freshwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Chemical	Bull trout	Lahontan cutthroat	Oregon chub	Lost River sucker	Shortnose sucker	Modoc sucker	Warner sucker	Hutton tui chub	Foskett speckled dace	Borax Lake chub	Vernal pool fairy shrimp	Short-tailed albatross	Marbled murrelet	Western snowy plover
Aluminum														
Freshwater	NLAA (LAA)	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Ammonia														
Freshwater	NLAA (LAA)	NLAA	NLAA (LAA)	LAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Tributyltin														
Freshwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA		
Saltwater	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Table 2. USEPA effects determinations and USFWS findings for listed species and critical habitats addressed in this Biological and Conference Opinion.

Species and Critical Habitat	Status	USEPA Effects Determination*	USFWS Finding	Notes
Mammals				
Columbian white-tailed deer (Columbia River DPS) (*Odocoileus virginianus leucurus*)	Endangered	NLAA	Concurrence	
Grey wolf (*Canis lupus*)	Endangered	NLAA	Concurrence	
Birds				
Brown pelican (*Pelecanus occidentalis*)	Delisted	NLAA	N/A	Delisted in 2009, not addressed further in this Biological Opinion
Marbled murrelet (*Brachyramphus marmoratus*)	Threatened	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Marbled murrelet critical habitat	Designated	NLAA	Concurrence	
Short-tailed albatross (*Phoebastria albatrus*)	Endangered	NLAA	Concurrence	

Species and Critical Habitat	Status	USEPA Effects Determination*	USFWS Finding	Notes
Western snowy plover (*Charadrius alexandrinus nivosus*)	Threatened	NLAA	Concurrence	
Western snowy plover critical habitat	Designated	NLAA	Concurrence	
Amphibians				
Columbia spotted frog (*Rana luteiventris*)	Candidate	NLAA	Concurrence	
Oregon spotted frog (*Rana pretiosa*)	Candidate	NLAA	Concurrence	
Fish				
Bull trout (*Salvelinus confluentus*)	Threatened	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Bull trout critical habitat	Designated	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Borax Lake chub (*Gila boraxobius*)	Endangered	NLAA	Concurrence	
Borax Lake chub critical habitat	Designated	NLAA	Concurrence	
Hutton tui chub (*Gila bicolor* ssp.)	Threatened	NLAA	Concurrence	
Foskett speckled dace (*Rhinichthys osculus* ssp.)	Threatened	NLAA	Concurrence	
Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*)	Threatened	NLAA	Concurrence	
Lost River sucker (*Deltistes luxatus*)	Endangered	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Lost River sucker critical habitat	Proposed	LAA	Formal conference recommended	Effects of the proposed action are addressed in this Conference Opinion
Shortnose sucker (*Chasmistes brevirostris*)	Endangered	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Shortnose sucker critical habitat	Proposed	LAA	Formal conference recommended	Effects of the proposed action are addressed in this Conference Opinion

Species and Critical Habitat	Status	USEPA Effects Determination*	USFWS Finding	Notes
Modoc sucker (*Catostomus microps*)	Endangered	NLAA	Concurrence	
Modoc sucker critical habitat	Designated	NLAA	Concurrence	
Warner sucker (*Catostomus warnerensis*)	Threatened	NLAA	Concurrence	
Warner sucker critical habitat	Designated	NLAA	Concurrence	
Oregon chub (*Oregonichthys crameri*)	Threatened	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Oregon chub critical habitat	Designated	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Invertebrates				
Vernal pool fairy shrimp (*Branchinecta lynchi*)	Threatened	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Vernal pool fairy shrimp critical habitat	Designated	LAA	Formal consultation required	Effects of the proposed action are addressed in this Biological Opinion
Plants				
Macdonald's rockcress (*Arabis macdonaldiana*)	Endangered	NLAA	Concurrence	
Applegate's milk-vetch (*Astragalus applegatei*)	Endangered	NLAA	Concurrence	
Bradshaw's lomatium (*Lomatium bradshawii*)	Endangered	NLAA	Concurrence	
Cook's lomatium (*Lomatium cookii*)	Endangered	NLAA	Concurrence	
Cook's lomatium critical habitat	Designated	NLAA	Concurrence	
Golden Indian paintbrush (*Castilleja levisecta*)	Threatened	NLAA	Concurrence	
Water howellia (*Howellia aquatilis*)	Threatened	NLAA	Concurrence	
Howell's spectacular thelypody (*Thelypodium howellii* ssp. *spectabilis*)	Threatened	NLAA	Concurrence	
Large-flowered woolly meadowfoam (*Limnanthes floccosa* ssp. *grandiflora*)	Endangered	NLAA	Concurrence	

Species and Critical Habitat	Status	USEPA Effects Determination*	USFWS Finding	Notes
Large-flowered woolly meadowfoam critical habitat	Designated	NLAA	Concurrence	
MacFarlane's four o'clock (*Mirabilis macfarlanei*)	Threatened	NLAA	Concurrence	
Nelson's checker-mallow (*Sidalcea nelsoniana*)	Threatened	NLAA	Concurrence	
Rough popcorn flower (*Plagiobothrys hirtus*)	Endangered	NLAA	Concurrence	
Western lily (*Lilium occidentale*)	Endangered	NLAA	Concurrence	
Willamette daisy (*Erigeron decumbens* var. *decumbens*)	Endangered	NLAA	Concurrence	
Willamette daisy critical habitat	Designated	NLAA	Concurrence	
*NLAA = Not Likely to Adversely Affect; LAA = Likely to Adversely Affect				

BIOLOGICAL AND CONFERENCE OPINION

1 DESCRIPTION OF THE PROPOSED ACTION

Under 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. Water quality standards define the water quality goals for a waterbody by designating the use or uses, setting criteria necessary to protect the uses, and preventing or limiting degradation through antidegradation provisions. Section 303(c)(2)(B) of the CWA specifically requires that States adopt numeric criteria for all toxic pollutants for which criteria have been published under Section 304(a), the discharge of which could reasonably be expected to interfere with designated uses. USEPA publishes criteria documents as guidance to States. States consider these criteria documents, along with the most recent scientific information, when adopting regulatory criteria.

Once adopted by the State, water quality standards are submitted to USEPA for approval or disapproval. The approval or disapproval of the standards hinges upon USEPA's review to determine whether: (1) the analyses performed are adequate, (2) the designated uses are appropriate, and (3) the criteria are protective of those uses. State standards are then reviewed and revised, where appropriate, on a triennial basis. This public process allows for new technical and scientific data to be incorporated into the standards.

The Federal action that is the subject of this consultation is USEPA's proposed approval of portions of the following Oregon toxics criteria for aquatic life (Oregon Administrative Rules [OAR] 340-041-0033) which were submitted by the State to USEPA on July 14, 2004. All of the subject criteria are equivalent to USEPA's 2002 Recommended Water Quality Criteria (EPA-822-R-02-047). Table 1-1 lists the criterion for each chemical being addressed in this consultation. The majority of these revised criteria are lower concentrations than the previous criteria (Table 1-1).

Table 1-1. Oregon water quality criteria for aquatic life[1] addressed in this Biological Opinion.

	Criteria Freshwater Acute Criteria[2] (µg/L)		Freshwater Chronic Criteria[2] (µg/L)		Saltwater Acute Criteria[2] (µg/L)		Saltwater Chronic Criteria[2] (µg/L)	
	Previous	Proposed	Previous	Proposed	Previous	Proposed	Previous	Proposed
Aluminum	unregulated	750	unregulated	87		-		-
Ammonia (@pH 8 & 20 °C)	6	5.6[3] 8.4[4]	0.76[3] 1.08[4]	1.7		-		-
Arsenic	360	340	190	150	69	69	36	36
Lindane	2	0.95		-		-		-
Cadmium[5]	3.9	2.0	1.1	0.25	43	40	9.3	8.8
Chromium III[5]	1700	570	210	74		-		-
Chromium VI[5]	16	16	11	11	1100	1100	50	50
Copper[5]	18	13	12	9	2.9	4.8	2.9	3.1
Dieldrin	2.5	0.24	0.0019	0.056		-		-
alpha-Endosulfan	unregulated	0.22	unregulated	0.056	unregulated	0.034	unregulated	0.0087
beta-Endosulfan	unregulated	0.22	unregulated	0.056	unregulated	0.034	unregulated	0.0087
Endrin	0.18	0.086	0.0023	0.036		-		-
Heptachlor epoxide	unregulated	0.52	unregulated	0.0038	unregulated	0.053	unregulated	0.0036
Lead[5]	82	65	3.2	2.5	140	210	5.6	8.1
Nickel[5]		470		52		74		8.2
Pentachlorophenol (@pH 7.8)	20	19	13	15	13	13	unregulated	7.9
Selenium	260	12.82[6] 185.9[7]	35	5.0	410	290	54	71
Silver[5]	4.1	3.2	0.12	0.10	2.3	1.9		-
Tributyltin	unregulated	0.46	unregulated	0.063	unregulated	0.37	unregulated	0.01
Zinc[5]	120	120	110	120	95	90	86	81

[1]*The concentration for each compound listed is the criterion not to be exceeded in waters of the State in order to protect aquatic life. The acute criteria refer to the average concentration for 1 hour and the chronic criteria refer to the average concentration for 96 hours, and these criteria should not be exceeded more than once every 3 years. Chronic criteria for metals are expressed as dissolved concentrations in the water column.*
[2]*Freshwater criteria for metals are a function of hardness in the water column, but saltwater criteria are not.*
[3]*Salmonids present*
[4]*Salmonids absent*
[5]*Hardness @100 mg/L $CaCO_3$*
[6] *Selenate*
[7] *Selenite*

An acute criterion is also known as a Criterion Maximum Concentration (CMC), defined as an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect. A chronic criterion is also known as a Criterion Continuous Concentration (CCC), defined as an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. The acute (CMC) and chronic (CCC) criteria numbers, along with the frequency and duration of exposure (see Appendix 1), comprise the aquatic life criteria. These criteria represent the basis for administering water quality programs under the water quality-based approach to pollution control, including

monitoring to determine whether waters are attaining designated uses, 303(d) listing of impaired waters, and the development and implementation of Total Maximum Daily Loads (TMDLs).

USEPA's approval of the Oregon water quality criteria for aquatic life will not directly affect listed species; rather effects to threatened and endangered species will occur in the future as the State implements the standards through listing of waters on the 303(d) list which exceed criteria, calculation of TMDLs, issuing or reissuing permits (i.e., National Pollutant Discharge Elimination System or NPDES; see Appendix 1 for further discussion of NPDES permits), and accomplishing nonpoint source controls. These effects are considered indirect effects of USEPA's proposed action, and are analyzed in this Biological Opinion. USEPA also has the authority to approve or disapprove each TMDL completed by the State under separate action, and these State activities are outside the scope of this action.

Although there are many waters in the State of Oregon that currently do not meet the toxics criteria (i.e., have levels of toxics above the proposed criteria), we assume that the numeric criteria will be met in each waterbody for the purpose of our analysis in this Biological Opinion. Implementation of the criteria is not part of our analysis as we do not have the technical capabilities to address the many complexities and unknowns of all possible effects. Therefore, we conducted our effects analysis based on the potential for the proposed numeric criteria to result in effects to the aquatic ecosystem and the listed species dependent on the ecosystem for their survival and recovery.

1.1 Scope of the Effects of the Action Analysis

In assessing the likely effects of the proposed action on listed species and designated and proposed critical habitat, USFWS acknowledges that the chain of causation is several links long. The USEPA, as the Federal action agency, must first approve the proposed State water quality standards. Once approved by USEPA, the State then adopts these standards into its CWA-related regulations. Following the adoption, criteria become the numeric interpretation of the narrative standard "no toxics in toxic amounts" which the CWA established as the national policy for water quality. The criteria define legal limits for ambient water quality required for protecting designated uses. Therefore, criteria inform discharge limits for point source discharges (NPDES permits), set limits for ambient water quality in waters influenced by nonpoint discharges, establish legal limits for water quality monitoring programs to determine whether designated uses are protected or a water body is impaired, and set legal ambient water quality targets for restoring impaired waters through the TMDL process. Thus, the chain of causation is co-mingled between USEPA's approval of State standards and the subsequent implementation by the State of numerous CWA programs that regulate discharges, monitor pollutants, and mitigate unacceptable water pollution.

For the purposes of determining the effects of the proposed action on listed species or critical habitat, the USEPA and USFWS considered the likelihood of exposure to water pollutants based on our ability to identify all current and future point and nonpoint source discharges. For example, while the standards being proposed apply to the waters of the State designated for fish and aquatic life use, in some waters the actual exposure of a listed species or critical habitat may be below levels allowed by the standard and required to protect the designated use. The USEPA

acknowledged this in their BE on the proposed action for a number of species and critical habitats, finding that determinations of NLAA would be appropriate in areas where the exposure potential for the listed species or critical habitat from anthropogenic sources of discharge of substances associated with the proposed standards are so unlikely as to represent a discountable chance of occurring.

In this Biological and Conference Opinion, we have followed USEPA's lead and have 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. The identification of likely discharges of toxics at issue in this opinion that are likely to result in exposure to listed species or critical habitats is a prerequisite to a causative link between the proposed action and adverse effects to listed species and critical habitats. For example, extremely remote areas such as isolated springs and seeps in the desert, or high elevation waters within designated wilderness areas, and other similarly situated areas are extremely unlikely to be impacted by discharges subject to the proposed water quality standards. For that reason, we have determined that while effects of the proposed action on listed species or critical habitats occurring in these areas are legally possible, they are extremely unlikely to occur, and, on that basis, we therefore consider them to be discountable. In such cases, we did not conduct an exposure/response analysis to assess the sensitivity of listed species to individual water pollutants but will rely solely on our ability to identify all current and future point and nonpoint discharges to determine whether effects of the proposed action are not likely to adversely affect listed species or critical habitat in that portion of the species' range or the designated critical habitat.

This exposure-based approach to assessing the water quality criteria means that USEPA may need to reinitiate this consultation if the underlying assumptions about the likely exposure of listed species or critical habitats to the pollutants at issue herein change in the future based on new information and subsequent exposure of certain species (or portions of the range of a particular species) or critical habitats to pollutants covered under the proposed action is then considered likely or known to occur. The USEPA will need to validate the assumptions relied upon in this document by ensuring there will be regular monitoring of the ambient water quality of these water bodies for the subject pollutants.

Table 1-2 lists the threatened and endangered species addressed in this Biological and Conference Opinion and the chemicals for which adverse effects are anticipated. Other than the bull trout, all listed threatened and endangered aquatic species under the authorities of USFWS reside strictly in freshwater. Therefore, the saltwater criteria included in the proposed action will have no effect on the Oregon chub, Lost River sucker, shortnose sucker, and the vernal pool fairy shrimp.

Table 1-2. Chemicals with anticipated adverse effects to listed species and critical habitat evaluated in this Biological and Conference Opinion.

Species and Critical Habitat	Chemicals with Anticipated Adverse Effects
Marbled murrelet	Selenium
Marbled murrelet critical habitat	(none)
Bull trout	Arsenic, Cadmium, Chromium III, Copper, Lead, Nickel, Selenium, Silver, Zinc, Aluminum, Ammonia
Bull trout critical habitat	Arsenic, Cadmium, Copper, Lead, Nickel, Selenium, Silver, Zinc, Ammonia
Shortnose sucker	Arsenic, Cadmium, Chromium III, Chromium VI, Copper, Lead, Nickel, Selenium, Silver, Zinc, Pentachlorophenol, Ammonia
Shortnose sucker proposed critical habitat	Arsenic, Cadmium, Chromium III, Chromium VI, Copper, Lead, Nickel, Selenium, Silver, Zinc, Pentachlorophenol, Ammonia
Lost River sucker	Arsenic, Cadmium, Chromium III, Chromium VI, Copper, Lead, Nickel, Selenium, Silver, Zinc, Pentachlorophenol, Ammonia
Lost River sucker proposed critical habitat	Arsenic, Cadmium, Chromium III, Chromium VI, Copper, Lead, Nickel, Selenium, Silver, Zinc, Pentachlorophenol, Ammonia
Oregon chub	Arsenic, Chromium III, Chromium VI, Copper, Lead, Nickel, Selenium, Silver, Zinc, Ammonia
Oregon chub critical habitat	Arsenic, Chromium III, Chromium VI, Copper, Lead, Nickel, Selenium, Silver, Zinc, Ammonia
Vernal pool fairy shrimp	Chromium VI, Copper, Lead, Selenium, Zinc, Pentachlorophenol
Vernal pool fairy shrimp critical habitat	Chromium VI, Copper, Lead, Selenium, Zinc, Pentachlorophenol

For the purposes of this consultation and conference, the term of the proposed action is for 25 years. The triennial review of water quality standards is not comprehensive, and States' resources typically restrict them to reviewing a limited number of standards in each triennial review process. Therefore, we do not expect these standards to be revisited within the next several review cycles but believe it is important to revisit these standards when new information arises. For example, national guidance for several of the criteria in this consultation has been updated since the consultation was initiated. In addition, exposure to the chemicals at criteria concentrations is based on current baseline information which we believe will not change dramatically over 25 years. Beyond 25 years it is difficult to predict what to expect regarding the future landscape of Oregon. Consequently, we consider this Biological and Conference Opinion to be valid for the next 25 years. However, to the extent that any of the reinitiation criteria at 50 CFR 402.16 are triggered, reinitiation of formal consultation will be required that results in the USFWS's issuance of a new Biological and Conference Opinion.

1.2 Conservation Measures

Under section 303(d) of the 1972 CWA, States are required to develop lists of impaired waters that do not meet water quality standards. Furthermore, the law requires that States establish priority rankings for waters on the lists and develop TMDLs for these waters.

As described in 40 CFR 130.4 the States must establish appropriate monitoring methods and procedures necessary to compile and analyze data on the quality of waters of the U.S. Furthermore, the State's water monitoring program shall include collection and analysis of physical, chemical, and biological data, as well as quality assurance and control programs to assure scientifically valid data for uses including developing and reviewing water quality standards, TMDLs, wasteload allocations and load allocations, and assessing compliance with NPDES permits. Therefore, inherent in the action is the monitoring requirement by ODEQ for evaluating water quality criteria which are part of the water quality standards.

1.3 Action Area

The action area is defined as all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR 402.02). In delineating the action area, we evaluated the farthest reaching physical, chemical, and biotic effects of the action on the environment.

The action area for this consultation consists of all surface waters of the State of Oregon for which the aquatic life toxics criteria have been proposed. Water quality standards apply to all surface waters of the State, defined as all lakes, bays, ponds, impounding reservoirs, springs, rivers, streams, creeks, estuaries, marshes, inlets, canals, the Pacific Ocean within the territorial limits of the State of Oregon, and all other bodies of surface water, natural or artificial, inland or coastal, fresh or salt, public or private (except those private waters which do not combine or affect a junction with natural surface or underground waters), which are wholly or partially within or bordering the State or within its jurisdiction [OAR 340-041-0006 (14)]. USEPA's approval action does not apply to, and thus the action area does not include, any waters within Indian Country.

1.4 Analytical Framework for the Jeopardy and Adverse Modification Determinations

1.4.1 The Jeopardy Determination

In accordance with policy and regulation, the jeopardy analysis in this Biological and Conference Opinion relies on four components: (1) the "*Status of the Species*," which evaluates each listed species rangewide condition, the factors responsible for that condition, and its survival and recovery needs, (2) the "*Environmental Baseline*," which evaluates the condition of each listed species in the action area, the factors responsible for that condition, and the relationship of the action area to its survival and recovery, (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 each listed species, and (4) "*Cumulative Effects*," which evaluates the effects of future, non-Federal activities in the action area on each listed 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 each listed 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 listed species in the wild.

The jeopardy analysis in this Biological and Conference Opinion emphasizes consideration of the rangewide survival and recovery needs of the listed species and the role of the action area in the survival and recovery of the listed species. It is within this context that we evaluate 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 are recognized for purposes of making jeopardy determinations. The preamble to the most recent final listing rule for the bull trout (USFWS 1999a, p. 58930) further discusses application of the jeopardy standard relative to this species in terms of relying upon recovery units:

> 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. Formal establishment of bull trout recovery units will occur during the recovery planning process.

National policy, via the Director's March 6, 2006, memorandum on "Recovery Units and Jeopardy Determinations under Section 7 of the Endangered Species Act" (H. Dale Hall, in litt., 2006) and page 4-38 of the *Endangered Species Consultation Handbook* (USFWS and NMFS 1998), establishes that:

When an action 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.

This framework and approach are used herein to make the jeopardy determination for the bull trout.

Similarly, the final recovery plan for the marbled murrelet establishes six Conservation Zones, which are considered as recovery units for purposes of jeopardy determinations as described

above for the bull trout. This framework and approach are used herein to make the jeopardy determination for the marbled murrelet.

1.4.2 The Adverse Modification Determination

This Biological and Conference 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.

In accordance with policy and regulation, the adverse modification analysis in this Biological and Conference Opinion relies on four components: (1) the "*Status of Critical Habitat*," which evaluates the range-wide condition of designated critical habitat for each listed 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, and (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 each listed species' critical habitat is 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 each listed species.

The analysis in this Biological and Conference Opinion places an emphasis on using the intended rangewide recovery function of each listed 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 STATUS OF THE SPECIES AND CRITICAL HABITAT

This section identifies those species or designated critical habitat likely to be adversely affected by the proposed action, which will be considered further in the remaining sections of the Biological Opinion.

2.1 Bull Trout

2.1.1 Species Description

2.1.1.1 Taxonomy

The bull trout is a native char found in the coastal and intermountain west of North America. Dolly Varden (*Salvelinus malma)* and bull trout were previously considered a single species and were thought to have coastal and interior forms. However, Cavender (1978) described morphometric, meristic and osteological characteristics of the two forms, and provided evidence of specific distinctions between the two. In 1980, the American Fisheries Society formally recognized bull trout and Dolly Varden as separate species (Robins et al. 1980). Despite an overlap in the geographic range of bull trout and Dolly Varden in the Puget Sound area and along the British Columbia coast, there is little evidence of introgression (Haas and McPhail 1991). The Columbia River Basin is considered the region of origin for the bull trout. From the Columbia, dispersal to other drainage systems was accomplished by marine migration and headwater stream capture. Behnke and Benson (1980) postulated dispersion to drainages east of the continental divide may have occurred through the North and South Saskatchewan Rivers (Hudson Bay drainage) and the Yukon River system. Marine dispersal may have occurred from Puget Sound north to the Fraser, Skeena and Taku Rivers of British Columbia.

2.1.1.2 Species Description

Bull trout have unusually large heads and mouths for salmonids. Their body colors can vary tremendously depending on their environment, but are often brownish green with lighter (often ranging from pale yellow to crimson) colored spots running along their dorsa and flanks, with spots being absent on the dorsal fin, and light colored to white under bellies. They have white leading edges on their fins, as do other species of char. Bull trout have been measured as large as 41 inches (103 centimeters) in length, with weights as high as 32 pounds (14.5 kilograms) (Fishbase 2012). Bull trout may be migratory, moving throughout large river systems, lakes, and even the ocean in coastal populations, or they may be resident, remaining in the same stream their entire lives (USFWS 2012b). Migratory bull trout are typically larger than resident bull trout (USFWS 1998a)

2.1.1.3 Current legal status, including listing history

The coterminous U.S. population of the bull trout (*Salvelinus confluentus*) was listed as threatened on November 1, 1999 (USFWS 1999a). The threatened bull trout generally occurs in the Klamath River Basin of south-central Oregon; the Jarbidge River in Nevada; the Willamette River Basin in Oregon; Pacific Coast drainages of Washington, including Puget Sound; major rivers in Idaho, Oregon, Washington, and Montana, within the Columbia River Basin; and the St. Mary-Belly River, east of the Continental Divide in northwestern Montana (Cavender 1978, pp. 165-166; Bond 1992, p. 4; Brewin et al. 1997, pp. 209-216; Leary and Allendorf 1997, pp. 715-720).

Throughout its range, the bull trout are threatened by the combined effects of habitat degradation, fragmentation, and alterations associated with dewatering, road construction and

maintenance, mining, grazing, the blockage of migratory corridors by dams or other diversion structures, poor water quality, entrainment (a process by which aquatic organisms are pulled through a diversion or other device) into diversion channels, and introduced nonnative species (USFWS 1999a). Although all salmonids are likely to be affected by climate change, bull trout are especially vulnerable given that spawning and rearing are constrained by their location in upper watersheds and the requirement for cold water temperatures (Battin et al. 2007, Rieman et al. 2007, Porter and Nelitz. 2009, pp. 4-8). Poaching and incidental mortality of bull trout during other targeted fisheries are additional threats.

The bull trout was initially listed as three separate Distinct Population Segments (DPSs) (USFWS 1998b; USFWS 1999b, p. 17112). The preamble to the final listing rule for the U.S. coterminous population of the bull trout discusses the consolidation of these DPSs with the Columbia and Klamath population segments into one listed taxon and the application of the jeopardy standard under section 7 of the Act relative to this species (USFWS 1999a):

2.1.2 Critical habitat Description

2.1.2.1 Current legal status of the critical habitat

The USFWS published a final critical habitat designation for the coterminous U.S. population of the bull trout on October 18, 2010 (USFWS 2010d); 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 scope of the designation involved the species' coterminous range, which includes the Jarbidge River, Klamath River, Columbia River, Coastal-Puget Sound, and Saint Mary-Belly River population segments (also considered as interim recovery units)[1]. Rangewide, USFWS designated reservoirs/lakes and stream/shoreline miles as bull trout critical habitat (Table 2-1). Designated bull trout critical habitat is of two primary use types: (1) spawning and rearing, and (2) foraging, migration, and overwintering.

[1] The USFWS 5-year review (USFWS 2008a, p. 9) identifies six draft recovery units. Until the bull trout draft recovery plan is finalized, the current five interim recovery units are in affect for purposes of section 7 jeopardy analysis and recovery. The adverse modification analysis does not rely on recovery units.

Table 2-1. 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
**Pine Creek Drainage which falls within Oregon*				

The 2010 revision increases the amount of designated bull trout critical habitat by approximately 76% for miles of stream/shoreline and by approximately 71% for acres of lakes and reservoirs compared to the 2005 designation.

The final rule also identifies and designates as critical habitat approximately 822.5 miles (1,323.7 km) of streams/shorelines and 16,701.3 acres (6,758.8 ha) 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. No unoccupied habitat was included in the 2005 designation. These unoccupied areas were determined by USFWS to be essential for restoring functioning migratory bull trout populations based on currently available scientific information. These unoccupied areas often include lower main stem 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 (Act), 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 USFWS, or (3) waters where impacts to national security have been identified (USFWS 2010d). Excluded areas are approximately 10% of the stream/shoreline miles and 4% of the lakes and reservoir acreage of designated critical habitat. Each excluded area is identified in the relevant Critical Habitat Unit (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.

2.1.2.2 The primary constituent elements (PCEs)

The conservation role of bull trout critical habitat is to support viable core area populations (USFWS 2010d, p. 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 foraging, migratory, and overwintering habitats, outside of core areas, that are important to the survival and recovery of bull trout.

Thirty-two CHUs within the geographical area occupied by the species at the time of listing are designated under the revised 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 biological features associated with Primary Constituent Elements (PCEs) 5 and 6, which relate to breeding habitat.

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 (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 (Rieman and McIntyre 1993, pp. 22-23; MBTSG 1998, pp. 48-49), (3) are large enough to incorporate genetic and phenotypic diversity, but small enough to ensure connectivity between populations (Rieman and McIntyre 1993, pp. 22-23; Hard 1995, pp. 314-315; Healey and Prince 1995, p. 182; MBTSG 1998, pp. 48-49), and (4) are distributed throughout the historic range of the species to preserve both genetic and phenotypic adaptations (Rieman and McIntyre 1993, p. 23; Hard 1995, pp. 321-322; MBTSG 1998, pp. 13-16; Rieman and Allendorf 2001, p. 763).

Within the designated critical habitat areas, the PCEs for bull trout are those habitat components that are essential for the primary biological needs of foraging, reproducing, rearing of young, dispersal, genetic exchange, or sheltering. Based on our current knowledge of the life history, biology, and ecology of this species and the characteristics of the habitat necessary to sustain its essential life-history functions, we have determined that the PCEs, as described within USFWS 2010d are essential for the conservation of bull trout. A summary of those PCEs follows.

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 36 to 59 °F (2 to 15 °C), 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 departure 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.

The revised PCE's are similar to those previously in effect under the 2005 designation. The most significant modification is the addition of a ninth PCE to address the presence of nonnative predatory or competitive fish species. Although this PCE applies to both the freshwater and marine environments, currently no nonnative fish species are of concern in the marine environment, though this could change in the future.

Note that only PCEs 2, 3, 4, 5, and 8 apply to marine nearshore waters identified as critical habitat. Also, lakes and reservoirs within the CHUs also contain most of the physical or

biological features necessary to support bull trout, with the exception of those associated with PCEs 1 and 6. Additionally, all except PCE 6 apply to foraging, migratory, and overwintering habitat designated as critical habitat.

Critical habitat includes the stream channels within the designated stream reaches and has a lateral extent as defined by the bankfull elevation on one bank to the bankfull elevation on the opposite bank. Bankfull elevation is the level at which water begins to leave the channel and move into the floodplain and is reached at a discharge that generally has a recurrence interval of 1 to 2 years on the annual flood series. If bankfull elevation is not evident on either bank, the ordinary high-water line must be used to determine the lateral extent of critical habitat. The lateral extent of designated lakes is defined by the perimeter of the waterbody as mapped on standard 1:24,000 scale topographic maps. The USFWS assumes in many cases this is the full-pool level of the waterbody. In areas where only one side of the waterbody is designated (where only one side is excluded), the mid-line of the waterbody represents the lateral extent of critical habitat.

In marine nearshore areas, the inshore extent of critical habitat is the mean higher high-water (MHHW) line, including the uppermost reach of the saltwater wedge within tidally influenced freshwater heads of estuaries. The MHHW line refers to the average of all the higher high-water heights of the two daily tidal levels. Marine critical habitat extends offshore to the depth of 33 feet, relative to the mean low low-water (MLLW) line (zero tidal level or average of all the lower low-water heights of the two daily tidal levels). This area between the MHHW line and minus 10 m MLLW line (the average extent of the photic zone) is considered the habitat most consistently used by bull trout in marine waters based on known use, forage fish availability, and ongoing migration studies and captures geological and ecological processes important to maintaining these habitats. This area contains essential foraging habitat and migration corridors such as estuaries, bays, inlets, shallow subtidal areas, and intertidal flats.

Adjacent shoreline riparian areas, bluffs, and uplands are not designated as critical habitat. However, it should be recognized that the quality of marine and freshwater habitat along streams, lakes, and shorelines is intrinsically related to the character of these adjacent features, and that human activities that occur outside of the designated critical habitat can have major effects on physical and biological features of the aquatic environment.

Activities that cause adverse effects to critical habitat are evaluated to determine if they are likely to "destroy or adversely modify" critical habitat by no longer serving the intended conservation role for the species or retaining those PCEs that relate to the ability of the area to at least periodically support the species. Activities that may destroy or adversely modify critical habitat are those that alter the PCEs to such an extent that the conservation value of critical habitat is appreciably reduced (USFWS 2010d, p. 63943; USFWS 2004a, Vol. 1. pp. 140-193, Vol. 2. pp. 69-114). The USFWS's evaluation must be conducted at the scale of the entire critical habitat area designated, unless otherwise stated in the final critical habitat rule (USFWS and NMFS 1998, p. 4-39). Thus, adverse modification of bull trout critical habitat is evaluated at the scale of the final designation, which includes the critical habitat designated for the Klamath River, Jarbidge River, Columbia River, Coastal-Puget Sound, and Saint Mary-Belly River

population segments. However, we consider all 32 CHUs to contain features or areas essential to the conservation of the bull trout (USFWS 2010d, pp. 63901, 63944). Therefore, if a proposed action would alter the physical or biological features of critical habitat to an extent that appreciably reduces the conservation function of one or more critical habitat units for bull trout, a finding of adverse modification of the entire designated critical habitat area may be warranted (USFWS 2010d, p. 63943).

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 (USFWS 2010d, p. 71240). This condition reflects the condition of bull trout habitat. The decline of bull trout is primarily due to habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, past fisheries management practices, impoundments, dams, water diversions, and the introduction of nonnative species (USFW 1998; USFWS 1999 p. 17112).

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 (Rieman and McIntyre 1993, p. 7; Dunham and Rieman 1999, p. 652), (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), (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), (4) in the Coastal-Puget Sound region where amphidromous bull trout occur, degradation of mainstem foraging, migratory, and overwintering habitat, and the degradation and loss of marine nearshore foraging and migration habitat due to urban and residential development, and (5) degradation of foraging, migratory, and overwintering habitat resulting from reduced prey base, roads, agriculture, development, and dams.

2.1.2.3 Effects of Climate Change on Bull Trout Critical Habitat

One objective of the final critical habitat rule was 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).

2.1.2.4 Identification of the potential effects of toxics or water temperature levels to alteration of the PCEs

Many of the PCEs for bull trout may be affected by the presence of toxics or increased water temperatures within the environment. The effects will vary greatly depending on a number of factors which include which toxic substance is present, the amount of temperature increase, the likelihood that critical habitat would be affected (probability), and the severity and intensity of any effects that might occur (magnitude).

2.1.3 Life history

2.1.3.1 Reproduction

The iteroparous reproductive strategy of bull trout has important repercussions for the management of this species. Bull trout require passage both upstream and downstream, not only for repeat spawning but also for foraging. Most fish ladders, however, were designed specifically for anadromous semelparous salmonids (fishes that spawn once and then die, and require only one-way passage upstream). 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. Additionally, in some core areas, bull trout that migrate to marine waters must pass both upstream and downstream through areas with net fisheries at river mouths. This can increase the likelihood of mortality to bull trout during these spawning and foraging migrations.

Growth varies depending upon life-history strategy. Resident adults range from 6 to 12 inches total length, and migratory adults commonly reach 24 inches or more (Pratt 1985, Goetz 1989). The largest verified bull trout is a 32-pound specimen caught in Lake Pend Oreille, Idaho, in 1949 (Simpson and Wallace 1982).

Bull trout typically spawn from August through November during periods of increasing flows and decreasing water temperatures. Preferred spawning habitat consists of low-gradient stream reaches with loose, clean gravel (Fraley and Shepard 1989). Redds are often constructed in stream reaches fed by springs or near other sources of cold groundwater (Goetz 1989, Pratt 1992, Rieman and McIntyre 1996). Depending on water temperature, incubation is normally 100 to 145 days (Pratt 1992). After hatching, fry remain in the substrate, and time from egg deposition to emergence may surpass 200 days. Fry normally emerge from early April through May, depending on water temperatures and increasing stream flows (Pratt 1992, Ratliff and Howell 1992).

Early life stages of fish, specifically the developing embryo, require the highest inter-gravel dissolved oxygen (IGDO) levels, and are the most sensitive life stage to reduced oxygen levels. The oxygen demand of embryos depends on temperature and on stage of development, with the greatest IGDO required just prior to hatching.

A literature review conducted by the Washington Department of Ecology (WDOE 2002) indicates that adverse effects of lower oxygen concentrations on embryo survival are magnified as temperatures increase above optimal (for incubation). In a laboratory study conducted in

Canada, researchers found that low oxygen levels retarded embryonic development in bull trout (Giles and Van der Zweep 1996, as cited in Stewart et al. 2007). Normal oxygen levels seen in rivers used by bull trout during spawning ranged from 8 to 12 mg/L (in the gravel), with corresponding instream levels of 10 to 11.5 mg/L (Stewart et al. 2007). In addition, IGDO concentrations, water velocities in the water column, and especially the intergravel flow rate, are interrelated variables that affect the survival of incubating embryos (ODEQ 1995). Due to a long incubation period of 220+ days, bull trout are particularly sensitive to adequate IGDO levels. An IGDO level below 8 mg/L is likely to result in mortality of eggs, embryos, and fry.

2.1.3.2 Population structure

Bull trout exhibit both resident and migratory life history strategies. Both resident and migratory forms may be found together, and either form may produce offspring exhibiting either resident or migratory behavior (Rieman and McIntyre 1993). Resident bull trout complete their entire life cycle in the tributary (or nearby) streams in which they spawn and rear. The resident form tends to be smaller than the migratory form at maturity and also produces fewer eggs (Fraley and Shepard 1989, Goetz 1989). Migratory bull trout spawn in tributary streams where juvenile fish rear 1 to 4 years before migrating to either a lake (adfluvial form), river (fluvial form) (Fraley and Shepard 1989, Goetz 1989), or saltwater (anadromous form) to rear as subadults and to live as adults (Cavender 1978, McPhail and Baxter 1996). Bull trout normally reach sexual maturity in 4 to 7 years and may live longer than 12 years. They are iteroparous (they spawn more than once in a lifetime). Repeat- and alternate-year spawning has been reported, although repeat-spawning frequency and post-spawning mortality are not well documented (Leathe and Graham 1982, Fraley and Shepard 1989, Pratt 1992, Rieman and McIntyre 1996).

Migratory forms of bull trout may develop when habitat conditions allow movement between spawning and rearing streams and larger rivers, lakes or nearshore marine habitat where foraging opportunities may be enhanced (Frissell 1993, Goetz et al. 2004, Brenkman and Corbett 2005). For example, multiple life history forms (e.g., resident and fluvial) and multiple migration patterns have been noted in the Grande Ronde River (Baxter 2002). Parts of this river system have retained habitat conditions that allow free movement between spawning and rearing areas and the mainstem Snake River. Such multiple life history strategies help to maintain the stability and persistence of bull trout populations to environmental changes. Benefits to migratory bull trout include greater growth in the more productive waters of larger streams, lakes, and marine waters; greater fecundity resulting in increased reproductive potential; and dispersing the population across space and time so that spawning streams may be recolonized should local populations suffer a catastrophic loss (Rieman and McIntyre 1993, MBTSG 1998, Frissell 1999). In the absence of the migratory bull trout life form, isolated populations cannot be replenished when disturbances make local habitats temporarily unsuitable. Therefore, the range of the species is diminished, and the potential for a greater reproductive contribution from larger size fish with higher fecundity is lost (Rieman and McIntyre 1993).

Whitesel et al. (2004) noted that although there are multiple resources that contribute to the subject, Spruell et al. (2003) best summarized genetic information on bull trout population structure. Spruell et al. (2003) analyzed 1,847 bull trout from 65 sampling locations, four located in three coastal drainages (Klamath, Queets, and Skagit Rivers), one in the Saskatchewan

River drainage (Belly River), and 60 scattered throughout the Columbia River Basin. They concluded that there is a consistent pattern among genetic studies of bull trout, regardless of whether examining allozymes, mitochondrial DNA, or most recently microsatellite loci. Typically, the genetic pattern shows relatively little genetic variation within populations, but substantial divergence among populations. Microsatellite loci analysis supports the existence of at least three major genetically differentiated groups (or evolutionary lineages) of bull trout (Spruell et al. 2003). They were characterized as:

1 - "Coastal", including the Deschutes River and all of the Columbia River drainage downstream, as well as most coastal streams in Washington, Oregon, and British Columbia. A compelling case also exists that the Klamath Basin represents a unique evolutionary lineage within the coastal group.

2 - "Snake River", which also included the John Day, Umatilla, and Walla Walla rivers. Despite close proximity of the John Day and Deschutes Rivers, a striking level of divergence between bull trout in these two systems was observed.

3 - "Upper Columbia River" which includes the entire basin in Montana and northern Idaho. A tentative assignment was made by Spruell et al. (2003) of the Saskatchewan River drainage populations (east of the continental divide), grouping them with the upper Columbia River group.

Spruell et al. (2003) noted that within the major assemblages, populations were further subdivided, primarily at the level of major river basins. Taylor et al. (1999) surveyed bull trout populations, primarily from Canada, and found a major divergence between inland and coastal populations. Costello et al. (2003) suggested the patterns reflected the existence of two glacial refugia, consistent with the conclusions of Spruell and the biogeographic analysis of Haas and McPhail (2001). Both Taylor et al. (1999) and Spruell et al. (2003) concluded that the Deschutes River represented the most upstream limit of the coastal lineage in the Columbia River Basin.

2.1.3.3 Population Dynamics

Although bull trout are widely distributed over a large geographic area, they exhibit a patchy distribution, even in pristine habitats (Rieman and McIntyre 1993). Increased habitat fragmentation reduces the amount of available habitat and increases isolation from other populations of the same species (Saunders et al. 1991). Burkey (1989) 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 for local populations may be low and probability of extinction high (Burkey 1989, 1995).

Metapopulation concepts of conservation biology theory have been suggested relative to the distribution and characteristics of bull trout, although empirical evidence is relatively scant (Rieman and McIntyre 1993, Dunham and Rieman 1999, Rieman and Dunham 2000). A metapopulation is an interacting network of local populations with varying frequencies of migration and gene flow among them (Meffe and Carroll 1994). For inland bull trout,

metapopulation theory is likely most applicable at the watershed scale where habitat consists of discrete patches or collections of habitat capable of supporting local populations; local populations are for the most part independent and represent discrete reproductive units; and long-term, low-rate dispersal patterns among component populations influences the persistence of at least some of the local populations (Rieman and Dunham 2000). Ideally, multiple local populations distributed throughout a watershed provide a mechanism for spreading risk because the simultaneous loss of all local populations is unlikely. However, habitat alteration, primarily through the construction of impoundments, dams, and water diversions has fragmented habitats, eliminated migratory corridors, and in many cases isolated bull trout in the headwaters of tributaries (Rieman and Clayton 1997, Dunham and Rieman 1999, Spruell et al. 1999, Rieman and Dunham 2000).

Human-induced factors as well as natural factors affecting bull trout distribution have likely limited the expression of the metapopulation concept for bull trout to patches of habitat within the overall distribution of the species (Dunham and Rieman 1999). However, despite the theoretical fit, the relatively recent and brief time period during which bull trout investigations have taken place does not provide certainty as to whether a metapopulation dynamic is occurring (e.g., a balance between local extirpations and recolonizations) across the range of the bull trout or whether the persistence of bull trout in large or closely interconnected habitat patches (Dunham and Rieman 1999) is simply reflective of a general deterministic trend towards extinction of the species where the larger or interconnected patches are relics of historically wider distribution (Rieman and Dunham 2000). Recent research (Whiteley et al. 2003) does, however, provide genetic evidence for the presence of a metapopulation process for bull trout, at least in the Boise River Basin of Idaho.

2.1.3.4 Ecology / Habitat Characteristics

Bull trout have more specific habitat requirements than most other salmonids (Rieman and McIntyre 1993). Habitat components that influence bull trout distribution and abundance include water temperature, cover, channel form and stability, valley form, spawning and rearing substrate, and migratory corridors (Fraley and Shepard 1989, Goetz 1989, Hoelscher and Bjornn 1989, Sedell and Everest 1991, Howell and Buchanan 1992, Pratt 1992, Rieman and McIntyre 1993, Rieman and McIntyre 1995, Rich 1996, Watson and Hillman 1997). Watson and Hillman (1997) concluded that watersheds must have specific physical characteristics to provide the habitat requirements necessary for bull trout to successfully spawn and rear and that these specific characteristics are not necessarily present throughout these watersheds. Because bull trout exhibit a patchy distribution, even in pristine habitats (Rieman and McIntyre 1993), bull trout should not be expected to simultaneously occupy all available habitats (Rieman et al. 1997).

Migratory habitat links seasonally used areas for all bull trout life histories. The ability to migrate is important to the persistence of bull trout (Rieman and McIntyre 1993, Rieman et al. 1997). Migrations facilitate gene flow among local populations when individuals from different local populations interbreed or stray to nonnatal streams. Local populations that are extirpated by catastrophic events may also become reestablished by bull trout migrants. However, it is important to note that the genetic structuring of bull trout indicates there is limited gene flow among bull trout local populations, which may encourage local adaptation within individual

populations, and that reestablishment of extirpated populations may take a long time (Rieman and McIntyre 1993, Spruell et al. 1999). Migration also allows bull trout to access more abundant or larger prey, which facilitates growth and reproduction. Additional benefits of migration and its relationship to foraging are discussed below under "Diet."

Cold water temperatures play an important role in determining bull trout habitat quality, as these fish are primarily found in colder streams (below 59 °F), and spawning habitats are generally characterized by temperatures that drop below 48 °F in the fall (Fraley and Shepard 1989, Pratt 1992, Rieman and McIntyre 1993).

Thermal requirements for bull trout appear to differ at different life stages. Spawning areas are often associated with cold-water springs, groundwater infiltration, and the coldest streams in a given watershed (Pratt 1992, Rieman and McIntyre 1993, Baxter et al. 1997, Rieman et al. 1997). Optimum incubation temperatures for bull trout eggs range from 35 to 39 °F whereas optimum water temperatures for rearing range from about 46 to 50 °F (McPhail and Murray 1979, Goetz 1989, Buchanan and Gregory 1997). In Granite Creek, Idaho, Bonneau and Scarnecchia (1996) observed that juvenile bull trout selected the coldest water available in a plunge pool, 46 to 48 °F, within a temperature gradient of 4 to 60 °F. In a landscape study relating bull trout distribution to maximum water temperatures, Dunham et al. (2003) found that the probability of juvenile bull trout occurrence does not become high (i.e., greater than 0.75) until maximum temperatures decline to 52 to 54 °F.

Although bull trout are found primarily in cold streams, occasionally these fish are found in larger, warmer river systems throughout the Columbia River basin (Fraley and Shepard 1989, Rieman and McIntyre 1993, Rieman and McIntyre 1995, Buchanan and Gregory 1997, Rieman et al. 1997). Availability and proximity of cold water patches and food productivity can influence bull trout ability to survive in warmer rivers (Myrick et al. 2002). For example, in a study in the Little Lost River of Idaho where bull trout were found at temperatures ranging from 46 to 68 °F, most sites that had high densities of bull trout were in areas where primary productivity in streams had increased following a fire (Bart L. Gamett, Fisheries Biologist, U.S. Forest Service, Salmon-Challis National Forest, Mackay, Idaho, pers. comm., June 20, 2002).

All life history stages of bull trout are associated with complex forms of cover, including large wood, undercut banks, boulders, and pools (Fraley and Shepard 1989, Goetz 1989, Hoelscher and Bjornn 1989, Sedell and Everest 1991, Pratt 1992, Thomas 1992, Rich 1996, Sexauer and James 1997, Watson and Hillman 1997). Maintaining bull trout habitat requires stability of stream channels and maintenance of natural flow patterns (Rieman and McIntyre 1993). Juvenile and adult bull trout frequently inhabit side channels, stream margins, and pools with suitable cover (Sexauer and James 1997). These areas are sensitive to activities that directly or indirectly affect stream channel stability and alter natural flow patterns. For example, altered stream flow in the fall may disrupt bull trout during the spawning period, and channel instability may decrease survival of eggs and young juveniles in the gravel from winter through spring (Fraley and Shepard 1989, Pratt 1992, Pratt and Huston 1993). Pratt (1992) indicated that increases in fine sediment reduce egg survival and emergence.

Bull trout are opportunistic feeders, with food habits primarily a function of size and life-history strategy. A single optimal foraging strategy is not necessarily a consistent feature in the life of a fish, because this strategy can change as the fish progresses from one life stage to another (i.e., juvenile to subadult). Fish growth depends on the quantity and quality of food that is eaten (Gerking 1994), and as fish grow, their foraging strategy changes as their food changes, in quantity, size, or other characteristics. Resident and juvenile migratory bull trout prey on terrestrial and aquatic insects, macrozooplankton, and small fish (Boag 1987, Goetz 1989, Donald and Alger 1993). Subadult and adult migratory bull trout feed on various fish species (Leathe and Graham 1982, Fraley and Shepard 1989, Donald and Alger 1993, Brown 1994). Bull trout of all sizes other than fry have been found to eat fish half their length (Beauchamp and VanTassell 2001). In nearshore marine areas of western Washington, bull trout feed on Pacific herring (*Clupea pallasi*), Pacific sand lance (*Ammodytes hexapterus*), and surf smelt (*Hypomesus pretiosus*) (Goetz et al. 2004).

Bull trout migration and life history strategies are closely related to their feeding and foraging strategies. Migration allows bull trout to access optimal foraging areas and exploit a wider variety of prey resources. Optimal foraging theory can be used to describe strategies fish use to choose between alternative sources of food by weighing the benefits and costs of capturing one source of food over another. For example, prey often occurs in concentrated patches of abundance ("patch model"; Gerking 1994). As the predator feeds in one patch, the prey population is reduced, and it becomes more profitable for the predator to seek a new patch rather than continue feeding on the original one. This can be explained in terms of balancing energy acquired versus energy expended. For example, in the Skagit River system, anadromous bull trout make migrations as long as 121 miles between marine foraging areas in Puget Sound and headwater spawning grounds, foraging on salmon eggs and juvenile salmon along their migration route. Anadromous bull trout also use marine waters as migration corridors to reach seasonal habitats in non-natal watersheds to forage and possibly overwinter (Goetz et al. 2004, Brenkman and Corbett 2005).

2.1.4 Status

2.1.4.1 Summary of historical status and distribution

The historical range of bull trout includes major river basins in the Pacific Northwest at about 41 to 60 degrees North latitude, from the southern limits in the McCloud River in northern California and the Jarbidge River in Nevada to the headwaters of the Yukon River in the Northwest Territories, Canada (Cavender 1978, Bond 1992). To the west, the bull trout's range includes Puget Sound, various coastal rivers of British Columbia, Canada, and southeast Alaska (Bond 1992). Bull trout occur in portions of the Columbia River and tributaries within the basin, including its headwaters in Montana and Canada. Bull trout also occur in the Klamath River basin of south-central Oregon. East of the Continental Divide, bull trout are found in the headwaters of the Saskatchewan River in Alberta and Montana and in the MacKenzie River system in Alberta and British Columbia, Canada (Cavender 1978, Brewin et al. 1997).

2.1.4.2 Current status and distribution of the listed species in rangewide (summary)

Each of the five interim recovery units 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. No new local populations have been identified and no local populations have been lost since listing.

Jarbidge River Interim Recovery Unit

The Jarbidge River 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 spawning adults, are estimated to occur in the core area. The current condition of the bull trout in this interim recovery unit is attributed to the effects of livestock grazing, roads, incidental mortalities of released bull trout from recreational angling, historic angler harvest, timber harvest, and the introduction of nonnative fishes (USFWS 2004b). The draft bull trout recovery plan (USFWS 2004b) identifies the following conservation needs for this interim recovery unit: (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 bull trout 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).

Klamath River Interim Recovery Unit

The Klamath River interim recovery unit currently contains three core areas and seven 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 (USFWS 2002c). Bull trout populations in this interim recovery unit face a high risk of extirpation (USFWS 2002c). The draft Klamath River bull trout recovery plan (USFWS 2002c) identifies the following conservation needs for this interim recovery unit: (1) maintain the current distribution of 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, (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 2,400 adults currently to 8,250 adults are needed to provide for the persistence and viability of the three core areas (USFWS 2002c).

Columbia River Interim Recovery Unit

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% of the Columbia River Basin, and presently occur in 45% of the estimated historical range (Quigley and Arbelbide 1997, p.1177). This interim recovery unit currently contains approximately 97 core areas and 527 local populations. About 65% of these core areas and local populations occur in central Idaho and northwestern Montana. The bull trout in the Columbia River interim recovery

unit have declined in overall range and numbers of fish (USFWS 1998b). The condition of the bull trout within these core areas varies from poor to good. All core areas have been subject to the combined effects of habitat degradation and fragmentation caused by the following activities: dewatering; road construction and maintenance; mining; grazing; the blockage of migratory corridors by dams or other diversion structures; poor water quality; incidental angler harvest; entrainment into diversion channels; and introduced nonnative species. Although some strongholds still exist with migratory fish present, bull trout generally occur as isolated local populations in headwater lakes or tributaries where the migratory life history form has been lost. Though still widespread, there have been numerous local extirpations reported throughout the Columbia River basin. In Idaho, for example, bull trout have been extirpated from 119 reaches in 28 streams (Bill Horton, in litt., 1995). The USFWS completed a 5-year status review and determined that, of the 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 2005a).

The draft Columbia River bull trout recovery plan (USFWS 2002e) 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) restore and maintain suitable habitat conditions for all bull trout life history stages and strategies, and (4) conserve genetic diversity and provide opportunities for genetic exchange.

Coastal-Puget Sound Interim Recovery Unit

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 interim recovery unit. This interim recovery unit currently contains 14 core areas and 67 local populations (USFWS 2004a). Bull trout are distributed throughout most of the large rivers and associated tributary systems within this interim recovery unit. Bull trout continue to be present in nearly all major watersheds where they likely occurred historically, although local extirpations have occurred throughout this interim recovery unit. Many remaining populations are isolated or fragmented and abundance has declined, especially in the southeastern portion of the interim recovery 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, poaching, incidental mortality from other targeted fisheries, and the introduction of nonnative species. The draft Coastal-Puget Sound bull trout recovery plan (USFWS 2004a) identifies the following conservation needs for this interim recovery 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 Interim Recovery Unit

This interim recovery unit currently contains six core areas and nine local populations (USFWS 2002d). Bull trout are widely distributed in the St. Mary-Belly River drainage and occur in

nearly all of the waters that it inhabited historically. Bull trout are found only in a 1.2-mile reach of the North Fork Belly River within the U.S. 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). 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). The draft St. Mary-Belly bull trout recovery plan (USFWS 2002d) identifies the following conservation needs for this interim recovery 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 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 interim recovery unit are comprised mostly of migratory fish, whose habitat is mostly in Canada.

Consulted-on Effects

Projects subject to section 7 consultation under the Act have occurred throughout the range of the bull trout. From the time of its listing in June 1998 until July 2012, the USFWS issued 421 biological opinions that addressed the effects of various Federal actions on the bull trout. All of these opinions included a determination that the proposed Federal action was not likely to jeopardize the continued existence of the bull trout, based on consideration of the range-wide and action area conditions and conservation needs of the bull trout, the effects of the action and any cumulative effects in the action area.

The 421 biological opinions referenced above involve various activity types (e.g., grazing, road maintenance, habitat restoration, timber sales, hydropower, etc.); many of these opinions involved multiple projects, including restoration actions for the bull trout. The geographic scale of projects analyzed in these biological opinions varied from individual actions (e.g., construction of a bridge or pipeline) within one basin, to multiple-project actions, occurring across several basins. Some large-scale projects affected more than one interim recovery unit of the bull trout.

For each of the actions considered in the above 421 biological opinions, the causes of adverse and any beneficial effects were identified as were the anticipated consequences for spawning streams and/or migratory corridors, if possible (in some cases, these consequences were unknown).

2.1.4.3 Threats, reasons for listing, current rangewide status

Bull trout distribution, abundance, and habitat quality have declined rangewide (Bond 1992, Schill 1992, Thomas 1992, Ziller 1992, Rieman and McIntyre 1993, Newton and Pribyl 1994, McPhail and Baxter 1996). Several local extirpations have been documented, beginning in the 1950s (Rode 1990, Ratliff and Howell 1992, Donald and Alger 1993, Goetz 1994, Newton and Pribyl 1994, Berg and Priest 1995, Light et al. 1996, Buchanan et al. 1997, WDFW 1998). Bull trout were extirpated from the southernmost portion of their historic range, the McCloud River in California, around 1975 (Moyle 1976, Rode 1990). Bull trout have been functionally extirpated

(i.e., few individuals may occur there but do not constitute a viable population) in the Coeur d'Alene River basin in Idaho and in the Lake Chelan and Okanogan River basins in Washington (USFWS 1998b).

These declines result from the combined effects of habitat degradation and fragmentation, the blockage of migratory corridors; poor water quality, angler harvest and poaching, entrainment (process by which aquatic organisms are pulled through a diversion or other device) into diversion channels and dams, and introduced nonnative species. Specific land and water management activities that depress bull trout populations and degrade habitat include the effects of dams and other diversion structures, forest management practices, livestock grazing, agriculture, agricultural diversions, road construction and maintenance, mining, and urban and rural development (Beschta et al. 1987, Chamberlain et al. 1991, Furniss et al. 1991, Meehan 1991, Nehlsen et al. 1991, Sedell and Everest 1991, Craig and Wissmar 1993, Frissell 1993, Henjum et al. 1994, McIntosh et al. 1994, Wissmar et al. 1994, MBTSG 1995a-e, 1996a-f; Light et al. 1996, USDA and USDI 1995).

2.1.4.4 Climate Change

Global climate change, and the related warming of global climate, have been well documented (WWF 2003, IPCC 2007, ISAB 2007). Evidence of global climate change/warming includes widespread increases in average air and ocean temperatures and accelerated melting of glaciers, and rising sea level. Given the increasing certainty that climate change is occurring and is accelerating (Battin et al. 2007, IPCC 2007), we can no longer assume that climate conditions in the future will resemble those in the past.

Patterns consistent with changes in climate have already been observed in the range of many species and in a wide range of environmental trends (Hari et al. 2006, ISAB 2007, Rieman et al. 2007). In the northern hemisphere, the duration of ice cover over lakes and rivers has decreased by almost 20 days since the mid-1800s (WWF 2003). The range of many species has shifted poleward and elevationally upward. For cold-water associated salmonids in mountainous regions, where their upper distribution is often limited by impassable barriers, an upward thermal shift in suitable habitat can result in a reduction in range, which in turn can lead to a population decline (Hari et al. 2006).

In the Pacific Northwest, most models project warmer air temperatures and increases in winter precipitation and decreases in summer precipitation. Warmer temperatures will lead to more precipitation falling as rain rather than snow. As the seasonal amount of snow pack diminishes, the timing and volume of stream flow are likely to change and peak river flows are likely to increase in affected areas. Higher air temperatures are also likely to increase water temperatures (ISAB 2007). For example, stream gauge data from western Washington over the past 5 to 25 years indicate a marked increasing trend in water temperatures in most major rivers.

Climate change has the potential to profoundly alter the aquatic ecosystems upon which the bull trout depends via alterations in water yield, peak flows, and stream temperature, and an increase in the frequency and magnitude of catastrophic wildfires in adjacent terrestrial habitats (Bisson et al. 2003).

All life stages of the bull trout rely on cold water. Increasing air temperatures are likely to impact the availability of suitable cold water habitat. For example, ground water temperature is generally correlated with mean annual air temperature, and has been shown to strongly influence the distribution of other chars. Ground water temperature is linked to bull trout selection of spawning sites, and has been shown to influence the survival of embryos and early juvenile rearing of bull trout (Rieman et al. 2). Increases in air temperature are likely to be reflected in increases in both surface and groundwater temperatures.

Climate change is likely to affect the frequency and magnitude of fires, especially in warmer drier areas such as are found on the eastside of the Cascade Mountains. Bisson et al. (2003) note that the forest that naturally occurred in a particular area may or may not be the forest that will be responding to the fire regimes of an altered climate. In several studies related to the effect of large fires on bull trout populations, bull trout appear to have adapted to past fire disturbances through mechanisms such as dispersal and plasticity. However, as stated earlier, the future may well be different than the past and extreme fire events may have a dramatic effect on bull trout and other aquatic species, especially in the context of continued habitat loss, simplification and fragmentation of aquatic systems, and the introduction and expansion of exotic species (Bisson et al. 2003).

Migratory bull trout can be found in lakes, large rivers and marine waters. Effects of climate change on lakes are likely to impact migratory adfluvial bull trout that seasonally rely upon lakes for their greater availability of prey and access to tributaries. Climate-warming impacts to lakes will likely lead to longer periods of thermal stratification and coldwater fish such as adfluvial bull trout will be restricted to these bottom layers for greater periods of time. Deeper thermoclines resulting from climate change may further reduce the area of suitable temperatures in the bottom layers and intensify competition for food (WWF 2003).

Bull trout require very cold water for spawning and incubation. Suitable spawning habitat is often found in accessible higher elevation tributaries and headwaters of rivers. However, impacts on hydrology associated with climate change are related to shifts in timing, magnitude and distribution of peak flows that are also likely to be most pronounced in these high elevation stream basins (Battin et al. 2007). The increased magnitude of winter peak flows in high elevation areas is likely to impact the location, timing, and success of spawning and incubation for the bull trout and Pacific salmon species. Although lower elevation river reaches are not expected to experience as severe an impact from alterations in stream hydrology, they are unlikely to provide suitably cold temperatures for bull trout spawning, incubation and juvenile rearing.

As climate change progresses and stream temperatures warm, thermal refugia will be critical to the persistence of many bull trout populations. Thermal refugia are important for providing bull trout with patches of suitable habitat during migration through or to make feeding forays into areas with greater than optimal temperatures.

There is still a great deal of uncertainty associated with predictions relative to the timing, location, and magnitude of future climate change. It is also likely that the intensity of effects will vary by region (ISAB 2007) although the scale of that variation may exceed that of States. For example, several studies indicate that climate change has the potential to impact ecosystems in nearly all streams throughout the State of Washington (Battin et al. 2007, ISAB 2007, Rieman et al. 2007). In streams and rivers with temperatures approaching or at the upper limit of allowable water temperatures, there is little if any likelihood that bull trout will be able to adapt to or avoid the effects of climate change/warming. Climate change will be an important factor affecting bull trout distribution. As its distribution contracts, patch size decreases and connectivity is truncated, bull trout populations that may be currently connected may face increasing isolation, which could accelerate the rate of local extinction beyond that resulting from changes in stream temperature alone (Rieman et al. 2007). Due to variations in land form and geographic location across the range of the bull trout, it appears that some populations face higher risks than others. Bull trout in areas with currently degraded water temperatures or at the southern edge of its range may already be at risk of adverse impacts from current as well as future climate change.

2.1.5 Conservation

2.1.5.1 Needs

The conservation needs of bull trout are often generally expressed as the four "Cs": cold, clean, complex, and connected habitat. Cold stream temperatures, clean water quality that is relatively free of fine sediment and contaminants, complex channel characteristics (including abundant large wood and undercut banks), and large patches of such habitat that are well connected by unobstructed migratory pathways are all needed to promote conservation of bull trout at multiple scales ranging from the coterminous to local populations (a local population is a group of bull trout that spawn within a particular stream or portion of a stream system). The recovery planning process for bull trout (USFWS 2002b, 2004a, 2004b) has also identified the following conservation needs: (1) maintenance and restoration of multiple, interconnected populations in diverse habitats across the range of each interim recovery unit, (2) preservation of the diversity of life-history strategies, (3) maintenance of genetic and phenotypic diversity across the range of each interim recovery unit, and (4) establishment of a positive population trend. It has also been recognized that bull trout populations need to be protected from catastrophic fires across the range of each interim recovery unit (Rieman et al. 2003).

Central to the survival and recovery of bull trout is the maintenance of viable core areas (USFWS 2002b, 2004a, 2004b). 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. Each of the interim recovery units listed above consists of one or more core areas. There are approximately 121 core areas recognized across the coterminous range of the bull trout (USFWS 2002b, 2004a, 2004b).

1 - Maintenance and restoration of multiple, interconnected populations in diverse habitats across the range of each interim recovery unit

Multiple local populations distributed and interconnected throughout a watershed provide a mechanism for spreading risk from stochastic events (Rieman and McIntyre 1993, Hard 1995, Healy and Prince 1995, Spruell et al. 1999, Rieman and Allendorf 2001). Current patterns in bull trout distribution and other empirical evidence, when interpreted in view of emerging conservation theory, indicate that further declines and local extinctions are likely (Rieman et al. 1997, Dunham and Rieman 1999, Rieman and Allendorf 2001, Spruell 2003). Based in part on guidance from Rieman and McIntyre (1993), bull trout core areas with fewer than five local populations are at increased risk of extirpation; core areas with between 5 to 10 local populations are at intermediate risk of extirpation; and core areas which have more than 10 interconnected local populations are at diminished risk of extirpation.

Maintaining and restoring connectivity between existing populations of bull trout is important for the persistence of the species (Rieman and McIntyre 1993). Migration and occasional spawning between populations increases genetic variability and strengthens population variability (Rieman and McIntyre 1993). Migratory corridors allow individuals access to unoccupied but suitable habitats, foraging areas, and refuges from disturbances (Saunders et al. 1991).

Because bull trout in the coterminous U.S. are distributed over a wide geographic area consisting of various environmental conditions, and because they exhibit considerable genetic differentiation among populations, the occurrence of local adaptations is expected to be extensive. Some readily observable examples of differentiation between populations include external morphology and behavior (e.g., size and coloration of individuals; timing of spawning and migratory forays). Conserving many populations across the range of the species is crucial to adequately protect genetic and phenotypic diversity of bull trout (Leary et al. 1993, Rieman and McIntyre 1993, Hard 1995, Healy and Prince 1995, Spruell et al. 1999, Taylor et al. 1999, Rieman and Allendorf 2001). Changes in habitats and prevailing environmental conditions are increasingly likely to result in extinction of bull trout if genetic and phenotypic diversity is lost.

2 - Preservation of the diversity of life-history strategies

The bull trout has multiple life history strategies, including migratory forms, throughout its range (Rieman and McIntyre 1993). Migratory forms appear to develop when habitat conditions allow movement between spawning and rearing streams and larger rivers or lakes where foraging opportunities may be enhanced (Frissell 1997). For example, multiple life history forms (e.g., resident and fluvial) and multiple migration patterns have been noted in the Grande Ronde River (Baxter 2002). Parts of this river system have retained habitat conditions that allow free movement between spawning and rearing areas and the mainstem of the Snake River. Such multiple life history strategies help to maintain the stability and persistence of bull trout populations to environmental changes. Benefits to migratory bull trout include greater growth in the more productive waters of larger streams and lakes, greater fecundity resulting in increased reproductive potential, and dispersing the population across space and time so that spawning

streams may be recolonized should local populations suffer a catastrophic loss (Frissell 1997, MBTSG 1998, Rieman and McIntyre 1993).

3- Maintenance of genetic and phenotypic diversity across the range of each interim recovery unit

Healy and Prince (1995) reported that, because phenotypic diversity is a consequence of the genotype interacting with the habitat, the conservation of phenotypic diversity is achieved through conservation of the sub-population within its habitat. They further note that adaptive variation among salmonids has been observed to occur under relatively short time frames (e.g., changes in genetic composition of salmonids raised in hatcheries; rapid emergence of divergent phenotypes for salmonids introduced to new environments). Healy and Prince (1995) conclude that while the loss of a few sub-populations within an ecosystem might have only a small effect on overall genetic diversity, the effect on phenotypic diversity and, potentially, overall population viability could be substantial (Healy and Prince 1995). This concept of preserving variation in phenotypic traits that is determined by both genetic and environmental (i.e., local habitat) factors has also been identified by Hard (1995) as an important component in maintaining intraspecific adaptability (i.e., phenotypic plasticity) and ecological diversity within a genotype (Hard 1995). He argues that adaptive processes are not entirely encompassed by the interpretation of molecular genetic data; in other words, phenotypic and genetic variation in adaptive traits may exist without detectable variation at the molecular genetic level, particularly for neutral genetic markers. Therefore, the effective conservation of genetic diversity necessarily involves consideration of the conservation of biological units smaller than taxonomic species (or DPSs). Reflecting this theme, the maintenance of local sub-populations has been specifically emphasized as a mechanism for the conservation of bull trout (Rieman and McIntyre 1993, Taylor et al. 1999).

4 - Establishment of a positive population trend

A stable or increasing population is a key criterion for recovery under the requirements of the Endangered Species Act. Measures of the trend of a population (the tendency to increase, decrease, or remain stable) include population growth rate or productivity. Estimates of population growth rate (i.e., productivity over the entire life cycle) that indicate a population is consistently failing to replace itself, indicate increased extinction risk. Therefore, the reproductive rate should indicate the population is replacing itself, or growing.

Since data of the total 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 index of a spawning adult population. The direction and magnitude of a trend in the index can be used as a surrogate for the growth rate of the entire population. For instance, a downward trend in an abundance indicator may signal the need for increased protection, regardless of the actual size of the population. A population which is below recovered abundance levels but moving toward recovery would be expected to exhibit an increasing trend in the indicator.

The population growth rate is an indicator of extinction probability. The probability of going extinct cannot be measured directly; it can, however, be estimated as the consequence of the population growth rate and the variability in that rate. For a population to be considered viable, its natural productivity should be sufficient to replace itself from generation to generation. Evaluations of population status will also have to take into account uncertainty in estimates of population growth rate or productivity. For a population to contribute to recovery, its growth rate must indicate that the population is stable or increasing for a period of time (USFWS 2002f, p. 16)

5 - Protect Bull Trout from Catastrophic Fires

Bull trout evolved under historic fire regimes in which disturbance to streams from forest fires resulted in a mosaic of diverse habitats. However, forest management and fire suppression over the past century have increased homogeneity of terrestrial and aquatic habitats, increasing the likelihood of large, intense forest fires in some areas. Because the most severe effects of fire on native fish populations can be expected where populations have become fragmented by human activities or natural events, an effective strategy to ensure persistence of native fishes against the effects of large fires may be to restore aquatic habitat structure and life history complexity of populations in areas susceptible to large fires (Gresswell 1999).

Rieman and Clayton (1997) discussed relations among the effects of fire and timber harvest, aquatic habitats, and sensitive species. They noted that spatial diversity and complexity of aquatic habitats strongly influence the effects of large disturbances on salmonids (Rieman and Clayton 1997). For example, Rieman et al. (1997) studied bull trout and redband trout responses to large, intense fires that burned three watersheds in the Boise National Forest in Idaho. Although the fires were the most intense on record, there was a mix of severely burned to unburned areas left after the fires. Fish were apparently eliminated in some stream reaches, whereas others contained relatively high densities of fish. Within a few years after the fires and after areas within the watersheds experienced debris flows, fish had become reestablished in many reaches, and densities increased. In some instances, fish densities were higher than those present before the fires or in streams that were not burned (Rieman and Clayton 1997). These responses were attributed to spatial habitat diversity that supplied refuge areas for fish during the fires, and the ability of bull trout and the redband trout to move among stream reaches. For bull trout, the presence of migratory fish within the system was also important (Rieman and Clayton 1997, Rieman et al. 1997).

In terms of conserving bull trout, the appropriate strategy to reduce the effects of fires on bull trout habitat is to emphasize the restoration of watershed processes that create and maintain habitat diversity, provide bull trout access to habitats, and protect or restore migratory life-history forms of bull trout. Both passive (e.g., encouraging natural riparian vegetation and floodplain processes to function appropriately) and active (e.g., reducing road density, removing barriers to fish movement, and improving habitat complexity) actions offer the best approaches to protect bull trout from the effects of large fires.

2.1.5.2 Summary of Current Status and Actions

Coastal-Puget Sound Interim Recovery Unit

Although the status of bull trout in Coastal-Puget Sound interim recovery unit has been improved by certain actions, it continues to be degraded by other actions, and it is likely that the overall status of the bull trout in this population segment has not improved since its listing on November 1, 1999. Improvement has occurred largely through changes in fishing regulations and habitat-restoration projects. Fishing regulations enacted in 1994 either eliminated harvest of bull trout or restricted the amount of harvest allowed, and this likely has had a positive influence on the abundance of bull trout. Improvement in habitat has occurred following restoration projects intended to benefit either bull trout or salmon, although monitoring the effectiveness of these projects seldom occurs. On the other hand, the status of this population segment has been adversely affected by a number of Federal and non-Federal actions, some of which were addressed under section 7 of the Act. Most of these actions degraded the environmental baseline; all of those addressed through formal consultation under section 7 of the Act permitted the incidental take of bull trout.

Section 10(a)(1)(B) permits have been issued for Habitat Conservation Plans (HCP) completed in the Coastal-Puget Sound population segment. These include: (1) the City of Seattle's Cedar River Watershed HCP, (2) Simpson Timber HCP, (3) Tacoma Public Utilities Green River HCP, (4) Plum Creek Cascades HCP, (5) Washington State Department of Natural Resources HCP, (6) West Fork Timber HCP (Nisqually River), and (7) Forest Practices HCP. These HCPs provide landscape-scale conservation for fish, including bull trout. Many of the covered activities associated with these HCPs will contribute to conserving bull trout over the long-term; however, some covered activities will result in short-term degradation of the baseline. All HCPs permit the incidental take of bull trout.

Columbia River Interim Recovery Unit

The overall status of the Columbia River interim recovery unit has not changed appreciably since its listing on June 10, 1998. Populations of bull trout and their habitat in this area have been affected by a number of actions addressed under section 7 of the Act. Most of these actions resulted in degradation of the environmental baseline of bull trout habitat, and all permitted or analyzed the potential for incidental take of bull trout. The Plum Creek Cascades HCP, Plum Creek Native Fish HCP, and Forest Practices HCP addressed portions of the Columbia River population segment of bull trout.

Klamath River Interim Recovery Unit

Improvements in the Threemile, Sun, and Long creeks local populations have occurred through efforts to remove or reduce competition and hybridization with nonnative salmonids, changes in fishing regulations, and habitat-restoration projects. Population status in the remaining local populations (Boulder-Dixon, Deming, Brownsworth, and Leonard creeks) remains relatively unchanged. Grazing within bull trout watersheds throughout the recovery unit has been curtailed. Efforts at removal of nonnative species of salmonids appear to have stabilized the Threemile Creek and positively influenced the Sun Creek local populations. The results of

similar efforts in Long Creek are inconclusive. Mark and recapture studies of bull trout in Long Creek indicate a larger migratory component than previously expected.

Although the status of specific local populations has been slightly improved by recovery actions, the overall status of Klamath River bull trout continues to be depressed. Factors considered threats to bull trout in the Klamath Basin at the time of listing—habitat loss and degradation caused by reduced water quality, past and present land use management practices, water diversions, roads, and nonnative fishes—continue to be threats today.

Jarbidge Interim Recovery Unit

While the overall status of the Jarbidge Interim Recovery Unit has not changed significantly since the original time of listing, numerous study efforts have been conducted to obtain more data on populations and distribution. Studies on distribution and genetic variation have been concluded. Bull trout presence has now been documented in Cougar Creek and Deer Creek. Temperature monitoring combined with GIS modeling has identified many thermal barriers that exist throughout the unit.

Both the U.S. Forest Service and U.S. Bureau of Land Management (USBLM) have implemented new road management plans that address road maintenance needs and improvements within the Jarbidge Canyon intended to reduce long-term sediment input into the West Fork Jarbidge River. This work is anticipated to improve foraging, migratory, and overwintering habitat within the West Fork Jarbidge River and result in positive long-term effects to bull trout abundance, distribution, and trend.

Saint Mary-Belly River Interim Recovery Unit

The overall status of bull trout in the Saint Mary-Belly River interim recovery unit has not changed appreciably since its listing on November 1, 1999. Extensive research efforts have been conducted since listing, to better quantify populations of bull trout and their movement patterns. Limited efforts in the way of active recovery actions have occurred. Habitat occurs mostly on Federal and Tribal lands (Glacier National Park and the Blackfeet Nation). Known problems due to instream flow depletion, entrainment, and fish passage barriers resulting from operations of the U.S. Bureau of Reclamation's (USBR's) Milk River Irrigation Project (which transfers Saint Mary-Belly River water to the Missouri River Basin) and similar projects downstream in Canada constitute the primary threats to bull trout and to date they have not been adequately addressed under section 7 of the Act. Plans to upgrade the aging irrigation delivery system are being pursued, which has potential to mitigate some of these concerns but also the potential to intensify dewatering. A major fire in August 2006 severely burned the forested habitat in Red Eagle and Divide Creeks, potentially affecting three of nine local populations and degrading the baseline.

2.1.5.3 State Conservation Actions

Idaho

Conservation actions by the State of Idaho include: (1) the development of a management plan for bull trout in 1993, (2) the approval of the State of Idaho Bull Trout Conservation Plan (Idaho Plan) in July 1996 (Batt 1996), (3) the development of 21 problem assessments involving 59 key

watersheds, (4) the implementation of conservation actions identified in the problem assessments, and (5) the implementation of more restrictive angling regulations.

Montana

Conservation actions by the State of Montana include: (1) development of the Montana Bull Trout Restoration Plan issued in 2000, which defines strategies for ensuring the long-term persistence of bull trout in Montana, (2) formation of the Montana Bull Trout Restoration Team (MBTRT) and Montana Bull Trout Scientific Group (MBTSG) to produce a plan for maintaining, protecting, and increasing bull trout populations, (3) the development of watershed groups to initiate localized bull trout restoration efforts, (4) funding of habitat restoration projects, recovery actions, and genetic studies throughout the state, (5) the abolition of brook trout stocking programs, and (6) restrictive angling regulations.

Nevada

Conservation actions by the State of Nevada include: (1) the preparation of a Bull Trout Species Management Plan that recommends management alternatives to ensure that human activities will not jeopardize the future of bull trout in Nevada (Johnson 1990), (2) implementation of more restrictive State angling regulations in an attempt to protect bull trout in the Jarbidge River in Nevada, and (3) the abolition of a rainbow trout stocking in the Jarbidge River.

Oregon

Since 1990, the State of Oregon has taken extensive action to address the conservation of bull trout, including: (1) establishment of bull trout working groups in the Klamath, Deschutes, Hood, Willamette, Odell Lake, Umatilla and Walla Walla, John Day, Malheur, and Pine Creek river basins for the purpose of developing bull trout conservation strategies, (2) establishment of more restrictive harvest regulations in 1990, (3) reduced stocking of hatchery-reared rainbow trout and brook trout into areas where bull trout occur, (4) angler outreach and education efforts are also being implemented in river basins occupied by bull trout, (5) research to further examine life history, genetics, habitat needs, and limiting factors of bull trout in Oregon, (6) reintroduction of bull trout fry from the McKenzie River watershed to the adjacent Middle Fork of the Willamette River, which is historical but currently unoccupied, isolated habitat, (7) establishment of a water temperature standard such that the 7-day-average maximum temperature of a stream having bull trout spawning and juvenile use (in specified subbasins) may not exceed 12 °C (53.6 °F) (OAR 340-041-0028), and (8) expansion of The Oregon Plan for salmon and watersheds (State of Oregon 1997) to include all at-risk wild salmonids throughout the State.

Washington

Conservation actions by the State of Washington include: (1) establishment of the Salmon Recovery Act (Engrossed Substitute House Bill [ESHB] 2496) and Watershed Management Act (ESHB 2514) by the Washington State legislature to assist in funding and planning salmon recovery efforts, (2) abolition of brook trout stocking in streams or lakes connected to bull trout-occupied waters, (3) changing angling regulations in Washington prohibit the harvest of bull trout, except for a few areas where stocks are considered "healthy," (4) collecting and mapping updated information on bull trout distribution, spawning and rearing areas, and potential habitat, and (5) adopting new emergency forest practice rules based on the "Forest and Fish Report"

process. These rules address riparian areas, roads, steep slopes, and other elements of forest practices on non-Federal lands.

2.1.5.4 Tribal Conservation Activities

Many Tribes throughout the range of the bull trout are participating on bull trout conservation working groups or recovery teams in their geographic areas of interest. Some tribes are also implementing projects which focus on bull trout or that address anadromous fish but also benefit bull trout (e.g., habitat surveys, passage at dams and diversions, habitat improvement, and movement studies).

2.1.6 Status of the Species in the Action Area (Oregon)

2.1.6.1 Current status and distribution of the listed species in the State of Oregon including population size, variability, and trend

Bull trout within Oregon face most if not all of the challenges described throughout the entire conterminous population. While the threats faced by bull trout may be the same across the State, individual core areas are threatened by greater or lesser degrees depending on their particular location and site specific conditions.

Water quality (including temperature), habitat fragmentation, invasive species competition and hybridization, and barriers that disrupt migration, genetic interchange, and foraging abound. A basic description of the status of bull trout across Oregon can be seen in Table 2-2 below.

Table 2-2. Status of bull trout within Oregon (USFWS 2008d)

Critical Habitat Unit	Core Area	Number of Local Populations	Life Style	Life History Form(s)	Population Size Range (estimated)	Population Trend	Primary Risks	Invasive Species Risk	Overall Risk
Klamath River	Upper Klamath Lake	3	Fluvial	Resident	50-250	Increasing	Water Quality, Habitat Fragmentation, Barriers	Brook Trout	At Risk
	Upper Sprague River	5	Fluvial	Resident Migratory	1,000-2,500	Stable	Dewatering Water Quality, Habitat Fragmentation	Brown Trout	At Risk
	Upper Sycan	1	Fluvial,	Resident	50-250	Declining	Water Quality, Habitat Fragmentation	Brook Trout	High Risk
Malheur River	Malheur River	2	Fluvial	Resident	50-250	Declining	Water Quality, Barriers	Brook Trout	High
Willamette River	Upper Willamette	4	Fluvial, Adfluvial	Migratory	300	Stable	Water Quality, Barriers	Brook Trout	High
Hood River	Hood River	2	Adfluvial Fluvial	Migratory	50-250	Declining	Barriers, Water Quality		High
Lower Deschutes River	Lower Deschutes River	5	Adfluvial Fluvial	Resident	1,000-2,500[2]	Increasing	Barriers Water quality		At Risk
Odell Lake	Odell Lake	1	Adfluvial[3]		<100	Stable	Water Quality	Brook Trout	High

[3] Naturally occurring adfluvial population in Oregon

Critical Habitat Unit	Core Area	Number of Local Populations	Life Style	Life History Form(s)	Population Size Range (estimated)	Population Trend	Primary Risks	Invasive Species Risk	Overall Risk
John Day River	Middle Fork John Day River	3	Fluvial	Resident Migratory	<100	Stable	Water Quality	SM bass	At Risk
	North Fork John Day River	7	Fluvial	Resident, Migratory	<100	Increasing	Water Quality	Brook Trout	At Risk
	Upper Mainstem	2	Fluvial	Resident, Migratory	100-250	Stable	Dewatering, Water Quality		At Risk
Umatilla River	Umatilla River	1	Fluvial	Resident Migratory	<100	Declining	Water Quality		High
[4]Walla Walla River	Walla Walla River	3	Fluvial	Resident Migratory	1,000-2,500	Stable	Dewatering		At Risk
[3]Grande Ronde	Grande Ronde	10	Fluvial Adfluvial[3]	Resident Migratory	50-250	Stable	Water Quality		At Risk
[5]Imnaha River	Imnaha River	4	Fluvial	Resident, Migratory	1,000-2,500	Stable	Declining Prey Base		At Risk
Powder River	Powder River	10	Fluvial, (Recent Adfluvial)	Resident	<500	Stable	Habitat Fragmentation, Water Quality Barriers	Brook Trout, Yellow Perch	High
[4]Hells Canyon Complex	Pine Indian –Wildhorse Creek	7	Fluvial, Adfluvial	Resident, Migratory	<500	Declining	Habitat Fragmentation, Water Quality Barriers	Brook Trout	Very High

[4] Portions of this Unit are located in Washington
[5] Portions of this Unit are located in Idaho

2.1.6.2 Current critical habitat designation within the State of Oregon

There are approximately 2,943.5 miles (4,737 Km) of stream designated as critical habitat for bull trout within Oregon. Of these about 1,081.4 miles are considered to be spawning and rearing habitat, with the remaining 1,862.1 miles considered foraging, migrating and overwintering habitat. Additionally there are some 14 lakes and reservoirs which total approximately 30,155.5 acres (12,245 Hectares) that are also designated as critical habitat for bull trout within Oregon.

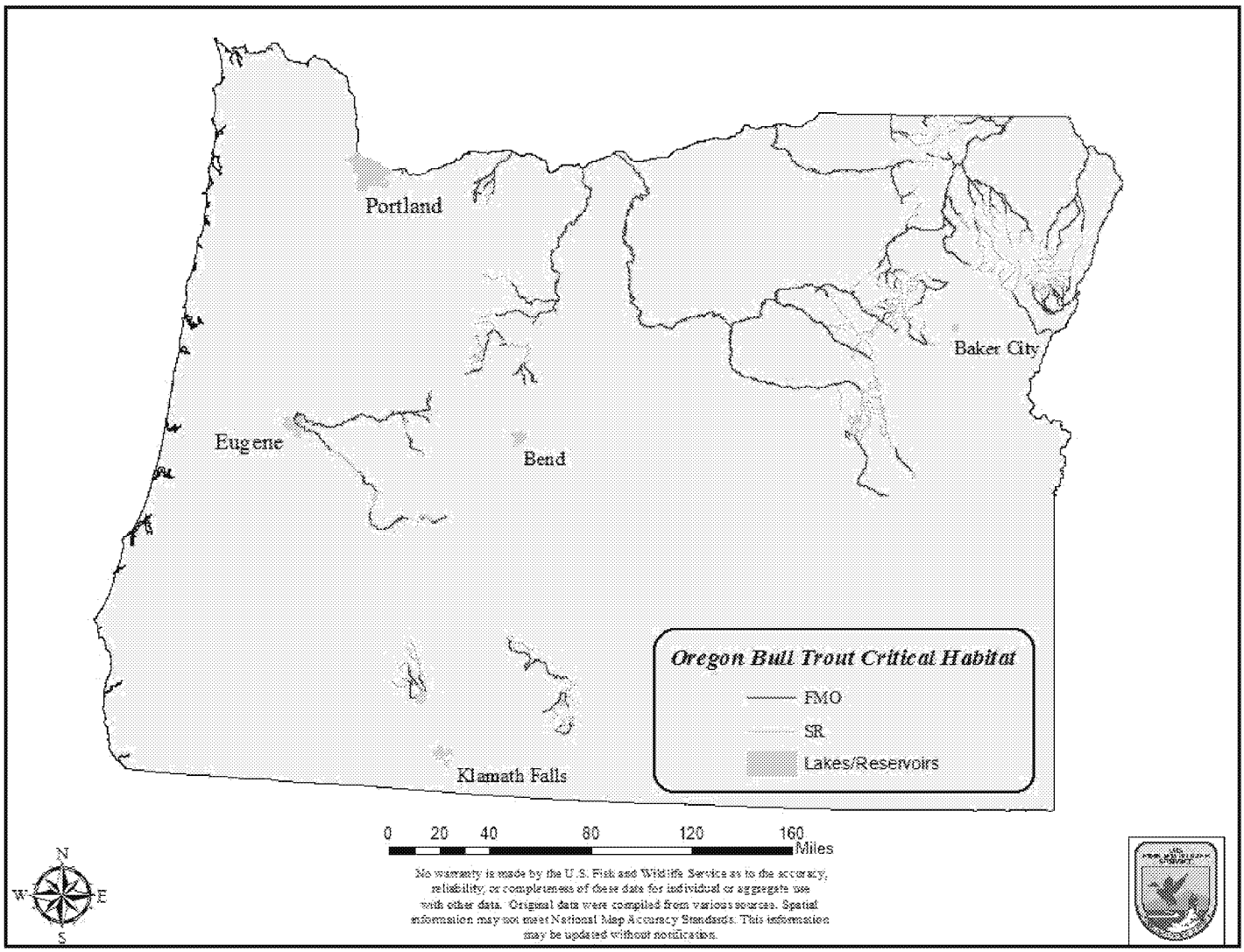


Figure 2-1. Bull Trout Critical Habitat within Oregon
(FMO – Foraging, migrating, and overwintering habitat; SR – Spawning and rearing habitat)

2.1.6.3 Threats specific to Oregon and discussion of threats relative to water quality (toxicity and temperature)

Bull trout within Oregon are still subject to all those threats outlined at the time of listing, and the new threats associated with climate change.

Increased stream temperatures and the presence of toxic substances both have tremendous potential to pose a threat to bull trout within Oregon. Habitat fragmentation combined with poor water quality and physical barriers have left most core areas for bull trout extremely vulnerable to decline.

Increased temperatures (those above 59 °F) pose as barriers to bull trout foraging and migration. Bull trout found within Oregon require high quality, cold water for spawning. Though it is generally accepted that temperatures ranging from 36 to 59 °F are acceptable for bull trout this can vary to some degree by core area, or local population. The Willamette National Forest reports that they have never observed bull trout spawning in temperatures greater than 7.5 °C (45.5 °F) (Ray Rivera, Fish Biologist, U.S. Forest Service, Willamette National Forest, Eugene, Oregon, pers. comm., 2011).

The presence of toxins within bull trout habitat may present variable direct and indirect threats for bull trout within many locations in Oregon. Depending on the toxin, these effects could range from death or injury, to reduced reproduction potential from disruption of homing abilities for migratory fish. Indirect threats to bull trout associated with decreases in already low numbers of prey species resulting from both direct mortality of the prey species, and habitat changes that reduce the carrying capacity of prey species habitat could occur.

2.2 Lost River Sucker and Shortnose Sucker

2.2.1 Species Description

2.2.1.1 Taxonomy

The Lost River sucker was first described by Cope (1879), as *Chasmistes luxatus*, based on specimens collected in Upper Klamath Lake. Shortly afterward, *Catostomus rex* was described from the Lost River and Tule Lake, in south-central Oregon and northern California, but has been regarded as a synonym of *D. luxatus*. Other authors have placed the Lost River sucker either in the genus *Deltistes* or *Catostomus,* but currently *Deltistes* is the generic epithet most widely used by fish taxonomists, and it is the name accepted by the American Fisheries Society and USFWS (Andreasen 1975, Miller and Smith 1981, Williams et al. 1985, USFWS 1988, Markle et al. 2005).

The shortnose sucker was described by Cope (1879) as *Chasmistes brevirostris*, based on specimens collected from Upper Klamath Lake. Fowler (1913) suggested that *C. brevirostris* should be transferred to the genus *Lipomyzon*, but this has not been adopted by later workers. Two additional nominal taxa, *C. stomias* and *C. copei,* were later described from Upper Klamath Lake and vicinity, but were synonymized with *C. brevirostris* by Miller and Smith (1981).

Molecular genetic evidence suggests that the genus *Chasmistes* is artificial and should be synonymized under *Catostomus* (Wagman and Markle 2000), but no formal revision has been published.

2.2.1.2 Species Description

Upper Klamath River Basin suckers can be difficult to identify owing to considerable phenotypic variability, effects of hybridization, and to growth- and gender-related morphological changes. The three upper Klamath Basin sucker species (the two aforementioned and the Klamath largescale sucker - *Catostomus snyderi)* are best distinguished based on multiple characters, since no one character is diagnostic. Diagnostic characters include: lip morphology, vertebral number, gill raker number, and head shape (Andreasen 1975, Markle and Simon 1993, Markle et al. 2005). Markle et al. (2005) recently provided a diagnostic key for the three upper Klamath River basin sucker species which is summarized below in Table 2-3.

Table 2-3. Comparative morphology of Klamath largescale suckers, Lost River suckers, and shortnose suckers, based on Markle et al. (2005).

Species	Maximum Body Length (cm)	Body Shape	Snout Shape	Lip Shape	Number of Gill Rakers	Number of Vertebrae
Klamath Largescale Sucker	55	Deep	Short and rounded	Large and papillose, no gap	29-40	40-46
Lost River Sucker	100	Elongate	Long and angular	Small with a gap	23-37	44-48
Shortnose Suckers	64	Elongate	Short and rounded	Small with a gap	30-45	41-45

The greatest difficulty in identification of upper Klamath Basin suckers is separating the two listed species. However; Markle et al. (2005), state that they can be "readily distinguished" by lip morphology and gill raker counts. Other information can be helpful in separating these species as well and Markle et al. (2005) point to some differences in local distributions, for example Klamath largescale sucker is less common in Upper Klamath Lake than the shortnose sucker, but is the only sucker found in the upper Williamson River. Also, spawning run time and location is somewhat different in the Sprague River with Klamath largescale sucker running mid-March to early April and many spawning upstream of Chiloquin, whereas the shortnose suckers run mid-April to early May and mostly spawn below Chiloquin.

Field identification of individual suckers can be problematic due to the high degree of morphological variability expressed by each species and the "mixing" of phenotypes caused by hybridization. Therefore, accurate identification is frequently dependent on the experience of the observer, but is not accurate in every case, thus many specimens are reported as, "species uncertain." In an effort to ensure Klamath sucker identifications are as accurate and consistent as

possible, Oregon State University (OSU), USGS, Klamath Tribes, USBR, and USFWS biologists have participated in informal sucker identification workshops.

2.2.1.3 Legal Status

The Lost River sucker and shortnose sucker were listed as endangered on July 18, 1988 (USFWS 1988). Both species are also listed as endangered by the States of Oregon and California. A recovery plan for Lost River sucker and shortnose sucker was finalized on March 17, 1993 (USFWS 1993b). Five-year reviews for the Lost River sucker and shortnose sucker were completed on July 19, 2007 (USFWS 2008f and USFWS 2007 b, c). A considerable amount of scientific information has been collected since the 1993 recovery plan and an updated, revised draft recovery plan for the Lost River sucker and shortnose sucker was released in 2011 (USFWS 2011a).

On September 9, 1991, USFWS received a 60-day notice of intent to sue from the Oregon Natural Resources Council (ONRC) for failure to prepare a recovery plan and to designate critical habitat for the Lost River sucker and shortnose sucker. On December 1, 1994, we published proposed critical habitat for Lost River sucker and shortnose sucker (USFWS 1994b); that proposal was never finalized. A settlement agreement was reached that stipulates USFWS submit a final rule designating critical habitat for the Lost River sucker and the shortnose sucker to the Federal Register no later than November 30, 2012 (Wood et al. v. Thorson et al., No. 91–cv–6496–TC [D. Or.]).

2.2.2 Critical habitat description

2.2.2.1 Current critical habitat status

Critical habitat was originally proposed by USFWS for the Lost River and shortnose suckers in 1994 (USFWS 1994b), but was not finalized. However, on December 7, 2011, USFWS re-proposed critical habitat for the two species (USFWS 2011d). The new proposal includes two proposed critical habitat units for each species, as described below.

- The Upper Klamath Lake Unit (Unit 1; Figure 2-1), situated in Klamath County, Oregon, includes Upper Klamath Lake and Agency Lakes, the Link River and upper Klamath River downstream to Keno Dam (known as the Keno Reservoir), as well as portions of the Williamson and Sprague Rivers, for a total of 90,415 acres and 119 RM. This unit is the same for both species with the following exceptions: (1) for Lost River suckers, the unit includes the lower Wood River and Crooked Creek and extends up the Sprague River to the Beatty Gap east of Beatty (near RM 75), and (2) for shortnose suckers, the unit does not include the Wood River or Crooked Creek and it extends up the Sprague River only as far as Braymill near RM 8.

- The Lost River Basin Unit (Unit 2; Figure 2-1) is situated in Klamath and Lake Counties, Oregon and Modoc County, California. It includes Clear Lake and its main tributary Willow Creek for both Lost River suckers and shortnose suckers, and Gerber Reservoir and its main tributaries for shortnose suckers only. Additionally there are differences in the amount of upstream proposed critical habitat in Willow Creek for the two species.

For the Lost River sucker, proposed critical habitat includes Willow Creek and its tributary Boles Creek upstream to Avanzino Reservoir in California. For shortnose suckers, proposed critical habitat extends up Willow Creek to Boles Creek and upstream past Fletcher Creek and includes Fourmile and Wildhorse Creeks in California and also includes Willow Creek to its East Fork in Oregon.

A summary of the differences for Lost River sucker and shortnose sucker is below:

	1994 Proposal Total	2011 Proposal Total
	880,000 acres (6 units)	241,438 acres, 274 miles (2 units)
Lost River Sucker:	424,000 acres	117,848 acres, 146 miles
Shortnose Sucker:	456,000 acres	123,590 acres, 128 miles

2.2.2.2 The primary constituent elements (PCEs)

The physical and biological features essential to the conservation of the species, which may require special management considerations or protection, include, but are not limited to:

(1) Space for individual and population growth and for normal behavior,
(2) Food, water, air, light, minerals, or other nutritional or physiological requirements,
(3) Cover or shelter,
(4) Sites for breeding, reproduction, or rearing (or development) of offspring, and
(5) Habitats that are protected from disturbance or are representative of the historical, geographical, and ecological distributions of a species.

Primary constituent elements (PCEs) are the specific elements of physical and biological features that are essential to the conservation of the species. Based on our current knowledge of the physical or biological features and habitat characteristics required to sustain the species' life-history processes, the PCEs specific to self-sustaining Lost River sucker and shortnose sucker populations are:

- PCE 1- *Water.* Areas of sufficient quantity and depth within lakes, reservoirs, streams, marshes, springs, groundwater sources, and refuge habitats with minimal physical, biological, or chemical impediments to connectivity. Water should exhibit depths ranging from less than 3 feet up to 15 feet to accommodate each life stage. Water quality characteristics should include temperatures of less than 28 °C; pH less than 9.8; dissolved oxygen levels greater than 4.0 mg/L; algal toxins (less than 1.0 µg/L); and un-ionized ammonia (less than 0.5 mg/L). Elements also include natural flow regimes that provide flows during the appropriate time of year or, if flows are controlled, minimal flow departure from a natural hydrograph.

- PCE 2 - *Spawning and Rearing Habitat.* Streams and shoreline springs with gravel and cobble substrate at depths typically less than 4 feet with adequate velocity to allow spawning to occur. Areas identified in PCE1 containing emergent vegetation adjacent to open water that provides habitat for rearing. This facilitates growth and survival of suckers, as well as protection from predation and protection from currents and turbulence.

- PCE 3- *Food.* Areas containing an abundant forage base, including an array of small aquatic invertebrates especially midges, cladocerans, and copepods.

2.2.2.3 Identification of the potential effects of toxics or water temperature levels to alteration of the PCEs

Potentially all of the PCEs for both of the listed suckers may be affected by the presence of toxics or increased water temperatures within the environment. The effects will vary greatly depending on a number of factors which include which toxic substance is present, the amount of temperature increase, the likelihood that critical habitat would be affected (i.e., probability), and the severity and intensity of any effects that might occur (i.e., magnitude).

2.2.3 Life History

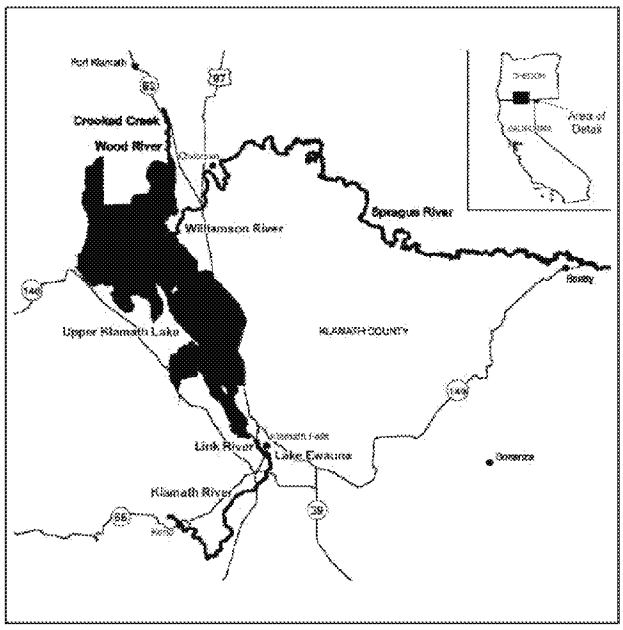
2.2.3.1 Reproduction

Both suckers spawn from February through May over gravel substrates in streams and rivers (Buettner and Scoppettone 1990). The Lost River sucker also spawns over rock and gravel substrates associated with shallow, spring-influenced areas along the eastern shore of Upper Klamath Lake (Barry et al. 2007b). Females broadcast their eggs where they fall into crevices between gravel or they are buried slightly.

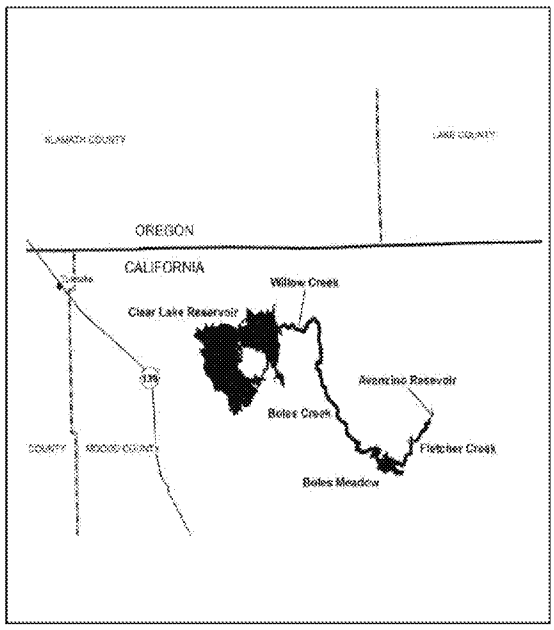
Both sucker species grow rapidly in their first 5 to 6 years, reaching sexual maturity sometime between 4 and 6 years for shortnose sucker and 4 and 9 years for Lost River sucker (Perkins et al. 2000b). Lost River sucker and shortnose sucker have been aged to 55 and 33 years, respectively. Females produce a large number of eggs, 44,000 to 200,000 per year for Lost River sucker and 18,000 to 70,000 per year for shortnose sucker when they spawn (Buettner and Scoppettone 1990). Larger, older females produce substantially more eggs and, therefore, can contribute relatively more to recruitment than a recently matured female. However, only a small percentage of the eggs survive to become larvae. Because adult are potentially long-lived and fecund these life history traits should make the species less sensitive to larval and juvenile mortality, but under current conditions adult survival is low and there is inadequate recruitment, as will be discussed later. Lost River sucker and shortnose sucker do not normally die after spawning and can spawn many times during their lifetime.

2.2.3.2 Population Structure

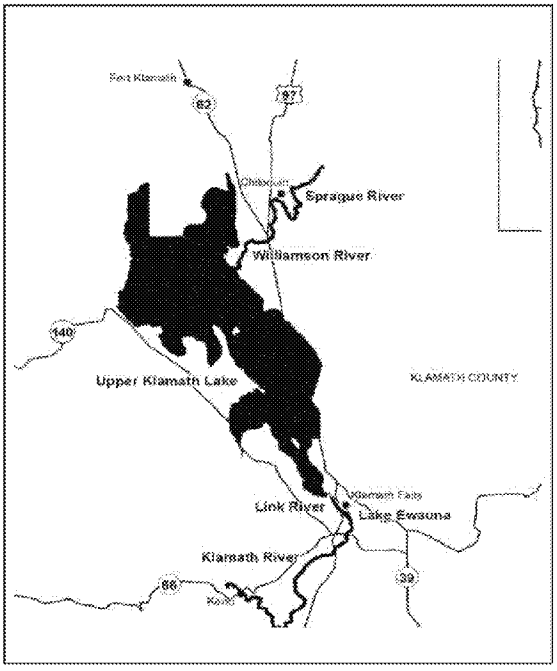
The Lost River sucker population in the Upper Klamath Lake appears to consist of two distinct stocks: (1) Several thousand Lost River sucker and a few shortnose sucker that spawn along shoreline springs, and (2) tens of thousands Lost River sucker fish that spawn in the Williamson and Sprague Rivers (Perkins et al. 2000b). Mark-recapture data show that the two stocks maintain a high degree of fidelity to spawning areas and therefore seldom interbreed (Hayes et al. 2002, Barry et al. 2007a, b). Shortnose sucker spawning is primarily confined to the Williamson River system, so there is only one substantial population or stock in Upper Klamath Lake.



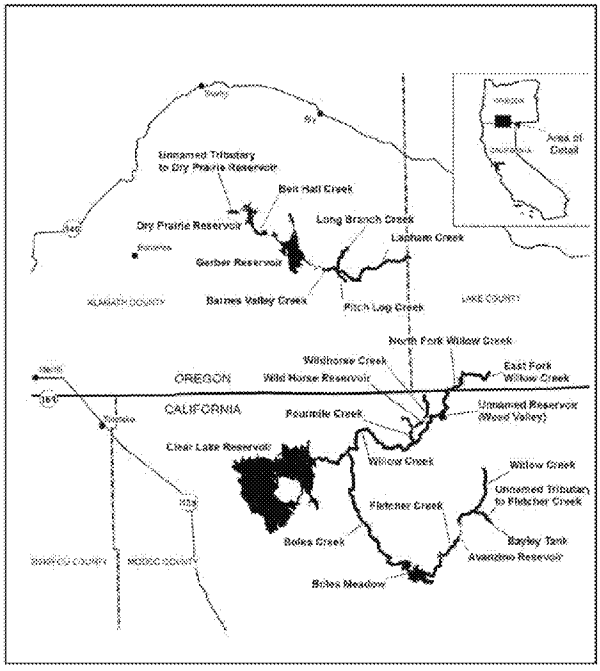
Lost River Sucker PCH Unit 1



Lost River Sucker PCH Unit 2



Shortnose Sucker PCH Unit 1



Shortnose Sucker PCH Unit 2

Figure 2-2. Critical habitat units for Lost River and shortnose suckers proposed in 2011 (USFWS 2011d).

2.2.3.3 Ecology Habitat Characteristics

The Lost River sucker and shortnose sucker have complex life histories that include stream/river, lake, marsh, and shoreline spring habitats and both species use a number of different aquatic habitats through their lives. Adults primarily occupy open water habitats with depths of 3 feet to 15 feet, but appear to prefer depths from 5 feet to 10 feet (Peck 2000, Reiser et al. 2001, Banish et al. 2009). Lost River sucker and shortnose sucker are generally limited to lake habitats when not spawning, although river-resident fish have been documented, especially in the Lost River system (Buettner and Scoppettone 1991).

Soon after hatching, when larvae reach about 0.2 to 0.6 inches total length (TL) and are mostly transparent with a small yolk sac, they move out of the gravel and into the water column (Buettner and Scoppettone 1990). Larval suckers spend relatively little time in rivers/streams before drifting downstream to the lakes by mid-July (Cooperman and Markle 2003, 2004; Ellsworth et al. 2009); however, some instream rearing has been observed in the Sprague River and elsewhere. Larval habitat is generally in shallow water along the shoreline in both vegetated and unvegetated habitats (Buettner and Scoppettone 1990, Cooperman and Markle 2004, Crandall et al. 2008). Juvenile suckers also occupy a wide variety of near and off-shores habitat in Upper Klamath Lake including emergent wetlands and non-vegetated areas with sand, mud, gravel, and cobble substrates (Buettner and Scoppettone 1990; Simon et al. 1996, 1998; Simon and Markle 2001, 2004; Hendrixson et al. 2007a, b; Burdick et al. 2008) and move offshore into the lake as they grow and move southward. Water quality, especially dissolved oxygen concentrations are likely to affect distributions (Burdick et al. 2009, Burdick and VanderKooi 2010).

2.2.4 Status

2.2.4.1 Historical status and distribution

Prior to settlement Lost River suckers and Shortnose suckers occurred in UKL, Tule Lake, Lower Klamath Lake, and presumably Clear Lake, as well as their tributaries. However, at the time of listing, these species were known from UKL and its tributaries and outlet (Klamath Co., Oregon), including a "substantial population" of shortnose sucker in Copco Reservoir (Siskiyou Co., California), as well as collections of both species from Iron Gate Reservoir (Siskiyou Co., California) and J.C. Boyle Reservoir (Klamath Co., Oregon), and Lost River sucker from Sheepy Lake and Lower Klamath Lake (Siskiyou Co., California). Remnants or highly hybridized populations were also stated to occur in the Lost River system (Klamath Co., Oregon, and Modoc and Siskiyou Co., California) including both species in Clear Lake Reservoir (Modoc Co., California) and Lost River sucker in Tule Lake (Siskiyou Co., California; USFWS 1988, p. 27130).

Although not stated explicitly, the reference in the listing to "highly hybridized populations" in the Lost River Basin probably refers to shortnose suckers within Gerber Reservoir (Klamath Co., Oregon). Spawning likely occurred throughout the Upper Klamath Lake drainage in both rivers and springs along shoreline of the lake (Andreasen 1975, Stein 1982, NRC 2004). Spawning also occurred in significant numbers in the Lost River system (Howe 1969, Bendire 1889), some of which in the Big Springs area near Bonanza, Oregon.

These two fishes were once very abundant and were important seasonal foods of Native Americans and white settlers in the upper Klamath River basin prior to about 1900 (Howe 1969, Cope 1879, Gilbert 1898). Sucker spawning migrations occurred in the spring at a critical time when winter food stores had been exhausted. The Klamath and Modoc Indians dried suckers for later use. It was estimated that the aboriginal harvest at one site on the Lost River may have been 50 tons annually (Stern 1966). Settlers built a cannery on the Lost River and suckers were also processed into oil and salted for shipment. In 1900, the Klamath Republican newspaper reported that "mullet," as suckers were referred to, were so thick in the Lost River that a man with a pitch fork could throw out a wagon load in an hour. The first reference to sport fishing of "mullet" appears to be a 1909 reference to sportsmen snagging "mullet" in the Link River at Klamath Falls.

In 1959, suckers were made a game species under Oregon State law and snagging suckers in the Williamson and Sprague River was popular with locals and out-of-town sportsmen (Bragg 2001, Markle and Cooperman 2002). In the 1960s Oregon Department of Fish and Wildlife (ODFW) estimated 100,000 pounds of suckers per year (catch 12,500 fish) were harvested. ODFW data indicated from 1966 through 1978, an approximate 50% decline in catches (from 3.5-5.6 suckers per angler before the 1969 bag limit, to 1.5-3.0 afterwards). More than 3,000 suckers were taken in the snag fishery in 1968 (Golden 1969). Numbers of harvested suckers from spawning runs in the Sprague and lower Williamson Rivers increased from 1.2 fish per hour in 1966 to 4.7 fish/hour in 1969 and then, from 1969 on, there was a steady decline to 0.8 fish/hour in 1974 (Andreasen 1975). Average weight of suckers caught in the fishery declined about 40% from 1966 to 1974 (from 7.5 to 4.9 pounds), and declines continued to the time of listing. By 1985, Bienz and Ziller (1987) estimated the harvest had dropped by about 95%, and based on this information, the game fishery was terminated in 1987, just prior to federal listing (USFWS 1988).

2.2.4.2 Current status and distribution

At the time of listing, Lost River sucker and shortnose sucker were known from Upper Klamath Lake and its tributaries (Klamath Co., Oregon), the Lost River (Klamath Co. Oregon, and Modoc and Siskiyou Co., California), Clear Lake (Modoc Co., California), the Klamath River above Keno (Klamath Co., Oregon), and Klamath Rivers below Keno (Klamath Co., Oregon, and Siskiyou Co., California; USFWS 1988; Figure 2-3). The largest populations occur in Upper Klamath Lake and Clear Lake and their tributaries. Clear Lake and Upper Klamath Lake are also the primary reservoirs for the Klamath Project, so the status of the species is primarily determined by their status in the action area.

Currently the total area of occupied lake habitat for Lost River sucker and shortnose sucker is about 118,000 acres, of which approximately 65% is in Upper Klamath Lake, which covers approximately 77,000 acres at full pool. The remaining habitat is in Clear Lake (25,000 acres); Tule Lake sump 1A (9,000 acres); Gerber Reservoir (4,000 acres); and Keno Reservoir (2,500 acres). Disjunct 'populations' also occupy areas outside of their historic distribution in Copco, Iron Gate, and J.C. Boyle reservoirs on the Klamath River (NRC 2004). Additional information about the distribution of the Lost River sucker and shortnose sucker is present in the 2008 Klamath Project biological opinion (USFWS 2008g).

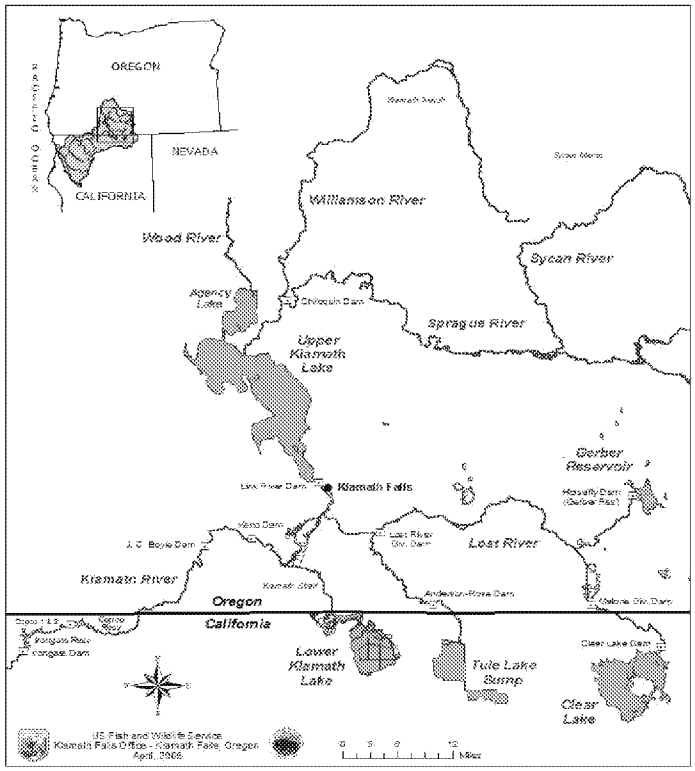


Figure 2-3. Map of the Upper Klamath River Basin showing primary water bodies.

Upper Klamath Lake and Tributaries

Upper Klamath Lake, in Klamath County, at approximately 80,000 acres, is the largest natural lake is the largest natural lake in Oregon. Adult Lost River suckers and shortnose suckers are widely distributed in Upper Klamath Lake during the fall and winter (USFWS 2002a, NRC 2004). In the spring months, they congregate in the north end of the lake near Goose Bay and Modoc Point prior to moving into tributaries or shoreline spawning areas for spawning (Hendrixson et al. 2003). During summer months adults are primarily found in the northern portion of the lake above Bare Island (Peck 2000, Banish et al. 2009). Reasons for this summer distribution appear to be related to better water quality near spring-fed Pelican Bay and the Williamson River (Reiser et al. 2001, USFWS 2002a, Banish et al. 2009). During the summer and early fall, Upper Klamath Lake water quality conditions periodically deteriorate to stressful and even lethal levels for suckers as a result of decomposition of massive algae blooms and resultant low levels of dissolved oxygen (Perkins et al. 2000a, Loftus 2001, Wood et al. 2006, Morace 2007). Multi-year radio telemetry studies have documented Lost River sucker and shortnose sucker concentrating in or near Pelican Bay during periods of deteriorating water quality, presumably to seek refuge at areas of better water quality (Peck 2000, Banish et al. 2009).

A discrete group of Lost River sucker and shortnose sucker are known to spawn at sites along the eastern shoreline of Upper Klamath Lake below Modoc Rim, a prominent fault scarp (Andreasen 1975, Buettner and Scoppettone 1990, Shively et al. 2000, Hayes and Shively 2001, Barry et al.

2007b). All of the sites are associated with spring seeps that affect water temperature, which is warmer than the adjacent lake during the spawning season. Current spawning populations at the shoreline springs of Upper Klamath Lake are likely to number in the thousands of Lost River sucker and less than 100 shortnose sucker (Barry et al. 2007b, Janney et al. 2007).

Known areas of concentrated Lost River sucker spawning in the Williamson and Sprague Rivers include the lower Williamson River from river mile (RM) 6 to the confluence of the Sprague River (RM 11), lower Sprague River below Chiloquin Dam, and in the Beatty Gap area of the upper Sprague River (RM 75) (Buettner and Scoppettone 1990, Tyler et al. 2007, Ellsworth et al. 2008). Other areas in the Sprague River watershed where Lost River sucker may spawn include the lower Sycan River and in the Sprague River near the Nine Mile area (Ellsworth et al. 2008). Shortnose suckers from Upper Klamath Lake currently spawn in the lower Williamson and Sprague Rivers (Tyler et al. 2007, Ellsworth et al. 2008). The few adult shortnose sucker captured at shoreline spawning areas in Upper Klamath Lake indicate that some shortnose sucker spawning is likely to still occur at these locations (Hayes et al. 2002; Barry et al. 2007a, b). Although species identification is not clear, a small number of suckers presumed to be shortnose sucker may spawn in the Wood River (USBR 2001). It is possible that sucker spawning may occur in other tributaries to Upper Klamath Lake; however, recent investigations have not located suckers in tributaries other than the Williamson, Sprague, and Wood Rivers.

In the Williamson River, larval out-migration begins in April and is generally completed by July. Downstream movement takes place mostly at night and near the water surface (Klamath Tribes 1996, Tyler et al. 2003), but movement does occur along the river margin during the day, at least in the lower river. Once in the lake, larvae disperse to near-shore areas where they move into shallow, near-shore, vegetated and unvegetated habitats (Cooperman 2004; Cooperman and Markle 2004; Crandall et al. 2008; Erdman and Hendrixson 2009, 2010).

In Upper Klamath Lake, larval suckers are first captured by researchers in early April during most years, with peak catches occurring in late May to early July (Cooperman and Markle 2000, Crandall et al. 2008, Erdman and Hendrixson 2009). Larval habitat is generally along the shoreline, in water 0.5-3 feet deep (Buettner and Scoppettone 1990, Cooperman and Markle 2004). Larval density in near-shore habitats is usually low (less than $10/m^2$), but densities as high as 120 larvae/m^2 have been documented (Buettner and Scoppettone 1990; Markle and Simon 1993; Simon and Markle 2001, 2004; Simon et al. 1995, 1996).

Larval density is generally higher within and adjacent to emergent vegetation than it is in areas devoid of vegetation (Klamath Tribes 1996, Cooperman and Markle 2004, Crandall et al. 2008, Erdman and Hendrixson 2009). Nevertheless, studies at the Williamson River delta have shown that some sucker larvae also use unvegetated areas (Erdman and Hendrixson 2009, 2010). This is especially the case for larval Lost River sucker that showed no preference based on depth or vegetation, whereas larval shortnose sucker were 1.4 times more numerous in shallow vegetated habitats than unvegetated habitats (Erdman and Hendrixson 2009). At the delta, highest catches of larvae in vegetated habitats was in areas dominated by hardstem bulrush (*Scripus acutus*; Erdman and Hendrixson 2009).

Emergent vegetation is believed important to larval suckers because it provides habitat for prey such as zooplankton, macroinvertebrates, and periphyton (the community of microscopic organisms that live on submerged surfaces in aquatic environments) (Klamath Tribes 1996, Cooperman and Markle 2004, Crandall et al. 2008). At the Williamson River delta, Erdman and Hendrixson (2009) found that larvae collected in shallow vegetated habitats had fuller guts than those collected in unvegetated habitats, suggesting that more food was available in vegetated habitats. Emergent wetlands also may provide cover from predators such as fathead minnows (Markle and Dunsmoor 2007) and yellow perch. Additionally, larvae that reside in wetlands are less likely to be swept down lake via wind-driven currents and be entrained into the Link River where most are lost from the population (Markle et al. 2009). Water temperature can be higher in emergent vegetation than in unvegetated areas and this is likely to increase larval growth rates and shorten the time they are most vulnerable to predators (Crandall et al. 2008, Erdman and Hendrixson 2009).

Juvenile (age 0 or young-of-the-year) suckers use relatively shallow water with depths of less than 4 feet deep that are vegetated or un-vegetated (Buettner and Scoppettone 1990). Juvenile suckers use a wide variety of near-shore habitat including emergent wetlands and non-vegetated areas and mud-bottom off-shore habitat (Terwilliger 2006; VanderKooi et al. 2006; Hendrixson 2007a, b). Juvenile Lost River sucker appear to be less dependent on inshore habitats than are shortnose suckers. As they grow during the summer, many juvenile suckers move offshore and down the lake and into the Link River and Keno Reservoir (Gutermuth et al. 2000).

Keno Reservoir

Keno Reservoir is the first of five reservoirs on the Klamath River and is located just downstream of Upper Klamath Lake, near Klamath Falls, Oregon. Fisheries surveys in Keno Reservoir have been conducted infrequently and have generally been short in duration (Hummel 1993, Piaskowski 2003, PacifiCorp 2004). The only intensive monitoring effort was conducted by Terwilliger et al. (2004). Larvae and age-0 suckers were generally most abundant in the upper part of Keno Reservoir and decreased downstream. Juvenile and sub-adult and adult suckers are less common; although, Reclamation biologists have recently captured adult suckers in Lake Ewauna, which is located at the upper end of the Keno Reservoir, indicating that a population of several thousand resides there (Kyger and Wilkens 2011).

The relatively low number of adult suckers residing in Keno Reservoir appears to be related primarily to poor water quality in the summer (Piaskowski 2003). The Keno Reservoir is listed by the State of Oregon for not meeting Clean Water Act section 303(d) water quality standards for pH, chlorophyll-a, and dissolved oxygen and dissolved oxygen concentrations in the Keno Reservoir regularly reach stressful and lethal levels for suckers during July and August (Piaskowski 2003, Deas and Vaughn 2006, USBR 2007, Kirk et al. 2010, Sullivan et al. 2011). Fish die-offs, including juvenile suckers, have been a regular occurrence in the reservoir (William Tinniswood, in litt., 2006).

Gerber Reservoir

Gerber Reservoir is a small reservoir with a surface area of approximately 4,000 acres and is located in southeastern Klamath County, Oregon (Figure 2-4). Monitoring in Gerber Reservoir

has documented a substantial shortnose sucker population (or shortnose sucker x Klamath largescale suckers (*Catostomus snyderi*), exhibiting multiple size classes and presumably multiple age classes. Data from 2004 to 2006 indicate a lower frequency of larger adults compared to those from 2000 (Piaskowski and Buettner 2003, Barry et al. 2007c, Leeseberg et al. 2007). Such a change in length frequency suggests relatively good recruitment but low adult survivorship (USFWS 2002a). Lost River suckers do not occur in Gerber Reservoir probably because none were apparently upstream of dam when it was constructed (USBR 2001, Piaskowski and Buettner 2003, Barry et al. 2007c, Leeseberg et al. 2007).

While the population of shortnose sucker in Gerber Reservoir appears to have more frequent recruitment than some other populations, the problems of restricted distribution and lack of genetic connectivity with other populations still exist (USFWS 2002a). A high degree of hybridization between shortnose sucker and Klamath longnose suckers are thought to occur in Gerber Reservoir (Markle et al. 2005). However, until the status of these fish has been resolved, USFWS considers the Gerber sucker population to be shortnose sucker.

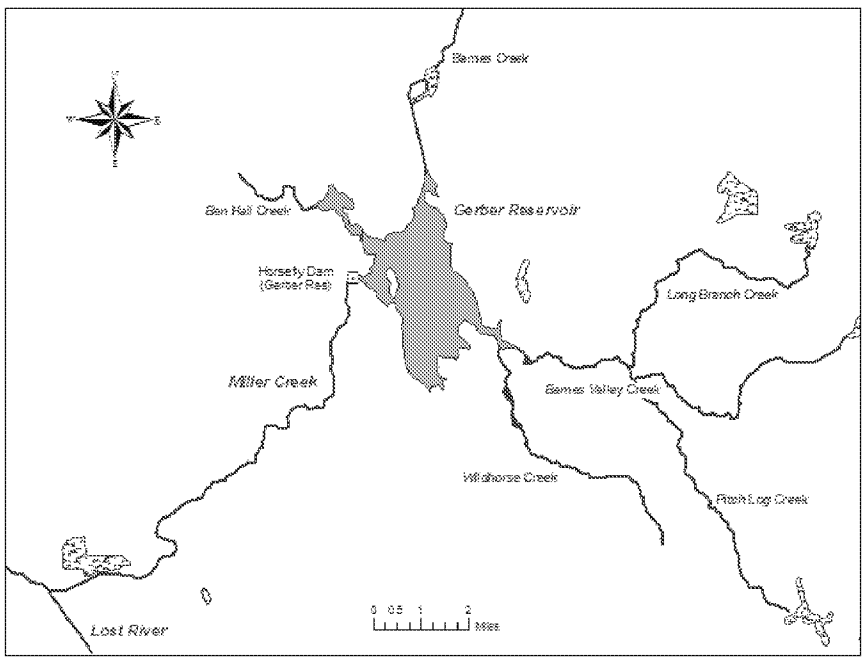


Figure 2-4. Vicinity map for the Gerber Reservoir area in Klamath County, Oregon.

Adult Demography and Population Trends in Upper Klamath Lake

The size of Upper Klamath Lake and the scarcity of Lost River and shortnose suckers make it difficult to accurately estimate their abundance in the lake. Lost River sucker and shortnose sucker demography (demography is the study of population parameters) has been monitored since the mid-1990s using mark-recapture methods and passive integrated transponder tags. It is also difficult to determine the age of Lost River and shortnose suckers. The length of fish is often used to estimate age of individuals, and the distribution of fish in various age classes is often estimated by assessing the number of fish of various lengths.

Lost River and shortnose sucker populations transformed from ones dominated by old fish with little size diversity and consistently poor recruitment in the late 1980s and early to mid-1990s, to populations dominated by smaller young adult fish and very few remaining large individuals by the late 1990s (Janney et al. 2008). This marked shift in size structure to smaller individuals suggests that substantial recruitment in these populations occurred sometime during the mid-1990s from fish born in the early 1990s. In recent years, populations of both species exhibited a slight increasing trend in length (i.e., 1 to 1.5 cm increase in median fork length per year) while the number of age classes in the population decrease (Janney and Shively 2007, Janney et al. 2008). Decreasing relative abundance of younger fish suggests that populations are comprised mostly of similarly-aged, older individuals, and that recent substantial recruitment is lacking.

One way ecologists determine if a population is increasing or decreasing is by evaluating survival, mortality, and recruitment, and calculating a variable called lambda (λ) over time. When λ is greater than 1, the population is increasing, and when it is less than 1 the population is decreasing. Several sources of information can be used to derive λ, such as regular counts of individuals, recruitment rates, and survival rates. This information can be determined from regular sampling, but it must be quantified several times before accurate assessments of trends in abundance can be made. When λ is known from a number of samplings conducted over a long period of time, the relative change in the size of a population can be determined, which is a variable known as Δt.

When the USGS Lost River sucker and shortnose sucker capture-recapture program began in 1995, few fish were captured and tagged and mean estimates of survival and λ had little precision or statistical rigor, and Δt could not be accurately calculated. As the number of tagged fish increased each year, precision of these estimates improved and in 2001 (for shortnose sucker Williamson-Sprague River spawning fish) and 2002 (for shoreline- spawning Lost River sucker) information became adequately precise to accurately calculate λ, relatively accurate estimates of Δt became possible.

Results from passive integrated transponder tag studies in Upper Klamath Lake from 2002 to 2007 show that annual survival probabilities of shoreline springs spawning Lost River sucker ranged from 0.80 to 0.95, with a mean of 0.90, and Δt over the period was 0.56 for males and 0.75 for females. These Δts show that the abundance of male and female, shoreline-spring-spawning Lost River sucker in 2007 was 56 % and 75%, respectively, of their 2002 abundance. Estimates for river-spawning shortnose sucker show that annual survival probabilities were lower for this subpopulation. From 2001 to 2007 annual survival probabilities of river-spawning shortnose sucker were more variable than for Lost River sucker and ranged from 0.68 to 0.94, with a mean of 0.82. Over this period, Δts of male and female, river-spawning shortnose sucker were 0.42 and 0.49, respectively. Similar data are not currently available for Upper Klamath Lake Lost River sucker river-spawning fish or for Clear Lake Lost River sucker and shortnose sucker populations. This information will be forthcoming in the future when the number of tagged fish is sufficiently large to provide relatively precise, statistically rigorous estimates.

Demography of Lost River sucker and shortnose sucker Populations in Clear Lake

Historically, large Lost River sucker and shortnose sucker spawning migrations occurred from Tule Lake up the Lost River to near Olene and Big Springs near Bonanza (Howe 1969, USFWS 2002a). Clear Lake currently supports the only substantial populations of shortnose sucker and Lost River sucker in the Lost River subbasin. Less is known about shortnose suckers and Lost River suckers in Clear Lake than those in Upper Klamath Lake because monitoring studies have been sporadic over the past 35 years, and studies similar those conducted by Janney et al. (2008) in Upper Klamath Lake were not initiated in Clear Lake until 2006 (Barry et al. 2009). Data collected by Koch et al. (1975) and Andreasen (1975) suggested both populations were in decline; however, monitoring from 1989-2000 indicated that populations were relatively large and had diverse age structures (Buettner and Scoppettone 1991, USBR 1994, Scoppettone et al. 1995, USFWS 2002a). In Upper Klamath Lake, 15 age classes were documented in the shortnose sucker population during 1989 and nine during 1993. Similar data are not available for Lost River sucker in Clear Lake, because they appear to be less abundant than shortnose sucker.

Summarizing historical and recently collected data, Barry et al. (2009) observed that populations of both species in Clear Lake have undergone major demographic changes during the past 15 years. Populations in the mid-1990s showed little evidence of recruitment and consisted mostly of large and presumably older suckers. The abundance of large suckers decreased in the late 1990s and early 2000s, and current populations are mostly ones recruited into the adult population in the late 1990s (Barry et al. 2009). Length-frequencies from 2005 – 2009 studies found little evidence of shortnose sucker recruitment and that recruitment into the Lost River sucker population had been relatively consistent over the period. Variability in age class structure, longevity, and abundance Lost River suckers and shortnose suckers in Clear Lake is poorly understood in comparison with populations in Upper Klamath Lake. Several more years of sampling and analysis are needed before data are sufficient to discern current status of these populations and their demographic health.

Consulted-on Effects

Other Federal actions affecting the Lost River sucker and shortnose sucker or their proposed critical that have been consulted on under section 7, or that required a section 10 (a) permit include: operations of the USBR's Klamath Project; a Federal Energy Regulatory license for operation of PacifiCorps' Klamath Hydroelectric Project; the proposed removal of four Klamath River dams by the USBR; forest and rangeland management actions by the U.S. Forest Service and USBLM; U.S. Army Corps of Engineers, section 404 permits; energy development projects; transportation projects, ecosystem recovery projects; and others. None of the section 7 consultations still in effect for the Lost River sucker or shortnose sucker reached a jeopardy determination or adverse modification of their proposed critical habitat.

2.2.4.3 Threats

The reasons for listing as well as threats to the continued survival of Lost River sucker and shortnose sucker are well reviewed in a number of reports and peer reviewed articles (USFWS 1988; Markle and Cooperman 2002; NRC 2004; ISRP 2005; USFWS 2007b, c, 2008, 2011a, b; Rasmussen 2011). The major threats are discussed below.

Effects of Habitat Loss and Alteration

Loss and alteration of habitats (including spawning and rearing habitats) were major factors leading to the listing of both species (USFWS 1988) and continue to be significant threats to recovery. As noted above, both species use the spectrum of aquatic habitats during some stage of the life cycle, including river or stream habitats, open-water lake habitats, and the wetlands areas along banks and shores. However, negative impacts and alterations to each of these different habitats have occurred, and continue to threaten the recovery of these species. Suitable habitat has drastically declined due to conversion of wetlands to agricultural use and construction of irrigation and hydroelectric facilities, both of which drained lakes and wetlands, created barriers preventing access to spawning habitat, and caused mortality by entraining fish.

Effects of Nonnative Fishes

Nonnative fishes were identified as a potential threat at the time of listing through predation or as sources of exotic diseases/parasites, although no direct evidence was cited. Since then, controlled experiments have demonstrated that adult fathead minnows prey on sucker larvae (Markle and Dunsmoor 2007). In Upper Klamath Lake negative relationships between fathead minnow population size and larval sucker survival rates (i.e., higher fathead minnow populations are associated with lower sucker survival rates) have been observed (Markle and Dunsmoor 2007). Likewise, as indirect evidence, higher larval survival rates were also associated with greater water depth and shoreline vegetative cover, habitat which help larvae avoid predation (Markle and Dunsmoor 2007). These data suggest that predation by highly-abundant fathead minnows may be an important threat to larval sucker survival, and that loss of emergent wetland habitat may exacerbate this. Other nonnative fishes may also pose a threat to Lost River sucker and shortnose sucker; however, little quantitative information exists to indicate their influence on sucker abundance and distribution.

Effects of Adverse Water Quality

Most water bodies currently occupied by Lost River and shortnose suckers do not meet water quality standards for nutrients, dissolved oxygen, temperature, and pH set by the States of Oregon and California (Boyd et al. 2002, Kirk et al. 2010). Lost River and shortnose suckers are relatively tolerant of degraded water quality conditions in comparison to species like trout and salmon. Suckers tolerate higher pH, temperature, and un-ionized ammonia concentrations, and lower dissolved oxygen concentrations than many other fishes (Saiki et al. 1999, Meyer and Hansen 2002, NRC 2004). Nevertheless, both species are regularly adversely affected by poor summer water quality in Upper Klamath Lake, Keno Reservoir, Lost River subbasin, and the hydropower reservoirs downstream in the Klamath River (NRC 2004). Adverse water-quality conditions, which have primarily occurred in summer, have caused multiple incidents of mass adult mortality (Perkins et al. 2000a). The primary cause of water-quality-related mortality appears to be caused by hypoxia (i.e., low levels of dissolved oxygen), but high concentrations of un-ionized ammonia resulting from elevated total ammonia concentrations and high pH, could also be a contributing factor (Perkins et al. 2000a). Additionally, in the fish die-offs that occurred in Upper Klamath Lake in the 1990s, disease outbreaks contributed to mortality and continued to affect suckers after the adverse water conditions had abated (Perkins et al. 2000a, NRC 2004).

Adverse water quality conditions in Upper Klamath Lake are attributed to high nutrient loading, especially phosphorus, and the presence of blue-green algae, *Aphanizomenon flos-aquae* (AFA). This alga (actually categorized as a bacterium known as "cyanobacterium") now dominates the algal community from June to November, and because of the high concentrations of nutrients, especially phosphorus, available, is able to reach seasonally high biomass levels that can lead to highly degraded water quality (Boyd et al. 2002, NRC 2004, Wood et al. 2006, Morace 2007). These conditions affect Lost River and shortnose suckers because rapid algal decay depletes dissolved oxygen levels and can create toxic conditions for suckers, especially when water temperatures are high and wind speeds low (Perkins et al. 2000a, Boyd et al. 2002, NRC 2004, Wood et al. 2006, Morace 2007).

Water quality remains one of the most important, if not the most important, proximate factor threatening sucker existence; however, the uncertainty surrounding many of the potential ultimate factors (i.e., the complex interactions of factors causing poor water quality), including wetland reduction, natural nutrient loads, nonpoint sources, and water management, also make it one of the most difficult threats to address.

Effects of Algal Toxins

Some cyanobacteria, such as *Microcystis aerginosa*, which is also present in Upper Klamath Lake, produce toxins that may directly result in mortality or may indirectly cause mortality through a combination of disease and stress produced by hypoxia (low dissolved oxygen), high pH, and high ammonia concentrations. Recent studies by USGS provide preliminary support for a hypothesis that juvenile suckers in Upper Klamath Lake are at risk from biotoxins produced by *M. aerginosa* (USGS 2010). The toxin, microcystin, attacks liver cells. Up to 50% of juveniles sampled showed evidence of liver damage, some severe enough that death would likely result. Microcystin levels in the water samples in 2008 were up to 17x higher than is considered safe for drinking water. Microcystin is an algal toxin that affects the liver and can lead to death. In a 2007 survey in Upper Klamath Lake, 49% of a sample of juvenile suckers collected at 11 shoreline sites exhibited indications of microcystin exposure (Vanderkooi et al. 2010). However, these data are preliminary and further investigations are required to determine the extent of microcystin exposure and of the effects. Additionally, the means by which the toxin is introduced into the body remains unknown, but there is some evidence that suggesting that the toxin is indirectly ingested when suckers consume midge larvae, which feed on the algae.

Effects of Pathogens and Parasites

Degraded water quality conditions may weaken fish and increase their susceptibility to disease and parasites (Holt 1997, Perkins et al. 2000a). Parasites and pathogens were not identified as important threats at the time of listing; however, new information indicates that pathogens and parasites likely contribute to low rates of sucker survival, especially during adverse water quality events (USFWS 2007b, c). A number of pathogens have been identified from moribund (dying) suckers, but Columnaris disease or "gill rot" seems to be the primary organism involved (J. Scott Foott, in litt., 1997; Holt 1997). It is caused by the bacterium *Flavobacterium columnare*, which can damage gills and produce body lesions, which leads to respiratory problems and an imbalance of internal salt concentrations, which provides an entry route for lethal systemic pathogens.

Anchor worm, an external, copepod parasite affects suckers and other fish in Upper Klamath Lake and its incidence on age-0 suckers appears to be increasing (ISRP 2005). From 1994-1996, the percent of age-0 suckers parasitized by anchor worms ranged from 0% to 7%, but by 1997-2000 it had increased to between 9% and 40%. Anchor worms now infect about half of age-0 shortnose suckers. Parasites like anchor worm may not directly cause death to suckers, but they can provide a route for pathogens to enter fish, since they create a wound, or can make fish more susceptible to predation (Robinson et al. 1998). The degree to which parasites threaten sucker survival and reduce productivity is unknown.

Effects of Entrainment Losses

Movement of fish into irrigation systems through unscreened diversions was identified as a threat to the suckers at the time of listing (USFWS 1988). At that time thousands of suckers, including some adults, were entrained into the A-Canal, the largest diversion in the upper basin located near the Link River Dam. Although some of these fish were salvaged, many likely died (NRC 2004). The impact of entrainment into the irrigation system of the Klamath Project was reduced by construction of screening facilities over the A-Canal; although larvae are still at risk. Fish screened from entering the A-Canal are returned via pipeline to the Link River above the dam (Marine and Gorman 2005). Further investigations are needed to determine the overall effects and stress on transferred fish and if fish expelled through the pipeline remain in Upper Klamath Lake or are subsequently entrained by flows through the Link River Dam (USFWS 2007a, b, c).

Substantial entrainment occurs at the river gates of the Link River Dam. Currently these gates have no structures to prevent drawing fish downstream. During the late summer of 2006, over 3,500 age-0 juvenile suckers were collected in the Link River just below the dam with intermittent sampling of a small fraction of the channel (Tyler 2007). The Committee on Endangered and Threatened Fishes in the Klamath River Basin of the National Research Council recommended screening to prevent downstream losses at Link River Dam (NRC 2004). Efforts to assess the impacts of this operation found significantly lower numbers of suckers were entrained during surface spill versus bottom spill experiments (Marine and Lappe 2009). Gutermuth et al. (2000) also documented tens of thousands of young suckers entrained at the PacifiCorp hydropower canals and turbines associated with the Link River Dam. Nonetheless, further research is required to better quantify the threats these structures pose to recovery.

Most suckers that pass through the gates at Link River Dam, or that survive passage through the hydroelectric facilities, are believed to be lost from the breeding population in Upper Klamath Lake. Most likely, these fish either die in poor summer water quality conditions in Keno Reservoir, or pass further downstream into reservoirs along the Klamath River, from which upstream passage is blocked. A fish ladder was constructed at Link River Dam in 2004 through which adult suckers have been documented moving upstream through Link River. As of 2008, only seven individuals had been documented as passing through the ladder (Korson et al. 2008). In 2010, at least 20 individuals were documented in the ladder during (Kyger and Wilkens 2011); additional untagged suckers likely also passed upstream through the ladder undetected.

In addition to major diversion point in the Keno Reservoir, the Lost River Diversion Channel, several hundred small, typically unscreened diversions in tributary streams and rivers and the lakes proper may also affect Lost River sucker and shortnose sucker. The influence of these diversions on sucker abundance and recovery is unknown.

Effects of Climate Change

Climate variability, such as fluctuations between wet and dry periods, is part of natural processes; however, climatic models and other information suggest that much of the recent trends is driven by anthropogenic pollutants, primarily CO_2 (Barnett et al. 2008). Since the 1950s, western North America generally has exhibited trends toward less snowfall, earlier snowmelt, and earlier peak spring runoff, much of which cannot be attributed to natural fluctuations (Hamlet et al. 2005, Stewart et al. 2005, Knowles et al. 2006). Furthermore, models indicate that these trends are likely to continue (Barnett et al. 2008). Perhaps the greatest foreseeable concern related to climate change is more intense summer heat waves that have the potential to create especially severe water-quality conditions in Upper Klamath Lake.

It is difficult to predict how such climatic changes will affect these species (Matthews and Marsh-Matthews 2003). Certainly these species have evolved under variable climates with relatively dry periods (Dicken and Dicken 1985, Negrini 2002); however, given the current lack of recruitment, lack of population connectivity even in wet years when it should be higher, degraded habitat including poor water quality, the overall low number of individuals, and other threats, we consider populations of these species to be highly vulnerable to negative impacts from climate change, either from distinct droughts or from extended periods of declining trends. If current trends continue into the future, important changes, which may threaten the continued existence of these species, e.g., further reductions in water quality, water-quality refuge availability, food-web alterations, and spawning run timing, are likely to occur (Dahm et al. 2003, Magoulick and Kobza 2003). Further reductions in water quality and reduced inflows as a result of climate change, if they occur, could be a serious threat to the survival and recovery of the Lost River and shortnose suckers, especially for populations in Upper Klamath Lake.

2.2.5 Conservation

2.2.5.1 Needs

The following eight conservation needs or recovery actions were identified in the draft recovery plan (USFWS 2011a).

1. Restore or enhance spawning and nursery habitat in Upper Klamath Lake and Clear Lake
2. Reduce the negative impacts of poor water quality
3. Clarify and reduce the effects of introduced species on all life stages by conducting and applying appropriate scientific investigations
4. Reduce the effect of loss of individuals by entrainment

5. Establish a refugial program to increase the numbers of populations
6. Increase juvenile survival and recruitment to spawning populations
7. Maintain and increase the number of recurring, successful spawning populations
8. Establish a Klamath Basin Sucker Recovery Implementation Program

2.2.5.2 Current Actions

The USFWS has worked with other agencies and stakeholders to recover the endangered suckers since 1994. Important cooperators include the USBR, National Resources Conservation Service (NRCS), The Nature Conservancy (TNC), ODEQ, National Fish and Wildlife Conservation Fund, Klamath Water Users, and Modoc Irrigation District. Approximately 300 on-the-ground restoration projects, including 90 wetland, 130 riparian, 45 in-stream, 25 upland, and 15 fish passage projects have been funded and implemented in the Upper Klamath River Basin that directly or indirectly benefit Lost River and shortnose suckers since 2009. Many of the projects included elements of more than one category of restoration project type. These projects have had significant cost share from multiple sources, including Federal programs such as Partners for Fish and Wildlife, Hatfield, Jobs in the Woods, and Oregon Resources Conservation Act programs, as well as state and private grants, and contributions from landowners.

Major sucker recovery oriented projects completed include: screening of the main irrigation diversion on the Klamath Project (A-Canal) in 2002 and the outlet to Clear Lake Dam in 2003, and screening of Modoc Irrigation District's diversion on the Williamson River in 2007, and the Geary Canal diversion in Howard Bay on Upper Klamath Lake in 2009; construction of a new fish ladder at Link River Dam in 2004; restoration of Williamson River Delta approximately 6,000 acres between 2000 and 2008, restoration of the lower 3 miles of the Wood River in 1999; and removal of Chiloquin Dam in 2008, a major impediment to upstream migration of listed suckers. Removal of Chiloquin Dam provides improved upstream passage to spawning areas.

It is too early to assess the efficacy of these projects to support recovery, and some project modification may be required for the full benefit of each program to be realized. This is particularly true with the project screening the A-Canal. Under present design, fish screened from entering the A-Canal are delivered via pipeline to Upper Klamath Lake at a point that is upstream of the Link River Dam. Investigations are needed to determine if these suckers remain in Upper Klamath Lake or pass downstream into Lake Ewauna and possibly are lost to the spawning population because of poor water quality conditions in the lake during the summer and apparently are having difficulties moving upstream past the dam.

The NRCS completed a large number of projects under the 2002 Farm Bill to improve water quality and water conservation. This has resulted in restoration of over 2,200 acres of wetland habitat and conservation of over 6,700 acre-feet of on-farm water. Conservation systems on over 70,000 ac have been planned, and practices have been applied to over 30,000 acres to manage soil, water, air, plants, and animals on private lands.

The Sprague River, the primary spawning habitat for suckers in Upper Klamath Lake and the largest tributary to the Williamson River, is listed as water quality impaired for nutrients, temperature, sediment, and DO under the section 303(d) of the Clean Water Act. In 2002,

ODEQ completed a TMDL process for the Sprague River and Upper Klamath Lake (Boyd et al. 2002). Water quality management plans were developed which provide targets and guidance on improvements water quality in the Sprague River and Upper Klamath Lake. Many wetland and riparian restoration projects are now designed to address TMDL issues.

In 2004, Oregon State University Agricultural Extension Service and the Klamath Watershed Council (now called the Klamath Watershed Partnership) began a series of monthly meetings with rural landowners in the Sprague River Valley to discuss watershed restoration goals. With the help of USFWS, NRCS and the Klamath Soil and Water Conservation District, this effort has effectively connected landowners with appropriate state and federal resource conservation programs. As a result, more than 70% of the private lands within the Sprague River Valley are partnering with local, state and federal agencies on land conservation and natural resource actions. The efforts of the Klamath Watershed Partnership have brought additional fiscal partners (e.g., Oregon Department of Agriculture, Klamath County, and Oregon Watershed Enhancement Board) into the conservation partnership. These partnership-forming actions will continue and build on themselves and enable more restoration to be done in the future.

The tributaries in the Wood River Valley supply a large portion of the inflow to Upper Klamath Lake. This valley also supports about half of the livestock in the Upper Basin and is responsible for approximately 30% of the external phosphorus loading to the lake. Because of this, it was identified by ODEQ as a priority water quality impaired area. The Klamath Basin Rangeland Trust has been active in the Wood River Valley encouraging landowners to adopt sustainable land and water management practices. Since 2002, the number of landowners who partner with this trust on conservation and restoration activities has increased to include approximately 50% of the agricultural lands in the watershed.

Klamath River Basin stakeholders signed the Klamath Basin Restoration Agreement in February 2010. The agreement is intended to result in effective and durable solutions which will restore native fishes throughout the Klamath Basin including listed suckers; establish reliable water and power supplies which sustain agricultural uses, communities, and National Wildlife Refuges; and contribute to the public welfare and the sustainability of all Klamath Basin communities. With authorization and appropriation of funds from federal and state governments and implementation of the Agreement substantial progress should be made toward the recovery of Lost River and shortnose suckers.

2.2.6 Status of the Species in the Action Area

2.2.6.1 Current Status and Distribution of the Listed Species in the State of Oregon including Population Size, Variability, and Trend

The current status and distribution of the Lost River sucker and shortnose sucker in the State of Oregon, including population size, variability, and trends, are described above in Section 2.2.4.2.

2.2.6.2 Status of Proposed Critical Habitat within the Action Area (Oregon):

The status of proposed critical habitat within the action area is described above in Section 2.2.2.1.

2.2.6.3 Threats Specific to Oregon and Discussion of Threats Relative to Water Quality

Threats to the Lost River and shortnose suckers specific to Oregon and discussion of threats relative to water quality are described above in section 2.2.4.3.

2.3 Oregon Chub

2.3.1 Species Description

2.3.1.1 Taxonomy

The Oregon chub was first described in scientific literature in 1908 (Snyder 1908), however it was not identified as a unique species until 1991 (Markle et al. 1991). The genus O*regonichthys* is endemic to the Umpqua and Willamette Rivers of western Oregon. In the past, the common name "Oregon chub" has been used to refer to all *Oregonichthys* from both of these drainages. However, the Umpqua River form of *Oregonichthys (O. kalawatseti)* was formally described by Markle et al. (1991), and is taxonomically distinct from *Oregonichthys* in the Willamette River which retains the earlier name of *O. crameri.* Use of the term "Oregon "chub" therefore refers only to *O. crameri.*

2.3.1.2 Species Description

The Oregon chub is a small minnow (Family: Cyprinidae) with an olive-colored back grading to silver on the sides and white on the belly. Scales are relatively large with fewer than forty occurring along the lateral line and scales near the back are outlined with dark pigment (Markle et al. 1991). While young of the year range in length from 7 to 32 millimeters (mm) (0.3 to 1.3 inches), adults can be up to 90 mm (3.5 inches) in length (Pearsons 1989). The species is distinguished from its closest relative, the Umpqua chub (*Oregonichthys kalawatseti*), by Oregon chub's longer caudal peduncle (the narrow part of a fish's body to which the tail is attached), mostly scaled breast, and more terminal mouth position (Markle et al. 1991).

2.3.1.3 Current legal status

The USFWS listed the Oregon chub as an endangered species in 1993, (USFWS 1993a) and a final recovery plan for the Oregon chub was published in 1998, (USFWS 1998c). The Oregon chub recovery plan established the following criteria for downlisting the species from endangered to threatened status:

(1) Establish and manage 10 populations of at least 500 adults each;
(2) All of these populations must exhibit a stable or increasing trend for 5 years; and
(3) At least three populations must be located in each of the three subbasins of the Willamette River identified in the plan (Mainstem Willamette River, Middle Fork, and Santiam River).

The recovery plan established the following criteria for delisting (i.e., removing the species from the List of Endangered and Threatened Wildlife):

(1) Establish and manage 20 populations of at least 500 adults each;
(2) All of these populations must exhibit a stable or increasing trend for 7 years;
(3) At least four populations must be located in each of the three subbasins (Mainstem Willamette River, Middle Fork, and Santiam River); and
(4) Management of these populations must be guaranteed in perpetuity.

In 2008, USFWS completed a 5-year review of the Oregon chub, concluding that downlisting criteria had been met and the species should be downlisted to threatened status (USFWS 2008b). The final rule designating critical habitat (USFWS 2010a, b) and the final rule to downlist Oregon chub were published in 2010 (USFWS 2010c).

2.3.2 Critical Habitat Description

2.3.2.1 Current legal status of critical habitat

Critical habitat was designated for Oregon chub in 2010 (USFWS 2010a, b). In the final rule, USFWS determined that 25 units totaling approximately 132 acres in Benton, Lane, Linn and Marion Counties met the proposed definition of critical habitat (Figure 2-5). Land ownership of the proposed critical habitat is as follows: 32.9 acres private, 30.11 acres state, 66.3 acres Federal and 2.8 acres other public lands.

2.3.2.2 The Primary Constituent Elements

The Primary Constituent Elements (PCEs) of Oregon chub critical habitat are the habitat components that provide the following:

1. Off-channel water bodies such as beaver ponds, oxbows, side-channels, stable backwater sloughs, low-gradient tributaries, and flooded marshes, including at least 500 continuous square meters (m^2) (0.12 acres) of aquatic surface area at depths between approximately 0.5 and 2.0 meters (m) (1.6 and 6.6 feet)
2. Aquatic vegetation covering a minimum of 250 m^2 (0.06 acres) (or between approximately 25% and 100%) of the total surface area of the habitat. This vegetation is primarily submergent for purposes of spawning, but also includes emergent and floating vegetation, and algae, which are important for cover throughout the year. Areas with sufficient vegetation are likely to also have the following characteristics.
    a. Gradient less than 2.5%;
    b. No or very low water velocity in late spring and summer;
    c. Silty, organic substrate; and
    d. Abundant minute organisms such as rotifers, copepods, cladocerans, and chironomid larvae.
3. Late spring and summer subsurface water temperatures between 15 and 25 °C (59 and 78 °F), with natural diurnal and seasonal variation.

4. No or negligible levels of nonnative aquatic predatory or competitive species. Negligible is defined for the purpose of this rule as a minimal level of nonnative species that will still allow the Oregon chub to continue to survive and recover.

2.3.3 Life History

2.3.3.1 Reproduction

Oregon chub reach maturity at about 2 years of age (Scheerer and McDonald 2003, p. 78) and in wild populations can live up to 9 years. Most individuals over 5 years old are females (Scheerer and McDonald 2003, p. 68). Oregon chub spawn from May through August; individuals are not known to spawn more than once a year. Spawning activity has only been observed at water temperatures exceeding 16 °C (61 °F). Males over 35 mm (1.4 inches) have been observed exhibiting spawning behavior (Pearsons 1989, p. 4). Egg masses have been found to contain 147-671 eggs (Pearsons 1989, p. 17).

2.3.3.2 Population Structure

Research suggests there may be risks associated with isolating populations that previously interacted with a larger network of interacting populations (Meffe and Vrijenhoek 1988; Burkey 1989). Isolating populations that would normally experience gene exchange can result in a general decline in local genetic diversity and a corresponding increase in divergence among populations within a drainage system (Meffe and Vrijenhoek 1988). Burkey (1989) 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 for local populations may be low and probability of extinction high (Burkey 1989, 1995). Multiple local populations distributed and interconnected throughout a watershed provide a mechanism for spreading risk from stochastic events (Hard 1995, Healy and Prince 1995, Rieman and Allendorf 2001, Rieman and McIntyre 1993, Spruell et al. 1999). Migration and occasional spawning between populations increases genetic variability and strengthens population variability (Rieman and McIntyre 1993).

Effective population sizes of 500 to 5000 have been recommended for the retention of evolutionary potential (Franklin and Frankham 1998, Lynch and Lande 1998). According to the 2011 annual monitoring report 34 out of 63 populations (54%) had less than 500 fish (Bangs et al. 2011), and therefore do not have sufficiently large effective population sizes to retain optimal evolutionary potential. Increased homozygosity of deleterious recessive alleles is thought to be the main mechanism by which inbreeding depression decreases the fitness of individuals within local populations (Allendorf and Ryman 2002). Hedrick and Kalinowski (2000) provide a review of studies demonstrating inbreeding depression in wild populations with very small effective population sizes.

The USFWS' Abernathy Fish Technology Center conducted a genetic analysis on Oregon chub that will be used to guide future restoration efforts. The report suggests that four genetically distinct groups of Oregon chub exist and these groups corresponded to the subbasins of the Willamette River. The report supports the current approach for chub reintroductions using a

donor population for a given reintroduction from within the same subbasin as the reintroduction site. The report authors examined genetic diversity within and among 20 natural and four introduced populations at 10 microsatellite loci and observed moderate levels of diversity with the exception of one population that displayed signs of a genetic bottleneck (Shetzline Pond) (Ardren et al. 2008).

2.3.3.3 Ecology / Habitat Characteristics

Oregon chub are found in slack water off-channel habitats such as beaver (*Castor Canadensis*) ponds, oxbows, side channels, backwater sloughs, low gradient tributaries, and flooded marshes. These habitats usually have little or no water flow, are dominated by silty and organic substrate, and contain considerable aquatic vegetation providing cover for hiding and spawning (Pearsons 1989, p. 27; Markle et al. 1991, p. 289; Scheerer and McDonald 2000, p. 1). The average depth of habitat used by Oregon chub is less than 1.8 m (6 ft), and summer water temperatures typically exceed 16 °C (61 °F).

Adult chub seek dense vegetation for cover and frequently travel in the mid-water column in beaver channels or along the margins of aquatic plant beds. Larval chub congregate in shallow near-shore areas in the upper layers of the water column, whereas juveniles venture farther from shore into deeper areas of the water column (Pearsons 1989, p. 16). In the winter months, Oregon chub can be found buried in the detritus or concealed in aquatic vegetation (Pearsons 1989, p. 16). Fish of similar size school and feed together. In the early spring, Oregon chub are most active in the warmer, shallow areas of the ponds.

Oregon chub are obligatory sight feeders (Davis and Miller 1967, p. 32). They feed throughout the day and stop feeding after dusk (Pearsons 1989, p. 23). Chub feed mostly on water column fauna. The diet of Oregon chub adults collected in a May sample consisted primarily of minute crustaceans including copepods, cladocerans, and chironomid larvae (Markle et al. 1991, p. 288). The diet of juvenile chub also consists of minute organisms such as rotifers and cladocerans (Pearsons 1989, p. 2).

Of the known Oregon chub populations, the sites with the highest diversity of native fish, amphibian, and reptile species have the largest populations of Oregon chub (Scheerer and McDonald 2000, p. 24). Beavers appear to be especially important in creating and maintaining habitats that support these diverse native species assemblages (Scheerer and Apke 1998, p. 45).

2.3.4 Status

2.3.4.1 Historical status and Distribution

Historically, Oregon chub were found throughout the Willamette River drainage from Oregon City to Oakridge. Records note collections from the Clackamas River, Molalla River, Mill Creek, Luckiamute River, North Santiam River, South Santiam River, Calapooia River, Long Tom River, Muddy Creek, McKenzie River, Coast Fork Willamette River, Middle Fork Willamette River drainages, and the mainstem Willamette River. Oregon chub were distributed throughout the Willamette River Valley (Snyder 1908) in off-channel habitats such as beaver ponds, oxbows, stable backwater sloughs, and flooded marshes. These habitats usually have little

or no water flow, have silty and organic substrate, and have an abundance of aquatic vegetation and cover for hiding and spawning. In the last 100 years, these habitats have largely disappeared because of changes in seasonal flows resulting from the construction of dams throughout the basin, channelization of the Willamette River and its tributaries, and agricultural practices. This loss of habitat combined with the introduction of nonnative species to the Willamette Valley resulted in a sharp decline in Oregon chub abundance.

At the time of listing in 1993, there were only eight known populations of Oregon chub. These locations represented a small fraction (estimated as 2% based on stream miles) of the species' formerly extensive distribution within the Willamette River drainage. Since the time of listing, several Oregon chub populations have been extirpated, a number of new populations have been discovered, and there have been a number of successful introductions (Bangs et al. 2011).

2.3.4.2 Current Status and distribution

In 2011, ODFW confirmed the continued existence of Oregon chub at 56 locations in the North and South Santiam River, McKenzie River, Middle Fork and Coast Fork Willamette River, and several tributaries to the mainstem Willamette River downstream of the Coast Fork/Middle Fork Willamette River confluence (Bangs et al. 2011). These included 40 naturally occurring and 16 introduced populations. Seven new populations of Oregon chub were discovered in connected sloughs in the Middle Fork Willamette, McKenzie, and North Santiam drainages (Bangs et al. 2011). Twenty-nine populations have an estimated abundance of over 500 fish; 15 of these populations also have exhibited a stable or increasing trend over the last 7 years (Bangs et al. 2011). The distribution of these sites is shown in Table 2-4.

Table 2-4. Distribution of Oregon chub populations meeting recovery criteria (Bangs et al. 2011).

Subbasin	No. of populations	No. of large populations (over 500)	No. of large populations with stable/increasing trend	Total chub in subbasin	Size range of populations
Santiam	17	8	3	15,785	1 to 3,030
Mainstem Willamette*	11	7	3	162,555	20 to 96,810
Middle Fork Willamette	26	14	9	52,869	1 to 23,000
Coast Fork Willamette	4	0	0	292	2 to 130

**includes McKenzie River subbasin*

Although certain populations of Oregon chub have remained relatively stable from year to year, substantial fluctuations in population abundance are normal. For instance, the largest known population at Ankeny National Wildlife Refuge had an estimated abundance of 21,790 chub in 2010 and increased to 96,810 chub in 2011.

Consulted-on Effects

Other Federal actions affecting Oregon chub, or designated critical habitat, that required formal section 7 consultation with our office include: the USFWS issuance of section 10(a)(1)(A) recovery permits; statewide transportation improvement projects; BPA transmission line replacement; research, restoration, and species enhancement; aquatic restoration on U.S. Forest Service, USBLM, and U.S. Army Corps of Engineers lands; and the continued operation and maintenance of the U.S. Army Corps of Engineers Willamette River Basin Flood Control Project. None of the completed section 7 consultations reached a jeopardy finding for Oregon chub, nor a finding of adverse modification of designated critical habitat.

2.3.4.3 Threats

Historically, the mainstem of the Willamette River was a braided channel with many side channels, meanders, oxbows, and overflow ponds that provided habitat for the chub. Periodic flooding of the river created new habitat and transported the chub into new areas to create new populations. The construction of flood control projects and dams, however, changed the Willamette River significantly and prevented the formation of chub habitat and the natural dispersal of the species. Other factors responsible for the decline of the chub include habitat alteration; the proliferation of nonnative fishes; desiccation of habitats; sedimentation resulting from timber harvesting in the watershed; and possibly the demographic risks that result from a fragmented distribution of small, isolated populations.

Most populations of chub are currently isolated from other chub populations due to the reduced frequency and magnitude of flood events and the presence of migration barriers such as impassible culverts and permanent, high beaver dams. Managing Oregon chub in isolation may have genetic consequences (DeHaan et al. 2010, p. 20). Burkey (1989) 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 for local populations may be low and probability of extinction high (Burkey 1989, 1995). A genetic analysis completed in 2010 shows that while gene flow is limited among Oregon chub populations, most of the populations in isolated ponds are currently genetically viable and have remained so over several years (1997 to 2005) (DeHaan et al. 2010). However, the data were collected over only a 3 to 4-generation time period and it may be too soon to see evidence of negative genetic effects. Additionally, genetic data from historic populations (pre-Willamette project) is not available to compare with these results.

Excessive siltation from ground disturbing activities in the watershed, such as logging upstream of Oregon chub habitat, can degrade or destroy Oregon chub habitat. The threat of siltation due to logging in the watershed has been identified at five sites: Green Island North Channel, Finley Gray Creek Swamp, East Fork Minnow Creek Pond, Buckhead Creek, and Wicopee Pond (Paul Scheerer, ODFW, Corvallis, Oregon, pers. comm., 2008). In the 1990s, a large part of the Minnow Creek Watershed in the Middle Fork Willamette River subbasin was logged; flood events in the watershed in 1996, 1997, and 1998 caused accelerated sedimentation in the beaver pond at East Fork Minnow Creek Pond, and over half of the open water wetted area of the Oregon chub habitat there was lost as sediment filled the pond (P. Scheerer, pers. comm., 2009).

The Oregon chub population in East Fork Minnow Creek Pond declined dramatically following these floods and the resulting sedimentation (P. Scheerer, pers. comm., 2009).

2.3.4.4 Climate Change

Climate change presents substantial uncertainty regarding the future environmental conditions in the Willamette Basin and is expected to place an added stress on the species and its habitats. The Intergovernmental Panel on Climate Change (IPCC) has concluded that recent warming is already strongly affecting aquatic biological systems; this is evident in increased runoff and earlier spring peak discharge in many glacier- and snow-fed rivers (IPCC 2007, p. 8). Projections for climate change in North America include decreased snowpack, more winter flooding, and reduced summer flows (IPCC 2007, p. 14). Projections for climate change in the Willamette Valley in the next century include higher air temperatures that will lead to lower soil moisture and increased evaporation from streams and lakes (CLI and NCCSP 2009, p. 9). While there is high uncertainty in the total precipitation projections for the region, effective precipitation (precipitation that contributes to runoff) may be reduced significantly even if there is no decline in total precipitation (CLI and NCCSP 2009, p. 9).

Although climate change is almost certain to affect aquatic habitats in the Willamette Basin (CLI and NCCSP 2009, p. 1), there is great uncertainty about the specific effects of climate change on the Oregon chub. The USFWS has developed a strategic plan to address the threat of climate change to vulnerable species and ecosystems; goals of this plan include maintaining ecosystem integrity by protecting and restoring key ecological processes such as nutrient cycling, natural disturbance cycles, and predator-prey relationships (USFWS 2009b; p. 21). The Oregon chub recovery program will strive to achieve these goals by working to establish conditions that allow populations of Oregon chub to be resilient to changing environmental conditions and to persist as viable populations into the future. Our recovery program for the species focuses on maintaining large populations distributed across the species' entire historical range in a variety of ecological settings (e.g., across a range of elevations). This approach is consistent with the general principles of conservation biology. In their review of minimum population viability literature, Traill et al. (2009, p. 3) found that maintenance of large populations across a range of ecological settings increases the likelihood of species persistence under the pressures of environmental variation and facilitates the retention of important adaptive traits through the maintenance of genetic diversity. Maintaining multiple populations across a range of ecological settings, as described in the recovery plan, will also increase the likelihood that at least some of these populations persist under the stresses of a changing climate.

2.3.5 Conservation

2.3.5.1 Needs

The recovery strategy has focused on improving Oregon chub habitats in isolation due to the loss and fragmentation of suitable habitats and the threats posed by nonnative fishes. Increasing the abundance and distribution of Oregon chub in isolation has proven to be effective at halting the decline of Oregon chub populations and in meeting the recovery criteria for downlisting. However, managing Oregon chub in isolation does not allow genetic transfer between populations and may have future genetic consequences. Floodplain connectivity at many sites

near mainstem rivers is not well understood. Recent hydrological data were collected by ODFW at sites that are influenced by the operation of dams in the Willamette Basin to determine the point of connectivity at each site and the duration of floodplain connection. They found that several sites connect to the river more frequently or for longer periods than previously known. Although, it is not known whether Oregon chub are moving between these habitats during high water events, the study shows that the mechanism for dispersal does exist. Genetic studies are needed to determine whether the populations in these periodically connected sites are operating as a metapopulation.

Additionally, some populations are persisting even in the presence of nonnatives, although these populations are less abundant than populations without nonnatives present. Understanding what habitat characteristics allow Oregon chub to coexist with nonnatives in these connected habitats will be useful in determining whether chub can be reintroduced in connected habitats.

2.3.5.2 Current Actions

The Oregon Chub Working Group was formed in 1991 and has been proactive in conserving and restoring habitat for the Oregon chub and raising public awareness of the species since before the Federal listing in 1993 (USFWS 2008b, p. 11).

In 1992, an interagency Conservation Agreement for the Oregon Chub in the Willamette Valley, Oregon was completed and signed by USFWS, U.S. Forest Service, USBLM, ODFW, and Oregon Parks and Recreation Department (USFWS 1998c). The purpose of the coordinated plan was to facilitate Oregon chub protection and recovery and to serve as a guide for all agencies to follow as they conduct their missions.
In February 1997, a Memorandum of Understanding was signed by USFWS and the City of Salem to protect and enhance the population of Oregon chub located in the drinking water treatment facility at Geren Island in the North Santiam River until a formal Habitat Conservation Plan is developed.

In 1996, a no-spray agreement with the Oregon Department of Transportation was formalized to protect Oregon chub sites located in the Middle Fork Willamette River drainage adjacent to Highway 58 in Lane County. The agreement prohibits spraying of herbicides in the vicinity of Oregon chub sites and limits vegetation control to mechanical methods if necessary.
The USFWS has completed three individual safe harbor agreements (SHA) for Oregon chub. To streamline the process for landowners to enter into a SHA in the future, a programmatic SHA was prepared by USFWS and ODFW in 2009 (USFWS 2009b). Under a SHA, property owners who undertake management activities that attract listed species onto their property or that increase the numbers or distribution of listed species already present on their property will not incur future property-use restrictions. SHAs provide assurances to the property owner that allow alterations or modifications to enrolled property, even if such action results in the incidental take of the covered listed species or, in the future, returns the species back to an originally agreed-upon baseline condition.

In 2008, USFWS signed a biological opinion on the continued operation and maintenance of the Willamette River Basin Project and effects to Oregon chub, bull trout, and bull trout critical

habitat (USFWS 2008c). To address specific terms and conditions outlined in the opinion, ODFW initiated a study in 2009 to determine the current status of chub populations, fish assemblages, and habitat conditions in habitats potentially affected by the operation of Willamette River Basin Project dams. They are assessing relationships between pond bathymetry, pond elevations, pond temperatures, river flow levels, site connectivity, and fish assemblages. Data from this study will be used to provide the U.S. Army Corps of Engineers with flow management recommendations that will contribute to Oregon chub recovery and minimize incidental take of chub.

The improvement in status of Oregon chub is due largely to the implementation of actions identified in the Oregon chub recovery plan. This includes habitat restoration, the discovery of many new populations as a result of ODFW's surveys of the basin, and the establishment of additional populations via successful reintroductions within the species' historical range. Introduced populations have been established in suitable habitats with low connectivity to other aquatic habitats to reduce the risk of invasion by nonnative fishes.

2.3.6 Status of the Species in the Action Area

2.3.6.1 Current status and distribution of the listed species in the State of Oregon including population size, variability, and trend

The Oregon chub is endemic to the Willamette River Valley of western Oregon; therefore, the status of the species throughout its range, as discussed above, constitutes the status of the species in the Action Area.

2.3.6.2 Current critical habitat designation within the State of Oregon

The Oregon chub is endemic to the Willamette River Valley of western Oregon; therefore, the status of the critical habitat, as discussed above, constitutes the status of the species in the Action Area. See Figure 2-5.

2.3.6.3 Threats specific to Oregon and discussion of threats relative to water quality (toxicity and temperature)

The analysis of threats in the final rule to list the Oregon chub as an endangered species and the recovery plan for the species discussed numerous potential threats to water quality in Oregon chub habitats. Many Oregon chub populations occur near rail, highway, and power transmission corridors; near agricultural fields; and within public park and campground facilities; prompting concern that these populations could be threatened by chemical spills and runoff (USFWS 1998c, p. 14; P. Scheerer, pers. comm., 2008). In the 18 years since listing, a few of these concerns have been realized, and are discussed in the paragraphs below.

Water quality investigations at sites in the Middle Fork and Mainstem Willamette River subbasins have found some adverse effects to Oregon chub habitats. Nutrient enrichment may have caused the crash of the Oregon chub population at Oakridge Slough on the Middle Fork. The slough is downstream from the Oakridge Sewage Treatment Plant and has a thick layer of decaying organic matter, which may limit the amount of useable habitat available to the chub (Buck 2003, p. 2). In the late 1990s, the Oregon chub population in Oakridge Slough peaked at

nearly 500 individuals; since then, the population is now extirpated (Bangs et al. 2011). Increased nitrogen and phosphorus concentrations have been detected in the slough; while the nutrient concentrations are not believed to be directly harmful to Oregon chub, the elevated nutrient levels may have resulted in eutrophication of the pond, with associated anoxic conditions unsuitable for chub, or increased plant and algal growth that severely reduced habitat availability (Buck 2003, p. 12).

Studies at William L. Finley National Wildlife Refuge have found evidence of elevated levels of nutrients and pesticides in Oregon chub habitats (Materna and Buck 2007, p. 67). Water samples were collected in 1998 from Gray Creek Swamp, which is home to a large population of Oregon chub. Analyses detected three herbicides, although all were below criteria levels recommended for protection of aquatic life; however, one form of nitrogen (total Kjeldahl N) exceeded USEPA criteria levels recommended for protection of aquatic life in the WillametteValley (Materna and Buck 2007, p. 67). The source of the contamination is likely agricultural runoff from farm fields adjacent to the Refuge (Materna and Buck 2007, p. 68).

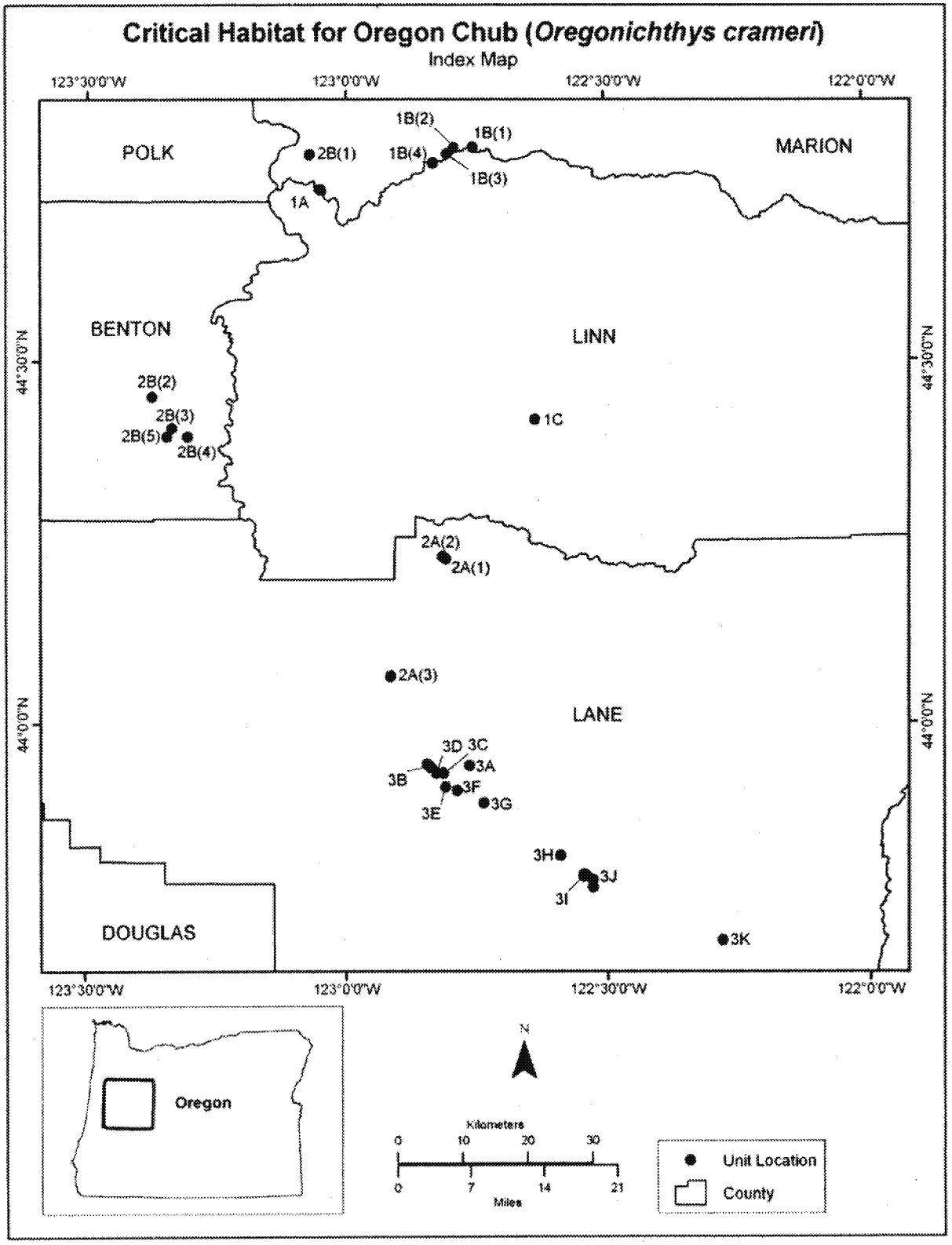


Figure 2-5. Locations of Oregon chub critical habitat.

2.4 Vernal Pool Fairy Shrimp

2.4.1 Species Description

2.4.1.1 Taxonomy

The vernal pool fairy shrimp (*Branchinecta lynchi*) (fairy shrimp) is a member of the aquatic crustacean order Anostraca, in the Branchinectidae family. This family includes only one genus with approximately 50 species of fairy shrimp (Rogers 2006, p. 35). It was first described by Eng, Belk, and Eriksen (Eng et al. 1990). The type specimen was collected in 1982 at Souza Ranch, Contra Costa County, California. Although not yet described, the vernal pool fairy shrimp had been collected between 1874 and 1941, when it was identified as the Colorado fairy shrimp by Linder (Eng et al. 1990).

2.4.1.2 Species Description

The shrimp is a thin "shrimp like" invertebrate that ranges in size from 0.4 to 1.0 inch (Eng et al. 1990). Fairy shrimp have delicate elongate bodies which are encased in a translucent shell, which can be whitish or dusky colored. They have large stalked compound eyes, no carapace, and 11 pairs of orange-colored swimming legs. Fairy shrimp swim or glide gracefully upside down by means of complex beating movements of their legs that pass in a wave-like anterior to posterior direction.

Fairy shrimp have two pairs of antennae. The second pair of antennae in the adult females is cylindrical and elongate, but the males' antennae are greatly enlarged and specialized for clasping the females during copulation. The females carry the fertilized eggs in a discernible oval or elongate ventral brood sac.

2.4.1.3 Current Legal Status

The vernal pool fairy shrimp was listed as threatened on September 19, 1994. Along with vernal pool fairy shrimp, three other fairy shrimp were listed in same listing package (USFWS 1994a). The USFWS listed the fairy shrimp as a threatened species primarily due to the present or threatened destruction, modification, or curtailment of their habitat or range. They determined that "the habitat of these animals is imperiled by a variety of human-caused activities, primarily urban development, water supply/flood control activities, and conversion of land to agricultural use. Habitat loss occurs from direct destruction and modification of pools due to filling, grading, discing, leveling, and other activities, as well as modification of surrounding uplands that alters vernal pool watersheds."

A final recovery plan for vernal pool ecosystems of California and Southern Oregon, including the three other fairy shrimp and 11 vernal pool plants, was published in 2005 (USFWS 2005c). The recovery plan sets out specific goals, objectives and tasks to direct recovery efforts for fairy shrimp. The Jackson County vernal pool area, known as the Klamath Mountains Vernal Pool Region, is composed of three core areas (Agate Desert, White City and Table Rocks) and is considered a Zone 2 area (USFWS 2005c). The Recovery Plan's strategy outlines a 6-step process which is anticipated to guide the species to reach recovery. The six steps include habitat

protection; adaptive habitat management, restoration, creation and monitoring; status surveys; research; and participation and outreach (USFWS 2005c, pp. III-88 – III-93).

The USFWS is preparing the final recovery plan for the listed species of the Rogue Valley Vernal Pool and Illinois Valley Wet Meadow Ecosystems (USFWS 2006). This draft recovery plan sets out specific goals, objectives and tasks to direct recovery efforts for Cook's desert parsley and large-flowered woolly meadowfoam. The recovery plan will out specific goals, objectives and tasks to direct recovery efforts for vernal pool fairy shrimp in Oregon.

2.4.2 Critical Habitat Description

2.4.2.1 Current Legal Status

A final rule designating critical habitat for 4 vernal pool crustaceans and 11 vernal pool plant species, including the vernal pool fairy shrimp, was published in the on August 6, 2003 (USFWS 2003) and updated on February 10, 2006 (USFWS 2006). Total area of critical habitat in these two states is 597,821 acres and 7,574 acres in Oregon.

Fairy shrimp is the only species addressed in USFWS's 2003 critical habitat designation regarding vernal pool species occurring in Oregon. Four critical habitat units in Oregon are designated as essential to the conservation of fairy shrimp, and there are 29 units in California. The Oregon units are comprised of 7,574 acres in Jackson County. These units occur approximately 125 miles north of the nearest unit designated for this species in California. The USFWS identified critical habitat areas essential to the conservation of fairy shrimp to reflect the species geographic distribution and varying habitat types and species associations across its range. Maintaining fairy shrimp across their full geographic distribution would make the species less susceptible to environmental variation or negative impacts associated with human disturbances or natural catastrophic events across the species entire range at any one time (Helm 1998).

Variation in environmental conditions such as precipitation amount, precipitation timing, and temperature, influence vernal pool species including hatching and reproduction of fairy shrimp from year to year (Eriksen and Belk 1999, Helm 1998).

2.4.2.2 The primary constituent elements

The PCEs of critical habitat for fairy shrimp are the habitat components that provide:

- Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools described below, providing for dispersal and promoting hydroperiods of adequate length in the pools;
- Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum of 18 days, in all but the driest years; thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal

basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands;

- Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding; and
- Structure within the pools described above, consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter.

2.4.2.3 Identification of the potential effects of toxics or water temperature levels from alteration of the PCEs

Water quality and temperature are not directly addressed in the PCEs; however the introduction of toxics into and the alteration of water temperature of a vernal pool system could potentially disrupt two PCE's: sources of food and structure within pools. Nearly all fairy shrimp feed on algae, bacteria, protozoa, rotifers, and bits of detritus (Pennak 1989). These food sources and structure could be sensitive to the addition of herbicides into the system. The alteration of temperatures could potentially decrease food source and organic material availability in a vernal pool system, although, the degree of effect of these alterations has not been quantified in a vernal pool system (USFWS 2007d).

In Oregon, the habitat structure of the fairy shrimp was observed becoming degraded due to deposition of timber industry wood waste (log deck debris), planting of intermediate wheat grass, accumulation of dead organic matter (including thatch from dead, dry wetland plants), and deposition of biosolids (treated sewage) or other pollutants (ONHP 1999, USFWS 2006). Plant decomposition and algal growth can lead to the depletion of dissolved oxygen in natural waters, and can increase sediment toxicity to food sources (Lee 2007). Depletion of dissolved oxygen can cause complete mortality of other gill-breathing aquatic organisms (Horne and Goldman 1994). These factors could contribute to degradation of the fairy shrimp PCEs, although the degree of risk is unknown.

2.4.3 Life History

2.4.3.1 Reproduction

Male fairy shrimp use their antennae to clasp the females during copulation. The females will either drop fertilized eggs to the pool bottom or eggs will remain in the brood sac until the female dies and sinks. The "resting" or "summer" eggs, also called diapause eggs, are not actually eggs, but cysts capable of withstanding heat, cold, and prolonged desiccation. After the eggs are fertilized, the embryo undergoes additional development to the nauplius or metanauplius stage before entering diapause.

When the pools refill in the same or subsequent seasons some, but not all, of the cysts may hatch. Branchiopods respond to inherent variability in climatic conditions by producing eggs with different diapause characteristics in each clutch. Some hatch after a single wet and dry cycle; while others may go through several wet/dry cycles before they hatch. The cyst bank in the soil may also be comprised of individuals from several years of breeding. The species typically produces only one clutch of eggs each year and then dies (Belk 1977).

2.4.3.2 Population structure

This species has an ephemeral life history which can be made up of individuals that are of similar age-classes. Data suggests that gene flow between pools is common and increases local population sizes by uniting pools within vernal pool complexes (Fugate 1998). Branchinecta species in North America have populations that appear to exchange approximately 3-100 migrants per generation (most species exchange above 20) at 0.62 miles separation, and 0.1-0.2 per generation at 621-1242 miles separation (Fugate 1998).

In general, the fairy shrimp has a sporadic distribution within the vernal pool complexes, with most pools being uninhabited by the species (USFWS 1994a). In the Rogue River Valley, pool complexes have been documented with fairy shrimp occupation between 10 to 40%.

Simovich and Fugate (1992) reported that the fairy shrimp typically is found at low population densities. Only rarely does the fairy shrimp co-occur with other fairy shrimp species, but where it does, the fairy shrimp is never the numerically dominant one (Eng et al. 1990).

2.4.3.3 Ecology / Habitat Characteristics

Fairy shrimp occurs only in cool-water pools. Fairy shrimp first hatch at the bottom of the vernal pool when water temperatures reach 10 °C, typically during cold-weather winter storms (Helm 1998, Eriksen and Belk 1999). Under optimal conditions they undergo a series of molts before reaching maturity in about 2½ weeks, when they are approximately 0.2 inches - 0.8 inches in length. They have been reported to live anywhere from 2 to 4½ months, depending on many environmental factors (Eriksen and Belk 1999). The time to maturity and reproduction is temperature dependent, varying between 18 days and 147 days, with a mean of 39.7 days (Helm 1998). When the temporary pools dry, offspring persist in suspended development as desiccation-resistant embryos (commonly called cysts) in the pool substrate until the return of winter rains and appropriate temperatures allow some of the cysts to hatch (Eriksen and Belk 1999). Immature and adult shrimp are known to die off when water temperatures rise to approximately 24 °C (Helm 1998). Fairy shrimp often disappear early in the season due to warmer water conditions long before the vernal pools dry up.

Fairy shrimp are endemic to vernal pools and live within an ephemeral freshwater environment which supports plant species that tolerate the changing abiotic and biotic conditions within and surrounding the pools (Robbins and Vollmar 2002). The fairy shrimp are ecologically dependent on seasonal fluctuations in their habitat, such as absence or presence of water during specific times of the year, availability of ephemeral plant and animal matter, duration of inundation, and other environmental factors that include specific salinity, conductivity, dissolved solids, and pH levels. They are sporadic in their distribution, often inhabiting only one or a few pools in

otherwise more widespread vernal pool complexes. Although the species has been collected from large vernal pools it tends to occur in smaller, shallower pools, frequently measuring less than 0.05 acres and with an average depth of 2 inches (Helm 1998). Genetic characteristics, as well as ecological conditions, indicate that populations are defined by pool complexes rather than by individual vernal pools.

Nearly all fairy shrimp feed on algae, bacteria, protozoa, rotifers, and bits of detritus (Pennak 1989). Fairy shrimp play an important role in the community ecology of ephemeral water bodies (Loring et al. 1988) and are fed upon by migratory waterfowl (Krapu 1974), many species of insects, amphibians, crustaceans and other vertebrates. This species is an important link in the food web, particularly as a supply of energy for migratory birds.

Fairy shrimp populations occur in ephemeral freshwater habitats, such as vernal pools and swales. None are known to occur in running or marine waters or other permanent bodies of water. Vernal pools occur in an environment influenced by geology, geomorphology, soils, topography, hydrology, and local climate, which in turn influences vegetation and wildlife use (Silveira 1998).

The distribution of vernal pools is highly discontinuous and some of the aquatic invertebrates that are found in this habitat occur only in specific geographic areas. Due to local topography and geology, the pools are usually clustered into pool complexes (Holland and Jain 1988). Pools within a complex typically are separated by distances on the order of meters and may form dense, interconnected mosaics of small pools or a sparser scattering of larger pools.

Although the vernal pool fairy shrimp has a relatively wide range, the majority of known populations inhabit vernal pools with clear to tea-colored water, most commonly in grass or mud bottomed swales, or basalt flow depression pools in unplowed grasslands, but one population occurs in sandstone rock outcrops and another population in alkaline vernal pools (Collie and Lathrop 1976). They are ecologically dependent on seasonal fluctuations in their habitat, such as absence or presence of water during specific times of the year, duration of inundation, and other environmental factors that include specific salinity, conductivity, dissolved solids, and pH levels.

2.4.4 Status

2.4.4.1 Historical Status and Distribution

In Oregon, fairy shrimp are thought to have occupied all vernal pool systems that occurred within in Jackson County's Agate Desert of the Rogue River Valley prior to conversion to agricultural lands, military bases, water treatment facilities, mining operations, and industrial, commercial, and residential developments (ONHP 1997). The former vernal pool systems roughly occurred where soils are mapped as the Agate-Winlo soils series in the vicinity of northern Medford, White City, and along the Rogue River between White City and Shady Cove (Petersen 2006).

At the time of its listing, the vernal pool fairy shrimp was known to occur only in California, extending from Tulare County in the south to Shasta County in the north. In 1998, fairy shrimp

were discovered in vernal pools in Jackson County, Oregon, in an area north of Medford known as the Agate Desert. Prior to this discovery, the most northerly known location for the species was south of Mount Shasta, California, some 80 miles south of the Agate Desert (Eng et al. 1990).

2.4.4.2 Current Status and Distribution

The species occurs in disjunctive fragmented habitats distributed across the Central Valley of California from Shasta County to Tulare County and the central and southern Coast Ranges from northern Solano County to Ventura County, California (Eng et al. 1990). Additional disjunctive populations have been identified in southern California and in Oregon. In southern California the distribution limited with populations occurring in three areas in Riverside County. In Oregon, the species' distribution is equally limited to the vicinity of an approximately 32 square mi area within the Agate Desert in Jackson County.

Consulted-on Effects

Other Federal actions affecting vernal pool, or designated critical habitat, that required formal section 7 consultation with our office include: the USFWS issuance of section 10(a)(1)(A) recovery permits; issuance of Army Corps of Engineers section 404 permits; statewide and local transportation improvement projects; research; restoration on USBLM and USBR lands; grazing related projects; gravel mining; various housing and commercial development projects; and utility improvement projects. None of the completed section 7 consultations reached a jeopardy finding for vernal pool fairy shrimp, nor a finding of adverse modification of designated critical habitat.

2.4.4.3 Threats

The greatest threat for vernal pool fairy shrimp survival is the elimination/pollution of vernal pools. Like the other species of vernal pool branchiopods, the number of fairy shrimp populations has declined primarily because of destruction or degradation of vernal pools through development of urban, suburban, and agricultural projects. In addition to direct habitat loss, vernal pool fairy shrimp populations have declined from of a variety of activities that degrade existing vernal pools by altering pool hydrology (water regime). Vernal pool hydrology can be altered by a variety of activities, including the construction of roads, trails, ditches, or canals that can block the flow of water into, or drain water away from, the vernal pool complex (USFWS 2005c).

Other threats include fragmentation and habitat degradation. In most vernal pool regions, many vernal pool fairy shrimp occurrences are located on remnant vernal pool habitat adjacent to residential, industrial or commercial development, to infrastructure, or to deep-ripped vineyards or orchards (CNDDB 2007). As discussed above, development can result in the loss of hydrological connections that sustain the remnant vernal pools, resulting in premature drying of pools before the life cycle of the shrimp is completed (USFWS 2006). Alternatively, in remnant habitat parcels, vernal pools can be subject to increased periods of inundation due to nearby irrigation, outfall discharge, or runoff from development (CNDDB 2007). Increased periods of inundation can reduce habitat suitability for the vernal pool fairy shrimp as they are commonly found in only the smaller shorter-lived pools (Eriksen and Belk 1999). Small unprotected and

protected parcels may be particularly susceptible to degradation by nonnative species, particularly when the parcels are located in a matrix of development and are associated with chronically-disturbed transportation corridors (Zedler and Black 2004). The vernal pools on the Agate Desert Preserve in Oregon are threatened by the indirect effects of adjacent land use, including alteration of hydrology (USFWS 2006). In addition, development of off-site banks may not meet the functional equivalence of the original wetlands ecosystems (Wacker and Kelly 2004).

Other threats include pesticides. Several commonly used pesticide formulations, including the herbicides ingredients diuron and glyphosate, have been found to be toxic to *Thamnocephalus platyurus*, a fairy shrimp easily hatched in lab beakers and commonly used to test water quality and toxicity (Brausch et al. 2006). Pesticide effects can include the effects of the surfactants formulated with the active ingredient. For example, polyethoxylated tallowamine (POEA) is a surfactant that is commonly used in herbicide formulations to increase the efficacy of active ingredients. POEA use has increased recently with the advent of "Roundup-Ready" crops. Brausch and Smith (2007) used *T. platyurus* to assess the acute toxicity of POEA and found it to be extremely toxic at low concentrations.

The most commonly used herbicide ingredient on the western U.S. region is glyphosate and its use in other herbicide brand names. It is often used as a post-emergent, which can be applied at five different times throughout the growing season. It is used in high concentrations, with up to 4.5 kg active ingredient/hectare as the recommended application rate. However; the surfactant ingredient commonly used in glyphosate formulations, POEA, is exceedingly toxic to *Thamnocephalus platyurus* (Brausch and Smith 2007, Relyea 2005), and likely to fairy shrimp as well.

2.4.4.4 Climate Change

Climate change has the potential to be a threat to the fairy shrimp through changes in vernal pool inundation patterns and temperature regimes. Climate scientists predict that climate will become warmer within the 21st century (Field et al.1999), although there is still uncertainty about particular regional effects of warming. Geographical responses would include latitudinal and elevation shifts in species ranges. The ability of vernal pool fairy shrimp to survive is likely to depend in part on their ability to disperse to pools where conditions are suitable (Bohanak and Jenkins 2003; Bonte et al. 2004). Loss and fragmentation of vernal pool habitat is thought to decrease the probability of successful dispersal. Therefore, species range shifts that may be required by climate change may be more difficult due to factors such as the loss of potential habitat from development, occupation of potential habitat by nonnative species, and lack of appropriate soil substrates (Field et al. 1999). Remnant suitable habitats, even within conservation banks, may be too far apart to allow dispersal or natural re-colonization after a disturbance (Field et al. 1999). Existing fairy shrimp preserves may not provide the full range of conditions needed to sustain the species during variable climatic conditions (Pyke 2004, 2005b).

Vernal pools are particularly sensitive to slight increases in evaporation or reductions in rainfall due to their shallowness and seasonality (Field et al. 1999). It is expected that winters in Oregon may become warmer and wetter, while El Niño frequency and intensity may increase. Even

modest increases in average temperature could result in more runoff in winter with less runoff in spring and summer, more winter flooding, and drier summer soils, thereby altering the seasonality and duration of vernal pool hydration (Cayan et al. 2005, Field et al. 1999). Vernal pool crustaceans have developed life history strategies to survive drought periods. They are, however, adapted to complete their life cycles within limited temperature ranges and require a minimum length of inundation to reach maturity and reproduce. Although fairy shrimp mature relatively fast, they are able to produce more eggs when water conditions are suitable for a longer period of time (see Eriksen and Belk 1999, Helm 1998). Climate change is expected to lead to increased variability in precipitation (McLaughlin et al. 2002), and increased loss of soil moisture due to evaporation and transpiration of water from plants (Field et al. 1999); this may exacerbate effects due to drought. Drought-mediated decreases in water depth and inundation period could increase the frequency at which pools dry before shrimp have completed their life cycle, or cause pool temperatures to more often exceed temperatures suitable for hatching and persistence of the species.

The species active in vernal pools is expected to change over time (Field et al. 1999, Pyke 2005b). Presence of fairy shrimp appears to be associated with precipitation patterns and other climate factors, including temperature and aridity (Eriksen and Belk 1999, Graham 2003, Jones and Stokes 2006). Although the specific effects of climate change on the vernal pool fairy shrimp are unknown, the effect of warming temperatures on winter storm events and pool conditions have the potential to adversely affect this species. The shrimp may disappear from some areas to be replaced by more tolerant species. Changes in precipitation could alter marginal pools towards more or less favorable periods of inundation (Pyke 2005a). Climate change may also result in the alteration of vernal pool habitats through changes to water chemistry of pools due to increases in carbon dioxide from the atmosphere (Field et al. 1999) and decreased precipitation (Graham 2003), thereby affecting suitability of pools for specific species (Eriksen and Belk 1999, Graham 2003). Inter-annual population fluctuations might be amplified by changes in precipitation and could lead to rapid extinctions of individual populations, even where populations are already known to fluctuate widely (McLaughlin et al. 2002). It is also possible that fairy shrimp species could be buffered from extinction due to the presence of cyst banks in the soil, although the extent of this protection is unknown (Bohanak and Jenkins 2003).

2.4.5 Conservation

2.4.5.1 Needs

The USFWS published a final recovery plan for vernal pool ecosystems of California and Southern Oregon in 2005 (USFWS 2005c). The recovery plan sets out specific goals, objectives and tasks to direct recovery efforts for fairy shrimp. The Jackson County vernal pool area, known as the Klamath Mountains Vernal Pool Region, is composed of three core areas (Agate Desert, White City and Table Rocks) and is considered a zone 2 area (USFWS 2005c). For species such as fairy shrimp, which are widely distributed, protection of zone 2 core areas will significantly contribute to recovery of fairy shrimp. Recovery recommendations within zones 2 and 3 are considered more flexible than Zone 1 and recovery criteria specific to Zone 2 core areas may be modified on a case-by-case basis as information becomes available.

The USFWS is preparing the final recovery plan for the listed species of the Rogue Valley Vernal Pool and Illinois Valley Wet Meadow Ecosystems (USFWS 2006). In addition to fairy shrimp, this recovery plan will set out specific goals, objectives and tasks to direct recovery efforts for two federally listed plants, Cook's lomatium (*Lomatium cookii*) and large-flowered woolly meadowfoam (*Limnanthes floccosa* ssp. *grandiflora*). In Oregon, the overall recovery goal for fairy shrimp is to protect 85% of the 5,000 acres of remaining suitable vernal pool habitat (4,250 acres). The Klamath Mountain vernal pool region should be a significant contributor to the recovery of fairy shrimp in its range. As such USFWS has developed the following guidance for vernal pool conservation in Oregon:

- A minimum of 4,300 acres of vernal pool habitat should be protected through ownership, management; or conservation easement or agreement.
    - A minimum of 1,500 acres should be protected in the form of conservation banks.
        - Conservation banks should be a minimum of 150 acres. Banks can be composed of discontinuous parcels, with a core parcel of a minimum of 100 acres. Banks smaller than 100 acres can be approved on a case-by-case basis.
        - Conservations banks or bank parcels shall be located within the historical range of the agate-winlo soil type. Existing vernal pool habitat can be conserved, restored and enhanced. Re-creation of vernal pool habitat (re-creation of surface topography where the underlaying duripan layer is intact), may be considered on a case-by-case basis.
        - Creation of vernal pools (making vernal pool habitat in upland area) is considered a possible option at this time. Consideration of creating vernal pools will be on a case-by-case basis.
    - A minimum of 1,100 acres should be protected in the form of conservation easements or agreements.
    - This conservation effort should be distributed throughout the Agate Desert and Table Rocks area. Table 2-5 depicts the 17 focal areas USFWS recommends for conservation efforts. The arrangement of the protected parcels shall allow for a network of protected parcels with a maximum distance of 2 miles from the nearest parcel.

Approximately 2,330 acres is currently protected in four critical habitat units (CHUs) (Table 2-6). Management plans for several of these protected areas need to be finalized and implemented. This includes approximately:

- 180 acres of conservation banks in various stages of establishment,
- 140 acres in conservation easements,

- 550 acres administered by federal agencies,
- 800 acres in state ownership,
- 660 acres owned by TNC

Approximately 2,000 acres still need to be protected to meet the recovery objective for the Klamath Mountain Vernal Pool region,

Table 2-5. USFWS focal areas for vernal pool conservation and recovery in Jackson County (USFWS 2011c).

Focal Area	Area	Acreage needed	Acreage currently protected
1	Table Rocks (upper and lower)	400	892
2	Sam's Valley	400	100
3	Shady Cove	400	0
4	Staley Road	200	0
5	Eagle Point north	400	140
6	Eagle Point south	100	0
7	Dutton Road	100	34
8	Brigham-Brown/Avenue H	100	0
9	Highway 140 corridor	100	8
10	Dry Creek, north of Hwy. 140	400	0
11	Agate Reservoir	200	154
12	Corey Road	100	0
13	Denman north (Military Slough)	400	400
14	Denman south (Hall Tract)	300	300
15	Table Rock Road/Antelope Road	300	61
16	Kirtland Road	200	20
17	Truax Road/Newland Road	300	224
Total Acres		4,400	2,330

2.4.5.2 Current Actions

Vernal Pool Wetland Conservation Efforts in the Agate Desert

In 2003, USFWS awarded a Habitat Conservation Plan Planning Assistance grant to the Oregon Department of State Lands (DSL). DSL, in partnership with Jackson County, Oregon, used the funding to establish a forum to discuss economic develop and habitat conservation in and around White City, an unincorporated community located on the Agate Desert landform. The "Agate Desert Project" grew out of this effort to protect the remaining pools, with the hopes of conserving the vernal pool ecosystem and recovering the listed species to the point that they could be delisted. The Project also intended to provide new economic opportunities and allow the local communities to grow. Because most of the immediate development pressure has been occurring in and around White City, the Agate Desert Project focused the project efforts within the White City Study Area (the Unincorporated Urban Growth Boundary and a ½ mile buffer

around the boundary). In 2005, work on a Wetland Conservation Plan Inventory (WCPI) was initiated within the approximately 10,000-acre White City Study Area.

Field work for the WCPI was conducted by a consultant, hired by DSL, in May and June of 2005 (ESA 2007). Prior to field work, the consultant reviewed several offsite mapping sources including draft mapping to DSL Local Wetland Inventory (LWI) standards, which covered part but not all of the subject study area (DEA 2001), aerial photographs, National Wetland Inventory (NWI) maps and soil mapping. These sources were overlaid in Geographic Information System (GIS) mapping format to produce a preliminary wetlands map for the study area. The primary goal of field work was to ground-truth the preliminary mapping at the best available level of detail given both site access and timing/logistical constraints. Perimeter ("fenceline") and offsite (no onsite or "fenceline" viewing possible) methods were used to obtain a three-tiered framework of best available site evaluation level for the WCPI.

The approximately 10,000-acre study area included approximately 3,600 acres of vernal pool habitat identified in 59 individual vernal pool complexes. Not all vernal pool habitat was identified or characterized due to logistic and time constraints.

Habitat Conservation

Strategies to protect and recover the vernal pool fairy shrimp rely primarily on preservation and restoration/creation of vernal pool habitat (USFWS 2005c). In Oregon, approximately 2,333 acres of vernal pool habitats, including mitigation banks, have been set aside for the fairy shrimp (Tables VPFS-1, VPFS-2). The shrimp is protected on TNC's Agate Desert, Rogue River Plains, and Whetstone Savannah Preserves, and receives some protection on USBLM, USBR, and State lands (USFWS 2006) in Oregon.

The acreage of fairy shrimp habitat protected by the following entities is described below:

- USBLM to implement a management plan to restore and preserve 432 acres of vernal pool habitat on top of the Table Rocks,
- USBR to implement a management plan for 154 acres of vernal pool habitat near Agate reservoir,
- Oregon Department of Transportation (ODOT) to implement a management plan for an eight acre special management area along highway 140 and a long-term management plan for an 80-acre conservation bank adjacent to the TNC's whetstone Savannah; and,
- ODFW to implement a management plan for the 720 acres of vernal pool habitat within the Denman wildlife management area.

Species and habitat surveys, wetland function and value assessments and restoration efforts are currently underway on these land parcels as part of efforts to conserve and protect critical habitat and the associated PCEs.

Currently, acreage managed by TNC is considered more secure than some of the other parcels due to the long term management objectives of TNC. Approximately 197 acres of habitat exists on the TNC preserves (all designated as CHU #2) and supports the some of the largest

populations of the listed species. An additional 100 acres is under a conservation easement between TNC and a private landowner (CHU #1). However, development plans have been made for lands immediately surrounding the parcels.

In May 2009, TNC announced the purchase of 1,710 acres on and adjacent to Table Rocks. This large acquisition includes approximately 460 acres of vernal pool habitat on Upper and Lower Table Rock. All 460 acres of vernal pool habitat was included within CHU #4. This acquisition results in the entire CHU #4 (892 acres) considered protected (Table 2-6).

Table 2-6. Acres of vernal pool fairy shrimp critical habitat in Oregon, by CHU, by ownership, providing habitat protection for vernal pool fairy shrimp.

Land Parcel	CHU#	Ownership/ Management	Acres Of Critical Habitat
Agate Reservoir	CHU 2	USBR	154
Table Rocks	CHU 4	USBLM/TNC	892
Agate Desert Preserve	CHU 3	TNC	53
Whetstone Preserve	CHU 3	TNC	144
Rogue River Plains Preserve	CHU 1	TNC	100
Wildlands Bank	CHU 1	Wildlands, Inc.	100
Denman Wildlife Area[6]	CHU 3	ODFW	720
Highway 140 SMA	CHU 2	ODOT	8
ODOT Conservation Bank	CHU 3	ODOT	80
Jackson County School District #9	CHU 2	Jackson County School District #9	25
Hornecker	CHU 2	Private	5
ODOT	CHU 2	ODOT	4
City of Medford	CHU 3	City of Medford	3
Bear Creek Valley Sanitary Authority	CHU 3	City of Medford	5
Total			2,293

In 2009, Wildlands, Inc., proposed to USFWS, DSL, and the Corps to establish a 255-acre property as a conservation and mitigation bank. This property contains approximately 100 acres of vernal pool habitat and is located within fairy shrimp critical habitat (Table 2-6). In 2011, Wildlands Inc., DSL, the Corps, and USFWS signed an instrument that established the bank. Credits can now be sold to entities that are required to compensate for vernal pool wetland impacts under the Clean Water Act, rather than perform stand-alone wetland mitigation.

[6]. This Area is composed of the Military Slough Tract (1178 acres) and the Hall Tract (620 acres). A 12-acre wetland mitigation site is also located within the Military Slough Tract for impacts associated with the filling of 3 acres of wetlands at the Medford Airport.

In order to meet USFWS's vernal pool species recovery goals (including the fairy shrimp), a programmatic consultation pursuant to section 7(a)(2) of the Endangered Species Act of 1973, as amended (16 U.S. C. 1531 et seq.) was recently completed to cover federal agency activities related to conservation and development within vernal pool habitat in the Middle Rogue River Valley. The federal consultation requirements for conservation activities proposed by federal, state, and some private entities will be streamlined by the consultation.

The suite activities will be limited to those involving relatively minor impacts, but the overall effect of these activities and associated impacts is expected to be supportive of recovery of listed vernal pool species.

The primary objectives of this consultation will be the identification, establishment, and implementation of an overall conservation strategy for vernal pool habitats and species. This strategy includes:

- Habitat protection, restoration and enhancement targets for vernal pool complex;
- Guidance, best management practices, and performance standards for vernal pool complex and vernal pool species restoration and management;
- Prioritized use of credits available from conservation banks, mitigation banks, or equivalent conservation or mitigation projects (in conjunction with on-site best management practices and performance standards) as the most credible method to minimize the conservation effects and significance of unavoidable impacts to listed vernal pool species that might result from otherwise lawful development activities in vernal pool complex;
- Thresholds, criteria, ratios and other factors that will determine appropriate establishment of and credit allocation to banks or equivalent conservation or mitigation projects, and appropriate use of these credits to address the adverse impacts of development projects;
- Allowing Corps or DSL-approved wetland banks and credits established consistent with the above factors to also serve as species conservation banks and credits under the Act;
- Streamlined regulatory review and approval of certain conservation and development activities within vernal pool complex via utilization of a single suite of standards and processes by the Corps, DSL, and USFWS, and programmatic permitting and compliance products by each agency, including this programmatic Opinion produced by USFWS, and a Regional General Permit and General Permit produced by the Corps and DSL, respectively; and
- Common expectations and reasonable certainty for landowners, developers, habitat managers, and others engaged in vernal pool complex development and conservation activities about the standards and requirements for engaging in these activities.

Collectively, the components above represent USFWS's preferred conservation strategy for listed vernal pool complex species in Jackson County, Oregon. The PCS is based on and is intended to support USFWS's broader final recovery plan for vernal pool ecosystems of

California and Southern Oregon (USFWS 2005c), which established specific goals, objectives and tasks to direct recovery efforts for fairy shrimp, including that the described recovery criteria and actions for the Klamath Mountains Vernal Pool Region (which covers Jackson County) should be tailored over time to address more locally specific information and conditions.

2.4.6 Status of the Species in the Action Area (Oregon)

2.4.6.1 Current status and distribution of the listed species in the State of Oregon

Originally, vernal pool habitat covered approximately 21,000 acres in the Agate Desert-Rogue River Valley region. Of the remaining 8,032.4 acres of vernal pool topography (intact, weakly expressed, or altered or weak), 59.2% is in the "intact topography/hydrology and altered vegetation" class and another 15.7% is in the "intact topography/hydrology and severely altered vegetation" class. This means that a total of 6,019.8 acres (74.9 %) has intact topography and hydrology. Only 2,012.6 acres (25.6 %) of the remaining habitat has altered or weakly expressed topography and hydrology (Table 2-6). In the in the Agate Desert-Rogue River Valley region, the fairy shrimp only occurs in Jackson County. Fairy shrimp are known from at least 12 areas within the four designated critical habitat units. Two of these populations occur at each of the tops of the Table Rocks, Lower and Upper Table Rock, while the other 10 occur within the Rogue River Valley terraces.

In Oregon, approximately 2,293 acres of vernal pool habitats, including mitigation banks, have been set aside as conservation for the fairy shrimp (Table 2-5).

Vernal pool habitat inhabited by fairy shrimp range from 10% to 40% occupation within the pools. There is significant variability from pool complex to pool complex, depending on degree of the habitat quality and topographic/hydrologic condition.

The national Clean Water Act and state environmental quality laws lend some protection for fairy shrimp habitat. Such laws can be enforced on privately owned lands, but only if certain parameters are present. For example, the size of wetland filling and presence of federally listed species or designated critical habitat help determine whether a federal or state agency will regulate a wetland fill permit or a storm-water discharge.

A trend analysis or detailed population study has not been conducted within the Agate Desert-Rogue River Valley region.

2.4.6.2 Current critical habitat designation within the State of Oregon

Four critical habitat units were designated within Jackson County, Oregon, totalling 7,574 acres (USFWS 2006) (Table 2-6). Of this area, 432 acres are federally-owned land, and the remaining 7,142 acres are located on privately-owned lands. Critical habitat Units 1, 2, 3, and 4 include selected areas that occur on the Agate Desert area north of White City, in areas generally surrounding the eastern half of White City, in areas generally surrounding the western half of White City, and the summits of Upper and Lower Table Rock respectively.

The following critical habitat unit descriptions are taken from USFWS's final rule designating critical habitat for vernal pool species in California and southern Oregon (USFWS 2003):

Unit 1A, B, C, D, E, F, and G, North Agate Desert Unit, Jackson County (2,130 ac)

This unit consists of seven subunits, all located to the north of Little Butte Creek. This unit represents the northern limit of the species' distribution. It is of sufficient size to sustain the natural ecosystem processes (e.g., fires) that have historically influenced vernal pool habitat, and is separated from the nearest other unit designated for Oregon, Unit 4, by over 2 miles. Three of the subunits are west of the Rogue River, and the remaining four are to the east. All but one of these subunits is located to the south of U.S. Route 234 (Sam's Valley Highway). The one remaining unit is located to the east of the Rogue River, about 1.5 miles north of the confluence with Reese Creek.

Unit 2A, B, C, D, and E, White City East Unit, Jackson County (2,251 ac)

This unit consists of five subunits, located east of U.S. Route 62 (Crater Lake Highway) and south and southeast of Dutton Road. This unit provides the easternmost extent of the species' range in Oregon. It represents a significant component of the species' original range in the State and is of a sufficient size to sustain the natural ecosystem processes (e.g., fires) that have historically influenced vernal pool habitat. The largest and easternmost of the subunits occurs just to the east and north of Agate Lake. It is separated by more than 1 mile from Unit 3, White City West, and by approximately 3.5 miles from the North Agate Desert Unit.

Unit 3A, B, and C, White City West Unit, Jackson County (2,301 ac)

This unit consists of three subunits, located west of Agate Road, south of the Rogue River, and east of Bear Creek. This unit contains the least fragmented intact examples of the original Agate Desert mounded vernal pool grassland habitat. It is of sufficient size to sustain the natural ecosystem processes (e.g., fires) that have historically influenced vernal pool habitat; it is separated from the White City East Unit by more than 1 mile and from the Table Rocks Unit by over 1.5 miles. Taken together, the designated Agate Desert units (Units 1–3) comprise a functional vernal pool complex consisting of vernal pools, mounded grassland and associated uplands, where natural processes, including connectivity, function within or near the natural range of variability. Each of the three designated Agate Desert units is essential to the conservation of vernal pool fairy shrimp in the Agate Desert area.

Unit 4A and B, Table Rocks Unit, Jackson County (892 ac)

This unit consists of two subunits, located on two flat-topped mesas known as Upper and Lower Table Rocks, situated north and west of the Rogue River. These rimrock features are remnants of ancient lava flows that filled portions of the Rogue River nearly 10 million years ago. Subsequent erosion of softer geologic layers has left these harder andesite (volcanic rock) formations rising some 800 feet above the present Rogue Valley. Vernal pools on the Table Rocks differ from those of the Agate Desert, in that they are formed over an impervious layer of bedrock. This unit represents a unique habitat for vernal pool fairy shrimp in Oregon. The Table Rocks Unit is disjunct from the North Agate Desert Unit by over 2 miles, and from the White City West Unit by approximately 1.5 miles.

The vernal pool habitat within these four units was selected based on information provided by a wetland function and values assessment and habitat integrity analysis completed in 1999. Information describing the physical (i.e., parcel size, presence of intact hydrology) and biological (i.e., species diversity, presence and composition of native vegetation) condition of the vernal pool habitat, species inventory information detailing the presence fairy shrimp, Cook's lomatium (*Lomatium cookii*) and large-flowered woolly meadowfoam (*Limnanthes pumila ssp. grandiflora*), and parameters describing the potential long term sustainability of habitat (defensibility of the parcel, ownership, and positioning of the parcel relative to nearby habitat parcels) was used to identify specific parcels for inclusion as critical habitat.

2.4.6.3 Threats specific to Oregon and discussion of threats relative to water quality (toxicity and temperature)

The USFWS listed the fairy shrimp as a threatened species in Oregon primarily due to the present or threatened destruction, modification, or curtailment of their habitat or range. We determined that "the habitat of these animals is imperilled by a variety of human-caused activities, primarily urban development, water supply/flood control activities, and conversion of land to agricultural use. Habitat loss occurs from direct destruction and modification of pools due to filling, grading, disking, levelling, and other activities, as well as modification of surrounding uplands that alters vernal pool watersheds."

Section 404 of the Clean Water Act requires parties to obtain a permit from the Corps prior to discharging dredge or fill material into "waters of the United States." As defined in Part 232 of the Clean Water Act, however, normal farming, silviculture, and ranching activities, including, plowing, seeding, cultivating, minor drainage, and harvesting for the production of food, fiber, and forest products are exempt from the Clean Water Act.

One exception to this exemption is the utilization of "deep-ripping" activities. "Deep-ripping" activities within vernal pool habitat always require a permit pursuant to section 404 of the Clean Water Act. Within the vernal pool area, ripping is primarily used to convert agricultural lands for development (ripping breaks the impermeable soil layer allowing water to drain and eliminating wetlands). It is possible that deep ripping and other development activities are occurring illegally in vernal pool habitat without a 404 permit. However, there is no way to determine the extent of such activities within the study area. (IEI 2010).

Alterations of temperatures could be detrimental to fairy shrimp in the Agate Desert-Rogue River Valley. The life-cycle of fairy shrimp is dependent on cool water conditions in vernal pools during the winter and spring months. If temperatures rise above 24 °C in the temporary pools, before fairy shrimp have a chance to breed, juveniles and adults will die off and reproduction may not occur (Helm 1998, Eriksen and Belk 1999).

The use of glyphosate is likely detrimental to fairy shrimp. Brausch and Smith (2007) used the surrogate fairy shrimp species, *Thamnocephalus platyurus* to assess the acute toxicity of POEA, the ingredient often included as a surfactant to the commonly used herbicide ingredient, glyphosate, and found it to be extremely toxic to this species at low concentrations (Relyea 2005, Brausch and Smith 2007).

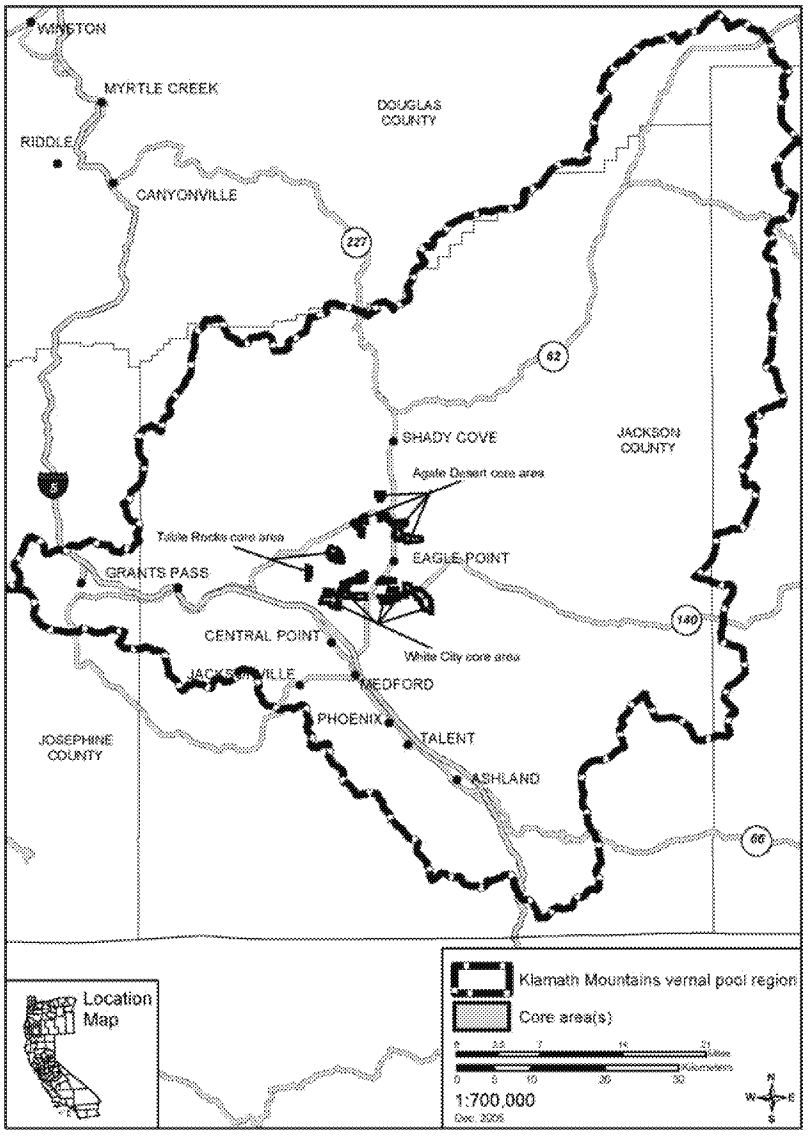


Figure III-4. Klamath Mountains vernal pool region.

III-31

Figure 2-6. Klamath Mountain Vernal Pool Region (from USFWS 2005c)

2.5 Marbled Murrelet

2.5.1 Species Description

2.5.1.1 Taxonomy

The murrelet (*Brachyramphus marmoratus*) is diving seabird of the Alcidae family.

2.5.1.2 Species description

The murrelet is a small robin-sized seabird. Males and females have sooty-brown upperparts with dark bars. Underparts are light, mottled brown. Winter adults have brownish-gray upperparts and white scapulars. The plumage of fledged young is similar to that of adults in winter. Chicks are downy and tan colored with dark speckling.

2.5.1.3 Current legal status, including listing history

The marbled murrelet (murrelet) was listed as a threatened species on September 28, 1992, in Washington, Oregon, and northern California (USFWS 1992a). Since the species' listing, USFWS has completed two 5-yr status reviews of the species: September 1, 2004 (USFWS 2004d) and June 12, 2009 (USFWS 2009a). The 2004 5-year review determined that the California, Oregon, and Washington distinct population segment of the murrelet did not meet the criteria outlined in the USFWS's 1996 Distinct Population Segment (DPS) policy (USFWS and NMFS 1996, USFWS 2004d). However, the 2009 5-year review concluded the 2004 analysis of the DPS question was based on a flawed assumption regarding discreteness at the international border with Canada (USFWS 2009a, pp. 3-12). The legal status of the murrelet remains unchanged from the original designation.

2.5.2 Critical habitat Description

2.5.2.1 Current legal status of the critical habitat

The final rule designating murrelet critical habitat (USFWS 1996a) was issued May 24, 1996. On October 5, 2011, the final rule revising critical habitat for the murrelet was published (USFWS 2011b).

2.5.2.2 The primary constituent elements (PCEs)

The physical or biological features associated with marbled murrelet critical habitat focused on individual trees with potential nesting platforms, and forested areas within 0.8 kilometers (0.5 miles) of individual trees with potential nesting platforms that had a canopy height of at least one-half the site potential tree height (the average maximum height for trees given local growing conditions).

2.5.2.3 Identification of the potential effects of toxics or water temperature levels to alteration

Effects from the proposed project are limited to in water areas. No effects are anticipated to forest environments adjacent to water bodies. Therefore, no effects to murrelet critical habitat are expected.

2.5.3 Life history

2.5.3.1 Reproduction

Marbled murrelets produce one egg per nest and usually only nest once a year, however re-nesting is documented. Nests are not built, but rather the egg is placed in a small depression or cup made in moss or other debris on the limb. Incubation lasts about 30 days, and chicks fledge after about 28 days after hatching. Both sexes incubate the egg in alternating 24-hour shifts. The chick is fed up to eight times daily, and is usually fed only one fish at a time. The young are semiprecocial, capable of walking but not leaving the nest. Fledglings fly directly from the nest to the ocean.

Generally, estimates of murrelet fecundity are directed at measures of breeding success, either from direct assessments of nest success in the terrestrial environment, marine counts of hatch-year birds, or computer models. Telemetry estimates are typically preferred over marine counts for estimating breeding success due to fewer biases (McShane et al. 2004, p. 3-2). However, because of the challenges of conducting telemetry studies, estimating murrelet reproductive rates with an index of reproduction, referred to as the juvenile ratio (Ŕ),[7] continues to be important, despite the debate over use of this index (see discussion in Beissinger and Peery 2007, p. 296).

Although difficult to obtain, nest success rates[8] are available from telemetry studies conducted in California (Peery et al. 2004, Hebert and Golightly 2006) and Washington (Bloxton and Raphael 2006). In northwest Washington, Bloxton and Raphael (2005, p. 5) documented a nest success rate of 0.20 (2 chicks fledging from 10 nest starts). In central California, murrelet nest success is 0.16 (Peery et al. 2004, p. 1098) and in northern California it is 0.31 to 0.56 (Hebert and Golightly 2006, p. 95). No studies or published reports from Oregon are available.

Unadjusted and adjusted values for annual estimates of murrelet juvenile ratios suggest extremely low breeding success in Conservation Zone 4 (mean ratio for 2000-2011 of 0.046, range 0.01 to 0.1, Crescent Coastal Research 2012a, p. 11), northern California (0.003 to 0.029 - Long et al. 2008, pp. 18-19; Crescent Coastal Research 2012a, p. 11), central California (0.035 and 0.032 - Beissinger and Peery 2007, pp. 299, 302), and in Oregon (0.0254 - 0.0598 - Crescent Coastal Research 2008, p. 13). Estimates for **Ŕ** (adjusted) in the San Juan Islands in Washington have been below 0.15 every year since surveys began in 1995, with three of those years below 0.05 (Raphael et al. 2007b, p. 16).

These current estimates of **Ŕ** are assumed to be below the level necessary to maintain or increase the murrelet population. Demographic modeling suggests murrelet population stability requires a minimum reproductive rate of 0.18 to 0.28 (95 % confidence interval) chicks per pair per year (USFWS 1997a; Beissinger and Peery 2007, p. 302). The estimates for Ŕ discussed above from individual studies, as well as estimates for the listed range (0.02 to 0.13; Table 2-9) are all below

[7] The juvenile ratio (Ŕ) for murrelets is derived from the relative abundance of hatch-year (0-1 yr-old) to after-hatch-year (1+ yr-old) birds (Beissinger and Peery 2007, p. 297) and is calculated from marine survey data.
[8] Nest success here is defined by the annual number of known hatchlings departing from the nest (fledging) divided by the number of nest starts.

the lowest estimated value (0.18) identified as required for population stability (USFWS 1997a, Beissinger and Peery 2007, p. 302).

The current estimates for **Ŕ** also appear to be well below what may have occurred prior to the murrelet population decline. Beissinger and Peery (2007, p. 298) performed a comparative analysis using historic data from 29 bird species to predict the historic **Ŕ** for murrelets in central California, resulting in an estimate of 0.27 (95% CI: 0.15 - 0.65). Therefore, the best available scientific information of current murrelet fecundity from model predictions, and from juvenile ratios and trend analyses based on of population survey data appear to align well. Both indicate that the murrelet reproductive rate is generally insufficient to maintain stable population numbers throughout all or portions of the species' listed range.

2.5.4 Population structure

Murrelets are long-lived seabirds that spend most of their life in the marine environment, with breeding adult birds annually nesting in the forest canopy of mature and old-growth forests from about March 24 through September 15. Murrelets have a naturally low reproductive rate. Murrelets lay just one egg, and are thought to usually first breed at age 3.

2.5.4.1 Recovery Zones

The Plan identified six Conservation Zones (Figure 2-7) throughout the listed range of the species: Puget Sound (Conservation Zone 1), Western Washington Coast Range (Conservation Zone 2), Oregon Coast Range (Conservation Zone 3), Siskiyou Coast Range (Conservation Zone 4), Mendocino (Conservation Zone 5), and Santa Cruz Mountains (Conservation Zone 6). Recovery zones are the functional equivalent of recovery units as defined by USFWS policy (USFWS 1997a, p. 115).

Recovery Zones in Oregon

Conservation Zone 3 (Oregon Coast Range Zone)

This zone extends from the Columbia River, south to North Bend, Coos County, Oregon. Conservation zone 3 includes waters within 2 km (1.2 miles) of the Pacific Ocean shoreline and extends inland a distance of up to 56 km (35 miles) from the Pacific Ocean shoreline and coincides with the zone 1 boundary line. This zone contains the majority of murrelet sites in Oregon. Murrelet sites along the western portion of the Tillamook State Forest are especially important to maintaining well-distributed murrelet populations. Maintaining suitable and occupied murrelet habitat on the Elliot State Forest, Tillamook State Forest, Siuslaw National Forest, and USBLM-administered forests is an essential component for the stabilization and recovery of murrelets (USFWS 1997a). Beissinger and Peery (2003) estimated a 2.8 to 13.4 percent population decline for this zone.

Conservation Zone 4 (Siskiyou Coast Range Zone)

The Siskiyou Coast Range zone extends from North Bend, Coos County, Oregon south to the southern end of Humboldt County, California. It includes waters within 1.2 miles of the Pacific Ocean shoreline (including Humboldt and Arcata bays) and, generally extends inland a distance

of 56 km (35 miles) from the Pacific shoreline. This zone contains populations in Redwood National Park and several state parks. It contains nesting habitat on private lands in southern Humboldt County and at lower elevations in the western portions of Smith River National Recreation Area (USFWS 1997a). Beissinger and Peery (2003) estimated a 2.5 to 13.2 percent population decline for this zone.

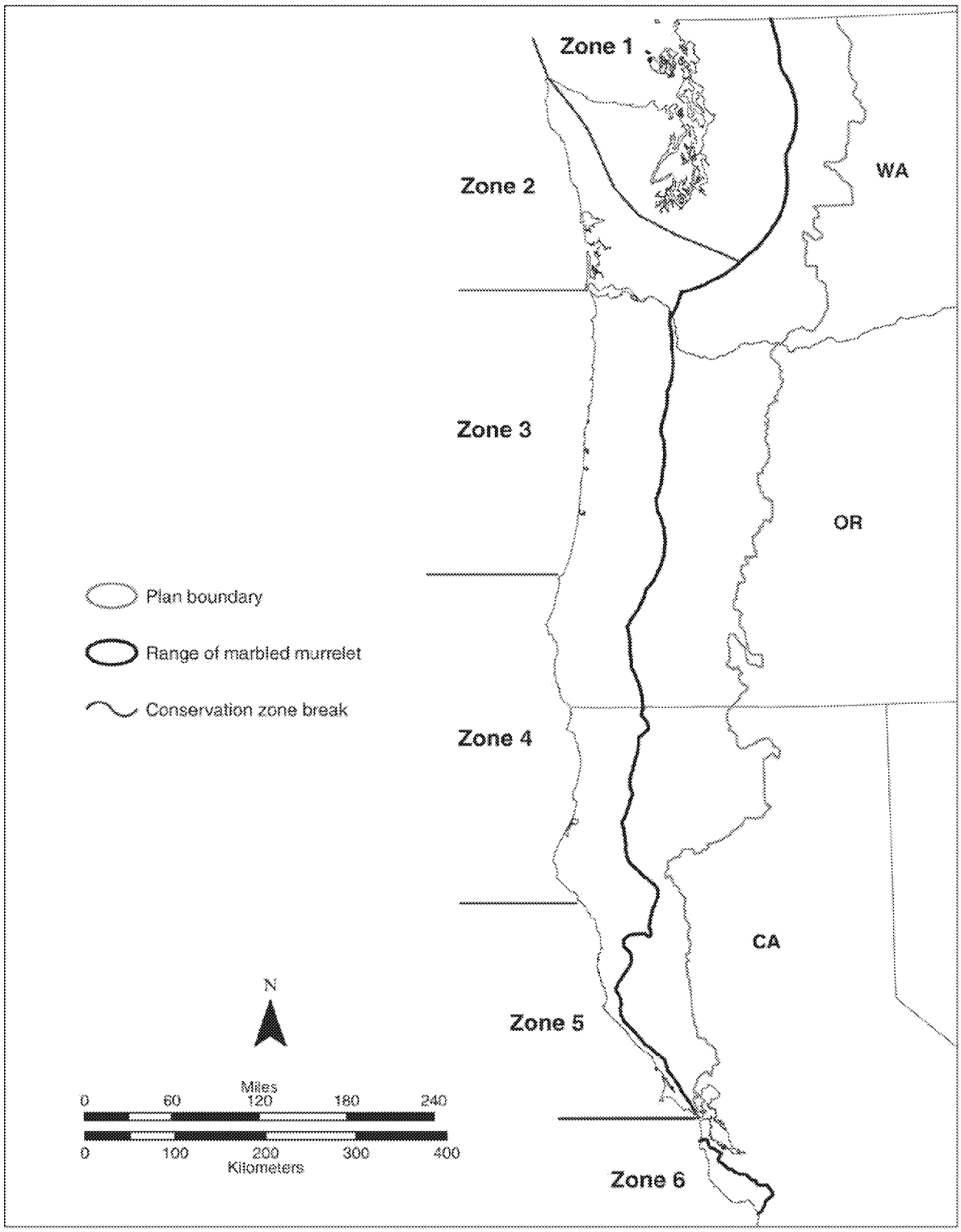


Figure 2-7. The six geographic areas identified as Conservation Zones in the recovery plan for the marbled murrelet (USFWS 1997a). Note: "Plan boundary" refers to the Northwest Forest Plan. Figure adapted from Huff et al. (2006, p. 6).

2.5.4.2 Ecology / Habitat Characteristics

Marbled murrelets are long-lived seabirds that spend most of their life in the marine environment, but use old-growth forests for nesting. Courtship, foraging, loafing, molting, and preening occur in near-shore marine waters. Throughout their range, marbled murrelets are opportunistic feeders and use prey of diverse sizes and species. They feed primarily on fish and invertebrates in near-shore marine waters although they have also been detected on rivers and inland lakes.

Murrelets spend most of their lives in the marine environment where they forage in near-shore areas and consume a diversity of prey species, including small fish and invertebrates. In their terrestrial environment, the presence of platforms (large branches or deformities) used for nesting is the most important characteristic of their nesting habitat. Murrelet habitat use during the breeding season is positively associated with the presence and abundance of mature and old-growth forests, large core areas of old-growth, low amounts of edge habitat, reduced habitat fragmentation, proximity to the marine environment, and forests that are increasing in stand age and height. Additional information on murrelet taxonomy, biology, and ecology can be found in Ralph et al. 1995 McShane et al. 2004, and Piatt et al. 2007.

Aquatic Habitat Use

Marbled murrelets are usually found within 5 miles (8 km) from shore, and in water less than 60 meters deep (Ainley et al. 1995, Burger 1995, Strachan et al. 1995, Nelson 1997, Day and Nigro 2000, Raphael et al. 2007a). In general, birds occur closer to shore in exposed coastal areas and farther offshore in protected coastal areas (Nelson 1997). Courtship, foraging, loafing, molting, and preening occur in marine waters.

Marbled murrelets are wing-propelled pursuit divers that forage both during the day and at night (Carter and Sealy 1986, Gaston and Jones 1998, Henkel et al. 2003, Kuletz 2005). Marbled murrelets can make substantial changes in foraging sites within the breeding season, but many birds routinely forage in the same general areas and at productive foraging sites, as evidenced by repeated use over a period of time throughout the breeding season (Carter and Sealy 1990, Whitworth et al. 2000, Becker 2001, Hull et al. 2001, Mason et al. 2002, Piatt et al. 2007). Marbled murrelets are also known to forage in freshwater lakes (Nelson 1997). Activity patterns and foraging locations are influenced by biological and physical processes that concentrate prey, such as weather, climate, time of day, season, light intensity, up-wellings, tidal rips, narrow passages between island, shallow banks, and kelp (*Nereocystis* spp.) beds (Ainley et al. 1995, Burger 1995, Strong et al. 1995, Speckman 1996, Nelson 1997).

Juveniles are generally found closer to shore than adults (Beissinger 1995) and forage without the assistance of adults (Strachan et al. 1995). Kuletz and Piatt (1999) found that in Alaska, juvenile marbled murrelets congregated in kelp beds. Kelp beds are often associated with productive waters and may provide protection from avian predators (Kuletz and Piatt 1999). McAllister (Strachan et al. 1995) found that juveniles were more common within 328 feet (100 m) of shorelines, particularly where bull kelp was present.

Marbled murrelets usually feed in shallow, near-shore water less than 98 feet (30 m) deep (Huff et al. 2006), but are thought to be able to dive up to depths of 157 feet (47 m) (Mathews and Burger 1998). During the non-breeding season, marbled murrelets disperse and can be found farther from shore (Strachan et al. 1995). Although little information is available outside of the nesting season, limited information on winter distribution also suggests they do move farther offshore (Craig Strong, Biologist, Crescent Coast Research, Crescent City, California, pers. comm., 2007). During the winter, and where available, there may be a general opportunistic shift from exposed outer coasts into more protected waters (Nelson 1997); for example many marbled murrelets breeding on the exposed outer coast of Vancouver Island appear to congregate in the more sheltered waters within the Puget Sound and the Strait of Georgia in fall and winter (Burger 1995). In many areas, marbled murrelets also undertake occasional trips to inland nesting habitat during the winter months (Carter and Erickson 1992). Throughout the listed range, marbled murrelets do not appear to disperse long distances, indicating they are year-round residents (McShane et al. 2004).

Throughout their range, marbled murrelets are opportunistic feeders and use prey of diverse sizes and species. They feed primarily on fish and invertebrates in marine waters although they have also been detected on rivers and inland lakes (Carter and Sealy 1986, USFWS 1992a). In general, small schooling fish and large pelagic crustaceans are the main prey items. Pacific sand lance (*Ammodytes hexapterus*), northern anchovy (*Engraulis mordax*), immature Pacific herring (*Clupea harengus*), capelin (*Mallotus villosus*), Pacific sardine (*Sardinops sagax*), juvenile rockfishes (*Sebastas* spp.), and surf smelt (Osmeridae) are the most common fish species taken. Squid (*Loligo* spp.), euphausiids, mysid shrimp, and large pelagic amphipods are the main invertebrate prey. Marbled murrelets are able to shift their diet throughout the year and over years in response to prey availability (Becker et al. 2007). However, long-term adjustment to less energetically-rich prey resources (such as invertebrates) appears to be partly responsible for poor marbled murrelet reproduction in California (Becker and Beissinger 2006).

Breeding adults exercise more specific foraging strategies when feeding chicks, usually carrying a single, relatively large (relative to body size) energy-rich fish to their chicks (Burkett 1995, Nelson 1997), primarily around dawn and dusk (Nelson 1997, Kuletz 2005). Freshwater prey appears to be important to some individuals during several weeks in summer and may facilitate more frequent chick feedings, especially for those that nest far inland (Hobson 1990). Becker et al. (2007) found marbled murrelet reproductive success in California was strongly correlated with the abundance of mid-trophic level prey (e.g., sand lance, juvenile rockfish) during the breeding and postbreeding seasons. Prey types are not equal in the energy they provide; for example parents delivering fish other than age-1 herring may have to increase deliveries by up to 4.2 times to deliver the same energy value (Kuletz 2005). Therefore, nesting marbled murrelets that are returning to their nest at least once per day must balance the energetic costs of foraging trips with the benefits for themselves and their young. This may result in marbled murrelets preferring to forage in marine areas in close proximity to their nesting habitat. However, if adequate or appropriate foraging resources (i.e., "enough" prey, or prey with the optimum nutritional value for themselves or their young) are unavailable in close proximity to their nesting areas, marbled murrelets may be forced to forage at greater distances or to abandon their

nests (Huff et al. 2006). As a result, the distribution and abundance of prey suitable for feeding chicks may greatly influence the overall foraging behavior and location(s) during the nesting season, may affect reproductive success (Becker et al. 2007), and may significantly affect the energy demand on adults by influencing both the foraging time and number of trips inland required to feed nestlings (Kuletz 2005).

Nesting Biology

Incubation is shared by both sexes, and incubation shifts are generally 1 day, with nest exchanges occurring at dawn (Nelson 1997, Bradley 2002). Hatchlings appear to be brooded by a parent for 1 or 2 days and then left alone at the nest for the remainder of the chick period while both parents spend most of their time foraging at sea. Both parents feed the chick (usually a single fish carried in the bill) and the chick typically receives 1 to 8 meals per day (mean 3.2) (Nelson 1997). About two-thirds of feedings occur early in the morning, usually before sunrise, and about one-third occur at dusk. Feedings are sometimes scattered throughout the day (Hamer and Nelson 1995). Chicks fledge 27 to 40 days after hatching, at 58%-71% of adult mass (Nelson 1997). Fledging has seldom been documented, but it typically appears to occur at dusk (Nelson 1997, Jones 2001).

Nest Tree Characteristics

Lank et al. (2003) states that murrelets "occur during the breeding season in near-shore waters along the north Pacific coastline from Bristol Bay in Alaska to central California", nesting in single platform trees generally within 20 miles of the coast and older forest stands generally within 50 miles of the coast. Unlike most auks, murrelets nest solitarily on mossy platforms of large branches in old-forest trees (Lank et al. 2003). Suitable murrelet habitat may include contiguous forested areas with conditions that contain potential nesting structure. These forests are generally characterized by large trees greater than 18 inches diameter at breast height, multi-storied canopies with moderate canopy closure, sufficient limb size and substrate (moss, duff, etc.) to support nest cups, flight accessibility, and protective cover from ambient conditions and potential avian predators (Manley 1999, Burger 2002, Nelson and Wilson 2002). Over 95% of measured nest limbs were ≥15 cm diameter, with limb diameter ranges from 7-74 cm diameter (Burger 2002). Nelson and Wilson (2002) found that all 37 nest cups identified were in trees containing at least seven platforms. All trees were climbed, however, and ground-based estimates of platforms per tree in the study were not analyzed. Lank et al. (2003) emphasizes that murrelets do not select nest sites based on tree species, but rather they select those individual trees that offer suitable nest platforms. Nest cups have been found in deciduous trees, albeit rarely and nest trees may be scattered or clumped throughout a forest stand.

Nest Stand Characteristics

Nest stands are typically composed of low elevation conifer species. In California, nest sites have been located in stands containing old-growth redwood and Douglas-fir, while nests in Oregon and Washington have been located in stands dominated by Douglas-fir, western hemlock, and Sitka spruce. Murrelets appear to select forest stands greater than 123.6 acres (50 ha) (Burger 2002), but nest in stands as small as 1 acre (Nelson and Wilson 2002). In surveys of mature or younger second-growth forests in California, murrelets were only found in forests

where there were nearby old-growth stands or where residual older trees remained (USFWS 1992a, Singer et al. 1995).

At the stand level, vertical complexity is correlated with nest sites (Meekins and Hamer 1998, Manley 1999, Waterhouse et al. 2002, Nelson and Wilson 2002), and flight accessibility is probably a necessary component of suitable habitat (Burger 2002). Some studies have shown higher murrelet activity near stands of old-forest blocks over fragmented or unsuitable forest areas (Paton et al. 1992, Rodway et al. 1993, Burger 1995, Deschesne and Smith 1997, Rodway and Regehr 2002), but this correlation may be confounded by ocean conditions, distance inland, elevation, survey bias and disproportionately available habitat. Nelson and Wilson (2002) found that potential nest platforms per acre were a strong correlate for nest stand selection by murrelets in Oregon.

Adjacent forests can contribute to the conservation of the murrelet by reducing the potential for windthrow during storms by providing area buffers and creating a landscape with a higher probability of occupancy by murrelets (USFWS 1996a, Burger 2001, Meyer et al. 2002, Raphael et al. 2002). Trees surrounding and within the vicinity of a potential nest tree(s) may provide protection to the nest platform and potentially reduce gradations in microclimate (Chen et al. 1993).

Consulted on effects from October 1, 2003 to May 23, 2012 (affected habitats are summarized in Table 2-7).

Table 2-7. Aggregate Results of All Suitable Habitat (Acres) Affected by Section 7 Consultation for the Marbled Murrelet; Summary of Effects By Conservation Zone and Habitat Type From October 1st, 2003 to May 23, 2012.

Conservation Zone[1]	Authorized Habitat Effects[2]		Reported Habitat Effects[2]	
	Stands[3]	Remnants[4]	Stands[3]	Remnants[4]
Puget Sound	-69	0	-1	0
Western Washington	-33	0	-12	0
Outside CZ Area in WA	0	0	0	0
Oregon Coast Range	-702	-150	-137	0
Siskiyou Coast Range	-1,765	0	-137	0
Outside CZ Area in OR	-2	0	0	0
Mendocino	0	0	0	0
Santa Cruz Mountains	0	0	0	0
Outside CZ Area in CA	0	0	0	0
Total	**-2,571**	**-150**	**-287**	**0**

Notes:

1. Conservation Zones (CZ) six zones were established by the 1997 Recovery Plan to guide terrestrial and marine management planning and monitoring for the Marbled Murrelet (USFWS 1997a).
2. Habitat includes all known occupied sites, as well as other suitable habitat, though it is not necessarily occupied. Importantly, there is no single definition of suitable habitat, though the Marbled Murrelet Effectiveness Monitoring Module is in the process. Some useable working definitions include the Primary Constituent Elements as defined in the Critical Habitat Final Rule, or the criteria used for Washington State by Raphael et al. (2002).
3. Stand: A patch of older forest in an area with potential platform trees.
4. Remnants: A residual/remnant stand is an area with scattered potential platform trees within a younger forest that lacks, overall, the structures for marbled murrelet nesting.

Landscape Characteristics

Studies have determined the characteristics of murrelet nesting habitat at a landscape-scale using a variety of methods, including predictive models, radio telemetry, audio-visual surveys, and radar. McShane et al. (2004, pg. 4-103) reported, "At the landscape level, areas with evidence of occupancy tended to have higher proportions of large, old-growth forest, larger stands and greater habitat complexity, but distance to the ocean (up to about 37 miles [60 km]) did not seem important." Elevation had a negative association in some studies with murrelet habitat occupancy (Burger 2002). Hamer and Nelson (1995) sampled 45 nest trees in British Columbia, Washington, Oregon, and California and found the mean elevation to be 1,089 feet (332 m).

Multiple radar studies (e.g., Burger 2001, Cullen 2002, Raphael et al. 2002, Steventon and Holmes 2002) in British Columbia and Washington have shown that radar counts of murrelets are positively associated with total watershed area, increasing amounts of late-seral forests, and with increasing age and height class of associated forests. Murrelet radar counts are also negatively associated with increasing forest edge and areas of logged and immature forests (McShane et al. 2004). Several studies have concluded that murrelets do not pack into higher densities within remaining habitat when nesting habitat is removed (Burger 2001, Manley et al. 2001, Cullen 2002).

There is a relationship between proximity of human-modified habitat and increased avian predator abundance. However, increased numbers of avian predators does not always result in increased predation on murrelet nests. For example, Luginbuhl et al. (2001, pg. 565) report, in a study using simulated murrelet nests, that "Corvid numbers were poorly correlated with the rate of predation within each forested plot". Luginbuhl et al. (2001, pg. 569), conclude, "that using measurements of corvid abundance to assess nest predation risk is not possible at the typical scale of homogenous plots (0.5-1.0 km^2 in our study). Rather this approach should be considered useful only at a broader, landscape scale on the order of 5-50 km^2 (based on the scale of our fragmentation and human-use measures)."

Artificial murrelet nest depredation rates were highest in western conifer forests where stand edges were close to human development (De Santo and Willson 2001, Luginbuhl et al. 2001), and Bradley (2002) found increased corvid densities within 3 miles of an urban interface, probably due to supplemental feeding opportunities from anthropogenic activities. Golightly et al. (2002) found extremely low reproductive success for murrelets nesting in large old-growth blocks of redwoods in the California Redwoods National and State Parks. Artificially high corvid densities from adjacent urbanization and park campgrounds are suspected to be a direct cause of the high nesting failure rates for murrelets in the redwoods parks.

If the surrounding landscape has been permanently modified to change the predators' numbers or densities through, for example, agriculture, urbanization, or recreation, and predators are causing unnaturally high nest failures, murrelet reproductive success may remain depressed. Because corvids account for the majority of depredations on murrelet nests and corvid density can increase with human development, corvid predation on murrelet habitat is a primary impact consideration. The threat of predation on murrelet populations (both nests and adults) appears to be greater than previously anticipated (McShane et al. 2004).

2.5.5 Status

2.5.5.1 Historical status and distribution

Murrelet abundance during the early 1990s in Washington, Oregon, and California was estimated at 18,550 to 32,000 birds (Ralph et al. 1995).

The historical breeding range of the marbled murrelet extends from Bristol Bay, Alaska, south to the Aleutian Archipelago, northeast to Cook Inlet, Kodiak Island, Kenai Peninsula and Prince

William Sound, south coastally throughout the Alexander Archipelago of Alaska, and through British Columbia, Washington, Oregon, to northern Monterey Bay in central California. Birds winter throughout the breeding range and also occur in small numbers off southern California.

At the time of listing, the distribution of active nests in nesting habitat was described as non-continuous (USFWS 1997a, p. 14). The at-sea extent of the species currently encompasses an area similar in size to the species' historic distribution, but with the extremely low density of murrelets in Conservation Zone 5, and the small population in Conservation Zone 6, the southern end of the murrelet distribution is sparsely populated compared to Conservation Zones 1-4.

2.5.6 Current status and distribution of the listed species rangewide

Based primarily on the results from the Northwest Forest Plan (NWFP) Effectiveness Monitoring (EM) Program, the 2010 murrelet population for the listed range (Table 2-8) is estimated at 16,691 birds (95% confidence interval: 13,075 – 20,307; Table 2-8). Based on the 2010 estimates, Conservation Zones 3 and 4 support approximately 65% of the murrelet population within the U.S., and consistently have the highest – at-sea densities during the nesting season (Falxa et al. 2011). Murrelets occur in the lowest abundance in Conservation Zones 5 and 6.

Table 2-8. Estimates of murrelet density and population size (95% confidence interval) in Conservation Zones 1 through 5 during the 2010 breeding season (Falxa et al. 2011), and in Conservation Zone 6 during the 2009 breeding season (Peery and Henry 2010). Conservation Zone 5 estimates are from 2008, because the zone was not surveyed in 2009 or 2010.

Conservation Zone	Density (birds/km^2)	Coefficient of Variation (% Density)	Population Size Estimates with 95% confidence interval			Survey Area (km^2)
			Number of Birds	Lower	Upper	
1	1.26	20.4	4393	2,689	6,367	3,497
2	0.18	25.7	1,286	650	1946	1,650
3	4.53	16.9	7,223	4,605	9,520	1,595
4	3.16	27.3	3,668	2,196	6,140	1,159
5	0.14	-	121	-	242	883
6	-	-	631	449	885	-
Zones 1-6	-	-	17,322	13,524	21,192	-

The at-sea distribution also exhibits discontinuity within Conservation Zones 1, 2, 5, and 6, where five areas of discontinuity are noted: a segment of the border region between British Columbia, Canada and Washington, southern Puget Sound, Washington, Destruction Island, Washington to Tillamook Head, Oregon, Humboldt County, California, to Half Moon Bay, California, and the entire southern end of the breeding range in the vicinity of Santa Cruz and Monterey Counties, California (McShane et al. 2004, p. 3-70).

The current breeding range of the marbled murrelet extends from Bristol Bay, Alaska, south to the Aleutian Archipelago, northeast to Cook Inlet, Kodiak Island, Kenai Peninsula and Prince William Sound, south coastally throughout the Alexander Archipelago of Alaska, and through British Columbia, Washington, Oregon, to northern Monterey Bay in central California. Birds winter throughout the breeding range and also occur in small numbers off southern California.

2.5.6.1 Trend

There are two general approaches that researchers use to assess murrelet population trend: at-sea surveys and population modeling based on demographic data. In general, USFWS assigns greater weight to population trend and status information derived from at-sea surveys than estimates derived from population models because survey information generally provides more reliable estimates of trend and abundance.

Marine Surveys

Researchers from the EM Program detected a statistically significant decline ($p < 0.001$) in the abundance of the population in Conservation Zones 1 through 5 combined, for the 2001-2010 sample period (Falxa et al. 2011). The estimated average annual rate of decline for this period

was 3.7% (95% confidence interval: -4.8% to -2.7%). This rate of annual decline suggests a total population decline of about 29% between 2001 and 2010 (Miller et al. In press).

At the scale of individual conservation zones, the murrelet population declined at an estimated average rate of 7.4% per year (95% confidence interval: -11.2 to -3.5) in Conservation Zone 1 (Falxa et al. 2011, Miller et al. In press). In that same analysis, statistically significant trends were not detected elsewhere at the single-zone scale, but evidence of a declining trend was strong in Zone 2 (6.5% rate of decline, $P = 0.06$). For Washington State (Conservation Zones 1 and 2 combined) there was a 7.31% (standard error = 1.31%) annual rate of decline in murrelet density for the 2001-2010 period (Pearson et al. 2011, p. 10), which equates to a loss of approximately 47% of the murrelet population since 2001.

In Conservation Zone 6, the 2008 population estimate for Conservation Zone 6 suggested a decline of about 55% from the 2007 estimate and a 75% decline from the 2003 estimate (Peery et al. 2008). However, in the most recent population estimate available, the 2009 estimate was similar to estimates from 1999-2003 (Peery and Henry 2010). Peery and Henry (2010) speculated that their 2009 results may have indicated murrelets in central California moved out of the survey area in 2007 and 2008, and then returned in 2009, or the higher estimate in 2009 may have been due to immigration from larger populations to the north. Results from 2010 and 2011 surveys from Zone 6 are currently not available.

Population Models

Prior to the use of survey data to estimate trend, demographic models were more heavily relied upon to generate predictions of trends and extinction probabilities for the murrelet population (Beissinger 1995, USFWS 1997a, Cam et al. 2003, McShane et al. 2004). However, murrelet population models remain useful because they provide insights into the demographic parameters and environmental factors that govern population stability and future extinction risk, including stochastic factors that may alter survival, reproductive, and immigration/emigration rates.

In a report developed for the 5-year Status Review of the Marbled Murrelet in Washington, Oregon, and California (McShane et al. 2004, p. 3-27 to 3-60), computer models were used to forecast 40-year murrelet population trends. A series of female-only, multi-aged, discrete-time stochastic Leslie Matrix population models were developed for each conservation zone to forecast decadal population trends over a 40-year period with extinction probabilities beyond 40 years (to 2100). The authors incorporated available demographic parameters (Table 2-9) for each conservation zone to describe population trends and evaluate extinction probabilities (McShane et al. 2004, p. 3-49).

McShane et al. (2004) used mark-recapture studies conducted in British Columbia by Cam et al. (2003) and Bradley et al. (2004) to estimate annual adult survival and telemetry studies or at-sea survey data to estimate fecundity. Model outputs predicted -3.1% to -4.6% mean annual rates of population change (decline) per decade the first 20 years of model simulations in murrelet Conservation Zones 1 through 5 (McShane et al. 2004, p. 3-52). Simulations for all zone populations predicted declines during the 20 to 40-year forecast, with mean annual rates of -2.1%

to -6.2% per decade (McShane et al. 2004, p. 3-52). These reported rates of decline are similar to the estimates of -4% to -7% per year reported in the Recovery Plan (USFWS 1997a, p. 5).

Table 2-9. Rangewide murrelet demographic parameter values based on four studies all using Leslie Matrix models.

Demographic Parameter	Beissinger 1995	Beissinger and Nur 1997*	Beissinger and Peery (2007)	McShane et al. 2004
Juvenile Ratio (Ŕ)	0.10367	0.124 or 0.131	0.089	0.02 - 0.09
Annual Fecundity	0.11848	0.124 or 0.131	0.06-0.12	-
Nest Success	-	-	0.16-0.43	0.38 - 0.54
Maturation	3	3	3	2 - 5
Estimated Adult Survivorship	85 % – 90%	85 % – 88 %	82 % - 90 %	83 % – 92 %

**In USFWS (1997a).*

McShane et al. (2004, pp. 3-54 to 3-60) modeled population extinction probabilities beyond 40 years under different scenarios for immigration and mortality risk from oil spills and gill nets. Modeled results forecast different times and probabilities for local extirpations, with an extinction risk[9] of 16% and mean population size of 45 individuals in 100 years in the listed range of the species (McShane et al. 2004, pp. 3-58).

2.5.6.2 Threats; including reasons for listing, current rangewide threats

When the murrelet was listed under the Endangered Species Act (USFWS 1992a, pp. 45333-45336) and threats summarized in the Recovery Plan (USFWS 1997a, pp. 43-76), several anthropogenic threats were identified as having caused the dramatic decline in the species.

- habitat destruction and modification in the terrestrial environment from timber harvest and human development caused a severe reduction in the amount of nesting habitat
- unnaturally high levels of predation resulting from forest "edge effects" ;
- the existing regulatory mechanisms, such as land management plans (in 1992), were considered inadequate to ensure protection of the remaining nesting habitat and reestablishment of future nesting habitat; and
- manmade factors such as mortality from oil spills and entanglement in fishing nets used in gill-net fisheries.

There have been changes in the levels of these threats since the 1992 listing (USFWS 2004d, pp. 11-12; USFWS 2009a, pp. 27-67). The regulatory mechanisms implemented since 1992 that affect land management in Washington, Oregon, and California (for example, the NWFP) and new gill-netting regulations in northern California and Washington have reduced the threats to murrelets (USFWS 2004d, pp. 11-12). The threat levels for the other threats identified in 1992

[9] Extinction was defined by McShane et al. (2004, p. 3-58) as any murrelet conservation zone containing less than 30 birds.

listing (USFWS 1992a, p. 45333-45336) including the loss of nesting habitat, predation rates, and mortality risks from oil spills and gill net fisheries (despite the regulatory changes) remained unchanged following USFWS's 2004, 5-year, range-wide status review for the murrelet (USFWS 2004d, pp. 11-12).

However, new threats were identified in USFWS's 2009, 5-year review for the murrelet (USFWS 2009a, pp. 27-67). These new stressors are due to several environmental factors affecting murrelets in the marine environment. These new stressors include:

- Habitat destruction, modification, or curtailment of the marine environmental conditions necessary to support murrelets due to:
    - elevated levels of polychlorinated biphenyls in murrelet prey species;
    - changes in prey abundance and availability;
    - changes in prey quality;
    - harmful algal blooms that produce biotoxins leading to domoic acid and paralytic shellfish poisoning that have caused murrelet mortality; and
    - climate change in the Pacific Northwest.
- Manmade factors that affect the continued existence of the species include:
    - derelict fishing gear leading to mortality from entanglement;
    - energy development projects (wave, tidal, and on-shore wind energy projects) leading to mortality; and
    - disturbance in the marine environment (from exposures to lethal and sub-lethal levels of high underwater sound pressures caused by pile-driving, underwater detonations, and potential disturbance from high vessel traffic;, particularly a factor in Washington State).

The USFWS also believes climate change is likely to further exacerbate some existing threats such as the projected potential for increased habitat loss from drought related fire, mortality, insects and disease, and increases in extreme flooding, landslides and windthrow events in the short-term (10 to 30 years). However, while it appears likely that the murrelet will be adversely affected, we lack adequate information to quantify the magnitude of effects to the species from the climate change projections described above (USFWS 2009a, p. 34).

Several threats to murrelets, present in both the marine and terrestrial environments, have been identified. These threats collectively comprise a suite of environmental stressors that, individually or through interaction, have significantly disrupted or impaired behaviors which are essential to the reproduction or survival of individuals. When combined with the species naturally low reproductive rate, these stressors have led to declines in murrelet abundance, distribution, and reproduction at the population scale within the listed-range.

Detailed discussions of the above-mentioned threats, life-history, biology, and status of the murrelet are presented in the Federal Register, listing the murrelet as a threatened species (USFWS 1992a); the Recovery Plan, Ecology and Conservation of the Marbled Murrelet (Ralph

et al. 1995); the final rule designating murrelet critical habitat (USFWS 1996a); the Evaluation Report in the 5-Year Status Review of the Marbled Murrelet in Washington, Oregon, and California (McShane et al. 2004); the 2004 and 2009, 5-year Reviews for the Marbled Murrelet (USFWS 2004d, USFWS 2009a), and the final rule revising critical habitat for the murrelet (USFWS 2011b).

2.5.7 Conservation

2.5.7.1 Needs

Reestablishing an abundant supply of high quality murrelet nesting habitat is a vital conservation need given the extensive removal during the 20th century. However, there are other conservation imperatives. Foremost among the conservation needs are those in the marine and terrestrial environments to increase murrelet fecundity by increasing the number of breeding adults, improving murrelet nest success (due to low nestling survival and low fledging rates), and reducing anthropogenic stressors that reduce individual fitness[10] or lead to mortality.

The overall reproductive success (fecundity) of murrelets is directly influenced by nest predation rates (reducing nestling survival rates) in the terrestrial environment and an abundant supply of high quality prey in the marine environment during the breeding season (improving potential nestling survival and fledging rates). Anthropogenic stressors affecting murrelet fitness and survival in the marine environment are associated with commercial and tribal gillnets, derelict fishing gear, oil spills, and high underwater sound pressure (energy) levels generated by pile-driving and underwater detonations (that can be lethal or reduce individual fitness).

General criteria for murrelet recovery (delisting) were established at the inception of the Plan and they have not been met. More specific delisting criteria are expected in the future to address population, demographic, and habitat based recovery criteria (USFWS 1997a, p. 114-115). The general criteria include:

- documenting stable or increasing population trends in population size, density, and productivity in four of the six Conservation Zones for a 10-year period and
- implementing management and monitoring strategies in the marine and terrestrial environments to ensure protection of murrelets for at least 50 years.

Thus, increasing murrelet reproductive success and reducing the frequency, magnitude, or duration of any anthropogenic stressor that directly or indirectly affects murrelet fitness or survival in the marine and terrestrial environments are the priority conservation needs of the species. The USFWS estimates recovery of the murrelet will require at least 50 years (USFWS 1997a).

[10] Fitness is measure of the relative capability of individuals within a species to reproduce and pass its' genotype to the next generation.

2.5.7.2 Current Actions

On Federal lands under the NWFP surveys are required for all timber sales that remove murrelet habitat. If habitat outside of mapped LSRs is found to be used by murrelets, then the habitat and recruitment habitat (trees at least 0.5 site potential tree height) within a 0.5-mile radius of the occupied behavior is designated as a new LSR. Timber harvest within LSRs is designed to benefit the development of late-successional conditions, which should improve future conditions of murrelet nesting habitat. Designated LSRs not only protect habitat currently suitable to murrelets (whether occupied or not), but will also develop future suitable habitat in large blocks.

2.5.8 Status of the Species in the Action Area (Oregon)

2.5.8.1 Current status and distribution of the listed species in the State of Oregon including population size, variability, and trend

Conservation Zone 3 (Oregon Coast Range Zone)

In 2011, the population for Conservation Zone 3 was estimated at 7,420 birds (95% confidence interval of 5,200 to 10,100 birds). The population estimate was close to the 2004 estimate and higher than the 5 past years (Crescent Coastal Research 2012b, page 4).

Consulted on effects within this Zone, since October 1, 2003 through May 23, 2012, have included removal of 852 acres of habitat consisting of 702 acres of suitable habitat and 150 acres of younger stands with remnant trees containing nesting structure. Federal NWFP lands effects include the removal of 236 acres of suitable habitat and 30 acres of younger stands with potential remnant trees containing nesting structure. Tribal lands effects include the removal of 276 acres of suitable habitat. Non-Federal lands effects include the removal of 190 acres of suitable habitat and 120 acres of younger stands with potential remnant trees containing nesting structure.

Conservation Zone 4 (Siskiyou Coast Range Zone)

In 2011, the population for Conservation Zone 4 was estimated at 6,465 birds (95% confidence interval of 3,270 to 10,450 birds). This is higher than all other years (Crescent Coastal Research 2012b, page 4). The Oregon portion of Zone 4 is 95.3 percent of the area (Crescent Coastal Research 2012b, page 5).

Consulted on effects within this Zone, since October 1, 2003 through May 23, 2012, have included 1,765 acres of habitat removed. Federal NWFP lands effects include the removal of 793 acres of suitable habitat. Tribal lands effects include the removal of 972 acres of suitable habitat.

2.5.8.2 Current critical habitat designation within the State of Oregon

Currently about 1,369,132 acres of critical habitat are designated with the State of Oregon on Federal, state, county and private lands.

2.5.8.3 Threats specific to Oregon and discussion of threats relative to water quality (toxicity and temperature)

Threats specific to Oregon and discussion of threats relative to water quality include changes in prey quality due to accumulation of toxins in prey species.

3 ENVIRONMENTAL BASELINE

Regulations implementing the Act (50 CFR 402.02) define the environmental baseline as the past and present impacts of all Federal, State, or private actions and other human activities 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 undergone section 7 consultation, and the impacts of State and private actions which are contemporaneous with the consultation in progress.

3.1 Status of the Species in the Action Area

We have provided an analysis of the status of each species and critical habitat within the action area in Chapter 2: Status of the Species. Additionally, the Exposure Analysis section which is included in Chapter 4: Effects Analysis provides detailed information on each species' current exposure to the chemicals addressed in this Biological Opinion.

3.1.1 Oregon Water Quality

USEPA's BE (2008) includes a detailed description of the current status of the environment taken largely from the *Oregon State of the Environment Report 2000* (Risser 2000). In this section, we excerpt relevant portions (quoted below) of the BE. Listed salmon are discussed in the excerpt; however, many of these environmental conditions are similar for bull trout, which is a salmonid, and the other aquatic species analyzed in this BO.

"Past and present actions within the State of Oregon have adversely impacted the habitat of ESA-listed species, bringing about changes in the amount and timing of water flows, reductions in water quality, introduction of nonnative species, and the alteration, elimination, and degradation of riparian and aquatic habitat."

3.1.1.1 Changes in Flows

"The quality and quantity of freshwater habitat in much of Oregon has been adversely impacted by water management activities and declines in water quality. Oregon's currently available water supplies are fully or often over-allocated during low flow months of summer and fall. In the Columbia Plateau ecoregion, less than 20% of instream water rights can expect to receive their full allocation 9 months of the year. In the Willamette Valley and Cascades ecoregions, more than 80% of the instream water rights can expect to receive their full allocation in the winter, but only about 25% in the early fall. Increased demand for water is linked to the projected 34% increase in human population in the State over the next 25 years. Depletion and storage of natural flows have altered natural hydrological cycles in basins occupied by listed species. This may cause juvenile salmon mortality through migration delay resulting from

insufficient flows or habitat blockages, loss of sufficient habitat due to dewatering and blockage, stranding of fish resulting from rapid flow fluctuations, entrainment of juveniles into poorly screened or unscreened diversions, and increased juvenile mortality resulting from increased water temperatures (Spence et al. 1996). Reduced flows have also negatively affected fish habitats because they have resulted in increased deposition of fine sediments in spawning gravels, decreased recruitment of new spawning gravels, and encroachment of riparian and exotic vegetation into spawning and rearing areas. Consumptive use of freshwater in the upper watersheds has reduced freshwater inflow to estuaries by as much as 60 to 80%, thus reducing the natural dilution and flushing of pollutants. Further, some climate models predict 10 to 25 percent reductions in late spring, summer, and early fall runoff amounts in the coming decades."

3.1.1.2 Reductions in Water Quality

"Water quality, as characterized by the Oregon Water Quality Index (Risser 2000), is typically poor, or very poor, except in the Cascades and Coast Range ecoregions and in the Blue Mountain ecoregion during high flows. Most of the waterbodies in Oregon are on the 303(d) list for not meeting temperature standards. Temperature alterations affect aquatic biota metabolism, growth rate, and disease resistance, as well as the timing of adult salmonid migrations, fry emergence, and smoltification. The loss of vegetative shading is the predominant cause of elevated summer water temperatures. Smaller streams with naturally lower temperatures that are critical to maintaining downstream water temperatures are most vulnerable to this effect. The same factors that elevate summer water temperature can decrease winter water temperatures and put salmon at additional risk. Widespread channel widening and reduced base flows further exacerbate seasonal water temperature extremes.

Generally, water quality in Oregon is poor during low flow periods, except in mountainous areas. Areas with excellent or good water quality occur most often in forested uplands. Poor or very poor water quality occurs most often in non-forested lowlands where land has been converted to agricultural and urban uses. Most ecoregions include some rivers and streams with excellent water quality and others with very poor water quality. Only the Cascades ecoregion has excellent water quality overall as shown by average Oregon Water Quality Index measurements. The Willamette Valley, Columbia Plateau, Northern Basin and Range, and the southern end of the Eastern Cascade Slope ecoregions have poor water quality indices.

Pollutants also degrade water quality. The effects of pesticides and fertilizers, especially nitrates, on water supplies and aquatic habitats are a significant concern. Water pollution of almost every category is increasing, as are hazardous waste emissions, air pollution, toxic releases, and waste generation. Sedimentation from logging, mining, urban development, and agriculture is a primary cause of salmon habitat degradation. In general, the effects of sedimentation on salmon are well documented and include (1) clogging and abrasion of gills and other respiratory surfaces, (2) adhering to the chorion of eggs, (3) providing conditions conducive to entry and persistence of disease-related organisms, (4) inducing behavioral modifications, (5) entombing early life stages, (6) altering water chemistry by the absorption of chemicals, (7) affecting usable habitat by scouring and filling pools and riffles and changing bedload composition, (8) reducing photosynthetic growth and primary production, and (9) reducing intergravel permeability and DO (Spence et al. 1996)."

3.1.1.3 Introduction of Nonnative Species

"More than 32 species of freshwater fish have been introduced into Oregon, and are now self-sustaining, making up approximately one-third of Oregon's freshwater fish fauna. Introduced species are frequently predators on native species, compete for food resources, and alter freshwater habitats. In 1998, introduced species were found to represent 5% of the number of species found in the upper Willamette River, but accounted for 60% of the observed species in the lower river near Portland."

3.1.1.4 Alteration, Elimination, and Degradation of Riparian and Aquatic Habitat

"Oregon contains approximately 114,500 miles of rivers and streams. No Statewide measurements exist of the area of riparian vegetation, although some estimates have been made for more localized regions. Using the conservative estimate of a 100-yard riparian corridor on each side of the stream, the total area of riparian habitats for flowing water in Oregon may be 22,900 square miles. That is equal to approximately 15% of the total area of the State. Healthy riparian areas retain the structure and function of natural landscapes as they were before the intensive land use and land conversion that have occurred over the past 150 to 200 years. However, land use activities have reduced the numbers of large trees, the amount of closed-canopy forests, and the proportion of older forests in riparian areas. In western Oregon, riparian plant communities have been altered along almost all streams and rivers.

In the western Cascades, Willamette Valley, Coast Range, and Klamath Mountains, riparian areas on privately owned land are dominated by younger forests because of timber harvest, whereas riparian areas on public lands have more mature conifers. Old coniferous forests now account for approximately 20% of the riparian forests in the Cascades, but only 3% in the Coast Range. Older forests historically occurred along most of the McKenzie River, but now account for less than 15% of its riparian forests. Along the mainstem of the upper Willamette River, channel complexity has been reduced by 80% and the total area of riparian forest has been reduced by more than 80% since the 1850s. Downstream portions of the Willamette River have experienced little channel change, but more than 80% of the historical riparian forest has been lost.

Beginning in the early 1800s, riparian areas in eastern and southern Oregon were extensively changed by such activities as trapping beaver, logging, mining, livestock grazing, agriculture, and associated water diversion projects. Very little of the once-extensive riparian vegetation remains to maintain water quality and provide habitats for threatened salmon. Dams have affected flow, sedimentation, and gravel patterns, which in turn have diminished regeneration and natural succession of riparian vegetation along downstream rivers. Introduced plant species pose a risk to some riparian habitat by dominating local habitats and reducing the diversity of native species. Improper grazing in riparian areas is another significant threat.

Human activities have had vast effects on the salmon populations in the Willamette River Basin. First, the Willamette River, once a highly braided river system, has been dramatically simplified through channelization, dredging, and other activities that have reduced rearing habitat (i.e., stream shoreline) by as much as 75%. In addition, the construction of 37 dams in the basin has blocked access to over 375 miles of stream and river spawning habitat. The dams also alter the

temperature regime of the Willamette and its tributaries, affecting the timing of development of naturally spawned eggs and fry. Water quality is also affected by development and other economic activities. Agricultural and urban land uses on the valley floor, as well as timber harvesting in the Cascade and Coast Ranges, contribute to increased erosion and sediment load in Willamette River Basin streams and rivers. Finally, since at least the 1920s, the Lower Willamette River has suffered municipal and industrial pollution."

4 EFFECTS OF THE ACTION

4.1 Exposure Analysis

4.1.1 Determinations

There are several steps in section 7 consultation, the first being a decision as to whether the proposed action may result in any effects to listed species or designated critical habitat. If it is determined that the action "may affect" a listed threatened or endangered species, the next step is to determine if the effect of the action is insignificant, discountable, or entirely beneficial, in which case it is not likely to adversely affect the species. Alternatively, if the action has effects that are beyond insignificant, discountable, or entirely beneficial, the action is likely to adversely affect (LAA) the listed species. For the purposes of this consultation, we consider this process of determining whether an action may affect a listed species similar to an ecological risk assessment.

USEPA (1998, p. 1) describes ecological risk assessment as a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. Ecological risk assessment is a process used to systematically evaluate and organize data, information, assumptions, and uncertainties in order to help understand and predict the relationships between stressors and ecological effects in a way that is useful for environmental decision making. The basic framework for ecological risk assessment includes problem formulation, analysis, and risk characterization (USEPA 1998, p. 3). The analysis stage of the assessment includes characterization of exposure and characterization of ecological effects. There can be varying levels of structure in the ecological risk assessment process and our approach uses ecological risk assessment concepts without a strict adoption of the formal framework.

The USEPA (2008) used a variety of risk assessment models to form the basis of their NLAA or LAA determinations in their BE. Within the description of our methodology (Appendix 1), we identify numerous uncertainties associated with the models used to make these determinations. The defined threshold for whether a listed species is affected is "any effects" (USFWS and NMFS 1998). Given this low threshold and the uncertainties with the risk assessment models, we are using an alternative approach to determinations of LAA or NLAA based on exposure potential.

Our concurrence or non-concurrence with USEPA's determination of NLAA is based on the likelihood that a particular species will be exposed to the subject chemical via anthropogenic sources (point and nonpoint sources) and adverse effects from aqueous or dietary exposure and

accumulation to the chemical. In areas where there is very low potential for co-occurrence of a listed species with the chemicals considered in this consultation, then approval of the criteria is not likely to adversely affect the listed species because the possibility the species will be exposed to the chemical is low and effects are considered discountable. The potential for co-occurrence is based on review of National Pollutant Discharge Elimination System (NPDES) permits, abandoned or active mines, and land-use activities that are nonpoint sources of pollutants. Alternatively, where there is potential for exposure to anthropogenic sources of a chemical, then effects from exposure to the chemical are analyzed to determine if the action is likely to adversely affect the listed species. Although potential sources of exposure could change in the future, conditions in remote parts of Oregon (especially eastern Oregon) are not anticipated to change appreciably in the foreseeable future. However, if conditions do change such that there is an increase in potential exposure, this constitutes new information which would trigger reinitiation of consultation. A conceptual flowchart of this approach is diagramed in Figure 4-1. Therefore, a NLAA/LAA is based on co-occurrence and adverse effects associated with criteria concentration.

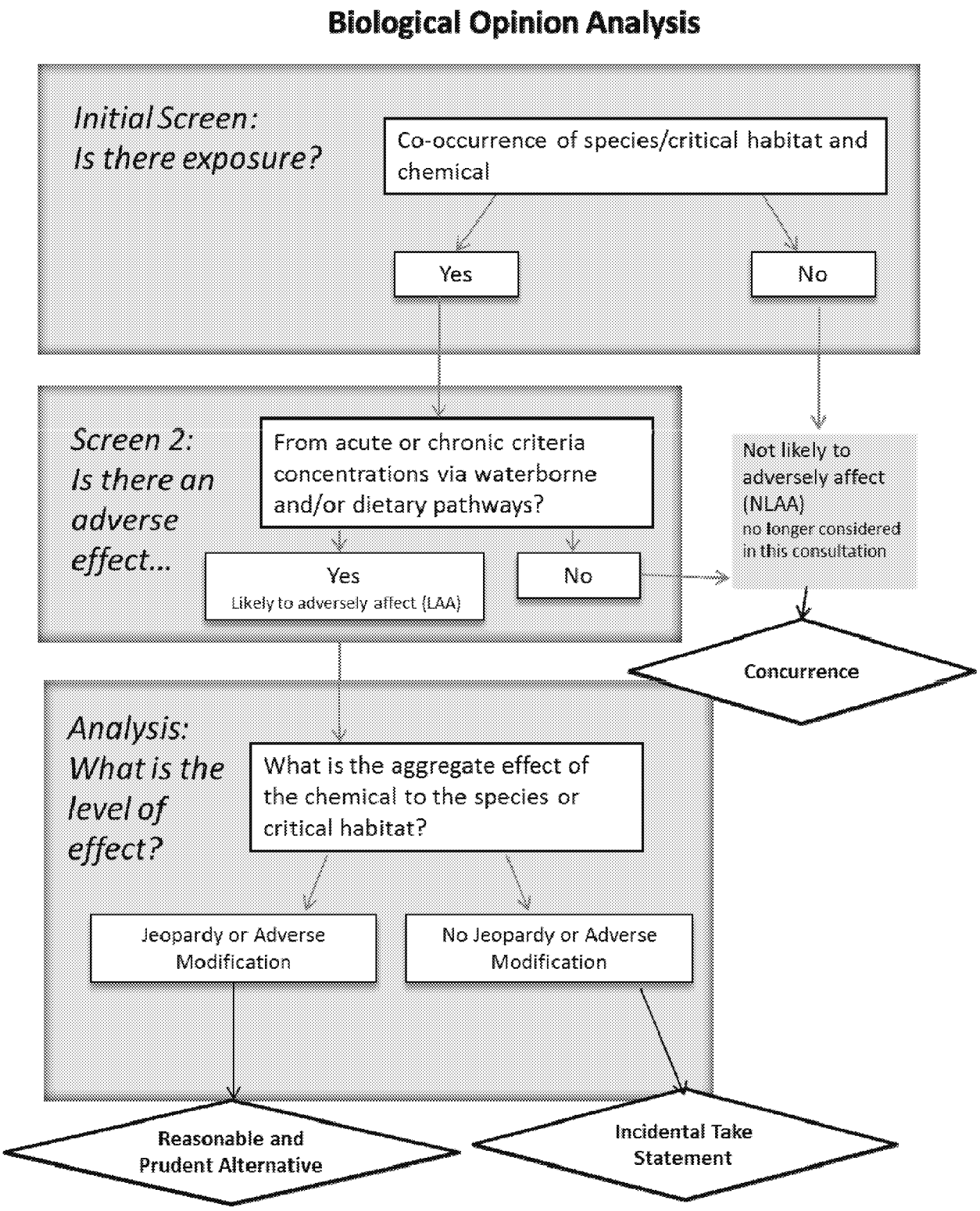


Figure 4-1. Flowchart describing the steps used in Biological Opinion determinations for this consultation.

4.1.1.1 Approach

As described above, where there is very low potential for co-occurrence of listed species with anthropogenic sources of the chemicals in this consultation, we consider the potential for adverse effects to be discountable and USEPA's approval of the criteria to be not likely to adversely affect the particular listed species. Where there are anthropogenic sources of a chemical in listed species habitat, we assume exposure to criteria chemicals at the Criterion Maximum Concentration (CMC) and Criterion Continuous Concentration (CCC). The CMC and CCC represent the basis for administering water quality programs under the water quality-based approach to pollution control, including monitoring to determine whether waters are attaining designated uses, 303(d) listing of impaired waters, and the development and implementation of TMDLs. Although we assume exposure to criteria chemicals at the CMC and CCC, we acknowledge that this is the upper end of the concentration a listed species would experience given that NPDES permit limits are based on low flow years and the CMC and CCC have temporal limits (not to be exceeded more than once every 3 years) associated with them. However, information is not available on actual concentrations of criteria chemicals in the aquatic habitats of listed species.

In Oregon, there are vast remote areas void of concentrated human population centers and anthropogenic sources of the chemicals considered in this consultation. In these areas, if a concentration of a chemical is at or above its criterion, then we assume it is naturally occurring. Several of the chemicals in this consultation are naturally-occurring elements (e.g., arsenic, copper, selenium, and silver) and background concentrations (i.e., no anthropogenic inputs) can exceed proposed water quality criteria. Listed species inhabiting waters where naturally-occurring elements are at elevated concentrations with no known anthropogenic source are considered to have adapted to withstand higher than criteria concentrations without adverse effects (e.g., Borax Lake arsenic concentrations). Consequently, we do not consider this type of situation to adversely affect listed threatened or endangered species.

Once we identify which listed species are likely to be exposed to criteria chemicals associated with an action (and for which a determination of LAA is appropriate), the next step of our analysis is to examine the best scientific and commercial data available to determine whether and how those listed species are likely to respond given their exposure. Depending on the level of response, both determinations of LAA or NLAA are possible in this step as well.

This exposure analysis explains: (1) how we determined distribution of the listed species, (2) how we determined the areas affected by the consultation chemicals, and (3) the degree of exposure likely from each chemical for each species. To determine if there is co-occurrence of a chemical and listed species we evaluated specific sources regulated by the water quality criteria, chemical-specific uses and listed species range and life history. This section presents our analysis for each specific chemical and species considered in the consultation.

Site-specific exposure assessments were conducted whenever possible. For example, in habitats of species with more limited distributions such as Oregon chub and Lost River and shortnose suckers, point source permits are easily identified and land uses known. In contrast, bull trout

has more widely distributed habitat with abundant point sources where it was not possible to identify all known specific point sources and thus site-specific exposure assessments were not possible. In this case, all known potential point sources of consultation chemicals that could be identified within 2 miles of listed species habitat or within 0.5 miles overland were identified and included in our analysis. Two miles was used as the best estimate of the exposure distance at which most toxins would be sufficiently diluted to reduce potential adverse effects. Distance measurements were conducted using ArcGIS. NPDES permits downstream of listed species habitat were not considered to be a source for species other than bull trout. Identification of nonpoint sources of chemicals was based on adjacent land-use activities (e.g., forestry, roads, agriculture) in proximity to the listed species habitat. Land-use activities were identified by aerial imagery, State and Federal agency reports (e.g., floodplain studies and annual reports), and communication with various State and local governmental staffs. On occasion, nonpoint sources were treated differently for different species due to differing habitat types. For example, Oregon chub ponds are typically isolated for a large part of the year and would not have the same dilution as a bull trout stream. Road use activities in Oregon chub habitat included major highways within 500 feet.

Appendix 1 presents the detailed method for evaluating contaminant sources used by the USFWS to complete our exposure analysis and effect determinations to inform the list of species and critical habitats for which ESA section 7(a)(2) determinations are made herein.

4.1.2 Species-Specific Exposure Analyses

We present both species- and chemical-specific exposure analyses in this consultation. The species-specific exposure analysis leading to NLAA determinations is presented followed by the species with LAA determinations. The chemical-specific exposure analyses are then conducted for those LAA species. This eliminates further analyses for those species and chemicals where the potential for adverse effects is determined to be discountable or insignificant due to lack of exposure to chemicals which are the subject of this consultation.

4.1.2.1 Not Likely to Adversely Affect (NLAA) Determinations

Birds

Short-tailed albatross

The short-tailed albatross is a large pelagic bird which is very few in number today but distributed widely throughout its historical foraging range of the temperate and subarctic North Pacific Ocean. The albatross is often found close to the U.S. coast, including the coastal waters of Oregon (USFWS 2000). Short-tailed albatross may be relatively common nearshore, but only where upwelling "hotspots" occur in proximity to the coast (Piatt et al. 2006, p. 393). In addition, this species is associated with continental shelf breaks and slope regions of the northwest Pacific Ocean (Piatt et al. 2006, Suryan et al. 2006), which occur well offshore in Oregon.

Breeding colonies of short-tailed albatross are limited to two Japanese islands, Torishima and Minami-kojima. At Torishima, birds arrive at the breeding colony in October and begin nest

building. Egg-laying begins in late October and continues through late November. The female lays a single egg, incubation involves both parents and lasts for 64 to 65 days, eggs hatch in late December and January, and by late May or early June, the chicks are almost full grown and the adults begin abandoning their nests (Hasegawa and DeGange 1982, as cited in USFWS 2008h). The chicks fledge soon after the adults leave the colony; by mid-July the breeding colony is totally deserted (Austin 1949, as cited in USFWS 2008h). Nonbreeders and failed breeders disperse from the breeding colony in late winter through spring (Hasegawa and DeGange 1982, as cited in USFWS 2008h). There is no detailed information on breeding activities on Minami-kojima, but it is likely to be similar to that on Torishima.

The likelihood that short-tailed albatross will be exposed to chemicals in this consultation in Oregon is discountable for two reasons. First, short-tailed albatross rarely come into shore where any point sources would be discharging and secondly, the albatross are only in Oregon during the non-breeding season, less than 6 months of the year. Consequently, USEPA determined that approval of the chemical criteria in this consultation is NLAA short-tailed albatross; we concur with this determination.

Western snowy plover

Plovers typically forage for small invertebrates in wet or dry beach sand, tide-cast kelp, low foredune vegetation, and near water seeps in salt pans. Prey species include mole crabs (*Emerita analoga*), crabs (*Pachygrapsus crassipes*), polychaetes (Neridae, *Lumbrineris zonata*, *Polydora socialis*, *Scoloplos acmaceps*), amphipods (*Corophium* spp., *Ampithoe* spp., *Allorchestes angustus*, and sand hoppers [Orchestoidea]), tanadacians (*Leptochelia dubia*), flies (Ephydridae, Dolichopodidae), beetles (Carabidae, Buprestidae, Tenebrionidae), clams (*Nutricola* spp.[formerly *Transennella*]), and ostracods (Reeder 1951, pp. 43-44; Tucker and Powell 1999, pp. 45-46).

Based on the bioaccumulation potential for each chemical (as discussed in Sections 4.3 and 4.5.4.1), selenium is the only chemical which presents potential adverse effects to western snowy plover through dietary exposure. However, snowy plover uptake of selenium from aquatic sources will be diluted due to their consumption of terrestrial invertebrates. In addition, studies show that snowy plovers are not as sensitive to adverse effects of selenium as compared to other shorebirds. Consequently, the western snowy plover is considered not likely to be adversely affected by the approval of the Oregon proposed selenium criterion of 5 µg/L (ppb).

The PCEs for the western snowy plover critical habitat are described in the final rule for critical habitat (USFWS 2012a). The PCE #2 describes habitat that supports small invertebrates, such as crabs, worms, flies, beetles, spiders, sand hoppers, clams, and ostracods, as a characteristic of habitat necessary to sustain the essential life history functions of the species. Given the variety of prey species consumed by western snowy plover we expect that there should not be any meaningful reduction in prey availability due to exposure to selenium. Therefore, exposure to selenium is NLAA to western snowy plover critical habitat (i.e., NLAA).

Fish and Invertebrates

Other than bull trout, all threatened and endangered aquatic species under the authority of USFWS in Oregon reside strictly in freshwater. Therefore, saltwater criteria will have no effect on Borax Lake chub, Hutton tui chub, Foskett speckled dace, Oregon chub, Warner sucker, Lost River sucker, shortnose sucker, Modoc sucker, and vernal pool fairy shrimp.

Lahontan cutthroat trout

Lahontan cutthroat trout habitat in Oregon is located within southeastern Oregon in Harney and Malheur Counties. The Lahontan cutthroat trout occurs in the following streams in southeastern Oregon: Willow Creek, Whitehorse Creek, Little Whitehorse Creek, Doolittle Creek, Fifteen Mile Creek (from the Coyote Lake Basin) and Indian, Sage, and Line Canyon Creeks, tributaries of McDermitt Creek in the Quinn River Basin (Nevada) (USFWS 2008e). There are no point source discharges identified within Lahontan cutthroat trout habitat in Oregon.

In streams inhabited by Lahontan cutthroat trout, there are several waterbodies included on the State's 303(d) list. Willow Creek is listed for dissolved oxygen and temperature under category 4A (water-quality limited, TMDL approved). Willow Creek is also listed as category 3B (potential concern) for arsenic and beryllium. Whitehorse Creek is listed as a category 3B stream for phosphate phosphorus. Indian and Sage Creeks are listed as category 5 (water-quality limited, TMDL needed) for temperature and a second, distinct Willow Creek is listed as category 4A for temperature. Denio and Van Horn Creeks are also listed under category 4A for temperature.

Although there are some stream reaches in Lahontan cutthroat trout habitat that are on the 303(d) list as either potential concern or water-quality limited, all but one are for chemicals outside the scope of this consultation. Willow Creek is listed as category 3B by ODEQ for arsenic, the one chemical which is considered in this Biological Opinion, based on 2 of 16 samples exceeding the criterion. Because there are no permits or mines in Lahontan cutthroat trout habitat, arsenic concentrations in these waters are considered to be the result of natural sources. Areas that are geothermally active can have high natural background concentrations of arsenic and waters in the Alvord Lake Basin show considerable variability in concentrations of arsenic with springs being extremely elevated (Whanger et al. 1977, p. 141). The Alvord Lake Basin has numerous geothermally active springs such as those associated with Borax Lake, known for its high arsenic concentrations. Whanger et al. 1977 (p. 139) also state that elevated arsenic levels in Oregon waters could be due to volcanic deposits. In addition, although beryllium is not a subject of this consultation, it is also listed as a potential concern for human health, not aquatic life.

According to the search of the MILO-2 database for metal mines that are current or past producers, there is one uranium and mercury mine site that is relatively close to Pike Creek (occupied by Lahontan cutthroat trout). This area has since been withdrawn from mineral development via the Steens Cooperative Protection Act of 2000. Further from any streams containing Lahontan cutthroat trout, two other mercury mines are identified, both categorized as small mines with underground work.

There are two exploration mining permits for uranium filed with the Oregon DOGAMI in the McDermitt Creek Basin, identified at 0.5 and 1.0 acres and limited to drilling. If this or a future operation involves chemical processing, it falls under OAR Division 37 for Chemical Process Mining requiring a lengthy permitting process. OAR Division 37 requires that chemical process mines provide an Operating Plan (632-037-0060), which must include monitoring, stable storage plans, and a fish and wildlife protection and mitigation plan. In addition, OAR Division 37 requires a Reclamation and Closure Plan (632-037-0070), which includes appropriate isolation and removal of waste material. If this mine expands to become operational, then there is the potential for an NPDES permit for metals to be issued.

Based on the remoteness of the Lahontan cutthroat trout distribution and the lack of any current point source discharges and nonpoint sources, there is very limited opportunity for Lahontan cutthroat trout to be exposed to chemicals in this consultation other than from natural sources.

Although cattle ranching is a primary land-use activity in the region of Lahontan cutthroat trout, several studies indicate that this would not produce a significant source of ammonia in nearby streams (Doran et al. 1981, Gary et al. 1983, Schepers and Francis 1982, Tiedemann et al. 1989). Gary et al. (1983, pp. 126-127) report that in rural areas, detectable ammonia nitrogen in stream water generally originates from leaking septic tanks, WWTP effluent runoff from barnyards and feedlots, and runoff from fertilized pastures and croplands. In their study, they found that manure deposits beyond the 3-meter limit would most likely never reach the stream except during major floods (Gary et al. 1983, p. 126). In addition, researchers found that in only one of four cattle grazing periods did the presence of cattle appear to increase the ammonia nitrogen concentration significantly (Gary et al. 1983, p. 127). Investigators have stated that sporadic additions of cattle manure and urine to stream water may go undetected because of almost immediate dilution (Gary et al. 1983, p. 126, Demal 1983, p. 31).

We conclude that any potential effects resulting from exposure to ammonia from cattle grazing will be discountable and insignificant. While livestock grazing is known to occur along many streams, Doran et al. (1981, p. 168) found that the levels of nitrogen added to grazing lands in Nebraska and Oklahoma through precipitation far exceeded those found in runoff from grazing land. Thus, other sources of ammonia added to the river basin by grazing are likely to be insignificant.

Cattle stocking rates in eastern Oregon are very low. In review of two USFWS biological opinions with grazing allotments for Lahontan cutthroat trout habitat areas (USFWS 1992b and c), the stocking rate was determined to be approximately one animal per 100 acres for 2 months or 0.01 animal units per month (AUM). In addition, topography in some Lahontan cutthroat trout streams is extremely steep (e.g., Steens Mountains) making it difficult for livestock to access streams containing Lahontan cutthroat trout. Finally, grazing units on Whitehorse and Willow Creeks in the Coyote Basin have large sections of streams that have been fenced to exclude cattle with only limited gaps left to allow for livestock watering. Consequently, we concur with USEPA's determination that the freshwater ammonia acute and chronic criteria are NLAA Lahontan cutthroat trout.

Borax Lake chub

Borax Lake chub is restricted to Borax Lake located in Harney County, Oregon. Borax Lake is privately owned by TNC and surrounded by USBLM public land designated as an Area of Environmental Concern.

No discharges are permitted in Borax Lake and there are no listings on the Oregon Integrated Report Database under any of the categories of water-quality limited waterbodies. There are no mines identified in the vicinity of Borax Lake.

Borax Lake is protected from point and nonpoint sources of pollutants and conditions are not expected to change in the foreseeable future. Any chemicals in the lake that are at or above criteria concentrations are naturally occurring. Therefore, there is no potential for exposure and no effect to the Borax Lake chub or its critical habitat from any of the chemicals in this consultation.

Hutton tui chub

The Hutton tui chub distribution includes two springs in the Alkali Subbasin, Hutton Spring and unnamed spring located 3/8 of a mile southeast of Hutton Spring. Hutton Spring has been diked and has a pool approximately 36 m^2 averaging 1.2-m deep (Scheerer and Jacobs 2007, p. 6) and is surrounded by bulrushes (*Scirpus* sp.). The unnamed spring is within 2,000 feet of Hutton Spring and significantly smaller in size with a diameter of approximately 11 feet and a depth of 2.4 feet.

Hutton Spring is privately owned and the habitat is in good condition primarily due to the landowner's stewardship which includes fenced exclusion of cattle. There are currently no permitted discharges into this wetland, nor is it listed under any of the categories of water-quality limited, nor are there any mines identified in the vicinity.

Alkali Lake Disposal Site is located about 1.5 miles south of Hutton Spring (Hart Crowser 2005, p. 10). According to an ecological risk assessment conducted by Hart Crowser (Hart Crowser 2005, p. 1), the disposal site of approximately 450 acres is where waste from the production of herbicides was buried between 1969 and 1971. The primary contaminants in soil and groundwater are phenols, chlorinated herbicides, dioxins, and furans (Hart Crowser 2005, p. 1). Long-term groundwater monitoring wells are in place and although there is a well-documented contaminant plume extending about 2,000 feet, the plume has not expanded during 10 years of relatively recent monitoring (Hart Crowser 2005, p. 1). In addition, no elevated contaminants were found in biota collected from the site. Numerous rounds of sampling have determined that Hutton Spring is well outside the extent of contamination from the Alkali Lake site (Hart Crowser 2005, p. 10).

Given the remote location of the springs and the lack of human population and roads in the area, we expect that the Hutton tui chub will only be exposed to the chemicals considered in this consultation if they are naturally occurring. Cattle are excluded from Hutton Spring by fencing and based on information presented above under Lahontan cutthroat trout, there is minimal chance that cattle use could create elevated ammonia concentrations from nonpoint sources.

Therefore, any exceedances of criteria for chemicals in this consultation would be from naturally-occurring metals as there are no known sources of contamination in the area.

Conditions in Hutton tui chub habitat are not expected to change in the foreseeable future (or during the life of this consultation, 25 years). Due to the lack of potential exposure from anything other than natural background sources, we concur with USEPS's determine that the action of approving criteria for all chemicals would be NLAA Hutton tui chub.

Foskett speckled dace

Foskett speckled dace exist in one spring (Foskett Spring) of those remaining within the vicinity of Coleman Lake. The USBLM acquired Foskett Spring and the surrounding 65 ha in 1987; 28 ha are fenced to exclude cattle. There are currently no permitted discharges into Foskett Spring, nor is it listed under any of the categories of water-quality limited waterbodies. Furthermore, no mines are identified in the vicinity of Foskett speckled dace habitat.

Given the remote location of the spring, USBLM ownership of the surrounding land, and the lack of point or nonpoint sources there is minimal probability of exposure to any of the subject chemicals in this consultation other than at natural background concentrations. There is no reason to expect that the conditions of this area will change within the life of this Biological Opinion. Due to the lack of potential exposure from any of the chemicals in this consultation, the potential for any adverse effects is considered discountable and we concur with USEPA's determination that this action is NLAA Foskett specked dace.

Warner sucker

When adequate water is present, distribution of the Warner sucker throughout the Warner Valley can include all the lakes, sloughs, and potholes. Resident populations of Warner sucker are found in Honey, Snyder, Twentymile and Twelvemile Creeks. The distribution appears to be discontinuous and centered around low gradient areas that form deep pools with protective cover.

There are no NPDES permits located in the Warner Valley. The ODEQ 2010 Integrated Report (ODEQ 2011b) lists the following streams for the Warner Lakes Basin that support Warner sucker as water-quality limited or of potential concern: Honey Creek for pH category 5 in summer, phosphate phosphorus category 3B in summer and temperature category 5 year-round; Little Honey Creek is listed as limited for temperature in summer; Snyder Creek category 5 for temperature year-round; Twelvemile Creek category 3B for alkalinity, beryllium, manganese and thallium, and category 5 for arsenic, silver and temperature; Twentymile Creek is listed for arsenic, dissolved oxygen, silver and temperature category 5 year-round, and beryllium, thallium and phosphate phosphorus category 3B. Beryllium, thallium, arsenic, manganese and silver are expected to be at natural occurring concentrations since there are no obvious anthropogenic sources in the Warner Valley. No mines are identified in the vicinity of Warner sucker habitat and the primary land use in the area is cattle ranching.

A majority of Warner sucker habitat is located in waterways managed by the Lakeview District of USBLM. With few exceptions, designated Warner sucker critical habitat is excluded from grazing and other land use authorizations analyzed in the Lakeview Resource Area Resource

Management Plan (USBLM 2003). The one exception is at the Deppy Creek/ Honey Creek confluence where a water gap allows stock access. The other exception is in the 0207 allotment on Twentymile Creek. This area is not occupied by Warner sucker and is an intermittent, rock-armored channel. Both these areas are covered by the programmatic opinion on grazing issued by USFWS in 1997 (USFWS 1997b). Although there is a slight chance that fish could be exposed to ammonia concentrations from livestock at the confluence of Deppy and Honey Creeks, physical disturbance from cattle entering the stream would eclipse any potential effects from elevated ammonia in this area. Given the remote location of the Warner Valley, USBLM ownership of the inhabited streams, and the lack of point or nonpoint sources, there is minimal probability of exposure to copper or ammonia other than at natural background concentrations. Due to the lack of potential exposure from anthropogenic sources of copper and ammonia, we concur with USEPA's determination that approval of copper and ammonia acute and chronic freshwater criteria is NLAA Warner sucker or its critical habitat because any potential effects will be discountable.

Modoc sucker

The present distribution of Modoc sucker in Oregon is limited to Thomas Creek. The entire known occupied habitat in Thomas Creek is in the Fremont National Forest and intermingled private lands above RM 19.6. Although it is possible that a few Modoc suckers occur downstream on private lands below the National Forest boundary, because Thomas Creek quickly takes on the warm water characteristics unlike what the species normally occupies, and is lacking habitat diversity once it reaches private lands, few Modoc suckers are likely to be present. Consequently, we consider the primary habitat for the species in Oregon to be Thomas Creek above RM 19.6.

Although Thomas Creek and East Branch Thomas Creek (at RM 9.0) are the receiving streams for the town of Lakeview's NPDES-permitted domestic sewage wastewater, the discharge is nearly 10 miles below the most downstream known location of the Modoc sucker. Thomas Creek is listed on ODEQ 2010 Integrated Report database as category 3B for phosphate phosphorus and category 5 for iron, dissolved oxygen, and temperature. All four parameters are listed for RM 0 to 35.9, which includes all of the known distribution of the Modoc sucker. There are no mines identified in the vicinity of Modoc sucker habitat.

Because the known distribution of the Modoc sucker is located in lands managed by the U.S. Forest Service or on associated private grazing lands located about 10 miles upstream of developed areas near Lakeview where any discharges containing copper are most likely to occur, we do not anticipate Modoc suckers would be exposed to effluent containing copper. There may be cattle in the area, yet we do not anticipate significant concentrations of ammonia given the low stocking rates and the results of studies presented previously (see Lahontan cutthroat trout exposure). Consequently, because there are no known anthropogenic sources of copper in Modoc sucker habitat and the potential for adverse effects is considered insignificant, we concur with USEPA's determination of NLAA for the proposed approval of acute and chronic freshwater criteria for copper. Furthermore, although there may be some exposure to ammonia for Modoc suckers inhabiting private grazing lands, the information presented previously for the Lahontan cutthroat trout exposure evaluation shows that any potential exposure would be

discountable. Therefore, we concur with USEPA's determination that approval of acute and chronic freshwater criteria for ammonia is NLAA for Lahontan cutthroat trout and its critical habitat.

4.1.2.2 Likely to Adversely Affect Determinations

Birds

Marbled murrelet

Other than the nesting season, marbled murrelets spend the majority of time in saltwater usually within 5 miles (8 km) of shore. Courtship, foraging, loafing, molting, and preening occur in marine waters. Marbled murrelets usually feed in shallow, near-shore water less than 98 feet (30 m) deep (Huff et al. 2006) but can be found farther from shore in the non-breeding season (Strachan et al. 1995). Marbled murrelets are considered year-round residents in Oregon (McShane et al. 2004).

Marbled murrelets are opportunistic feeders utilizing prey of diverse sizes and species. They feed primarily on fish and invertebrates in marine waters although they have also been detected on rivers and inland lakes (Carter and Sealy 1986, USFWS 1992a). In general, small schooling fish and large pelagic crustaceans are the main prey items. Marbled murrelets are able to shift their diet throughout the year and over years in response to prey availability (Becker et al. 2007). Freshwater prey appears to be important to some individuals during several weeks in summer. Because marbled murrelets spend the majority of time in marine environments, the saltwater criterion is applicable. Given their life history patterns, only bioaccumulative chemicals in this consultation have the potential to have any adverse effects on the marbled murrelets (see Effects of Selenium on Listed Species and Critical Habitats in Section 4.5.4.)

Fish and Invertebrates

Vernal pool fairy shrimp

In Oregon, vernal pool fairy shrimp are thought to have occupied all vernal pool systems that occurred in Jackson County's Agate Desert of the Rogue River Valley prior to conversion to agricultural lands, military bases, water treatment facilities, mining operations, and industrial, commercial, and residential developments (USFWS 2005c). The former vernal pool systems roughly occurred where soils are mapped as the Agate-Winlo soils series in the vicinity of northern Medford, White City, and along the Rogue River between White City and Shady Cove (Petersen 2006). The current distribution of vernal pool fairy shrimp in Oregon is limited to 7,574 acres within the Agate Desert in Jackson County (USFWS 2006, p. 7, 118) near the towns of White City and Eagle Point. The entire known habitat occurs on a system of intact to degraded mounded prairie uplands and intervening, seasonally inundated depressions or swales underlain by a hardpan that is from 2 to 22 inches below the surface. Together these features constitute vernal pool complexes. The vernal pool complex, often adjacent to industrial, agricultural, some rural, and some recreational areas, is typically degraded by past leveling, dumping, recreational or ditching practices.

In most vernal pool regions, many vernal pool fairy shrimp occurrences are often located on remnant vernal pool habitat adjacent to: (1) oak woodland habitats, (2) residential, industrial or commercial development, (3) infrastructure, or (4) deep-ripped vineyards or orchards (California Natural Diversity Database 2007). Even within mapped vernal pool fairy shrimp critical habitat vernal pools and vernal pool fairy shrimp occupancy of these pools is sporadic (Borgias 2006). The vernal pools on the Agate Desert Preserve in Oregon are predominantly threatened by development and the indirect effects of adjacent land use, including alteration of hydrology (USFWS 2006).

We do not consider ODEQ water-quality limited streams (303[d] listed) contribute to vernal pools because a vernal pool complex is a system that is influenced primarily by precipitation and shallow sub-soil water sources. Discharge permits (NPDES and WPCF) were searched via ODEQ databases (ODEQ 2011a). Currently 149 facilities with permits were found in the range of vernal pool fairy shrimp. However, because vernal pools are ephemeral and are filled primarily by precipitation, they do not qualify as receiving waters for NPDES permits. There are no mines listed in the MILO-2 database nearby vernal pool fairy shrimp habitat.

A large amount of vernal pool complex habitat (4,586 acres) is located within the vicinity of White City, which has had a history of military installations, timber processing plants, and industrial facilities. These facilities contain potential sources for several of the chemicals considered in this consultation.

Some amount of contamination may be present in the soils on 3 acres adjacent to the former Camp White military motor pool and dumping site in use from 1942 to 1952; (USFWS 2002g, pp. 3 and 6) (Figure 4-2). These chemicals could potentially have entered the adjacent vernal pool complex from runoff (TDC Environmental, LLC 2004) while Camp White was in operation in the 1940s and 1950s. The White Mountain Middle School was constructed on the former motor pool site in 2003 and recently the vernal pool and swale topography adjacent to the former motor pool site was restored and is protected. The federally-listed endangered plant, large-flowered woolly meadowfoam (*Limnanthes floccosa* ssp. *grandiflora*), has been documented throughout this restored site. All observed pools in the area do support aquatic invertebrates and algal matter, indicating that any chemical potentially affecting algal growth occurs in low concentrations. Although fairy shrimp presence at the site is confirmed, surveys have not been conducted since 2002.

In the next stage of White City development, lumber processing plants arrived and as lumber was brought to mills, it was treated and distributed to outside areas. After wood was processed, the remaining bark chips and leftover wood scraps were mixed with soils and buried in adjacent landfills (Borgias et al. 2009). Vernal pools adjacent to log deck debris landfills have visibly-altered hydrology as a result of elevated topography and fill leachate. Approximately 220 acres were used as log deck debris landfills in the White City area. Historic timber companies in the area commonly used pentachlorophenol. We are not aware that any chromated copper arsenate was used or buried in the landfill near the vernal pool complex (Vince Oredson, Wildlife Manager, Denman Wildlife Area, Oregon Department of Fish and Wildlife, White City, Oregon, pers. comm., 2011).

A 33-acre former solid waste landfill site occurs within mapped, but degraded vernal pool complex (Figure 4-2). This area was used by Camp White and the City of Medford from 1930 until 1962 and was used for automobile wrecking and wood mill ash disposal (ODEQ 2012a). Although detections of lead were reported from site testing, no tests for copper or chromium VI were conducted (ODEQ 2012a). Copper and chromium VI could potentially occur at the site from buried machinery or equipment, vehicle parts, and electrical wire. Log-deck waste was also spread at the site in 1993 and covered with additional fill. Vernal pools in the surrounding areas are very sparse, but none have been observed lacking aquatic invertebrates or having a lack of algal plant matter. This indicates that harmful levels of copper or other toxins are not present in these pools.

The 49-acre Jackson County Sports Park occurs within the mapped vernal pool complex (Figure 4-2). This recreation area includes heavily used dirt-tracks, accessed by off-road vehicles for recreational purposes. Historic agricultural and vehicle use degraded the functionality of the vernal pool complex at the site, even prior to listing of fairy shrimp. The majority of the site no longer functions as a vernal pool complex. The USEPA BE (2008) identifies zinc and copper in urban or road surface runoff and lead as associated with automotive vehicle use.

Copper from off-road vehicle brake pads and leaking vehicle fluids has the potential to enter the remaining vernal pool complex at the site. Copper, lead, and zinc associated with vehicle brake pads, fuel exhaust or oil leaks could enter pools, although the quantity cannot be determined and there are no impervious surfaces which occur on the property to channel or concentrate chemicals. However, we assume that copper, lead and zinc could be present, but would be transitory due to lack of impervious surfaces in the system. In addition, vernal pool fairy shrimp could be exposed to lead from this park.

Residential and industrial sectors have expanded onto the vernal pool complex on all sides of White City. The northern Agate Desert area supports predominantly livestock ranching mixed with some hay production. Hay production does not require heavy use of fertilizers or chemical pesticides but livestock grazing does not appear to use or release elevated levels of ammonia in nearby streams (see Lahontan cutthroat trout species-specific exposure analysis section above).

Based on the above information and information contained in the Environmental Baseline chapter of the BE (USEPA 2008), there is potential exposure to vernal pool fairy shrimp and its critical habitat for several of the chemicals subject to consultation including copper, lead, chromium VI, pentachlorophenol, and zinc. Chemicals in this consultation which do not have anthropogenic sources in vernal pool fairy shrimp habitat are aluminum, ammonia, arsenic, cadmium, nickel, silver, and selenium; we concur with USEPA's determination of NLAA for these chemicals to the vernal pool fairy shrimp and its critical habitat.

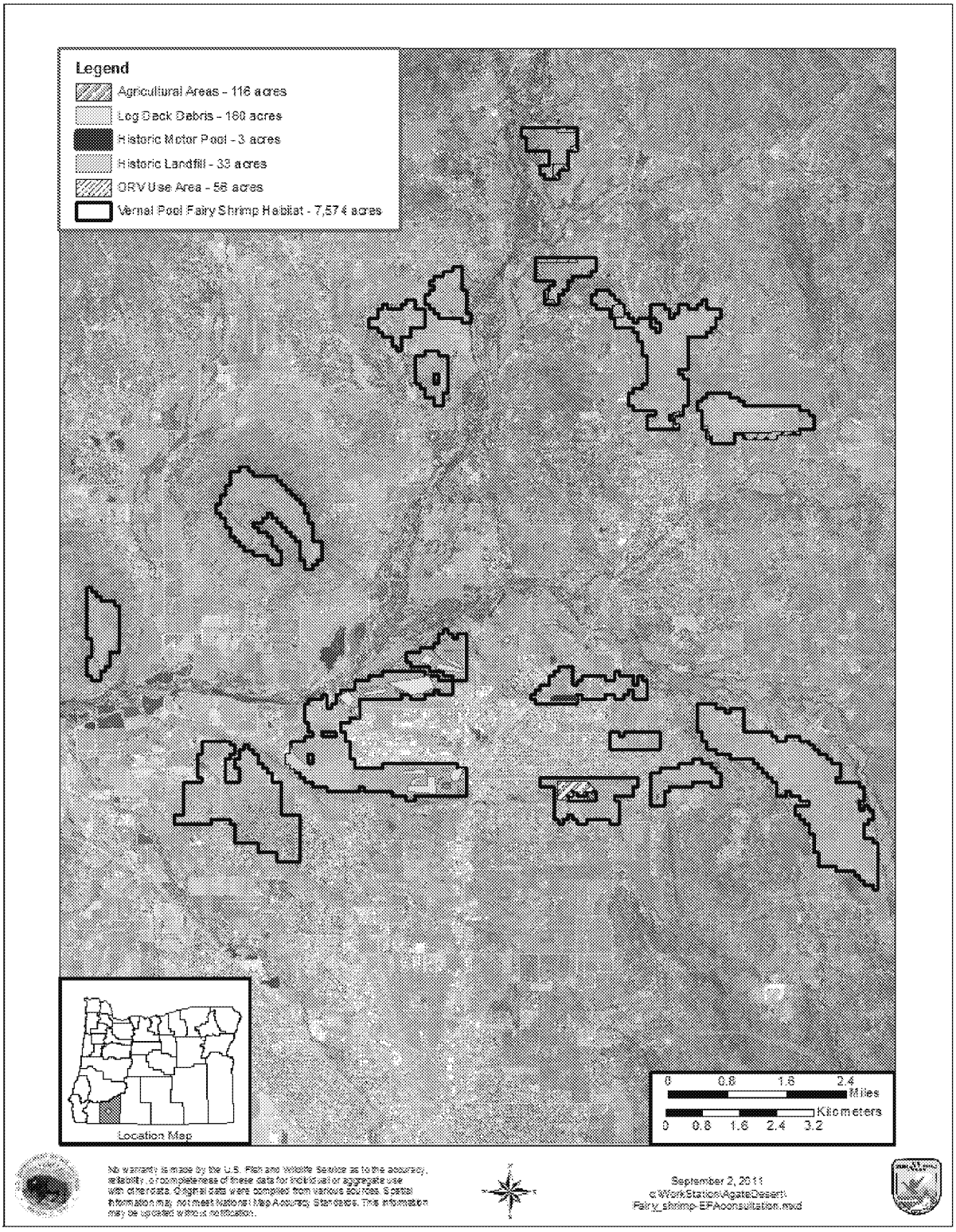


Figure 4-2. Map indicating the Vernal Pool Fairy Shrimp sites with potential exposure to chemicals included in this consultation.

Bull trout

Many point sources discharge into bull trout streams, or into streams that are directly connected to bull trout habitat. The various operations permitted in Oregon present a possible source of exposure to toxins for bull trout and their habitat. The ODEQ (http://www.deq.state.or.us/wq/assessment/assessment.htm) maintains a list of water-quality limited waters within the State. This 303(d) list includes sections of many streams inhabited by bull trout, and many others that are directly connected to bull trout habitat (Table 4-1). Mining exists on many streams occupied by bull trout, and on many streams with direct hydrologic connection to bull trout habitat (Table 4-1).

Besides NPDES permits and mine sites, other sources of exposure exist for bull trout including roads, forestry, and agriculture. Roads exist throughout the range of bull trout within Oregon. The degree of exposure will vary with the amount and type of vehicle traffic, the road surface, and the size and condition of each individual road. Forestry practices could be a source of exposure depending on the specific practice and location (proximity). Although USEPA reports that forestry activities account for approximately 9% of the water quality problems in forested areas, most of this is due to sediment (USEPA 2008). Most bull trout spawning and rearing habitat is located on Federal land managed by the U.S. Forest Service, 80% of spawning and rearing habitat within Oregon is on Federal land. Because these National Forests operate under management plans which include Best Management Practices (BMPs) it would be unlikely that they would pose a major source of exposure. Although these streams may be exposed to ammonia as a result of grazing leases, this exposure is localized and insignificant (see Section 4.4.1.4).

Various agricultural practices occur in areas within bull trout habitat. Most of these occur adjacent to feeding, migration, and overwintering habitat. Use of pesticides and fertilizers, along with organic wastes from farming and ranching are possible sources of exposure. Direct exposure could come from returns of irrigation water, runoff and chemigation. Almost all agriculture adjacent to bull trout habitat occurs within foraging, migratory, and overwintering habitat. Most of this habitat that could be exposed is composed of larger streams which hold greater flows. Also, the number of bull trout using foraging, migratory, and overwintering habitat is variable by time of year and stream temperature. Most agricultural activity that could present a source of exposure occurs during the mid- to late summer and early fall months when bull trout are not present. In bull trout habitat, temperatures are too cold in early to mid- spring or fall to support much agricultural production of forage, the major agricultural activity in bull trout habitat. Forage production requires irrigation during the summer which is concluded by early fall. Therefore, most exposure would be carried through irrigation returns during the summer months when stream temperatures are elevated and most bull trout are not present. Therefore, these activities are not considered to be a major source of exposure.

Given the wide distribution of bull trout in Oregon, for purposes of this exposure analysis the sources of chemicals considered in this consultation have been summarized in table form separated according to bull trout management units.

The probability of exposure of bull trout to toxins was determined as follows. All known potential sources that could specifically be identified that lay within 2 miles of bull trout habitat were included in the analysis (this included all sources that exist within 0.5 miles overland, and all other sources that exist within a 2-mile buffer of bull trout habitat, if those sources had hydrologic connection to the habitat). The resulting sources were then buffered and intersected with known bull trout habitat. The exposed sections of stream were identified and classified as to their habitat type, and are summarized in Table 4-1 and Table 4-2.

Two miles was used as the mixing zone for aquatic exposure as a best estimate of the distance at which most toxins would be sufficiently diluted to reduce their exposure potential. This distance could actually vary with the concentration of the exposure (magnitude), and the specific stream size and flow. Mathematical modeling to predict downstream effluent concentrations uses complex equations based on input parameters including river discharge, depth, effluent concentrations, wind speed and direction (Bathory et al. 2005, p. 19). In addition, forest buffers also enhance the in-stream processing of point source pollutants in small streams reducing their impact downstream (Sweeney et al. 2004, p. 14132). Given this is a Statewide consultation and there are widely variable conditions for the aquatic habitats of listed species and point source discharge locations, it is not possible to determine maximum length of downstream impacts. Therefore, we selected 2 miles as a reasonable distance for point sources. Although studies done in Idaho found bull trout with liver histology from exposure to toxins at a distance of over 4 miles from the source of exposure (Kiser et al. 2010, p. 306), their mobility makes it unclear if these fish had been foraging closer to the source at some point in time.

Our analysis assumes that all identified NPDES permits are sources of the chemicals considered in this consultation, unless otherwise noted. Although we understand that it is unlikely that all the chemicals would be released from every permit location (Table 4-2), we lack the resources to evaluate all the permits in bull trout habitat to determine which locations actually are sources of exposure. To err on the side of caution we assumed that all the chemicals would be released from every permit location.

Table 4-1. Baseline Stream Conditions and Potential Exposure to Metals by Bull Trout by basin.
(SR = spawning and rearing habitat; FMO = foraging, migratory and overwintering habitat.

Basin Name	Number of 303(d) Bull Trout Streams*[1]	Number of Metal Mines*	Number of current NPDES Permits*	Miles of SR habitat Exposed	Total SR Miles	Percent of SR Habitat Exposed	Miles of FMO Exposed	Total FMO Miles	Percent of FMO Habitat Exposed	Total Miles of Bull Trout Habitat Exposed	Total Miles of Stream Habitat
Grande Ronde River	89	85	12	69.6	245.8	28.3%	37.5	412.1	9.1 %	107.1	658
Hells Canyon Complex	19	42	2	35.6	88.2	40.3%	16.3	146.4	11.1%	51.9	234.6
Hood River	23	0	13	0	14.5	0%	12.7	65.1	19.5%	12.7	79.6
Imnaha River	12	45	0	46.8	78.4	59.7%	12	99.2	12%	58.8	177.6
John Day River	129	672	6	48.7	234.7	61.7%	108.2	460	24%	156.9	648.6
Klamath River	27	3	1	6.5	151	4.3 %	12.5	125.6	10%	19	276.6
Lower Deschutes River	40	0	5	4.4	72.4	6%	8.2	215.4	3.8%	12.6	287.8
Malheur River	22	2	0	0	101.4	0%	0	67.9	0%	0	169.3
Odell Lake	5	0	0	0	7.1	0%	0	9.9	0%	0	17
Powder River	30	303	2	38.4	99.4	39%	42	84.9	49%	80.4	184.3
Umatilla River	24	0	26	0	18.2	0%	37.5	113.4	36%	37.5	131.6
Upper Willamette River	69	5	48	5.1	15.9	32%	73	178.2	41%	78.1	194.1
Walla Walla River	10	2	3	5.7	75.1	8%	14.8	206	7%	20.5	281.1
Columbia River FMO	NA	3	215	NA	NA	NA	120	301.2	39.8%	120	301.2
Snake River FMO	NA	76	1	NA	NA	NA	65.1	107.7	60.4%	65.1	107.7
Totals:	499	1238	334	260.8	1124	36.3%	559.8	2312	25.7%	820.6	3436

**Includes those adjacent to habitat (within 0.5 miles), and those hydrologic connection to habitat (within 2 miles)*
[1]Provided for baseline information only

Table 4-2. Bull Trout Stream Exposure Potential from NPDES Permits.

Management Unit	Miles of SR Habitat Exposed	Percent of Total SR Habitat Exposed	Miles of FMO Exposed	Percent of Total FMO Habitat Exposed	Total Miles of Bull Trout Habitat Exposed
Grande Ronde River	0	0%	45.1	12.5%	45.1
Hells Canyon Complex	0	0%	14.6	16%	14.6
Hood River	0	0%	20.2	31%	20.2
Imnaha River	0	0%	0	0%	0
John Day River	2.6	1.1%	18.3	3.9%	20.9
Klamath River	4.9	3.2%	7	5.6%	11.9
Lower Deschutes River	4.4	6.1%	8.2	3.8%	12.6
Malheur River	0	0%	0	0%	0
Odell Lake	0	0%	0	0%	0
Powder River	9.6	9.7%	9.6	11.3%	19.2
Umatilla River	0	0%	40.5	22.7%	40.5
Upper Willamette River	5.1	32.1%	64.4	36.1%	69.5
Walla Walla River	4.1	10.5%	10.3	33%	14.4
Columbia River FMO	NA	NA	113.5	37.3%	113.5
Snake River FMO	NA	NA	7.8	7.2%	7.8
Totals:	**30.7**	**2.7%**	**359.5**	**16.9%**	**390.2**

Includes permits adjacent to habitat (within 0.5 miles), and those with hydrologic connection to habitat (within 2 miles)

Lost River sucker and shortnose sucker

The present distribution of Lost River and shortnose suckers in Oregon includes Upper Klamath Lake and its major tributaries (Williamson, Sprague, and Wood Rivers), all of the Lost River in Oregon, and the Oregon portion of the Klamath River. Additionally, the shortnose sucker occurs in the Gerber Reservoir and its major tributaries, Ben Hall and Barnes Valley Creeks in Oregon.

There are several NPDES-permitted discharges in the Upper Klamath Basin. Within the Oregon portion of the Klamath Basin, NPDES-permitted discharges are primarily located near Klamath Falls and they discharge to Upper Klamath Lake and the Keno Reservoir at the upper end of the Klamath River. The discharges include the following: (1) the Jeld-Wen Wood Products Manufacturing Plant in Klamath Falls discharges cooling water and has limitations on conventional parameters in addition to total copper, total lead, and total zinc, (2) the City of Klamath Falls discharges treated effluent from a STP into the upper end of the Keno Reservoir, (3) the South Suburban Sanitary District discharges domestic sewage into Lake Ewauna of the Klamath River and includes monitoring requirement for nutrients, ammonia, total phenols, and metals including arsenic, cadmium, chromium, copper, mercury, nickel, lead, silver, zinc and cyanide, (4) the Columbia Plywood Corporation discharges into the Keno Reservoir with limitations of biological oxygen demand, suspended solids, oil and grease, pH, flow, and temperature, (5) the Klamath County School District operates two small STPs that discharge treated effluent into an irrigation drainage ditch which ultimately drains into the Lost River in Oregon, and (6) the Klamath Irrigation District has an NPDES permit for aquatic weed control using the herbicide acrolein and was addressed in biological opinions dated February 9, 1995, and February 2, 1996 (USFWS 1995, 1996b).

There are also general stormwater permits for discharging into the Keno Reservoir near Klamath Falls for Champion Metal Products, Bettendorf Trucking, Klamath Auto Wrecking, Regional Disposal Company, and Klamath Falls Operations. There are also some stormwater permits for construction for discharging into Klamath and Lost Rivers including the City of Klamath Falls and Dove Hollow Development. Agricultural drains including the Straits Drain (at RM 239) and Lost River Diversion Channel closer to Klamath Falls (near RM 247), as well as many smaller drains, enter the Keno Reservoir. Permitted stormwater discharges from the urbanized area surrounding Klamath Falls also may contain copper and these enter the Keno Reservoir.

Copper-containing pesticides are used on some agricultural crops in the upper Klamath Basin to control certain diseases. Organometallic fungicides used in the upper basin containing copper hydroxide include Champion, Kocide, NuCop, and Ridomil. These products are registered for use on onions and potatoes and are applied by aircraft, boom, and sprinkler at rates varying from 1.5 to 2.5 pounds/acre, or 4 pints/acre, depending on the product. Data are not available for copper hydroxide application totals for Klamath County, but for adjacent Siskiyou County, California, which has similar crops where copper-based fungicides are used (primarily on onions and potatoes), 2,900 pounds were applied to 2,600 acres in 2008, the year of the most recent data (Pesticide Action Network 2011). Copper can enter sucker habitat from agricultural applications through wind drift, surface runoff, or through groundwater, and from around homes and businesses from nonpoint-source stormwater runoff or via groundwater discharges.

According to the State of Oregon's Pesticide Use Reporting System 2007 Annual Report (Oregon Department of Agriculture 2008), which had full reporting capabilities beginning in 2007, several copper-containing compounds were commonly used around homes in Oregon as wood preservatives, including copper naphthanate and copper ammonium carbonate.

Elevated ammonia concentrations have been reported in the Klamath Basin and sucker habitat for many years (Saiki et al. 1999). Therefore, we concur with USEPA's determination of LAA for acute ammonia freshwater criteria but do not concur with their NLAA determination for the chronic ammonia freshwater criteria for Lost River sucker and shortnose sucker and their proposed critical habitat.

The Klamath River/Agency Lake is listed as category 4A for aquatic weeds or algae and there is an approved TMDL for chlorophyll a, pH, and dissolved oxygen in summer. The Klamath River is listed as category 5 for ammonia, chlorophyll a, pH, temperature, and dissolved oxygen, and category 3 for phosphate phosphorus. Klamath River/Ewauna Lake is listed as category 5 for ammonia, dissolved oxygen, and pH, and as category 3B for phosphate phosphorus (summer). Klamath River/ Upper Klamath Lake is listed as category 4A for aquatic weeds or algae. An approved TMDL for chlorophyll a, pH, and dissolved oxygen is in place for Upper Klamath Lake.

Lost River/Lost River Reservoir is listed as category 5 for chlorophyll a and pH in summer and is on the 303(d) list for dissolved oxygen in summer. Lost River is listed as category 5 for ammonia, dissolved oxygen, and chlorophyll a (summer), and potential concern for arsenic, manganese, beryllium, and phosphate phosphorus (summer)

North Sprague River is listed as category 3B for alkalinity and phosphate phosphorus in summer. There is an approved TMDL for temperature in summer. The South Fork Sprague River is listed as category 5 for *E. coli* in summer and as category 3B for phosphate phosphorus in summer. There is also a TMDL approved for temperature in summer. The Sprague River has an approved TMDL for dissolved oxygen and temperature in summer; it is listed as category 4A for dissolved oxygen year-round and category 3B for phosphate phosphorus in summer.

There are no mines identified in the vicinity of Lost River sucker and shortnose sucker habitat.

There are numerous sources of chemicals in the Klamath Basin, which is habitat occupied by Lost River and shortnose suckers.

Oregon chub

Oregon chub occur only in the Willamette River drainage of western Oregon, including the North and South Santiam Rivers, McKenzie River, Middle Fork Willamette River (MFWR) and Coast Fork Willamette River (CFWR), and several tributaries to the mainstem Willamette River downstream of the CFWR/WFWR confluence (Bangs et al. 2009, p. 1).

Due to loss of habitat and the pervasive presence of predatory nonnative fish, recovery efforts have focused on managing chub in ponds which are isolated from any rivers. Some of these habitats do experience short periods of connectivity to surface flows annually, others are completely disconnected from the floodplain. Several of these streams that connect periodically to Oregon chub ponds are listed on ODEQ 303(d) list as water-quality limited as shown in Table 4-3.

Table 4-3. 303(d) listed streams connected to Oregon chub habitat.

Waterbody	Subbasin	Category 5 Water-quality limited	Category 4A Water-quality limited	Category 3B Potential Concern
North Santiam River	Santiam	temperature		alkalinity, iron
South Santiam River	Santiam	temperature		alkalinity, manganese
Muddy Creek	Mid-Willamette	temperature		iron, manganese
McKenzie River	Mid-Willamette	temperature		alkalinity, manganese, iron
Mohawk River	Mid-Willamette	temperature (9/15 - 6/15)		alkalinity, iron
MFWR	Middle Fork	temperature		alkalinity
Buckhead Creek	Middle Fork	temperature		
CFWR	Coast Fork	temperature, mercury, iron	dissolved oxygen, pH	alkalinity, manganese

(Streams listed for temperature are based on salmonids temperature criteria which are much cooler than preferred by Oregon chub [USFWS 2004c]).

Dieldrin is listed as a parameter of concern (3B) on Oregon's 303(d) list for segments including Fanno Creek (RM 0-13.9), Zollner Creek (RM 0-7.8), Pudding River (RM 0-35.4), Pringle Creek (RM 0-6.2), Kelly Creek (RM 0-3.6), and Johnson Creek (RM 0-23.7). In addition, there are two NPDES permits which have dieldrin listed as a parameter for monitoring and reporting, although not as a criteria limit. The first permit is for Covanta Marion, Inc., which is a combustion facility for municipal trash. The permit is for regulating discharge of boiler blowdown, cooling tower blowdown and demineralizer backwash (ODEQ, NPDES permit number 101240). The outfall discharges into RM 72 of the Willamette River, immediately downstream of Wheatland Ferry. The second permit is for Marion County and Brooks Community Sewer District which operates a wastewater treatment facility, which discharges to the Willamette River at RM 69.7 (ODEQ, NPDES permit number 101397). However, these will not impact Oregon chub because chub no longer occur in the Pudding River Watershed and all other creeks are not in proximity to Oregon chub habitat.

Within the Willamette Basin, the majority of NPDES-permitted discharges in the range of Oregon chub are located near the urban centers of Salem, Albany, Eugene and Springfield. We assessed NPDES permits based on the GIS layer provided on ODEQ website to determine the location of discharges in relation to known populations of Oregon chub. Three permits have been issued for discharges that occur within 2 miles upstream from Oregon chub sites, two of these are located in the MFWR subbasin and one is located in the Mid-Willamette River subbasin; they include:

- The City of Lowell discharges wastewater approximately 20 feet from Dexter Dam on the downstream end of Dexter Reservoir. Although permit limits are restricted to the standard conventional limits of most WWTPs, monitoring of biosolids is required for ammonia, arsenic, cadmium, copper, mercury, molybdenum, nickel, lead, selenium, and zinc. A reasonable potential analysis for ammonia was conducted in 2004 and found that there is not a reasonable potential to violate either the chronic or acute toxicity standard (ODEQ 2004). The following Oregon chub sites occur within 2 miles downstream of this discharge:
    - Elijah Bristow Island Pond, North Gravel Pit, Northeast Slough, and South Slough
    - Dexter Dam Slough
- The City of Westfir discharges wastewater into the MFWR at RM 0.1. Although permit limits are restricted to the standard conventional limits of most WWTPs, monitoring of treated effluent is required for ammonia. A reasonable potential analysis has not yet been completed for ammonia for this source. The following Oregon chub site occurs within 2 miles downstream of this discharge:
    - Buckhead Creek
- Monaco Coach, a manufacturer of motorcoaches and other recreational vehicles, operates a facility in the Mid-Willamette River subbasin and holds an individual NPDES permit for stormwater discharge. The chemicals associated with stormwater discharges that are subject of this consultation include arsenic, cadmium, chromium, copper, lead, nickel, and zinc (USEPA 2005b, p. 22). The following Oregon chub site occurs with 2 miles downstream of this discharge:
    - Muddy Creek

Agriculture. Several Oregon chub ponds are located adjacent to agricultural fields that may experience runoff during heavy rains. Fertilizers used in agriculture may be a potential source of metals including cadmium, lead, nickel, and copper (Curtis and Smith 2002). Fungicides may also be a potential source of copper (WSPMR 2012). The primary crops grown in these adjacent fields are hay, grass seed, and corn. The amount of cadmium, mercury, lead and nickel contained in approved fertilizers, agricultural amendments, agricultural mineral and lime products is regulated in Oregon (OAR 603-059-0100). Although high levels of these metals were not been identified in these agricultural products in Oregon, this regulation was developed in response to similar protective regulations developed in California and Washington to prevent the use of products containing industrial wastes with elevated levels of metals in Oregon (Matt

Haynes, Fertilizer Specialist, ODA, Salem, Oregon, pers. comm., 2011). Thus, exposure from fertilizer use at chronic or acute levels of these metals is unlikely. Limits are not in place for copper, but according to ODA, copper-based fungicides are not used in grass, grass seed, hay, and corn production (Rose Kachadoorian, Pesticide Registration Specialist, ODA, Salem, Oregon, pers. comm., 2011) and copper-based fertilizers are not typically used on crops in the Willamette Valley within the range of Oregon chub (M. Haynes, pers. comm., 2011). Therefore, we have determined that Oregon chub are not at risk of being exposed to chronic or acute levels of copper, cadmium, lead and nickel from agricultural sources.

These sites may also be exposed to ammonia from fertilizer runoff. As previously stated, the crops grown in fields adjacent to Oregon chub ponds are hay, grass seed, and corn. A review of fertilizer practices on farm fields on or adjacent to the Willamette Valley National Wildlife Refuges found that nitrogen-based fertilizers are applied to these crops in mid-January through April and August through November (Materna and Buck 2007). The resulting ammonia is rapidly taken up by plants and microorganisms and does not persist long in the environment (DHS 2004).

The Willamette Valley climate is characterized by cool, wet winters and warm, dry summers. Most rainfall occurs between October and March; thus, Oregon chub ponds adjacent to these fertilized agricultural fields may be at risk of exposure to chronic levels of ammonia during this time period if fertilizer was recently applied. The peak spawning period for Oregon chub is June through early August with eggs hatching after approximately 10 days. Therefore, eggs are not present in chub habitats during periods susceptible to runoff and will not be exposed to acute or chronic levels of ammonia due to agricultural runoff. Juveniles and adults, however, may be present during periods with runoff potential and may be exposed to acute or chronic levels of ammonia. The sites located adjacent to agricultural fields include:

> Cold Creek Slough, Gray Slough, Green's Bridge Slough, Harris Slough, Koenig Slough, Logan Slough, Dry Muddy Creek, Ezell Slough, Muddy Creek, Santiam Easement, Santiam I-5 Side Channels, Dunn Wetland, Ankeny Willow Marsh, Russell Pond, Finley Cheadle Pond, Hunsaker Slough, Finley Display Pond, Finley Beaver Pond, Shetzline Pond, Coast Fork Side Channels, St Paul Ponds, and Finley Gray Creek Swamp

A few Oregon chub sites are near fields grazed by cattle. However, Doran et al. (1981, p. 168) found that the levels of nitrogen added to grazing lands in Nebraska and Oklahoma through precipitation far exceeded those found in runoff from grazing land. So any ammonia added to the system by grazing is likely to be insignificant.

Forestry. Several Oregon chub sites are located adjacent to private or public forestland. Forests managed by the U.S. Forest Service operate under management plans which include BMPs that are protective of fish. The use of chemicals on private and State forest operations in Oregon is regulated by the Oregon Department of Forestry under OAR 629-620-0000 through 629-620-0800. These rules require operators to comply with product labels and additional protective measures to protect waters of the State, including leaving untreated vegetated buffers and limiting aerial applications near areas of standing open water larger than one-quarter acre. No copper-based pesticides are registered for use in Oregon for forest applications (WSPMR 2012). Regulation of cadmium, lead and nickel levels in fertilizers reduces the potential for these metals

to reach chronic or acute levels in sites adjacent to managed forests (OAR 603-059-0100). Based on these management practices, rules and regulations, we have determined that exposure to toxic chemicals considered in this consultation at acute or chronic levels from forestry practices is not likely to occur.

Roads. Roads are a potential source of copper, lead, and zinc due to the use of copper in vehicle brake pads, fuel exhaust, or oil leaks. As a result of the high annual rainfall in the Willamette Valley, these metals could enter waterways adjacent to roads. A few Oregon chub sites occur adjacent to major roads and highways and are at risk for exposure to copper, lead, and zinc; these sites include:

- Santiam I-5 Side Channels (Santiam River Subbasin)
- Hendrick's Bridge Slough (Mid-Willamette River Subbasin)
- Dexter Reservoir Alcove – PIT1 (MFWR Subbasin)
- Dexter Reservoir RV Alcove – DEX3 (MFWR Subbasin)
- East Fork Minnow Creek Pond (MFWR Subbasin)
- Buckhead Creek Ponds (MFWR Subbasin)
- Shady Dell Pond (MFWR Subbasin)

Urban stormwater runoff. Urban stormwater runoff has been identified as a major contributor of pollutants, including copper, to streams (Pitt et al. 1995). However, all Oregon chub sites are either located in rural areas upstream of population centers of the Willamette Valley or are managed at isolated sites that are not exposed to urban runoff.

Mining. We reviewed the MILO-2 GIS layer from the Oregon DOGAMI which identifies metal and industrial mining locations to determine the location of mines in relation to known populations of Oregon chub. No mines occur within 2 miles upstream from Oregon chub sites or within one-half mile upland of Oregon chub sites.

4.1.2.3 Summary.

Several of the listed aquatic or aquatic-dependent species in Oregon have extremely limited potential for exposure to the chemicals in this consultation due to their remote locations or limited occurrence in Oregon. Therefore, adverse effects are considered discountable and approval of the criteria is not likely to adversely affect these species or their designated critical habitats: short-tailed albatross, Borax Lake chub, Foskett speckled dace, Hutton tui chub, Lahontan cutthroat trout and Warner sucker, and western snowy plover. Further analysis of effects is therefore limited to those listed species and their designated or proposed critical habitats that have potential to be exposed to the chemicals at issue in this consultation, including bull trout, Oregon chub, Lost River and shortnose suckers, vernal pool fairy shrimp, and marbled murrelet.

4.1.3 Chemical-Specific Exposure Analyses

4.1.3.1 Chemicals with Not Likely to Adversely Affect Determinations

Several of the pesticides included in this consultation are no longer registered for use in the U.S. or have very restricted use. These chemicals, commonly referred to as legacy pesticides, are discussed below.

Dieldrin

Dieldrin is a chlorinated hydrocarbon insecticide originally developed as an alternative to DDT. It was used widely during the 1950s to the early 1970s. Due to concerns about damage to the environment and potentially to human health, USEPA banned all uses in 1974, with the exception of termite control, which it banned in 1987 (ATSDR 2002, p. 1). From 1972 until 1987 dieldrin was primarily applied to homes for termite protection (ODEQ 2008, p. 4-6). In 1987, the manufacturer voluntarily cancelled the registration for this remaining use (ODEQ 2008, p. 4-6).

Exposure to dieldrin only occurs from legacy effects (past usage) since this chemical has not been applied for over 30 years. Yet dieldrin is extremely persistent in the environment and was widely used as a domestic pesticide (ODEQ 2008, pp. 4-5–4-6). According to the BE, dieldrin is listed as a parameter of concern in nine waterbody segments on Oregon's 303(d) list. ODEQ's 2010 Integrated Report Database lists the following watersheds for dieldrin: Lower Columbia, Willamette, Lower Owyhee, and Lower Malheur Rivers. The two NPDES permits which have dieldrin listed as a parameter for monitoring and reporting, although not as a criteria limit, are Covanta Marion, Inc. (ODEQ, NPDES permit number 101240), which discharges into the Willamette River at RM 72, and Brooks Community Sewer District (ODEQ, NPDES Permit No. 101397), which discharges into the Willamette River at RM 69.7.

Results of dieldrin concentrations in biotic tissue samples are reported in the *Irrigation Drainwater Report for Detailed Study of Irrigation Drainage in the Klamath Basin, California and Oregon, 1990-1992* (MacCoy 1994, p. 99). Tissue samples were collected for aquatic invertebrates, fish, and bird eggs. The majority of the samples had dieldrin concentrations below detection limits (0.01 μg/g wet weight) with the exception of 4 western grebe egg samples which were at or just slightly above (0.03 μg/g wet weight) the detection limit.

Bull trout

From the list of 303(d)-listed waterbody segments, there appear to be 10 streams which could influence bull trout. Nine of these in the mid-Columbia River are foraging, migratory and overwintering habitats and one is directly connected to this habitat. Although dieldrin could theoretically be in bull trout surface waters, we believe that the occurrence will be very minimal. The streams identified on the 303(d) list are larger in size with high flows and bull trout are not spending significant portions of time in these streams. Even in the event that dieldrin is co-occurring in these systems with bull trout, the chemical will more likely be associated with the sediments as it has a low solubility in water (ODEQ 2008, p. 4-6) and sorbs to sediment (ODEQ 2008, p. 4-21). Sediment is not considered in water quality standards and is not part of this action.

Bull trout may be found in streams where dieldrin has been identified as a source of exposure from historic use. However, the potential for adverse effects is considered insignificant given that exposure would be limited due to seasonal in these streams, the fate of this chemical is most often associated with the sediments and not the water, and concentrations of this chemical are expected to be decreasing. Therefore, approval of the proposed dieldrin criteria is NLAA to bull trout and we do not concur with USEPA's determination that approval of dieldrin criteria is LAA to bull trout and its designated critical habitat.

Oregon chub

Although Oregon chub are found in the Willamette River, none of the streams identified in the 303(d) list or the NPDES permits could influence Oregon chub. These chub were historically found in the Pudding River watershed but no longer occur there.

ODEQ (2008) completed a TMDL for the Molalla-Pudding Subbasin which includes dieldrin. The TMDL states that dieldrin was one of the most frequently detected organochlorine compounds in a 1996 river basin water quality study but in 1997 (a later phase of the same study) dieldrin was not detected (ODEQ 2008, p. 4-20). ODEQ 2008 (p. 4-22) states that dieldrin concentrations should be declining since it is a legacy pesticide which, presumably, has not been applied to croplands for many years. Although a number of significant exceedances of the State of Oregon human health based criteria were observed in the 1990s, dieldrin has been detected only occasionally since that time (ODEQ 2008, p. 4-22).

Given that this chemical has not been approved for use for over 30 years and Oregon chub are not found in any areas identified as known sources of historic dieldrin contamination, adverse effects to Oregon chub are discountable and we concur with USEPA's determination that approval of the dieldrin criteria is NLAA Oregon chub and its designated critical habitat.

Lost River sucker and shortnose sucker

Results reported above for dieldrin concentrations from biotic tissue samples collected in the Klamath Basin (MacCoy 1994, p. 99) indicate that there does not appear to be continued persistence of this chemical in the system. Therefore, the potential for adverse effects is discountable and we concur with USEPA's determination that approval of the criteria is NLAA Lost River sucker or shortnose sucker or their proposed critical habitat.

Vernal pool fairy shrimp

There is unlikely to be any dieldrin in the vernal pool complex of Oregon based on the exposure analysis presented above. Therefore, we concur with USEPS's determination that approval of the criteria will NLAA vernal pool fairy shrimp or its designated critical habitat.

Aquatic-dependent birds

Potential exposure to aquatic birds from dieldrin is extremely unlikely and considered insignificant. Thus, we concur with USEPA's determination that approval of the criteria is NLAA marbled murrelet or western snowy plover or their designated critical habitat.

Endosulfan

Endosulfan was first registered as a pesticide in the U.S. in 1954. In 1991, the technical registrants amended labels to incorporate a 300-foot spray drift buffer for aerial applications between treated areas and waterbodies and in 2000 the technical registrants amended technical product labels to remove all residential use patterns (USEPA 2002a). USEPA took action to eliminate all uses of the insecticide endosulfan in the U.S. in June 2010 (USEPA 2010a). Most uses will be phased out by July 2012 with remaining crops phased out progressively until 2016 (USEPA 2010c).

Endosulfan is volatile, persistent, and has a high potential to bioaccumulate in aquatic and terrestrial organisms. A large body of scientific literature documents endosulfan's medium- and long-range transport on a global scale and subsequent accumulation in nearly all environmental media. Even though many countries had already banned the use of endosulfan, including Bahrain, Belize, Cambodia, Colombia, Jordan, Kuwait, Malaysia, New Zealand, Norway, and all 27 European Union member countries (Pesticide Action Network 2008), endosulfan is present in air, water, sediment, and biota thousands of miles from use areas (USEPA 2010a).

In Oregon, no streams are listed under the 303(d) list for endosulfan. Endosulfan is not listed as a constituent with effluent limits in the searched NPDES permit database. Since 1991 use of endosulfan has required a 300-foot spray buffer around waterbodies. Although the action to eliminate use was very recent, given the buffer requirement that has been in place for 20 years, we believe that there is limited exposure to threatened and endangered species.

According to the BE (USEPA 2008), endosulfan was used on apple crops in Oregon. The largest orchards in the State are in Umatilla, Hood River and Wasco Counties. These counties border the Columbia River. The BE also noted sizeable production of apples in Marion and Washington Counties in the Willamette River Valley and Jackson County in southern Oregon. Reports for 1998 indicated that approximately 12 to 33% of the apple acreages were treated that year.

Studies were undertaken in the 1990s to evaluate the distribution of pesticides in the Willamette Valley and Klamath Basin, two of the major agricultural areas in Oregon. USGS (Anderson et al. 1997) evaluated pesticide occurrence in the Willamette Valley and the Department of the Interior (National Irrigation Drainwater Study) investigated the Klamath Basin (MacCoy 1994). While both studies included persistent organochlorine pesticides in their list of analytes, neither included endosulfan indicating it was not a chemical considered of significant concern.

Bull trout

Bull trout could be in waters near the apple orchards areas listed above in Umatilla, Hood River, and Wasco Counties although not in Marion, Washington, or Jackson Counties. Agricultural areas where endosulfan may occur are used seasonally by bull trout. Given that a buffer requirement for waterbodies has been in place for over 20 years and the primary areas of exposure for bull trout are limited to agricultural areas where there is only seasonal use, any exposure to endosulfan is expected to be discountable and therefore NLAA bull trout or its

designated critical habitat. We do not concur with USEPA's determination that approval of the criteria is LAA to bull trout.

Oregon chub

Agriculture surrounding Oregon chub is limited to grass and hay production; therefore, endosulfan would not have been used in the vicinity of Oregon chub habitat. Consequently, exposure to endosulfan is considered insignificant and we concur with USEPA's determination that approval of the criterion is NLAA Oregon chub or its designated critical habitat.

Lost River sucker and shortnose sucker

Substantial agricultural land surrounds the sucker's habitat in the Klamath Basin. However, typical crops grown do not include apples. In addition, the 300-foot application buffer to waterbodies has been in place for over 20 years. Any exposure to Klamath Basin suckers would be discountable and we concur with USEPA's determination that approval of the criteria would be NLAA Lost River sucker or shortnose sucker or their proposed critical habitat.

Vernal pool fairy shrimp

The Middle Rogue River Valley, location of the vernal pool complex, supports agriculture and the fields in proximity to the vernal pool complex are predominantly livestock ranching mixed with some hay production. Hay production does not involve heavy cultivation or use of fertilizers or chemical pesticides. Therefore, since agricultural areas in the vicinity of vernal pool fairy shrimp do not include orchards, exposure to endosulfan is considered discountable and we concur with USEPA's determination that approval of the criteria is NLAA for the vernal pool fairy shrimp or its designated critical habitat.

Aquatic-dependent birds

Potential exposure to aquatic birds from endosulfan is extremely unlikely and considered insignificant. Thus, we concur with USEPA's determination that approval of the criteria are NLAA marbled murrelet or western snowy plover or their designated critical habitat.

Endrin

Production of endrin was banned by USEPA in 1986 (ATSDR 1996). There are no current releases of endrin in the U.S. The BE (USEPA 2008) states that endrin persists in the environment primarily through adsorption to sediments and soil particles. Although endrin has the potential to bioaccumulate, its prevalence in the aquatic environment has been decreasing for over 20 years. Persisting endrin is projected to be in sediments and potential exposure to listed species is limited to areas where it may be concentrated from past use; however, sediment is not considered in water quality standards and is not part of this action.

According to ODEQ's 2010 Integrated Report Database endrin is listed as potential concern in the Columbia River up to RM 142 (near Beacon Rock). This is based on previous data which found the compound elevated in river substrate sediment. It is also listed in the Owyhee River as a potential concern based on water data, although there are no listed species in the Owyhee River. In the Columbia River, bull trout use below Bonneville Dam is limited to winter foraging.

Because there is very limited occurrence of bull trout below the dam, exposure to endrin is expected to be minimal.

The Klamath Basin of Oregon is one of the most intensively farmed areas of the State which is part of an extensive USBR Irrigation Drainwater Project (USBR 2011). The *Irrigation Drainwater Report for Detailed Study of Irrigation Drainage in the Klamath Basin, California and Oregon, 1990-1992* (MacCoy 1994, p. 99) included concentrations of endrin in tissues of aquatic invertebrates, fish, and bird eggs. All tissue levels were below detection limits (0.01 µg/g wet weight). These data provide insight into the level of contamination remaining from historic use of this pesticide.

Due to the low probability that any listed species in Oregon will be exposed to this chemical, the potential for adverse effects is discountable and we concur with USEPA's determination that approval of the criteria is NLAA bull trout, Oregon chub, Lost River and shortnose suckers, vernal pool fairy shrimp, marbled murrelet, or western snowy plover or their designated or proposed critical habitat.

Heptachlor epoxide (alpha and beta)

Heptachlor epoxide is formed from the breakdown of heptachlor. Heptachlor was used extensively in the past for killing insects in homes, buildings, and on food crops but these uses were banned in 1988 (Syracuse Research Corporation 2007). Currently, it can only be used for fire ant control in underground power transformers. Although it is a persistent organic pollutant and can bioaccumulate, we believe the potential for exposure to listed species is low because it has not been used for over 30 years.

Heptachlor listings for ODEQ 2010 Integrated Report Database include potential concern for Columbia Slough and Pringle Creek in the Middle Willamette River where it is listed as Category 5 up to RM 2.8. No listed species reside in these waters.

The Klamath Basin of Oregon is one of the most intensively farmed areas of the State and part of a major USBR Irrigation Drainwater Project (USBR 2011). MacCoy (1994, p. 99) included concentrations of heptachlor epoxide in tissues of aquatic invertebrates, fish, and bird eggs. All sample tissue levels were below detection limits (0.01 µg/g wet weight), aside from one western grebe egg which had concentrations at the detection limit. These data provide confirmation of the lack of contamination remaining from historic use of this pesticide.

In conclusion, due to the low probability that any listed species will be exposed to heptachlor epoxide, the potential for adverse effects is determined to be discountable and we concur with USEPA's determination that approval of the criteria is NLAA all aquatic (bull trout, Oregon chub, Lost River and shortnose suckers, vernal pool fairy shrimp) and aquatic-dependent (marbled murrelet and western snowy plover) species or their designated or proposed critical habitat considered in this consultation.

Lindane

Lindane, a broad spectrum, organochlorine insecticide, was used to treat seeds of various agricultural crops for approximately 50 years. In 2006, USEPA issued final orders cancelling all remaining pesticide products containing lindane (USEPA 2006). Since 1995, lindane has been approved by the Food and Drug Administration for use in pharmaceutical products intended to control lice and scabies; however, medical professionals prefer to prescribe non-lindane chemicals such as permethrin.

Due to extensive past use, lindane is present in most environmental media and biological compartments and terrestrial and aquatic food chains. However, evidence suggests that concentrations have been gradually decreasing and that once exposure is minimized most data suggest that biotransformation, depuration, and elimination are relatively rapid (USEPA 2002b, p. 41). Overall, lindane seems to accumulate in the environment generally tending to biomagnify in lower trophic levels where biotransformation is minimal; however, because upper trophic levels are able to depurate and eliminate the compound, lindane does not appear to readily work its way up the food chain (USEPA 2002b, pp. 41-42).

Lindane is not included on the pollutant list in ODEQ's 2010 Integrated Report Database (ODEQ 2011b). Furthermore, lindane is not listed as a limited chemical or a chemical required for monitoring in the database of NPDES permits supplied by USEPA.

Given that the current application of lindane is limited to pharmaceutical uses and widespread agricultural use was stopped in 2006, USFWS believes the potential for exposure to listed species is limited. Non-lindane containing over-the-counter options are available for lice treatment as are natural options, so lice treatment would not automatically indicate lindane use. In addition, the amount of lindane used in a lice outbreak is considered relatively small, extremely localized, and largely diluted within wastewater treatment facilities. Although lindane may be present from historic uses, it is not listed under any water-quality limited streams in Oregon. The probability that any listed species will be exposed to lindane is highly unlikely; therefore, any adverse effects would be insignificant and we concur with USEPA's determination that approval of lindane is NLAA all aquatic (bull trout, Oregon chub, Lost River and shortnose suckers, vernal pool fairy shrimp) and aquatic-dependent (marbled murrelet and western snowy plover) species or their designated or proposed critical habitat considered in this consultation.

Pesticide Summary

The pesticides dieldrin, endrin, endosulfan, heptachlor epoxide (alpha and beta), and lindane are no longer approved for use on a widespread scale. Aside from very specialized use of lindane (pharmaceutical treatment of lice/scabies) and heptachlor epoxide (fire ant control in underground power transformers), uses of these pesticides have been cancelled or phased out. Due to the persistent nature of these chemicals, there may be areas where they have concentrated and continue to exist as is identified by contamination noted in either NPDES permits or 303(d) listings. However, these identified stream segments have very limited overlap with listed species and their habitats. In addition, these chemicals most commonly persist in sediments, which are not being considered in this consultation. The effects from these legacy chemicals to listed

species are considered to be insignificant or discountable and therefore we determine that the freshwater, saltwater, acute and chronic criteria for dieldrin, endrin, endosulfan, heptachlor epoxide (alpha and beta) and lindane are NLAA all species and their designated or proposed critical habitat considered in this consultation.

Tributyltin

Tributyltin (TBT) is used extensively as a fungicide and bacteriocide in underwater and anti-fouling paints (USEPA 2008, p. 4-22). USEPA (2008, p. 4-22) reports it is also used as a preservative for wood, textile, paper, leather, and glass; a rodent repellent, insecticide, and biocide; as a water repellant, antioxidant, curing agent, and corrosion inhibitor; and in flame resistant polyester. Toxipedia reports the main use of tributyltin is as a stabilizer in the manufacturing of plastic products (Toxipedia 2011). Organotins used as agricultural pesticides generally show low toxicity to non-target organisms and form harmless tin residues (Eisler 1989b, p. 8).

The use of organotin compounds in antifouling paints has been severely curtailed (Eisler 1989b, pp. 16 and 70). In the US, the use of tin-based antifoulant paints has been restricted since 1988 on small vessels (<25 m in length) (Meador 2011, p. 258). In addition, in 2001 a treaty adopted by the International Maritime Organization to prohibit the use of TBT by 2008 was adopted. These restrictions have resulted in declines in aqueous concentrations of TBT, although not necessarily a decline in sediment concentrations (Meador 2011, p. 260).

Tributyltin was not listed on Oregon's 2002 303(d) list (USEPA 2008, p. 4-23). ODEQ's 2010 Integrated Report Database (ODEQ 2011b) indicates some sections of the lower Columbia River are of potential concern and areas of the Southern Oregon Coast including Coos Bay. There are no listed species under USFWS jurisdiction within the Southern Oregon Coast Basin. Bull trout inhabit the lower Columbia River seasonally. Supporting data for the Columbia River status was based on elevated levels in sediments, which we are not consulting on. The database states that there is no indication of beneficial use impairment.

There is only one NPDES permit which includes tributyltin in either its effluent limits or monitoring requirements, Vigor Industrial. This facility is described as a ship repair yard located at Swan Island in Portland, discharging into the Willamette River at RM 8. Tributyltin is included in its effluent limits and monitoring requirements at one of its 10 outfalls. Tributyltin may also enter freshwater and saltwater aquatic systems from boat hulls that have used paint containing organotin as an antifouling ingredient. There are six listings for TBT as potential concern, three are coastal areas and three are in the lower Columbia River.

As one of the major sources of tributyltin to Oregon waters has been restricted for over 20 years, the potential for exposure is extremely limited. Although, TBT may remain present in sediments, this consultation addresses surface water concentrations only and regulation of sediment is not part of the action. Therefore, we believe that the potential for exposure to listed species is discountable and not likely to adversely affect the listed species under consideration in this consultation.

The USEPA BE (2008) states that tributyltin has low water solubility and it will bind strongly to suspended material such as organic material or inorganic sediments and precipitate to the bottom sediment. Therefore, the greatest risk is to bottom-dwelling organisms. The action under consideration in this consultation is the surface water concentrations of subject chemicals and not sediment accumulation.

Klamath suckers are considered to have essentially no exposure to tributyltin. Point source discharges are restricted to the Portland metropolitan area and Klamath-area lakes are used by boats that do not use anti-fouling paints (primarily recreational canoes). Tributyltin is more commonly associated with marine boats, but in the rare event that boats near Klamath Falls may contain tributyltin paint, boat docks are associated with individual private property owners, thus significant concentrations are not likely to build up.

None of the listed freshwater species considered in this consultation exist in habitats with abundant boat use or storage. Adult bull trout may, on rare occasions, forage in these areas seasonally, yet they do not feed on bottom-dwelling organisms which are more likely to accumulate tributyltin. Based on its uses, lack of any point sources other than one facility and low probability that migrating bull trout will occur in concentrated boat areas, we do not believe there will be exposure to listed aquatic species.

There are no sources of tributyltin in Oregon chub or vernal pool fairy shrimp habitats. Therefore, we concur with USEPA's determination that the action of approving the criteria is NLAA any aquatic species or their designated or proposed critical habitat in this consultation.

The two birds that could be affected by saltwater criteria are snowy plover and marbled murrelet. Neither of these species is found in the vicinity of the point source for tributyltin, although they could occur in coastal areas where antifouling paints are applied to boats. However, both plovers and marbled murrelets feed away from areas where they are likely to obtain tributyltin through the diet. Therefore, we concur with USEPA's determination that approval of the criteria is NLAA marbled murrelet or western snowy plover or their designated critical habitat.

4.1.3.2 Chemicals with Likely to Adversely Affect Determinations Based on Exposure

Aluminum

Aluminum is naturally present in soils as a water insoluble hydroxide which can dissolve at very low and high pH values. The BE (USEPA 2008) states that aluminum may come from industrial wastes but more likely from washwater of drinking water treatment plants. Mining may also be a source of aluminum in aquatic systems as acid rock discharges from mine tailings may contain elevated levels of metals, including arsenic, copper, lead, aluminum, manganese, and zinc (Center for Educational Technologies 2004, Lim et al. 2009).

According to ODEQ spreadsheets of NPDES dischargers provided by USEPA, only five facilities are listed for either discharge limits or monitoring requirements. Aluminum is not listed for any stormwater permits and is not expected to come from nonpoint sources. Four permits were identified when reviewing the spreadsheet. Northwest Aluminum Company

discharges into the Columbia River at RM 186 and Georgia Pacific Consumer Products discharges into the lower Columbia River at RM 41. Two other permits were listed in the spreadsheet in searching aluminum, Port of Portland (Reynolds Metals) and Arden Development Corp. However, aluminum was not listed as an effluent limit or monitoring requirement for either of these permits.

According to the BE, aluminum is not listed on the 303(d) list for any Oregon waterbody.

Point source discharges that list aluminum as a monitoring requirement or permit limit are restricted to the mainstem Columbia River. Bull trout forage, migrate and overwinter in the mainstem Columbia River. They also could be exposed from the numerous mines located near their occupied streams, particularly in the John Day and Powder River Basins. Because there are no mines in the vicinity of Oregon chub, vernal pool fairy shrimp, or Lost River or shortnose suckers, and these species are not found near the listed point sources of aluminum, any exposure is highly unlikely and considered discountable and we concur with USEPA's determination that approval of the criteria is NLAA. Coastal habitats of the marbled murrelet and western snowy plover lack sources of aluminum; therefore, potential adverse effects are considered discountable and we concur with USEPA's determination that approval of the criteria is NLAA either species. The only listed species in this consultation that could be exposed to more than discountable sources of aluminum is bull trout (see Section 4.4.1.3 for effects of aluminum on bull trout and its critical habitat). There are no adverse effects from aluminum anticipated to any designated or proposed critical habitat for the species addressed in this consultation.

Ammonia

Ammonia occurs naturally but is also produced by human activity (USEPA 2008, p. 4-23). Ammonia is used as a fertilizer and applied directly to soil in farm fields, lawns and landscape plants. Household and industrial cleaners contain ammonia. Ammonia does not last very long in the environment as it is rapidly taken up by living organisms and does not build up in the food chain. Ammonia can lead to eutrophication due to its nutrient properties.

Specific sources of ammonia include municipal waste treatment plants, septic seep, industrial point sources, agricultural and urban runoff from fertilizer, concentrated animal feeding operations, aquaculture, landfill leachate, atmospheric sources and loss of riparian vegetation (USEPA 2010b).

The BE (USEPA 2008, p. 4-24) notes that ammonia is listed as a parameter of concern in three waterbody segments on Oregon's 303(d) list including Klamath River, Klamath Strait and North Myrtle Creek. However, additional locations listed in Table 4-5 have been added to the list since the BE was written (ODEQ 2011b).

Bull trout, Oregon chub, Lost River sucker and shortnose sucker could be exposed to ammonia given that sources from agriculture and municipal waste treatment plants are widespread. Consequently, each of these species could be exposed to ammonia concentrations at criteria levels; see Section 4.4 for effects of ammonia on these listed species and their critical habitats. Vernal pool fairy shrimp will not be exposed as there are no NPDES point source discharges into

its habitat and agricultural activity is removed from the vernal pools; we concur with USEPA's determination that approval of the criteria will be NLAA this species or its designated critical habitat.

Arsenic

Distribution of arsenic in groundwater can be related to aquifer geology (Yang et al. 2009) and elevated arsenic levels found in certain Oregon waters are postulated to be due to volcanic deposits (Stoner et al. 1977). Stoner et al. (1977) identify several hot springs in eastern Oregon that have elevated arsenic concentrations. The USEPA BE states that, "High arsenic concentrations are widespread in bedrock areas in south-central and eastern Lane County, and Linn County."

Anthropogenic sources of arsenic include mining, smelting, manufacturing, electric power plants, pesticides, agricultural defoliants and reclamation. Arsenic can be found in discharges from municipal wastewater, stormwater, and wood preserving facilities. Stoner et al. (1977) indicate that the level of this element in municipal water supplies is very low and may be attributed to groundwater. Oregon revised its water quality criteria to protect human health on April 21, 2011, to account for the presence of arsenic in Oregon waters due to natural sources (Sturdevant 2011). Sorenson (1991, p. 62) states that there are a variety of sources of arsenicals including metal or strip mines, smelting operations, electric generating stations, manufacturing companies, agriculture (defoliants and pesticides).

The BE (USEPA 2008) states that 13 stream segments were listed for arsenic on the Oregon 2002 303(d) list for segments in the Warner Lakes Subbasin and Upper Willamette River Subbasin, and the Willamette River. Since there are no known anthropogenic sources of arsenic in the Warner Lakes Subbasin, we expect these concentrations are naturally occurring.

Since there are numerous point sources of arsenic throughout waterbodies in Oregon, all of the aquatic species with potential exposure could experience criteria concentrations; see Section 4.4 for effects of arsenic on these listed species and their critical habitats.

Cadmium

Anthropogenic sources of cadmium include smelter fumes and dusts, incineration products of fossil fuels, fertilizers, and municipal wastewater and sludge discharges (Eisler 1985a, p. 1). Mining and ore smelting are also significant sources of cadmium and it is frequently associated with industrial discharges and stormwater runoff (USEPA 2008, p. 4-9).

Cadmium occurs naturally in the aquatic environment, but is of no known biological use and is considered one of the most toxic of metals to fish (Sorensen 1991). Significant sources of cadmium (also frequently associated with other metals such as zinc, copper, and lead) from mining and ore smelting result from direct discharge of mine drainage as well as atmospheric deposition. Cadmium is also frequently associated with industrial discharges and stormwater runoff. Uses of cadmium include electroplating, pigments, plastic stabilizers, batteries, and electronic components.

Toxicity of cadmium to aquatic organisms varies with water hardness, alkalinity, the type and life stage of organisms, presence of organic matter, presence of other toxicants, and the duration of exposure (USEPA 1999, as cited in USEPA 2008).

Cadmium is listed as an effluent limit for several point source discharges including Bourne Mining Corporation (permit number 101091), City of Dallas (permit no. 101627), Bonneville Dam (permit number 102768), Valley Landfill, Inc. (permit number101545), City of Wilsonville (permit number 101888), and various stormwater permits. It is also listed as a monitoring requirement for several WWTPs/STPs.

As of 2008 (USEPA 2008 p. 4-10) only one stream segment in the South Umpqua River of Oregon was listed on the 303(d) list. No listed species under authorities of USFWS reside in this listed stream.

Because there are widespread sources of cadmium, there is potential for bull trout, Oregon chub, Lost River and shortnose suckers to be exposed to criteria concentrations; see Section 4.4 for effects of cadmium on these listed species and their critical habitats.

Chromium

Eisler (1986, p. 4) reports that thousands of tons of chromium ore and concentrates are imported annually for the production of stainless steels, chrome plated metals, pigments for inks and paints, and a wide variety of chemicals and principally used in the U.S. are metallurgy and chemical industries. Chromium is a significant component of wastes discharged from municipal waste treatment facilities (Eisler 1986, p. 5). In aquatic environments, the major sources of chromium are the electroplating and metal finishing industries and publicly-owned treatment plants, while relatively minor sources are iron and steel foundries, inorganic chemical plants, tanneries, textile manufacturing, and runoff from urban and residential areas (Eisler 1986, p. 2). The BE (USEPA 2008) includes the following additional sources of chromium in aquatic systems: mining, leaching from landfills, and runoff from urban and residential areas.

Chromium VI is the only form listed in the Oregon 2010 Integrated Database (ODEQ 2011b). The lower Columbia River and Koll Wetland (Tualatin River Subbasin) are listed as 303(d), and the lower Columbia River; Coos Bay, Clackamas River, Columbia Slough, Johnson Creek, Willamette River, McKenzie River, Rickreall Creek, Santiam River, Beaverton Creek, Amazon Diversion Canal, Calapooia River, Middle Fourth Lake, and the Yamhill River are listed as potential concern.

Total chromium in streams is more likely to be trivalent, while the dissolved is more likely to be hexavalent as chromium III tends to precipitate out. In Oregon, very few surface water samples collected have a dissolved concentration that exceeds the chronic criterion level. The highest total chromium levels are on the coast near Coos Bay, which is where chromite sediments have collected and are being mined by Oregon Resources Corporation (Bill Mason, Oregon Department of Environmental Quality, Salem, Oregon, pers. comm., 2012).

Numerous Oregon NPDES permits include chromium as either an effluent limit or monitoring requirement, including stormwater permits, potentially exposing bull trout, Oregon chub, Lost River and shortnose suckers, and vernal pool fairy shrimp to this chemical at criteria concentrations. See Section 4.4 for effects of chromium on these listed species and their critical habitats.

Copper

Copper occurs in natural waters and is a minor nutrient for both plants and animals at low concentrations. However, it can also be toxic to aquatic life at concentrations only slightly higher than background (USEPA 2008, p. 4-11). According to Eisler (1998a, p. 4), copper is the major toxic component in streams impacted by active placer mines. Sources of copper in aquatic ecosystems include inputs from waste discharges into saline waters, industrial discharges into freshwater and leaching of antifouling marine paints and wood preservatives. Atmospheric fallout is a significant source of copper to the aquatic environment in mining and industrial areas (Eisler 1998a, p. 4). Copper is used routinely in fungicidal sprays, wood preservatives and to control nuisance algae (Eisler 1998a, p. 5).

Copper is included in many NPDES permit effluent limits and as a monitoring requirement, including numerous WWTP permits. Stormwater discharges also commonly contain copper, with the highest concentrations observed at highway sites (compared to industrial, commercial, institutional, open space and residential) (USEPA 2005b, p. 23). Copper is a common component (up to 25%) of automotive brake pads (Sandahl et al. 2007, Schaap et al. 2006). Brakes account for anywhere from 35 to 60% of copper in California's urban watershed runoff (Copper Development Association, Inc. 2011). Both California and Washington have passed legislation mandating a reduction in the amount of copper used in automotive brake pads (Copper Development Association, Inc. 2011, Puget Sound Partnership 2010). All streams on the 303(d) list that overlap with listed species' habitats are included in Table 4-5.

Due to the widespread and numerous potential sources of copper, bull trout, Oregon chub, Lost River and shortnose suckers, and vernal pool fairy shrimp could be exposed to copper at criteria concentrations. See Section 4.4 for effects of copper on these listed species and their critical habitats.

Lead

The BE (USEPA 2008) states that lead is a naturally-occurring, ubiquitous compound in freshwater. Higher concentrations are associated with highly mineralized regions or anthropogenic sources including mining, smelting, automobiles and road surface runoff, and municipal and industrial discharges. Natural lead compounds are not usually mobile in normal groundwater or surface water because the lead leached from ores becomes adsorbed by ferric hydroxide or tends to combine with carbonate or sulfate ions to form insoluble compounds (Hem 1976, as cited in USEPA 2008). Lead is a monitoring requirement and effluent limit in numerous NPDES permits.

Due to the widespread and numerous potential sources of lead, bull trout, Oregon chub, Lost River and shortnose suckers, and vernal pool fairy shrimp could be exposed to this chemical at

criteria concentrations. See Section 4.4 for effects of lead on these listed species and their critical habitats.

Nickel

Nickel occurs naturally in rocks and soils throughout the U.S. The main anthropogenic sources of nickel in water are primary nickel production, metallurgical processes, combustion and incineration of fossil fuels, and chemical and catalyst production (Eisler 1998b). The USEPA BE (2008, p. 4-13) reports additional anthropogenic sources including: wastewater; manufacture of cement and asbestos; food processing; textile and fur fabrication; and laundries and car washes. Additional anthropogenic activities contributing to nickel in aquatic environments include mining, smelting, refining, alloy processing, scrap metal reprocessing, fossil fuel combustion, and waste incineration (Eisler 1998b, p. 1).

Nickel is listed as a monitoring requirement in most major WWTP NPDES permits. It is included as a permit limit for several facilities including Arden Development and Arden Inc. (Green Diamond Products), Northwest Aluminum Company, Oregon Metalgical Corporation, TDY Industries (Wah Chang), and U.S. Army Corp of Engineers Bonneville.

At the time the BE was completed (USEPA 2008), nickel was not listed as a contaminant of concern for any water segments on Oregon's 2002 303(d) list. Table 4-5. Sources of each chemical considered LAA for listed species and critical habitat in this consultation.Table 4-5 lists waterbodies identified since that time.

Due to the widespread and numerous potential sources of nickel, bull trout, Oregon chub, Lost River and shortnose suckers, and vernal pool fairy shrimp could be exposed to this chemical at criteria concentrations. See Section 4.4 for effects of nickel on these listed species and their critical habitats.

Pentachlorophenol

The major commercial application (about 80%) of technical grade preparations of PCP is in wood preservation formulations (Eisler 1989a, p. 3). This is due to the fungicidal and bactericidal properties of the chemical which inhibit growth of wood-destroying organisms. However, it is also used as an insecticide, fungicide, herbicide, molluscicide, and bactericide (Eisler 1989a, p. 1). As a wood preservative, only certified applicators can purchase pentachlorophenol (USEPA 2008, p. 4-21).

In 1987, pentachlorophenol use was restricted through the cancellation and restriction of certain uses other than as a wood preservative (USEPA 2011). This action cancelled its residential uses as an herbicide, defoliant, mossicide, and disinfectant. Currently, its registered commercial uses include treatment of utility poles, fences, shingles, walkways, building components, peers, docks and porches. The USEPA has received and granted requests from registrants of pesticide products containing pentachlorphenol to terminate certain uses of their products. The action leaves allowable use of pentachlorophenol limited to pressure and thermal treatment.

In reviewing the point source database, seven NPDES permits with limits or monitoring requirements for pentachlorophenol were found (Table 4-4).

Table 4-4. NPDES permits with pentachlorophenol as either a monitoring requirement or effluent limit.

Facility	Facility Description	Discharge Location
Georgia Pacific	Pulp and Paper Manufacturing	Columbia River RM 42
Georgia Pacific	Tissue Paper and Secondary Fiber Pulp and Paper	Willamette River RM 148
Boise White Paper	Bleached Draft Pulp and Paper Mill	Columbia River RM 86 Multnomah Channel RM 1
Pacific Wood Preserving	Wood Preserving	South Yamhill River RM 39
Jeld-Wen	Wood Products Manufacturing Plant	Upper Klamath Lake
Cascade Wood Products	Millwork and Wood Treating	Ditch to Military Slough RM 1.6
J.H. Baxter	Wood Preserving	Ditch to Amazon Creek (Upper Willamette River Subbasin)
Permapost Products	Wood Preserving	Rock Creek RM 1 (Tualatin River Subbasin)
McFarland Cascade Pole and Lumber	Wood Preserving	Ditch to Amazon Creek (Upper Willamette River Subbasin)

The only aquatic species in the vicinity of these discharges listed in Table 4-4 are bull trout, and Lost River and shortnose suckers. Bull trout occurrence in the Columbia River is rare and believed to be mainly sub-adults and adults moving from the Hood River Basin seasonally foraging in late winter and spring. Consequently, potential exposure to bull trout from pentachlorophenol is very limited and considered discountable. Lost River and shortnose suckers reside in Upper Klamath Lake where there is an NPDES permit for discharge of pentachlorophenol; therefore they are likely to be exposed. Vernal pool fairy shrimp may be exposed from buried waste (see previous discussion of such sources under species-specific exposure analysis section). None of the known Oregon chub populations occur within 2 miles downstream of a permitted discharge of pentachlorophenol. Thus, we expect that Oregon chub will not be exposed to pentachlorophenol

The BE (USEPA 2008) states that pentachlorophenol was listed as the parameter of concern for RM 0 to 24.8 of the Willamette River. In the ODEQ 2010 assessment (ODEQ 2011b), PCP was listed as a parameter of concern for miles 0 to 24.8 in the Willamette River. This is a heavily industrialized area and no listed species under the jurisdiction of USFWS reside in these waters.

Considering the sources of pentachlorophenol, bull trout, Lost River and shortnose suckers, and vernal pool fairy shrimp could be exposed to this chemical at criteria concentrations. See Section 4.4 for effects of pentachlorophenol on these listed species and their critical habitats.

Selenium

The USEPA BE (2008) states, "Natural selenium introduction occurs via water percolation through seleniferous soils (Dobbs et al. 1996). Typical background concentrations of selenium have been estimated at 0.25 µg/L (Wilber 1980), 0.1 to 0.3 µg/L (Lemly 1985), 0.2 µg/L (Lillebo et al. 1988), and 0.1 to 0.4 µg/L (average <0.2, Maier and Knight 1994). Discharge of subsurface agricultural drainage water to surface waters is a major pathway for the mass-loading of selenium into aquatic ecosystems, especially in heavily irrigated arid Western states (Presser 1994; Presser et al. 1994; Seiler 1997; Presser and Piper 1998; Skorupa 1998; Seiler et al. 1999). Sulfide ore mining (other than coal) such as uranium, copper, bentonite, and phosphoria is also a common source of artificially mobilized selenium (Talcott and Moller 1997; USDI-BOR/USFWS/GS/BIA 1998). Most commercial livestock feeding operations and dairies add supplemental selenium to the feeds, and liquid manure pits beneath feed barns contained elevated selenium concentrations. This animal waste is often discharged to surface water bodies without any prior waste treatment (Oldfield 1994). Although most municipal wastewater treatment systems process non-seleniferous wastewater, mass-loading of selenium to surface waters can occur from public wastewater treatment facilities (Pease et al. 1992; CRWQCB 1995)."

Although selenium is not listed as an effluent limit for any of the Oregon permits provided in the spreadsheet by USEPA (Appendix 2), it is a monitoring requirement for numerous facilities including City of Coos Bay for Coos Bay, Roseburg Urban Sanitary Authority, County Service District of Clackamas (#1), City of Medford, City of Newberg, City of Troutdale, etc. Therefore, we assume that there is exposure to bull trout, Oregon chub, and Lost River and shortnose suckers located within proximity to WWTP/STP NPDES permits. Vernal pool fairy shrimp should not be exposed to selenium as there are no known sources near occupied habitat.

The USEPA (2008) reported that no waterbodies on the Oregon 2002 303(d) list were identified for selenium contamination, which is consistent with ODEQ 2010 assessment (ODEQ 2011b; Table 4-5).

Considering the sources of selenium, bull trout, Oregon chub, and Lost River and shortnose suckers could be exposed to this chemical at criteria concentrations. See Section 4.5.4 for effects of selenium on these listed species and their critical habitats.

Silver

Silver occurs naturally in the aquatic environment in trace quantities in association with basalt and igneous rock sources and levels are elevated naturally in water from hot springs and steam wells. Anthropogenic sources of silver include mining, industrial and smelting wastes, hazardous waste sites, cloud seeding with silver iodide, WWTPs, jewelry manufacture, electroplating plants, and the production and disposal of photographic material (USEPA 2008).

Silver is listed as a parameter of concern in the Fifteen Mile, Twelvemile, and Twentymile Creeks in the Warner Lakes Subbasin (Table 4-5) which are waterbodies that may be inhabited by Foskett speckled dace and Warner sucker. Because there are no known anthropogenic sources of silver in the habitats of these two fish species, we expect this is a result of naturally-occurring silver.

Silver is a common monitoring requirement for WWTPs and STPs; however, only one facility includes it as an effluent limit, Arden (Green) Development, Inc. in Riddle, Oregon, Douglas County. The outfall of Arden Development is into Crawford Creek in the Umpqua Basin which has no listed species under USFWS jurisdiction.

Because there are no point source discharges into vernal pool fairy shrimp habitat and any silver would be naturally occurring, adverse effects are insignificant and we concur with USEPA's determination that approval of the criteria would be NLAA this species. As bull trout, Oregon chub, and Lost River and shortnose suckers are in habitats with WWTP/STP discharges, they are expected to be exposed at criteria concentrations. See Section 4.4 for effects of silver on these listed species and their critical habitats.

Zinc

Zinc is naturally occurring in the aquatic environment, typically as leachate from igneous rocks. Major sources of anthropogenic zinc discharges to the environment include electroplaters, smelting and ore processors, drainage from active and inactive mining operations, domestic and industrial sewage, combustion of fossil fuels and solid wastes, road surface runoff, corrosion of zinc alloys and galvanized surfaces and erosion of agricultural soils (Eisler 1993, p. 5).

Zinc is a common parameter listed in monitoring requirements for sewage treatment facilities and stormwater permits. Several NPDES permits also have effluent limits for zinc, including permits for industrial and mining facilities. According to the BE (USEPA 2008), zinc is listed as the parameter of concern in five small stream segments on Oregon's 2002 303(d) list. These include four creeks in the Lower Columbia, Middle Columbia-Hood, and Middle Willamette River Subbasins, as well as the Koll Wetland in the Tualatin River Subbasin. Table 4-5 contains waterbodies listed in ODEQ 2010 assessment (ODEQ 2011b).

Zinc sources are widespread through habitats of bull trout, Oregon chub, Lost River and shortnose suckers and vernal pool fairy shrimp. Therefore, these aquatic species may be exposed to zinc at criterion concentrations. See Section 4.4 for effects of zinc on these listed species and their critical habitats.

Table 4-5. Sources of each chemical considered LAA for listed species and critical habitat in this consultation.

Chemical	*NPDES Permits*	*303(d) Listings*	*Mining*
Aluminum	• Northwest Aluminum Co. → Columbia River • Georgia Pacific Consumer Products → Lower Columbia • Port of Portland (Reynolds Metals) • Arden Development Corp.	none	X
Ammonia	Numerous	5 – Klamath River 3B – Crooked River 3B- Odell Lake/Odell Creek 5 – Klamath Strait 5 – Lost River 5 – Klamath River/Ewauna River 3B – Grande Ronde River 3B – McKay Creek 4A – Umatilla River 3B – Wildhorse Creek 3B – Pine Creek 3B – Willow Creek 3B - Powder River 3B – Ecola Creek 3B – Netarts Bay 3B – Coquille River 3B – South Fork Coquille River 3B – Ashland Creek 3B – Diamond Lake 4B – North Myrtle Creek 3B – North Myrtle Creek 3B – North Fork Little Butte Creek/Fish Lake 5 – Arata Creek/Blue Lake 3B – Columbia Slough 3B – Gales Creek	
Arsenic	Numerous	3B – Columbia River (Lower Columbia – Clatskanie) 3B – Columbia River (Lower Columbia – Sandy; Lower Willamette) 3B – Clackamas River 3B – Willamette River 3B – McKenzie River 3B – Santiam River 3B – Beaverton Creek 3B – Amazon Diversion Canal 3B – Calapooia River 3B – Palmer Creek 3B – Yamhill River	X

Table 4-5. Sources of each chemical considered LAA for listed species and critical habitat in this consultation.

Chemical	*NPDES Permits*	*303(d) Listings*	*Mining*
Arsenic- cont.	Numerous	3B – Amazon Creek 3B – South Umpqua River 3B – North Umpqua River 3B – Sutherlin Creek 3B – Twenty Mile Creek 5 – Jordan Creek 5 – Owyhee River 5 – Middle Creek	
Cadmium	Numerous	3B – Columbia River 3B – Silver Creek 3B – Nehalem River 3B – Columbia Slough 3B - Beaverton Creek 3B – Amazon Creek 3B – Amazon Diversion Canal 3B – South Umpqua River 5 – South Fork Middle Creek 5 – Middle Creek	X
Copper	Numerous	3B – Columbia River (Lower Columbia; River Segment 0 – 35.2) 3B – Beaver Creek 3B – Lower Columbia (Lower Columbia – Clatskanie; Lower Willamette; River Segment 35.2 – 98) 3B – Columbia River (Lower Columbia – Sandy; Lower Willamette; River Segment 98 – 142) 3B – Middle Fork Hood River 3B – Silver Creek 3B – Coos Bay 3B – Clackamas River 3B – Johnson Creek 3B – Willamette River (Lower Willamette; River Segment 0-24.8) 3B – McKenzie River 3B – Rickreall Creek 3B – Willamette River (Middle Willamette; River Segment 24.8 – 54. 8) 3B – Milk Creek 3B – Santiam River 3B – Beaverton Creek 3B – Amazon Diversion Canal 3B – Calapooia 3B – Middle Fourth Lake 3B – Willamette River (Upper Willamette; River Segment 119.7 – 148.8) 3B – Willow Creek 3B – Yamhill River	X

Table 4-5. Sources of each chemical considered LAA for listed species and critical habitat in this consultation.

Chemical	*NPDES Permits*	*303(d) Listings*	*Mining*
Copper	Numerous	303(d) – Unnamed Creek (1111111111134) 5 – East Fork Hood River 5 – Evans Creek 5 – Hood River 303(d) - Overstreet Drain 303(d) – Unnamed (Fletcher Drain; 1171410437824) 5 – Sutherlin Creek 5 – Middle Creek 5 – South Fork Middle Creek 5 – Cook Creek 303(d) – Pringle Creek 303(d) – Koll Wetland 5 – Amazon Creek 5 – Amazon Creek Diversion Canal	
Lead	Numerous	3B – Columbia River (Lower Columbia – Clatskanie Lower; River Segment 0 – 35.2) 3B – Columbia River (Lower Columbia – Clatskanie; Lower Willamette; River Segment 35.2 – 98) 3B – Columbia River (Lower Columbia – Sandy; Lower Willamette; River Segment 98- 142) 3B – Nehalem River 3B – Deadwood Creek 3B – Coos Bay 3B – Willamette River 3B – Beaverton Creek 3B – Fanno Creek 3B – Amazon Creek 303(d) – Overstreet Drain 303(d) – Unnamed (Fletcher Drain) 1171410437824 5 – Sutherlin Creek 303(d) – Unnamed Creek (1111111111136) 5 – Cook Creek 4a – Columbia Slough 303(d) – Pringle Creek 303(d) – Koll Wetland 303(d) – Amazon Creek 5 – Amazon Creek Diversion Canal	X
Nickel	Numerous	3B – Columbia River (Lower Columbia – Clatskanie; River Segment 0 – 35.2) 3B – Beaver Creek 3B – Columbia River (Lower Columbia – Clatskanie; Lower Willamette; River Segment 35.2 – 98)	X (Metal Mining)

Table 4-5. Sources of each chemical considered LAA for listed species and critical habitat in this consultation.

Chemical	*NPDES Permits*	*303(d) Listings*	*Mining*
Nickel – cont.	Numerous	3B – Columbia River (Lower Columbia – Sandy; Lower Willamette; River Segment 98 – 142) 3B – Coos Bay 3B – South Fork Middle Creek 3B – Clackamas River 3B – Johnson Creek 3B – Willamette River (Lower Willamette; River Segment 0 – 24.8) 3B – McKenzie River 3B – Rickreall Creek 3B – Willamette River (Middle Willamette; River Segment 24.8 – 54.8) 3B – Santiam River 3B – Beaverton Creek 3B – Amazon Diversion Canal 3B – Calapooia River 3B – Middle Fourth Lake 3B – Willamette River (Upper Willamette; River Segment 119.7 – 148.8) 3B – Yamhill River 303(d) – Overstreet Drain 303(d) – Unnamed (Fletcher Drain 1171410437824) 5 – Sutherlin Creek 303(d) – Unnamed Creek (1111111111136) 5 – Cook Creek 4a – Columbia Slough 303(d) – Pringle Creek 303(d) – Koll Wetland 303(d) – Amazon Creek 5 – Amazon Creek Diversion Canal	
Pentachloro-phenol	• Georgia Pacific → Columbia River RM 42 • Georgia Pacific → Willamette River RM 148 • Boise White Paper→ Columbia River RM 86, Multnomah Channel RM 1 • Pacific Wood Preserving → South Yamhill River RM 39 • Jeld-Wen→ Upper Klamath Lake • Cascade Wood Products → Ditch to Military Slough RM 1.6	3B – Amazon Diversion Canal 303(d) – Willamette River	

Table 4-5. Sources of each chemical considered LAA for listed species and critical habitat in this consultation.

Chemical	*NPDES Permits*	*303(d) Listings*	*Mining*
Pentachloro-phenol – cont.	• J.H. Baxter → Ditch to Amazon Creek (Upper Willamette River Subbasin) • Permapost Products→Rock Creek RM 1 (Tualatin River Subbasin) • McFarland Cascade Pole and Lumber→Ditch to Amazon Creek (Upper Willamette River Subbasin)		
Selenium			X
Tributyltin	• Vigor Industrial → Willamette River RM 8	3B – Columbia River (Lower Columbia – Clatskanie; River Segment 0 – 35.2) 3B – Columbia River (Lower Columbia – Clatskanie; Lower Willamette; River Segment 35.2 – 98) 3B – Columbia River (Lower Columbia-Sandy; Lower Willamette; River Segment 98-142) 3B – Catching Slough 3B – Coos Bay 3B – North Slough	
Silver	Numerous	3B - Columbia River (Lower Columbia – Clatskanie; River Segment 0-35.2) 3B - Columbia River (Lower Columbia – Clatskanie; Lower Willamette; River Segment 35.2-98) 3B- Columbia River (Lower Columbia – Sandy; Lower Willamette; River Segment 98 -142) 3B – Beaverton Creek 3B – Amazon Diversion Canal	X
Zinc	Numerous	3B - Columbia River (Lower Columbia-Clatskanie; River Segment 0 -35.2) 3B- Beaver Creek 3B – Columbia River (Lower Columbia-Clatskanie; Lower Willamette; River Segment 35.2 – 98) 3B – Columbia River (Lower Columbia – Sandy; Lower Willamette; River Segment 98 – 142) 3B – Coos Bay 3B – Johnson Creek 3B – Willamette River (Lower Willamette; River segment 0 – 24.8) 3B – Rickreall Creek 3B – Willamette River (Middle Willamette;	X

Table 4-5. Sources of each chemical considered LAA for listed species and critical habitat in this consultation.

Chemical	*NPDES Permits*	*303(d) Listings*	*Mining*
Zinc – cont.	Numerous	River Segment 24.8 – 54.8) 3B – Mill Creek 3B – Beaverton Creek 3B – Amazon Creek 3B – Amazon Diversion Canal 3B – Middle Fourth Lake 3B – Yamhill River 303(d) – Unnamed Creek (1111111111134) 5 – Mitchell Creek 5 – Middle Creek 5 – South Fork Middle Creek 303(d) – Pringle Creek 303(d) – Koll Wetland	

4.2 Effects Analysis Methods Summary

In this consultation, the USEPA evaluation of potential adverse effects to listed species follows the standard ecological risk assessment paradigm (USEPA 1998) and their finding for each criterion is based upon multiple lines of evidence including assessments of water column, prey, multiple routes of exposure, wildlife-dependent effects, and exposure. Our process also follows ecological risk assessment processes, although the organization and certain technical aspects differ from that of the BE. The effects analysis of the Biological Opinion follows the analysis stage of risk assessment and includes characterizations of exposure and ecological effects. Methodologies used to characterize ecological effects are lengthy and complex. While our approach is outlined in section 4.1.1.1 of the Biological Opinion, the detailed description of these methodologies is included as Appendix 1.

Any risk assessment has areas of scientific uncertainty and relies on unproven assumptions to make risk determinations, especially in evaluations of listed species where toxicity data are sparse. In the BE methodology, we found the level of uncertainty associated with some determinations of adverse effects from proposed approval of the toxics water quality criteria was too high to confidently err on the side of caution for protection of the species. Within our assessment, we use the same models as USEPA with slight refinements to either reduce uncertainty or provide for a more conservative decision when faced with such uncertainty. These refinements give us greater confidence in the risk determinations related to vulnerable threatened and endangered species populations. A discussion of the uncertainties within the risk assessment models used in the BE is incorporated into Appendix 1 along with a description of our approach to refining them.

In analyzing the effects of the action on listed species, we have considered the best available laboratory information to estimate how these species or surrogates would respond from exposure to toxic chemicals in natural settings. We have estimated a level of effect from acute and chronic criteria, which is essential to our jeopardy determinations, adverse modification determinations

and incidental take estimates. A description of our approach is also incorporated into the methods section included in Appendix 1. The tables showing the estimated level of effect for each listed species and each chemical, models we used to estimate these effects, and intrepetration guidance for selecting the models are included in Appendices 3 to 6.

In order to complete our analysis of effects we must consider the frequency of the action. The CMC and CCC are described temporally as measures of 1 hour and 4 days, respectively, not to be exceeded more than once every 3 years. Specifically, any adverse acute effects are considered to occur one time for 1 hour every 3 years (consistent with the duration and frequency components of the CMC). Similarly, any adverse chronic effects are considered to occur one time for a period of 4 days every 3 years (consistent with the duration and frequency components of the CCC).

One way to estimate the total extent of the adverse effects, based on this recurrence interval of 3 years, is to multiply the estimated adverse effects in a 3-year period by the number of 3-year recurrence intervals that would occur during the length of time the action is in effect. The action under consideration in this consultation is in effect for 25 years (see section 1.1), so there will be 8.33 recurrence intervals (of 3 years each) over this time span. To illustrate further with a hypothetical example: if a level of effect for an endangered fish estimates that 10 fish are likely to die as a result of exposure to chemical X at criterion concentrations then over the 25-year period of the action a total of 83.3 fish will be estimated to die (10 fish x 8.33 recurrence intervals). Although this is one way to estimate the total extent of the adverse effects, we believe this multiplication misrepresents the effect to the population as it only takes into account the current population (level of effect based on percentage of current population) and not the population size when the next event is estimated to occur. There is no way of estimating what the population of a species will be in 3 years, 6 years, 9 years, etc. Consequently, our analysis of adverse effects for each species and each chemical is estimated as a percentage of the current population and expected to occur every 3 years; with this percentage and the current population estimate, we can estimate the number of individuals that are likely to be affected in the first 3-year interval of this action. Although we expect the adverse effects to be the same percentage of the future population, we cannot estimate the number of individuals affected since we do not know the likely population size in future 3-year intervals.

4.3 Chemical Modes of Action, Multiple Routes of Exposure, and Bioaccumulation

This section briefly describes the toxic modes of action of each chemical where co-occurrence with listed species is likely, and addresses multiple routes of exposure. Multiple routes of exposure refer to residues accumulated in tissue from both waterborne and dietary exposure at chronic criteria concentrations. In this section we evaluate the contribution of the dietary exposure pathway to adverse effects associated with tissue concentrations. In section 4.4 "Effects to Listed Species" we address the contribution of water-only exposure for these chemicals. Because of its unique characteristics in regards to bioaccumulation, selenium is also addressed separately in section 4.5.

4.3.1 Metals

4.3.1.1 Aluminum

Although aluminum is a common element in the environment, it is typically bound in soil and insoluble in water at normal pH (Gensemer and Playle 1999, pp. 316-318). Water solubility increases with decreasing pH and in alkaline and other conditions (Gensemer and Playle 1999, pp. 316-318; Wren and Stevenson 1991, p. 227). The USEPA BE (2008 p. 4-16) states:

> "Aluminum, like other metals, generally acts as a surface active toxicant, exerting its damage by binding to anionic sites on respiratory surfaces of aquatic animals, such as a fish gill (Wood et al. 1997). The physiological manifestation of these deleterious surface effects at the gill includes both ionoregulatory and respiratory effects. Ionoregulatory effects of aluminum predominate at low pH (e.g., less than pH 5.0) and include a mechanism similar to hydrogen ion toxicity alone, i.e., sodium uptake blockade (Playle et al. 1989). In moderately acidic water, it is generally the respiratory effects of aluminum that predominate. Respiratory effects are likely the result of the physical coating of the gills which occurs when aluminum-rich water passes into the more basic gill microenvironment (Gensemer and Playle 1999). Overall, the acute and chronic aluminum toxicity to fish species is substantially greater at low pH, particularly for salmonids. For many fish, aluminum toxicity increases with early life stage such that eggs and endogenously-feeding alevins are generally less sensitive than exogenous-feeding swim-up larvae (Buckler et al. 1985; DeLonay et al. 1993). Aluminum at extreme low pH (pH <5) can protect against the direct toxic effects of hydrogen ion. Several factors ameliorate aluminum toxicity at low pH, including, but probably not limited to: calcium ion (Brown 1983; Ingersoll et al. 1990), silicic acid (Birchall et al. 1989), fluoride (Wilkinson et al. 1990), and dissolved and natural organic matter (Parkhurst et al. 1990; Roy and Campbell 1997). Aquatic invertebrates are generally less sensitive to aluminum compared with fish."

Aluminum has little value to biological systems and no nutritional value (Gensemer and Playle 1999, p. 316). Aluminum does not biomagnify in food chains and, depending on certain ambient conditions, does not readily accumulate in tissues of aquatic organisms (Wren and Stevenson 1991, p. 227). Bioaccumulation is increased at lower pH and dose-dependent increases in accumulation have been observed in *Daphnia magna* (Havas 1985, p. 1747) and rainbow trout when exposed at high oral concentrations (Handy 1993, p 603). Handy (1993, p. 605) found very little mortality in trout at high doses, and suggest that regurgitation along with secretions of aluminum in the mucous was a form of detoxification for the trout. The authors also indicate that (p. 605) aluminum concentrations in prey species in the wild would have little adverse effect to trout. Wren and Stephenson (1991, p. 231) indicate that ambient concentrations of aluminum typically pose little threat to invertebrates in natural waters.

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore,

the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-67–5-69) compared the tissue-residue C_A calculated for arsenic of 4.4 mg/kg wet weight to available whole body residue-effects studies in fish and invertebrates to identify where effects were reported below the C_A. No studies were found that reported adverse effects to fish or invertebrates at residue concentrations below the C_A, and USEPA concluded that no adverse effects would occur to listed species based on multiple routes of exposure at the chronic criterion.

Based on USEPA's assessment, our review of the available literature regarding aluminum bioaccumulation, and the limited exposure of aluminum to listed species under our jurisdiction (see Exposure Analysis), we found no evidence that dietary aluminum would contribute any additional adverse effects beyond water only exposure to invertebrates or fish.

4.3.1.2 Arsenic

Arsenic occurs in natural waters as both inorganic and organic forms, and organic forms are reported to be much less toxic to aquatic biota than inorganic forms (Shaw et al. 2007b, p. 1532). The inorganic forms include arsenic III and arsenic V, and organic forms are methanearsonic acid and dimethylarsinic acid (USEPA 1985a, p. 1). USEPA (1985a, pp. 1-2) treated these compounds separately in the 1984 criteria document and identified acid-soluble forms III and V as the best measurements for expressing aquatic life criteria. We did not evaluate arsenic forms separately for the action under consideration in this consultation. Arsenic criteria are not hardness dependent and the conversion factor from dissolved to total is 1.000.

The BE (USEPA 2008, p. 5-112) reports that arsenic is a nutritionally essential element in the diet of birds and mammals at low concentrations. Arsenic is considered to be less toxic to aquatic organisms than other elements such as mercury silver and cadmium (see review paper Spehar et al. 1980, p. 54). Arsenic III appears to exert toxicity by reacting with sulfhydryl groups of proteins and inactivating enzymes (Webb 1966, p. 617; Peoples 1975, p. 9). Information is sparse on mode of action of other arsenicals, but arsenic V may uncouple oxidative phosphorylation (Eisler 1988a, p 8).

Arsenic is accumulated by freshwater organisms but is not reported to biomagnify along the aquatic food chain (see reviews by Eisler 1988a, p. 53 and Spehar et al. 1980, p. 61). Bottom feeding fish accumulate more arsenic than pelagic fish due to their association with sediments (Sorensen 1991, p. 71). Bioconcentration factors are relatively low for aquatic organisms, with the exception of algae (Eisler 1988a, p. 54). Spehar et al. (1980, p. 61) reported fish appear to be relatively tolerant to arsenic accumulation, and may have the ability to metabolize arsenic more than invertebrates.

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a

chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-113–5-114) compared the tissue-residue C_A calculated for arsenic of 6.6 mg/kg wet weight to available whole body residue-effects studies in fish and invertebrates to identify where effects were reported below the C_A. Two studies reported decreased juvenile survival and growth to bluegill at tissue concentrations of 0.97 mg/kg wet weight, and reduced growth in rainbow trout at 1.7 mg/kg wet weight (i.e., both values where below the C_A). In addition, USEPA (2008, pp. 5-113 – 114) identified studies on cladocerans where *Moina macrocopa* had reduced growth at 3.58 mg/kg wet weight and *D. magna* had 48% reduction in number of offspring when tissue concentrations were 5 mg/kg wet weight organic arsenic. In their secondary assessment, USEPA (2008, p. 5-515) found that all four reported studies on fish and invertebrates had higher exposure concentrations (ranging from 690 µg/L for bluegill to 18,000 µg/L for rainbow trout and over 1,000 µg/L for cladocerans) compared to the chronic criterion of 150 µg/L. These exposures resulted in bioaccumulation factors substantially lower than the 44 used by USEPA, and they considered that the tissue-residue C_A concentration was actually lower than all measured arsenic residues in aquatic biota associated with adverse effects after incorporating a revised bioaccumulation factor.

Studies of fish indicate arsenic can accumulate in tissues as exposure increases (Oladimeji et al. 1984, p. 734; Erikson et al. 2011, p. 112), but typically concentrations are lower compared to other metals such as cadmium and lead (Spehar et al. 1980, p. 61). Erickson et al. (2011, pp. 112-113) noted impacts to trout fed arsenic-exposed oligochaetes, but the exposure values from this bioaccumulation study as well others (Spehar et al. 1980, p. 57; Oladimeji et al. 1984, p. 737) were well above chronic criteria concentrations.

Based on USEPA's assessment and our review of the available literature regarding arsenic bioaccumulation, we found little supporting evidence that dietary arsenic would contribute any additional, significant adverse effects beyond water only exposure to invertebrates or fish.

4.3.1.3 Cadmium

While cadmium is a naturally occurring element, it is not biologically essential or beneficial and is highly toxic to aquatic organisms (Eisler 1985a, pp. 1 and 14). Cadmium concentrations are typically higher in biota near heavily urbanized and industrialized sites or near sites with evidence of historic mining pollution (Wayland and Scheuhammer 2011, p. 645). Cadmium could adversely affect food organisms since it is extremely toxic to some aquatic invertebrates (Eisler 1985a, Mebane 2006).

The USEPA BE (2008, p. 4-9) states:

> "Cadmium is bioconcentrated by organisms but is not biomagnified through the food chain (Eisler 1985a). Toxicity of cadmium to aquatic organisms varies with water hardness, alkalinity, the type and life stage of organisms, presence of organic matter,

> presence of other toxicants, and the duration of exposure (EPA 1999). Cadmium is a known teratogen, carcinogen, and a probable mutagen to freshwater organisms (Eisler 1985a). Effects of cadmium toxicity to freshwater organisms include spinal deformities, inhibited respiration, immune response, temporary immobility, decreased growth, inhibited reproduction, decreased survival, and population alterations (Sorensen 1991, Eisler 1985a, Brent and Herricks 1998, Sanchez-Dardon et al. 1999)."

Mebane (2006, p. 30) summarizes cadmium toxicity as follows:

> "Waterborne cadmium can cause severe, acute toxicological and physiological effects to aquatic organisms. At acute waterborne concentrations, cadmium severely disrupts calcium homeostasis, which ultimately leads to death. At the gill surface, cadmium competes with calcium for high affinity calcium-binding sites and once it enters the chloride cell irreversibly blocks calcium uptake. The cumulative effect of these two processes causes acute hypocalcaemia in freshwater fish (Wood and others, 1997). However, these effects can be altered by the calcium content of the water. Among various water-quality parameters that often influence the uptake and toxicity of metals, calcium seems to be of primary importance, with pH of secondary importance. Other water-quality parameters that influence the toxicity of other metals (such as copper) probably are less important in modifying cadmium toxicity (dissolved organic matter (DOM), magnesium, sodium, and alkalinity)."

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-141–5-144) compared the tissue-residue C_A calculated for cadmium of 0.0056 mg/kg (or 6 µg/g) wet weight to available whole body residue-effects studies in fish and invertebrates to identify where effects were reported below the C_A.
Only one study on Atlantic salmon reported adverse effects at a residue concentration of 5 µg/g which was below the C_A (USEPA 2008, pp. 5-141–5-144). Although this study showed effects to growth of alevins from dietary exposure at the C_A concentration, salmon exposed at higher concentrations and for longer duration did not show reduction in growth in this study (Rombough and Garside 1982, p. 2011). The USEPA did not conduct a secondary assessment on this study and made no conclusions regarding the data or its relevance to the consultation.

Mebane (2006, p. 69) reviewed bioaccumulation information on cadmium and concluded the data were insufficient to define bioaccumulation factors and predict risk to organisms based back calculating tissue concentrations. However, the author also concludes that bioaccumulation and associated effects to invertebrates or fish are unlikely to occur at the chronic concentration.

We conclude that additional adverse effects from accumulation of cadmium in tissues would not contribute significantly to the adverse effects of the overall toxicity. Based on residue effects data provided in the USEPA BE, the Mebane (2006) review of cadmium effects and tissue concentrations, and our review of the available literature regarding cadmium accumulation, we found little supporting evidence that dietary cadmium would contribute any additional adverse effects beyond water only exposure to invertebrates or fish.

4.3.1.4 Chromium

Chromium is a common naturally occurring element found in the environment, although concentrations can be augmented by a number of industrial processes (Eisler 1986, pp. 1-8). The more stable states of chromium include the trivalent (III) and hexavalent (VI) oxidation states (Eisler 1986, pp. 2-3). Chromium III is considered an essential nutrient while chromium VI is not. Chromium VI is the dominant dissolved form, as chromium III tends to precipitate out of solution (Eisler 1986, p. 5). The USEPA BE (2008, p. 4-10) states:

> "In freshwater environments, hydrolysis and precipitation are the most important processes in determining the environmental fate of chromium, while absorption and bioaccumulation are considered minor. Chromium (VI) is highly soluble in water and thus very mobile in aquatic systems (Ecological Analysts 1981).
>
> Chromium may be present in the environment in both inorganic and organic forms. Inorganic forms do not biomagnify; it is unknown whether organic forms of chromium biomagnify (Eisler 1986b). Chromium toxicity to aquatic biota is significantly influenced by abiotic variables such as water hardness, temperature, pH, salinity, species, life stage, and presence of mixtures (Eisler 1986b). Sensitivity to chromium varies widely, even among closely related species (Eisler 1986b). Effects of chromium toxicity to freshwater organisms include reduced survival in freshwater invertebrates (including molluscs), and reduced growth, reduced disease resistance, behavioral modifications, disrupted feeding, cell damage in the gills, osmoregulatory upset in outmigrating smolts, and reduced reproduction and survival in freshwater fish (Anestis and Neufeld 1986, Eisler 1986b and EPA 1999).
>
> Hexavalent chromium is more toxic than the trivalent form because its oxidizing potential is high and it easily penetrates biological membranes (Steven et al. 1976, Taylor and Parr 1978). At high concentrations, both forms of chromium can be a mutagen, teratogen, and carcinogen (Eisler 1986b). Although CrIII is the most common form found in nature, the known harmful effect of chromium is speculated to be related to the reduction of hexavalent chromium (CrVI) to CrIII intracellularly as it crosses the cell membrane and forms complexes with intracellular macromolecules (Danielsson et al. 1982; R.O.W. Sciences, 1997)."

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore,

the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-163–5-166) compared the tissue-residue C_A calculated for chromium III and chromium VI of 8.51 and 0.44 mg/kg wet weight, respectively, to available whole body residue-effects studies in fish and invertebrates to identify where effects were reported below the C_A. Although no toxicity studies of fish and invertebrates were found to have adverse effects below the C_A for chromium VI, two studies on invertebrates (a sand crab and a polychaete) and one study on Chinook salmon parr were found to have adverse effects below the C_A for chromium III. The fish and invertebrates in these studies were exposed using chromium VI, and USEPA stated it was unknown if the lowest observed effects residue values from these studies were associated with trivalent or hexavalent chromium and therefore, presumably, unknown which C_A to use for comparison.

The Chinook study found reduction in growth and other adverse effects in parr at a chromium VI dose of 24 µg/L (increased to 120 µg/L by day 105) over a period of 134 days (Farag et al. 2006, p. 251). The whole body concentration at the end of the test for this dose range (6.4 µg/g dry weight, or 1.28 µg/g wet weight based on 80% moisture as estimated by USEPA) was below the C_A for chromium III and above the C_A for chromium VI. Although the specific form of chromium in the whole body concentrations may be irrelevant in regards to estimating effects in this study, we do have concerns that 1) the dose for part of the test exceeded the chronic criterion of 74 µg/L; 2) the authors noted that "… while a trend existed, there was no longer a significant increase in whole body concentrations of Cr at day 134, indicating the limited usefulness of whole body Cr measurements with age", and 3) no significant differences in growth were observed at higher doses compared to reference fish (Farag et al. 2006, p. 251).

Based on USEPA's assessment and our review of the available literature regarding chromium bioaccumulation, we found little evidence that dietary cadmium would contribute significant or measureable adverse effects beyond water only exposure to fish. Too little information is available to determine if chromium accumulation in invertebrates would contribute to adverse effects to the vernal pool fairly shrimp, therefore, we rely on the direct exposure evaluation for assessing effects.

4.3.1.5 Copper

Copper is a naturally occurring element generally present in surface waters (Boyle 1979, p. 77). Although recognized as an essential micronutrient in living organisms (Stokes 1979, p. 358), at elevated concentrations copper may become toxic (Leckie and Davis 1979, p. 90). The USEPA BE (2008 p. 4-11) states:

> "The distinction between copper deficiency and toxicity is small in organisms such as algae and invertebrates that lack effective mechanisms to control absorption (EPA 1999). Copper is not strongly bioconcentrated in vertebrates but is more strongly bioconcentrated in invertebrates (EPA 1999). Toxicity of copper to aquatic organisms is dependent on pH, temperature, alkalinity, hardness, and concentrations of bicarbonate,

sulfide, and organic ligands (EPA1980b), as well as the type and life stage of exposed organism (EPA 1999). Copper is among the most toxic of the heavy metals to freshwater biota (Schroeder et al. 1966, Betzer and Yevich 1975). In general, mortality of tested aquatic species is greatest under conditions of low water hardness, starvation, elevated water temperatures, and among early developmental stages (Eisler 1998a). Effects of copper toxicity to freshwater organisms include valve closure, reduction in filtration rates, impaired structure and function of cellular membranes, and cardiac inhibition in mussels. Impaired disease resistance, disrupted migration (via avoidance behavior of Cu-contaminated areas), hyperactivity, impaired respiration, disrupted osmoregulation, pathology of kidneys, liver, and gills, impaired function of olfactory organs and brain, altered blood chemistry, and enzyme activity have been documented in fish (Eisler 1998a)."

Gills are the principal site of toxicity in freshwater organisms at exposure concentrations producing acute effects (Wood et al. 1997, pp. 33-36). Copper acts at the surface exerting damaging effects by binding to anionic sites on or in the gills. When the metal adheres at these anionic sites it blocks sodium uptake, an essential gill transport function. Toxicities of metals may be reduced in water with higher Ca2+ and DOC concentrations and H+ ions (Wood et al. 1997, pp. 36-37) as Ca2+ and H+ compete for the binding site, while DOC complexes with the copper. Brix et al. (2001, p. 1,848) indicated that temperature cold-water species appear to be more sensitive than temperate warm-water species, which in turn appear to be more sensitive than tropical species.

Eisler's review of copper (1998a) states that bioavailability and toxicity of copper to aquatic organisms is dependent on total concentration and speciation. Copper interacts with numerous compounds normally found in natural waters. Depending on chemical characteristics of the water, much of the copper may be associated with suspended solids and not available to biota. Copper toxicity to aquatic biota is related primarily to the dissolved cupric ion (Cu+2) which is largely complexed with carbonate, amino acids, or humic substances (Eisler 1998a). Copper tends to hydrolyze and precipitate at most pH values expected in natural waters, or combines with organic complexing agents. Sorensen (1991, p. 239) states, "Copper is highly complexed with carbonate and hydroxide ions in natural waters and, as a result, free Cu2+ ion is usually low in concentration…". In addition, she notes that most natural water systems contain a vast array of organic complexing agents which can substantially reduce free Cu2+ concentrations in aqueous systems including humic and fulvic acids, amino acids and suspended solids , and nitrogenous waste products. Absorption of the free Cu2+ by sand, gravel, and suspended clay is another important mechanism in natural water systems (Sorensen 1991, p. 241). Synthetic water solutions commonly used in toxicity testing are lacking many of these complexing agents normally present in an aquatic species natural environment thereby amplifying adverse effects in a laboratory study. The decreased bioavailability is likely one of the reasons that bioaccumulation has rarely been demonstrated for copper (Stokes 1979, p. 359), and it is not known to biomagnify (Stokes 1979, p. 376).

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the

chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, p. 5-217 – 5-221) compared the tissue-residue C_A calculated for copper of 2.6 mg/kg wet weight to available whole body residue-effects studies in fish and invertebrates to identify where effects were reported below the C_A. The USEPA BE states (p. 5-219) that "The residue effect studies on reproduction and growth indicate that the copper residue required to elicit toxicity in aquatic biota is higher than the amount of copper that could reasonably be expected to bioaccumulate in aquatic species exposed to the copper chronic criterion concentration (2.6 mg/kg wet weight) for all of the 18 species for which data exist." The copper tissue residues identified by USEPA (p. 5-218) as 10 mg/kg for the highest no effect residue for rainbow trout and 25 mg/kg as a toxic for invertebrates.

Based on USEPA's assessment and our review of the available literature regarding copper bioaccumulation, we find little supporting evidence that dietary arsenic would contribute any additional adverse effects beyond water only exposure to invertebrates or fish.

4.3.1.6 Lead

Lead is a nutritionally non-essential inorganic element with a potential to accumulate in tissues of aquatic animals (USEPA 2008 p. 5-330), yet biomagnification is negligible Eisler (1988b, p. 55). Like many metals, lead chemistry in freshwater is complex. Aquatic life criteria are hardness based, however, lead is most soluble and bioavailable under conditions of low pH, low organic content, low concentrations of suspended sediments, and low concentrations of calcium, iron, manganese, zinc and cadmium salts (Eisler 1988b, p. 9). Thus, aside from areas of point source discharges, solubility of lead in water is low, and most lead entering natural waters is precipitated to the sediment bed (Eisler 1988b, p. 10). Although highways are a source of lead contamination from automobile exhausts (Sorensen 1991, p. 97), lead is no longer used in gasoline and this source should be reduced. The USEPA BE (2008, p. 4-12) states:

> "The bioavailability of lead increases in environments with low pH, low organic content, and low metal salt content (Eisler 1988b). Toxicity of lead to aquatic organisms varies with water temperature, pH, water hardness, metal salt concentrations, organic matter, and suspended solid concentration (EPA 1999). Invertebrates tend to have higher bioconcentration factors than vertebrates (EPA 1999). Effects of lead toxicity to freshwater organisms include reduced growth, spinal curvature and other deformities, anemia, caudal fin degeneration, destruction of spinal neurons, enzyme inhibition, reduced swimming ability, increased mucus formation and coagulation over body and gills and destruction of respiratory epithelium, scale loss, muscular atrophy and paralysis, impaired reproduction, and reduced survival (Hodson et al. 1982, Eisler 1988b, Sorensen 1991, Farag et al. 1994). Organic lead compounds are generally more toxic than inorganic (Eisler 1988b)."

Grosell et al. (2006) studied the influence of calcium, dissolved organic carbon, and pH on chronic water-borne lead toxicity to larval fathead minnow. The authors found (p. 482) that these three water chemistry parameters clearly influenced chronic lead toxicity. Lead is chemical similar to calcium and is processed along with calcium and can replace calcium in bone tissue (Sorensen 1991, p. 95).

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-330–5-331) compared the tissue-residue C_A calculated for lead of 0.059 mg/kg wet weight to available whole body residue-effects studies in fish and invertebrates to identify where effects were reported below the C_A. The studies USEPA evaluated indicate that the lead tissue residue required to elicit toxicity in aquatic species is higher than the amount of lead that could reasonably be expected to bioaccumulate in aquatic species exposed to chronic lead criterion concentrations.

Based on USEPA's assessment and our review of the available literature regarding lead bioaccumulation, we conclude that dietary lead would not contribute any measureable or significant adverse effects beyond water only exposure to invertebrates or fish.

4.3.1.7 Nickel

Nickel is a nutritionally essential element for a number of species (Eisler 1998b, p. 45), and evidence indicates fish and invertebrates are included in this group (USEPA BE 2008, p. 5-364). The USEPA (2008, pp. 4-11 and 5-364) reported:

> "Nickel can be carcinogenic, may be mutagenic, and is not teratogenic. It is bioconcentrated and bioaccumulated by aquatic organisms (Eisler 1998b). Toxicity of nickel to aquatic organisms is dependent on water hardness, pH, ionic composition, chemical form, type and concentration of ligands, presence of mixtures, and availability of solid surfaces for adsorption (Eisler 1998b). Nickel interacts with many compounds to produce altered patterns of accumulation, metabolism, and toxicity (Eisler 1998b). Mixtures of metals containing nickel salts are more toxic to daphnids and fishes than are predicted on the basis of individual components (Enserink et al. 1991)."
>
> "Nickel has a potential to accumulate in tissues of aquatic animals, although as long as water column or dietary concentrations are not so high as to overwhelm homeostatic mechanisms, aquatic species are able to regulate their internal body burden of nickel. Aquatic organism exposure to nickel concentrations in excess of nutritional needs via other potential routes of exposure in addition to direct waterborne toxicity (e.g. dietary toxicity) may pose a threat to listed species. Waterborne nickel is believed to be the

> primary source of nutritionally required nickel (Muyssen et al. 2004), as nickel appears to be poorly assimilated by aquatic species through their diet (Watras et al. 1985)."

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-364–5-366) compared the tissue-residue C_A calculated for nickel of 2.4 mg/kg wet weight to available whole body residue-effects studies in fish and invertebrates to identify where effects were reported below the C_A. The studies USEPA evaluated indicate that the nickel tissue residue required to elicit toxicity in aquatic species is higher than the amount of nickel that could reasonably be expected to bioaccumulate in aquatic species exposed to chronic nickel criterion concentrations for all organisms evaluated except for one study on *Hyalella azteca* by Borgmann et al. (2001), which noted reduced growth at 1.15 mg/kg wet weight. USEPA (2008, p. 5-365) reports that the results from another residue-effects study (Keithly et al. 2004) with *Hyalella azteca* measured found no to low effects at 14.5 mg/kg wet weight, which is higher than the chronic C_A of 2.4 mg/kg. Too few studies are available to compare tissue residue effects concentrations for invertebrates related to vernal pool fairly shrimp (e.g., Family Daphnidae), but evidence suggests the effects from accumulation also would not be measureable at the chronic criterion, possibly regulated by invertebrates, and likely insignificant.

Based on USEPA's assessment and our review of the available literature regarding nickel bioaccumulation, we conclude that dietary nickel would not contribute any additional adverse effects beyond water only exposure to invertebrates or fish.

4.3.1.8 Selenium

Selenium, an element widely dispersed in nature and arising primarily from weathered rock, is nutritionally essential to many animals (Eisler 1985b, p. 4). The BE (USEPA 2008, p. 5-402) indicates that selenite, the hexavalent form of selenium, has higher toxicity to aquatic species compared to selenate, the tetravalent form. However, the USFWS California Toxics Rule (USFWS and NMFS 2000, p. 100) reports selenite as the more toxic form to aquatic organisms compared to selenate when considering traditional bioassay results for aqueous exposure only. The USEPA (2008, p. 4-14) states:

> "Of all the priority and non-priority pollutants, selenium has the narrowest range of what is beneficial for biota and what is detrimental. Aquatic and terrestrial organisms require 0.5 µg/g dry weight (dw) of selenium in their diet to sustain metabolic processes, whereas concentrations of selenium that are only an order of magnitude greater than the required level have been shown to be toxic to fish. Acute effects are observed after short exposure durations of typically 96 hours or less. Acute effects from the inorganic forms of selenium, selenite and selenate, require concentrations exceeding 300 µg/L,

> concentrations rarely reached in the environment. In contrast, toxic effects from long-term chronic exposure via diet and water can result in reduction of species in aquatic systems with aqueous concentrations less than 20 µg/L (Lemly 1985). As a result of the greater sensitivity to selenium from chronic exposures, water quality management practices over the last 10-15 years have focused on the control of chronic effects. Studies have shown that diet is the primary route of exposure that controls chronic toxicity to fish, the group considered to be the most sensitive to chronic selenium exposure (Coyle et al. 1993; Hamilton et al. 1990; Hermanutz et al. 1996).
>
> Effects of selenium toxicity to freshwater organisms range from physical malformations during embryonic development to sterility and death (Lemly and Smith 1987) and include reduced hatch, reduced growth, behavioral alterations (avoidance), shifts in species composition of freshwater algal communities, loss of equilibrium, lethargy, muscle spasms, protruding eyes, liver degeneration, reduction in blood hemoglobin, chromosomal aberrations, and reduced survival (Eisler 1985b). "

The most important aspect of selenium residues in aquatic food chains is not direct toxicity to the organisms themselves, but as the dietary source of selenium they provide to predatory species (Lemly 2002, p. 23). Most of the selenium in fish tissues results from selenium in the diet rather than water. A small increase in waterborne selenium will result in a disproportionately large elevation of selenium residues in biological tissues (Lemly 2002, p. 24). The author further states (p. 30) that studies show the most sensitive indicator of selenium impacts on centrarchids, fathead minnows, and probably salmonids and percichthyids is reproductive success and the most precise way to assess selenium status and potential reproductive impairment of adult fish is to measure selenium levels in gravid ovaries. For aquatic birds, most sensitive indicator of selenium toxicity is also reproductive failure.

See section 4.5 Selenium Effects Analysis for the complete discussion of effects to listed species which applies to all of the species and critical habitats considered in this Biological and Conference Opinion.

4.3.1.9 Silver

Eisler (1996, p. 3) reports silver has no known biological purpose and is one of the most toxic metals known to aquatic organisms. Elemental silver (Ag^0) and the monovalent ion (Ag^+) are the most common oxidation states, and soluble silver salts are more toxic than insoluble salts in water (Eisler 1996, p. 7). The soluble monovalent ion is the form posing the most environmental concern (Eisler 1996, p. 7). There is a high variability in rates of accumulation and elimination of silver in organisms, but generally less than 1% of the total silver intake is absorbed and retained (Eisler 1996, p. 7). USEPA (2008, p. 4-15) states that:

> "Silver is not known to be mutagenic, teratogenic, or carcinogenic (Eisler 1996). It bioconcentrates and may bioaccumulate (Eisler 1996). Toxicity of Ag may be altered by a number of factors including pH, organic carbon, cation exchange capacity, presence of mixtures (Ratte 1999), sulfides, and duration of exposure. Silver, as ionic Ag+, is one of the most toxic metals known to aquatic organisms in laboratory testing (Nebeker et al.

> 1983). Aquatic insects concentrate silver in relative proportion to environmental levels (Nehring 1976), and more efficiently than most fish species (Diamond et al. 1990). Effects of silver toxicity to freshwater algae and phytoplankton include growth inhibition and altered species composition and species succession (Eisler 1996). Effects of silver toxicity to freshwater invertebrates include inhibited feeding and coordination, reduced growth, elevated oxygen consumption, and reduced survival (Eisler 1996). Effects of silver toxicity to freshwater fish include inhibited ionic flux across gills, reduced growth, premature hatch, and reduced survival (Eisler 1996). Interspecies differences in the ability to accumulate, retain, and eliminate silver are large (Baudin et al. 1994)."

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-453–5-454) compared the tissue-residue C_A calculated for silver of 310 µg/kg wet weight to available whole body residue-effects studies in plants and invertebrates (no fish studies were evaluated) to identify where effects were reported below the C_A. The studies USEPA evaluated indicate that the silver residue required to elicit toxicity in aquatic species is higher than the amount of silver that could reasonably be expected to bioaccumulate in aquatic species exposed to the silver chronic criterion of 0.10 µg/L for all organisms evaluated except for two tests on cladocerans and two tests on copepods, which recorded decreased egg production and viability at tissue concentrations (USEPA 2008, p. 5-453). Although this data from surrogate invertebrates (cladocerans) indicates that vernal pool fairy shrimp may suffer adverse effects from silver at the chronic criterion concentration, there are no anthropogenic sources of silver in vernal pool fairy shrimp habitat (see section 4.1.2.2 Likely to Adversely Affect Determinations).

Information is sparse regarding silver bioaccumulation in fish and invertebrates. Studies available indicate silver tissue values associated with adverse effects are higher than the amount that would reasonably bioaccumulate when species are exposed at the chronic criteria. Therefore, we rely primarily on the direct, waterborne exposure evaluation for assessing adverse effects from silver to the listed fish.

4.3.1.10 Zinc

Zinc is an essential element for many organisms (Eisler 1993, p. 5). Although dietary zinc has direct benefits to aquatic organisms (Eisler 1993, p. 5; Canli 2005, p. 274), zinc deficiency can produce deleterious effects such as reduced growth and survival (Eisler 1993, p. 45). USEPA (2008, p. 4-15) states that:

> "Zinc bioconcentrates but does not biomagnify (EPA 1999). Zinc may be mutagenic and teratogenic (Eisler 1993). Toxicity of zinc to aquatic organisms is dependent on water

> hardness, pH, DO, presence of mixtures, and trophic level (Sorensen 1991, Eisler 1993). Zinc interacts with many chemicals to produce altered patterns of accumulation, metabolism, and toxicity; some interactions reduce toxicity and others increase toxicity (Eisler 1993). Most of the zinc introduced into aquatic environments is eventually partitioned into sediments (Eisler 1993). Zinc bioavailability from sediment is increased under conditions of high DO, low salinity, low pH, and high levels of inorganic oxides and humic substances. Effects of zinc toxicity to freshwater organisms include reduced growth, reduced populations, and reduced survival in algae species; reduced growth, activity, larval settlement, and reproduction, osmoregulatory impairment and reduced survival in freshwater invertebrates (including molluscs); and reduced growth, behavioral alteration (avoidance), reproduction impairment, increased respiration, decreased swimming ability, increased jaw and branchial abnormalities, hyperactivity, hyperglycemia, and reduced survival in freshwater fish (Eisler 1993)."

Canli (2005, p. 272-274) demonstrated that *Daphnia magna* accumulated and rapidly eliminated zinc, indicating a mechanism of regulation, and dietary zinc did not have adverse effects on reproduction, growth, and feeding. McGeer et al. (2003, p. 1,023) indicates that for identification of aquatic risk, body concentration does not seem to be a reliable indicator of the potential for adverse effects related to zinc.

Uptake of zinc by fish is altered by exposure time, exposure concentration, fish size, water hardness, length of acclimation to test solutions, feeding level and trophic level and this is only a partial list of the influences (Sorensen 1991, p.141). Eisler (1993, p. 17) states that intraspecies differences in zinc content vary with age, size, sex, season, and other modifiers.

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-498–5-501) compared the tissue-residue C_A calculated for zinc of 5.6 mg/kg wet weight to available whole body residue-effects studies in invertebrates and fish to identify where effects were reported below the C_A. The BE (p. 5-498) notes

> "The optimal dietary requirement of zinc for a number of fish species is in the range of 15 - 30 mg/kg dry weight in diet (Clearwater et al. 2002, Watanabe et al. 1997), equal to a wet weight concentration in diet of 3 - 6 mg/kg assuming an 80% water content in the diet."

The BE reports (p. 5-500) that the C_A of 5.6 mg/kg where adverse effects could occur is within the wet weight range of the optimal dietary requirement, and the C_A is lower than all reported

residues in fish and invertebrates associated with adverse effects on reproduction and growth in studies identified by USEPA.

Based on USEPA's assessment and our review of the available literature regarding zinc bioaccumulation, we conclude that dietary zinc would not contribute any additional adverse effects beyond water only exposure to invertebrates or fish.

4.3.2 Non-Metals

4.3.2.1 Ammonia

USEPA (2008, p. 5-95) states:

> "In water, total ammonia is composed of the ammonium ion NH_4^+ and un-ionized ammonia NH_3. Because un-ionized ammonia (UIA) is approximately100 times more toxic to aquatic life than the ammonium ion, the amount of UIA present greatly affects toxicity. The amount of UIA present in water is highly dependent on pH, increasing 10 fold for every unit increase in pH. UIA concentrations are also affected by temperature, increasing approximately 2-fold for every 10°C rise in temperature (USEPA 2009). Total ammonia is expressed as total ammonia nitrogen (TAN).
>
> Ammonia is the primary end product of nitrogen metabolism in fish (Smutna et al. 2002) and many aquatic invertebrates, formed by the deamination of amino acids and as a byproduct of glycolysis during periods of high muscular activity. Normally, excretory systems such as kidneys and gills serve to eliminate ammonia from aquatic animals. Changes in environmental conditions can easily alter the balance between endogenous ammonia concentrations and excretions. When the detoxification and excretory capabilities of aquatic animals are exceeded, ammonia acts as a toxicant.
>
> Ammonia is used directly as a nutrient in aquatic systems by bacteria such as Nitrosomonas, which oxidize ammonia to nitrite. Blue green algae can fix atmospheric nitrogen to produce ammonium. Many aquatic plants take up ammonium preferentially over nitrate as a nitrogen source (Walstad 2003). Although an extensive amount of literature is available on the nitrogen requirements of aquatic plants and animals, none of it appears to have evaluated the toxicity of ammonia or other forms of nitrogen in the diet of aquatic species."

As evident from the level of effect analyses, ammonia is more acutely toxic to aquatic organisms than chronically toxic (see Appendices 3 and 5). No bioaccumulation data were found by USEPA (2008, p. 5-95) for any aquatic species, as reported in their BE, so they were unable to conduct an analysis of multiple routes of exposure.

Based on lack of residue effects data, our review of the available ammonia effects data, and because the primary effects of ammonia are through water exposures, we conclude that dietary ammonia will not accumulate in tissues or have any additional adverse effects beyond water only exposure to invertebrates or fish.

4.3.2.2 Pentachlorophenol

Pentachlorophenol is a widely used pesticide mostly commonly associated with treating wood (Eisler 1989a, p. 5). Although the parent compound is practically insoluble in water, the sodium pentachlorophenate salt is readily water soluble and used in many industrial applications (Eisler 1989a, p. 5). Fish readily accumulate pentachlorophenol primarily through direct uptake from water rather than diet (Niimi and Cho 1983, p. 1,791). The USEPA (2008, p. 4-21) states:

> "Pentachlorophenol rapidly degrades in air, on land, and in water. Pentachlorophenol is teratogenic but evidence of its mutagenic or carcinogenic properties is incomplete (Williams 1982). It bioconcentrates, and bioaccumulates in predatory species (Eisler 1989a). Toxicity of PCP may be altered by a number of factors including pH, temperature, chemical composition (which congeners are present), organic matter, and presence of mixtures (Eisler 1989a). Toxicity of pure, reagent grade PCP is less than that of commercial PCP due to toxicity of some of the impurities present in commercial formulations (Cleveland et al. 1982). Polychlorinated dibenzodioxins and dibenzofurans are common impurities present in commercial PCP formulations; these compounds are known to bioaccumulate in the environment and are highly toxic to avian and mammalian wildlife (Giesy et al. 1994). Many of the available toxicity tests have been conducted with reagent grade PCP and may thus underestimate toxic effects of commercial PCP releases into the environment. In general, fish are more sensitive to PCP than are other aquatic organisms (USFWS 2000). Coldwater species are generally more sensitive than warmwater species in acute lethal toxicity tests (EPA 1995). Effects of PCP toxicity to algae include chlorosis inhibition, reduced cell numbers, reduced or inhibited growth, and reduced survival (Eisler 1989a). Effects of PCP toxicity to freshwater invertebrates include reduced populations, reduced locomotion or immobilization, abnormal larvae development, reduced reproduction (decreased production of eggs or young), decrease in periphyton biomass, larval drift, and suppression of community metabolism in invertebrates (Eisler 1989a). Effects of PCP toxicity to freshwater fish include reduced growth, increased alevin mortality, reduced food conversion efficiency, reduced ability to capture and consume prey, fin erosion, cranial malformations, reduced activity, reduced egg survival, rapid swimming at the water surface and increased opercular movements, loss of balance, and reduced survival (Eisler 1989a)."

Eisler (1989a, p. 14) reports that

> "Pentachlorophenol affects energy metabolism by partly uncoupling oxidative phosphorylation and increasing oxygen consumption, by altering the activities of several glycolytic enzymes and the citric acid cycle enzymes, and by increasing the consumption rate of stored lipid (Johansen et al. 1985; Brown et al. 1987; McKim et al. 1987). Collectively, these events could reduce the availability of energy for maintenance and growth and thereby reduce the survival of larval fish and the ability of prey to escape from a predator (Brown et al. 1985, 1987)."

The USEPA BE assessed multiple routes of exposure to fish and invertebrates by estimating a tissue residue-based assessment effects concentration (C_A) derived from the product of the

chronic criterion and bioconcentration factor. The C_A represents the highest concentration of a chemical in tissue of a listed species where effects are expected to be insignificant. Therefore, the adverse effects from bioaccumulation or multiple routes of exposure would be predicted to occur at the chronic criteria when the tissue concentration associated with an adverse effect is below the C_A.

The BE (USEPA 2008, pp. 5-391–5-392) compared the tissue-residue C_A calculated for pentachlorophenol of 11.5 mg/kg wet weight to available whole body residue-effects studies in invertebrates and fish to identify where effects were reported below the C_A. The USEPA identified only four studies with comparable residue effects data for fish and a sea urchin species. The BE (p. 5-391) notes

> "With the exception of one study demonstrating reduced growth [in largemouth bass] at tissue concentrations slightly lower [10.8 mg/kg wet weight] than the calculated criteria tissue concentration, all other studies indicate that the pentachlorophenol residue required to elicit toxicity in aquatic species is higher than the amount of pentachlorophenol that could reasonably be expected to bioaccumulate in aquatic species (11.5 mg/kg wet weight) exposed to chronic criterion concentrations (15 µg/L)."

Other studies USFWS reviewed included an experiment by (Rogers et al. 1988, p. 100) reported that Chinook salmon juveniles exposed to pentachlorophenol at 2 µg/L (well below the chronic criterion) for 62 days accumulated mean residues in tissue of 431 µg/g, but effects to the species were unknown.

Based on USEPA's assessment and our review of the available literature regarding pentachlorophenol bioaccumulation, we conclude that dietary pentachlorophenol would not contribute any additional adverse effects beyond water only exposure to fish. Too few data are available to evaluate tissue effects related to invertebrates, therefore, we rely on the direct exposure evaluation for assessing effects from pentachlorophenol to vernal pool fairy shrimp.

4.4 Effects of the Proposed Action on Listed Species and Critical Habitats

4.4.1 Effects of the Proposed Action on Bull Trout

4.4.1.1 Potential Exposure of Bull Trout to Metals and Non-Metals

Potential Exposure of Bull Trout to Metals

This portion of the analysis considers the potential for the metals addressed under the proposed action to be discharged by mining operations and NPDES Permits into habitats occupied by bull trout. It is unlikely that metals would be released from every mine and permit location, and unlikely all would be released during the same time period (Table 4-1), but because no information is available as to which locations may be a sources of exposure, and it is USFWS policy to give the benefit of the doubt to the listed species in cases of uncertainty (USFWS and

NMFS 1998), we have assumed for purposes of this analysis that all mines and NPDES Permits are sources of metal to which bull trout are likely to be exposed.

There is potential for bull trout to be exposed to most metals within its range in Oregon; approximately 24% of bull trout-occupied habitat in Oregon is subject to metal exposure. This translates to approximately 260.8 miles of spawning and rearing habitat, and 559.8 miles of foraging, migratory, and overwintering habitat.

Occupied habitat with potential exposure to metals is not equally distributed throughout Oregon. The percentage of habitats exposed (for spawning and rearing, and foraging, migratory and overwintering habitats) by river basin was calculated and the values can be found in Table 4-1. The percentage of exposed habitat was used to determine the level of effect. Some bull trout-occupied basins have no or very little potential for exposure and other basins have moderate to high potential for exposure to metals identified in this action.

Bull trout-occupied river basins with no potential for exposure to metals addressed under the proposed action include the Malheur River Basin and Odell Lake. While there is a small potential that bull trout could be exposed within these areas, such exposure would be highly unlikely based on the fact that there are no point source permits or mines currently and these remote areas are unlikely to see much development. Therefore, any potential exposure would be in an isolated area which would have little or no measurable influence on the system at large and effects would be localized and insignificant (i.e., the proposed action is NLAA the bull trout in these areas) because any effects are not likely to significantly disrupt their breeding, feeding, or sheltering behavior or cause injury or death of affected individuals.

Bull trout-occupied river basins with a low to moderate potential for exposure to metals include the Lower Deschutes, Klamath, and Walla Walla (Table 4-1). These are basins where 10% or less of bull trout spawning and rearing habitat or foraging, migratory and overwintering habitat could be potentially exposed to the metals addressed in the proposed action.

Approximately 40% of Snake River and Columbia River bull trout foraging, migratory, and overwintering habitats have a potential of exposure to metals, especially copper (moderate level, Table 4-1). However, the greatest potential for exposure would be during low flows (less dilution) when it would be unlikely for bull trout to be present because low flows most often occur during late summer and early fall when water temperature is unsuitable for bull trout and most adults would be in spawning and rearing habitats further upstream in the various river basins.

River basins where more than 10% of bull trout spawning and rearing habitat or foraging, migratory and overwintering habitat could be exposed to the metals identified in this action are: the Grande Ronde, Hells Canyon Complex, Hood, Imnaha, John Day, Powder, Umatilla, and the Upper Willamette. In the Hood and Umatilla river basins the only bull trout habitats potentially exposed to the metals addressed under the proposed action are those associated with foraging, migration and overwintering.

Chromium VI is considered an exception because ODEQ determined that chromium VI levels in water samples drop rapidly when groundwater reaches aerobic surface water (B. Mason, pers. comm., 2012). For that reason, there is little potential for bull trout to be exposed to chromium VI within the action area. The USEPA (2008, p. 4-10) states that Chromium VI is generally produced by industrial processes. The only area in Oregon where chromium VI has been identified as a parameter of concern for bull trout is an unnamed creek in the Lower Columbia River subbasin (USEPA 2008, p. 4-11). This creek drains into the lower Columbia River, which is identified as foraging, migratory and overwintering habitat, but where bull trout occurrences are very rare. Thus, there is very little likelihood that bull trout would be exposed to chromium VI at the criterion level identified in the proposed action.

Potential Exposure of the Bull Trout to Non-Metals

This portion of the analysis is based primarily on the potential that nonmetal toxics could be discharged by NPDES Permits. It is unlikely that nonmetal toxics would be released from every permit location (Table 4-2), but given uncertainty we are providing the benefit of doubt to the species in accordance with USFWS policy (USFWS and NMFS 1998) by assuming in this analysis that the locations of all NPDES Permits are sources of bull trout exposure to the nonmetal toxics addressed under the proposed action.

While Table 4-2 displays the potential for bull trout exposure to nonmetal toxics, best available information indicates that the possibility of bull trout being exposed to most of the nonmetal criteria levels identified in the proposed action is extremely low. Dieldrin and pentachlorophenol are two examples of nonmetal toxics where the possibility of bull trout being exposed is extremely low. The primary nonmetal toxic that is analyzed in this section is ammonia.

A total of 2.7% (30.7 miles) of the total bull trout-occupied spawning and rearing habitat and 16.9% (359.5 miles) of occupied foraging, migratory, and overwintering habitat in Oregon have the potential to be exposed to nonmetal toxics (Table 4-2). This habitat is not equally distributed throughout the State (Table 4-2). Bull trout in the Imnaha River, Malheur River, and Odell Lake are currently not exposed to the nonmetals described in this action. While there is a small potential that bull trout could be exposed within these areas, such exposure would be highly unlikely based on the fact that there are no point source permits or mines currently and these remote areas are unlikely to see much development. Therefore, any potential exposure would be in an isolated area which would have little or no measurable influence on the system at large and effects would be localized and insignificant (i.e., the proposed action is NLAA the bull trout in these areas) because any effects are not likely to significantly disrupt their breeding, feeding, or sheltering behavior or cause injury or death of affected individuals.

4.4.1.2 Analytical Approach

As discussed in the Exposure Analysis, this analysis uses all known potential sources of exposure within ½ mile of bull trout habitat overland, and within 2 miles if hydrologically connected to bull trout habitat. Wherever possible, an attempt was made to analyze the potential effects of bull trout chronic exposure to proposed criteria on bull trout reproduction, growth and survival. There were many instances where data were limited, or no data were available to

analyze one or more of the above parameters. Relevant data were included and analyzed when they were available.

Analysis of Eggs and Reproduction

There are few reliable data concerning the number and survival of bull trout eggs that are produced annually within the action area. Egg production varies widely by region, season, local population, life form, and size of female (Pratt 2003, p. 50). Natural survival rates for char eggs are generally quite low. Morita and Yokata (2002, p. 87) reported average survival rates for eggs to age 0+ individuals of about 4.5% for white-spotted char (*S. leucomaenis*) based on 7 years of data, and around 2.5% for closely related Dolly Varden (*S. malma*) based on 2 years of data.

Therefore, although it is likely that exposure to some of the toxics addressed herein will have adverse effects on bull trout reproduction we will not be able to quantify these effects. There is a lack of meaningful data to quantitatively translate these potential effects to eggs into their effects to bull trout populations. In some cases, where the possible reduction in the egg hatching rate is small (<10%), it is difficult to separate that effect from natural background processes.

Analysis of Juveniles and Recruitment

The natural survival ratio of juvenile to adult bull trout is not well understood. While NMFS has estimated that the ratio may be as low as 1,000 juveniles to 1 adult for *Oncorhynchus* spp. (Robert Anderson, NMFS, Portland, Oregon, pers. comm., 2011), this may be at the lower end of the natural curve of variability which may generally range from less than 10,000:1 up to about 10:1 depending on many variables (Jeff Dose, Forest Fisheries Biologist, U.S. Forest Service, Umpqua National Forest, Roseburg, Oregon, pers. comm., 2011). Schofield et al. (1991, p. 1) found that the annual survival rate of the closely related brook trout juveniles to be less than 20%. A study conducted in West Virginia found a survival rate of about 10% for combined juvenile and small adult (less than 120 mm) for brook trout (Petty et al. 2005, p. 574), but data were only available for a 2-year period. While there may not be a direct correlation of the survival rate of brook trout to that of bull trout, both species belong to the same genus and we assume brook trout data are more relevant than that of a salmonid of a different genus. A study being performed on the Walla Walla River system tagged juvenile bull trout and marked 253 individual fish ranging from 72 to 229 mm in length. They recaptured 20 fish that had been marked in 2007 and 25 fish in the second marking occasion that had been marked in the first capture occasion in 2009. No estimates for survival were given, but based on the data the 2-year survival appears to be about 10% (Budy et al. 2010, p. 46). A modeling exercise which was done for bull trout in the Metolius River system garnered numbers similar to those mentioned by Anderson (R. Anderson, pers. comm., 2011), in the 1,000 juvenile to 1 adult range.

While the data used in this analysis were developed to assist in analyzing potential effects to juvenile fish, we have very little data to aid us in deciphering the abundance of juvenile bull trout within the action area. Accurate juvenile bull trout data are difficult to collect because of the individual's small size, secretive nature, and ability to hide within stream flora and substrate. Therefore, although it is likely that exposure to some toxics will have adverse effects on juvenile bull trout growth and survival; we are unable to quantify these effects.

The best and most reliable data that we have access to are estimates of adult numbers by basin. Although our effects models were done to gauge effects to juvenile bull trout, it is widely believed that juveniles are more sensitive and susceptible to the influences of toxins than are adults. Since we have no reliable data for juvenile abundance we applied our effects data to the adult population numbers. While this may overestimate the extent of effects to the population, we do so in keeping with the Act's mandate to "give the species the benefit of the doubt" when the best available scientific and commercial data are not adequate to address an effect (USFWS and NMFS 1998).

Effects to Prey Species

Based on the body of literature we used in developing our models, many bull trout prey species are likely to be adversely affected by the proposed action. Since we are unable to accurately analyze all possible effects to bull trout prey species, we make the assumption that there may be a reduction in these prey species under some of the proposed standards. This reduction in prey may have adverse effects to bull trout. However, we have no way to quantify those effects.

4.4.1.3 Effects of Aluminum on Bull Trout

Proposed Acute Criterion for Aluminum

The proposed action would establish an acute aluminum criterion of 750 µg/L at 100 mg/L $CaCO_3$. The LC10 developed using direct data for bull trout exposure to aluminum is 2,434 µg/L (at 100 mg/L $CaCO_3$). The LC10 would equate to a potential lethality of up to 10%. This means that the LC10 is more than 3 times higher than the acute aluminum criterion. Although the difference is great we are unable to say with confidence that there will not be some adverse effects to bull trout even at a low aluminum concentration. Therefore, the acute exposure of bull trout to aluminum at the proposed acute criterion is considered significant, although we do not believe that these adverse effects are likely to rise to a level that would actually cause injury or mortality to bull trout because the LC10 is more than 3 times higher than the acute aluminum criterion.

Proposed Chronic Criterion for Aluminum

The proposed chronic aluminum criterion analyzed in this Biological and Conference Opinion is 87 µg/L (at 100 mg/L $CaCO_3$). Bull trout data were not available for the analysis on survival and reproductive effects of exposure to aluminum. Brook trout were used as a surrogate for bull trout because they are in the same genus (*Salvelinus*) and often co-occur in the same habitats as bull trout. Data for juvenile brook trout (Cleveland et al. 1990) were used for survival. Based on our analyses of the data, we found that brook trout exposed to significantly higher concentrations (266.8 µg/L and 305 µg/L) of aluminum for 7 days showed no increase in mortality, and no significant decrease in growth. On that basis, we conclude that the much lower proposed chronic aluminum criterion would have insignificant effects (i.e., effects that cannot be meaningfully measured, detected, or evaluated) on the survival and growth of individual bull trout.

Summary of Effects to Bull Trout caused by its Exposure to Proposed Acute and Chronic Criteria for Aluminum

Acute Aluminum Exposure

Based on the brook trout response data discussed above, bull trout exposed to the proposed acute criterion for aluminum are likely to `be adversely affected, but such effects are not likely to significantly disrupt their breeding, feeding, or sheltering behavior or cause injury or death of affected individuals.

Chronic Aluminum Exposure

Based on the brook trout response data discussed above, effects to bull trout from exposure to the proposed chronic criterion for aluminum are insignificant.

4.4.1.4 Effects of Ammonia on Bull Trout

Ammonia is a common contaminant associated with manufacturing and is often present in the environment. For example, it is the primary constituent contributing to the acute toxicity of zirconium processing effluent (Robinson et al. 1975, pp. 980, 982). It is also present in many agricultural chemicals and associated with livestock.

USEPA (2011) reported that salmonid exposure to ammonia can cause repression of the immune system, resulting in increased susceptibility to bacteria and parasitic diseases. This may lead to increased incidence of saprolegniosis, a fungal infection that attacks salmonids (Carballo et al. 1995, p. 2109). This is most likely due to a reduction in the fish's ability to renew mucus (which is inhibited in presence of ammonia) in order to shed fungal spores (Carballo et al. 1995, p. 2111). Further, in salmonids, ammonia exposure can lead to release of cortisol, which increases the plasma cortisol levels, and may cause an increase in metabolic rate, thus reducing metabolic reserves needed for muscle function during swimming (Shingles et al. 2001, p. 2695). Ortega et al. (2005, p. 1860) found that high concentrations of ammonia reduced appetite and feeding in rainbow trout.

Proposed Acute Criterion for Ammonia

The proposed acute exposure criterion is 5.6 mg/L as N at pH= 8.0. Our modeling indicates that the LC10 for bull trout exposure is 13.7 mg/L as N at pH =8. The LC10 is more than two times greater than the proposed criterion; it is unlikely that bull trout exposure to the proposed acute criterion would significantly disrupt its breeding, feeding, or sheltering behavior and not otherwise cause injury or death of affected individuals. The percentage of the bull trout population within Oregon that could be adversely affected cannot be determined. Giving bull trout the benefit of the doubt for their conservation, we assume there will a low level of adverse effects on individual fish, but these effects will not rise to levels that cause a significant disruption of breeding, feeding, or sheltering behavior or otherwise cause injury or death of affected individuals.

Proposed Chronic Criterion for Ammonia

The chronic criterion for ammonia under the proposed action is 1.7 mg/L as N (at pH = 8.0, 20°C, salmonids present or absent). Our analysis of this proposed criterion relies on modeling of

fathead minnow (*Pimephales promelas*) response data (Thurston et al. 1986) on reproduction, and smallmouth bass (*Micropterus dolomieu*) response data (Broderius et al. 1985) on growth and survival as surrogates. This information is used in our assessment of the bull trout's response to being exposed to chronic levels of ammonia at the proposed criterion because there is insufficient chronic toxicity data on the response of bull trout or other closely related species to such a concentration of ammonia.

Effects to Bull Trout Reproduction

Our modeling predicts a likely reduction in bull trout egg hatchability, number of eggs per spawn, or number of eggs per female by 9.65% (95% confidence limits = 0.0% – 21%). The effect of the reduction in eggs from the chronic exposure to ammonia will have some adverse effect on bull trout populations, but the level of effect cannot be quantified because of factors described in the *Analysis of Eggs and Reproduction* section above.

Effects to Bull Trout Growth

Our modeling indicates there is not likely to be a reduction in juvenile bull trout growth (reduction in mass, length, or biomass;) in juveniles (95% confidence limits = 0.00% – 3.54%) in response to bull trout exposure to the chronic criterion for ammonia.

Salmonid growth can be affected by exposure to ammonia. Robinson et al. (1975, pp. 975 and 983) found that low levels of ammonia (less than 3mg/L) stimulated growth in juvenile chinook salmon, while tests with concentrations above 4.4 mg/L of ammonia caused slower growth than the control. Further, Knights (1989, pp. 114) found that ammonia can inhibit growth in European eels (*Anguilla anguilla L)* through the increased expenditure of energy.

Because our modeling indicates that the proposed chronic criterion for ammonia exposure will not affect the growth of bull trout populations, only adverse effects to survival (see below) are presented in column C of Table 4-6.

Effects to the Survival of Individual Bull Trout

Our modeling results indicate that exposure to this chronic criterion concentration at 7.25 pH would reduce survival in bull trout juveniles by 16.8% (95% confidence limits = 9.08% – 25.5%) every 3 years during the 25-year term of the proposed action. Because most ammonia is absorbed through the gills, the concentration at the gill surface is believed to determine the true toxicity of ammonia solutions (Caglan et al. 2005, p. 339) (Robinson et al. 1975, p.981). Harm to bull trout (Table 4-6) is determined using the following formula: Effects per Modeling (column C) x % Exposed habitat (column D) x Estimated Average Adult Population (column E). See Table 4-2 for exposed habitat values.

Table 4-6. Type and estimated amount of bull trout mortality caused by exposure to the proposed chronic criterion level for ammonia.

A	B	C	D	E	F
Basin	Type of Effect	Reduction in survival of affected individuals	Percent Habitat Exposed	Estimated Average Adult Population Size	Mortality of Adult Bull Trout per 3-Year Period
Grande Ronde River	Survival	17%	7%	150	2
Hells Canyon Complex	Survival	17%	6%	300	3
Hood River	Survival	17%	25%	150	7
Imnaha River	Survival	17%	0%	1,750	0
John Day River	Survival	17%	3%	400	2
Klamath River	Survival	17%	4%	2,050	14
Lower Deschutes River	Survival	17%	4%	1,750	12
Malheur River	Survival	N/A	0%	150	0
Odell Lake	Survival	N/A	0%	50	0
Powder River	Survival	17%	10%	300	5
Umatilla River	Survival	17%	28%	50	3
Upper Willamette River	Survival	17%	36%	300	18
Walla Walla River	Survival	17%	5%	1,750	15
Totals:				9,150	81

Although Columbia River and Snake River foraging, migratory, and overwintering habitats encompass substantial areas where bull trout could be exposed to the proposed chronic criterion for ammonia, many of the bull trout utilizing these rivers also utilize the tributary river basins listed in Table 4-6. Chronic exposure of bull trout to ammonia at the proposed standard may occur within portions of Columbia River and Snake River foraging, migratory, and overwintering habitat and this exposure is likely to cause adverse effects to the affected individuals. However, because of the high flows within these two mainstem rivers, and the ability of bull trout to move through or avoid the highest concentrations of ammonia they encounter, the adverse effects to the bull trout associated with this exposure to chronic concentrations of ammonia in Columbia River and Snake River mainstem habitats are not likely to cause additional death or injury of affected bull trout that reside in the tributaries listed in Table 4-6.

Summary of Adverse Effects to Bull Trout caused by its Exposure to Ammonia at the Proposed Acute and Chronic Criteria

The following summary is based on a reasonable worst case scenario:

Bull Trout Exposure to the Acute Ammonia Criterion

Exposure to the acute ammonia criteria is likely to result in some degree of adverse effects to the bull trout by the disrupting feeding, overwintering, or migratory behavior to individual adult bull trout associated with 459.7 miles of occupied bull trout habitat (see Table 4-2 above) in Oregon for each 3-year period during the 25-year term of the proposed action. However, these effects are not likely to cause the death or injury of affected individuals based on a comparison between the proposed acute criterion and the ammonia LC10 for the bull trout.

Bull Trout Exposure to the Chronic Ammonia Criterion

Exposure to ammonia at the proposed criterion has insignificant effects (i.e., effects that cannot be meaningfully measured, detected, or evaluated) on the growth of individual bull trout, but is likely to kill up to 81 individual adult bull trout associated with 459.7 miles of bull trout habitat in Oregon for each 3-year period of the proposed action.

We are unable to quantify the exact number of bull trout eggs that may be adversely affected due to such exposure as it is not possible to accurately inventory this life stage within the action area at this time. However, we assume that some small portion of bull trout eggs is likely to be killed within 30.7 miles of spawning and rearing habitat (see Table 4-2) because of a 9.65% reduction in hatching of bull trout eggs where exposed to ammonia at the chronic criterion..

4.4.1.5 Effects of Arsenic on Bull Trout

Arsenic is considered a ubiquitous, naturally-occurring element that may be found at increased levels in water and soil. Arsenic can enter streams as a result of leaching from rock into ground water, and possible geothermal activity (Budiati 2008, p. 8). Lim et al. (2009, pp. 2865-2866) noted that arsenic has a great deal of solubility and mobility, and is commonly leached into water from mining operations. This is often because mining exposes arsenopyrite (FeAsS) to oxidation.

Once arsenic enters aquatic environments it tends to have a very complex chemistry and may form more than 20 different compounds. Arsenic may occur in both organic and inorganic compounds. Many of these compounds are toxic, in particular As_2O_3 (Budiati 2008, p. 8). Bacterial action often creates organic methyl derivatives of arsenic. Arsenite class compounds are more toxic and more readily absorbed into fish tissues than arsenate class compounds. These substances are considered to have a moderate to high toxicity to fish (Svobodova et al. 1993, pp. 28-29). Budiati (2008, p. 8) reported that the highest contribution of heavy metals found in salmon is from arsenic.

Fish may be able to acclimate to the presence of arsenic, thus reducing toxic effects. Rainbow trout pre-exposed to arsenic showed a significant (47%) increase in tolerance to arsenite (Dixon and Sprague 1981, pp. 579 and 585).

Proposed Acute Criterion for Arsenic

The proposed action would establish an acute standard for arsenic of 340µg/L. The LC10 developed for bull trout exposure to arsenic is 5,726 µg/L for juvenile fish. This means that the proposed standard would likely affect far less than 10% of the exposed bull trout population

because it is significantly lower than the LC10. In fact, the LC10 is nearly 17 times higher than the standard, based on our modeling results.

While certain river basins have greater potential for exposure than others, the criterion is so low when compared to the above LC10, that such exposure would be considered to have insignificant effects (i.e., effects that cannot be meaningfully measured, detected, or evaluated) on the exposed bull trout. While arsenic at higher concentrations may cause adverse effects to fish we find no evidence to support a conclusion that acute exposure to arsenic at such low concentrations (340µg/L) is likely to cause adverse effects to individual bull trout.

Proposed Chronic Criterion for Arsenic

The chronic arsenic criterion analyzed in this Biological and Conference Opinion is 150 µg/L. Exposure at this criterion was modeled for its effects on growth and survival of bull trout. Bluegill data were used for evaluating both growth and survival effects. This species was used as a surrogate in this analysis for bull trout because there are insufficient chronic toxicity data for bull trout or any other more closely related salmonid. We were unable to obtain toxicity data that was relevant to bull trout reproduction.

Effects to Bull Trout Growth

Our modeling results indicate no reduction in growth of bull trout (mean mass) when exposed to the chronic criteria of arsenic (95% confidence limits = 0.00%- 0.00%). There appears to be no direct correlation between chronic arsenic exposure at the proposed criterion and bull trout growth.

Effects to Bull Trout Survival

Modeling bull trout exposure to the proposed chronic criterion concentration for arsenic yielded the following result: effects to survival of juveniles in the family Salmonidae (which includes the bull trout) of 5.13% (95% confidence 0.00% – 13.8%, bluegill surrogate) every 3 years during the 25-year term of the proposed action. This translates to a likely mortality to individual bull trout of about 5%.

The presence of arsenic decreased the levels of cystic fibrosis transmembrane regulator, increased the levels of chlorine ions, and caused significant mortality of killifish (*Fundulus heteroclitus*) during acclimation from freshwater to seawater in tidal environments (Shaw et al. 2007a, p. R1058). Bull trout traveling into estuarine or near shore environments could experience this same condition. Although few to no bull trout in Oregon are found in estuarine or near shore habitats.

Table 4-7. Estimated amount of bull trout mortality caused by exposure to the proposed chronic criterion for arsenic.

A	B	C	D	E	F
Basin	Type of Effect	Reduction in Survival per Modeling	Percent Habitat Exposed	Estimated Average Adult Population Size	Maximum Mortality of Adult Bull Trout
Grande Ronde River	Survival	5.1%	16%	150	1
Hells Canyon Complex	Survival	5.1%	22%	300	4
Hood River	Survival	5.1%	16%	150	4
Imnaha River	Survival	5.1%	33%	1,750	29
John Day River	Survival	5.1%	24%	400	5
Klamath River	Survival	5.1%	6%	2,050	7
Lower Deschutes River	Survival	5.1%	4%	1,750	4
Malheur River	Survival	N/A	0%	150	0
Odell Lake	Survival	N/A	0%	50	0
Powder River	Survival	5.1%	44%	300	7
Umatilla River	Survival	5.1%	33%	50	6
Upper Willamette River	Survival	5.1%	40%	300	6
Walla Walla River	Survival	5.1%	7%	1,750	6
Totals:				9,150	79

Although Columbia River and Snake River foraging, migratory, and overwintering habitats encompass substantial areas where bull trout could be exposed to the proposed chronic criterion for arsenic, many of the bull trout utilizing these rivers also utilize the tributary river basins listed in Table 4-7. Chronic exposure of bull trout to arsenic at the proposed standard may occur within portions of Columbia River and Snake River foraging, migratory, and overwintering habitat and such exposure is likely to cause adverse effects to the exposed individuals. However, because of the high flows within these two rivers, and the ability of bull trout to move through or avoid the highest concentrations of arsenic they encounter within large streams, the adverse effects to the bull trout associated with their exposure to chronic concentrations of arsenic in Columbia River and Snake River mainstem habitats are not likely to cause additional death or injury of affected bull trout that reside in the tributaries listed in Table 4-7.

Summary of Effects to the Bull Trout caused by its Exposure to the Proposed Acute and Chronic Criteria for Arsenic

The following summary is based on a reasonable worst case scenario:

Acute Arsenic Exposure

Based on a comparison between the proposed criterion and the LC10 for arsenic, the effects to the bull trout caused by its exposure to the proposed acute criterion for arsenic are likely to be insignificant (i.e., effects that cannot be meaningfully measured, detected, or evaluated) and are not likely to cause the death or injury of exposed bull trout within the action area.

Chronic Arsenic Exposure

Effects to the growth of bull trout caused by its exposure to the proposed chronic criterion for arsenic are likely to be insignificant (i.e., effects that cannot be meaningfully measured, detected, or evaluated).

Based on the results of modeling bluegill response data, the effects to the bull trout caused by its exposure to the proposed chronic criterion for arsenic will result in the mortality of 79 individual adult bull trout over each 3-year period of the proposed action. The affected area includes surface waters along 820.6 miles of bull trout habitat that are likely to contain a concentration of arsenic at the proposed chronic concentration (see Table 4-7).

4.4.1.6 Effects of Cadmium on Bull Trout

Proposed Acute Criterion for Cadmium

The proposed action would establish an acute standard for cadmium of 2.0µg/L (at 100 mg/L as CaCO3). The LC10 developed using direct data for bull trout exposure to cadmium is 1.24 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish (Table 4-8). This result means that the proposed acute standard for cadmium would likely cause a reduction in bull trout survival of more than 10% of the exposed population every 3 years during the 25-year term of the proposed action. Using the mid-range (low + high/2) of the percent effects in the last column of Table 4-8, the estimated percent effect to exposed bull trout populations at the criterion concentration is 12% mortality (Table 4-8, Column B). The maximum level of mortality likely to occur is presented in Table 4-8 based on the following formula: 12% x % Exposed habitat % x Estimated Adult Population (exposed bull trout habitat is in Table 4-2).

Table 4-8. Estimated amount of bull trout mortality caused by exposure to the proposed acute criterion for cadmium.

A	B	C	D	E
Basin	% Reduction in Survival at Acute Criterion	Percent Habitat Exposed	Estimated Average Adult Population Size	Maximum Mortality of Adult Bull Trout (for each 3-year period during the term of the proposed action)
Grande Ronde River	12%	16%	150	3
Hells Canyon Complex	12%	22%	300	8
Hood River	12%	16%	150	3
Imnaha River	12%	33%	1,750	69
John Day River	12%	24%	400	12
Klamath River	12%	6%	2,050	15
Lower Deschutes River	12%	4%	1,750	8
Malheur River	N/A	0%	150	0
Odell Lake	N/A	0%	50	0
Powder River	12%	44%	300	19
Umatilla River	12%	28%	50	2
Upper Willamette River	12%	40%	300	14
Walla Walla River	12%	7%	1,750	15
Totals:			9,150	165

Chronic Cadmium Exposure

The proposed chronic cadmium criterion is 0.25 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. Direct data on the effects of bull trout exposure to cadmium (Hansen et al. 2002d) were used in this analysis. While Hansen et al. (2002d) did not test the precise chronic criterion level proposed herein (i.e., 0.25 µg/L), they did test for bull trout mortality, and for bull trout growth reduction at concentration levels equivalent 0.21 µg/L (at 100 mg/L $CaCO_3$), which is relatively close to the chronic criterion under consideration.

Effects to Bull Trout Growth

Hansen et al. (2002d, p. 171) concluded that bull trout exposed to cadmium at concentrations equivalent to 0.21 µg/L (at 100 mg/L $CaCO_3$) experienced a 12.4% reduction in growth (weight) from the control after 55 days of exposure, while bull trout exposed to a much higher concentration of cadmium [equivalent to 0.9 µg/L (at 100 mg/L $CaCO_3$)] experienced a 12.9% reduction in growth from the control. These results are somewhat ambiguous, as testing done at a concentration between these amounts [at 0.46. µg/L (at 100 mg/L $CaCO_3$)] showed only a 9% reduction in weight. While we have no direct data on bull trout response to the exact proposed acute criterion for cadmium, based on the above discussion, we conclude that a reduction in bull trout growth of about 13% (a reasonable worst case) is likely to occur every 3 years during the 25-year term of the proposed action when bull trout are subject to chronic exposure to cadmium at the proposed standard.

Chronic exposure to cadmium at the proposed level is considered to have adverse effects to all bull trout potentially exposed by reducing their fitness through a reduction in growth (Table 4-9, column G).

Effects to Bull Trout Survival

Hansen et al. (2002d, p. 171) found that bull trout exposed to cadmium at 0.21 µg/L (slightly lower than the proposed chronic criterion for cadmium) experienced a 2% reduction in survival from the control after 55 days of exposure, while bull trout exposed to a much higher concentration of cadmium (0.46 µg/L) experienced no reduction in survival from the control. In the Hansen et al. (2002d) study, mortality did not become evident until concentrations reached very high levels (1.84 µg/L). Although mortality of 2% at the lower concentration may be within normal variation for the test, we will use a conservative approach and assume a 2% reduction in survival of the exposed bull trout with the implementation of the proposed chronic criterion for cadmium population every 3 years during the 25-year term, of the proposed action (Table 4-9, Column F).

Although Columbia River and Snake River foraging, migratory, and overwintering habitats encompass substantial areas where bull trout could be exposed to the proposed chronic criterion for cadmium, many of the bull trout utilizing these rivers also utilize the tributary river basins listed in Table 4-9. Chronic exposure of bull trout to cadmium at the proposed standard may occur within portions of Columbia River and Snake River foraging, migratory, and overwintering habitats, and such exposure is likely to cause adverse effects to the exposed individuals. However, because of the high flows within these two rivers, and the ability of bull trout to move through or avoid the highest concentrations of cadmium they encounter within large streams, the adverse effects to the bull trout associated with their exposure to chronic concentrations of cadmium in Columbia River and Snake River mainstem habitats are not likely to cause additional death or injury of affected bull trout that reside in the tributaries listed in Table 4-9.

Table 4-9. Estimated amount of bull trout mortality caused by exposure to the proposed chronic criterion level for cadmium.

A	B	C	D	E	F	G
Basin	Type of Effect	Effects per Modeling	Percent Habitat Exposed	Estimated Average Adult Population Size	Mortality of Adult Bull Trout	Reduced Growth of Adult Bull Trout
Grande Ronde River	Survival	2%	16%	150	0	
	Growth	100[1]%		150		24
Hells Canyon Complex	Survival	2%	22%	300	1	
	Growth	100[1]%		300		66
Hood River	Survival	2%	16%	150	0	
	Growth	100[1]%		150		24
Imnaha River	Survival	2%	33%	1,750	12	
	Growth	100[1]%		1,750		578
John Day River	Survival	2%	24%	400	2	
	Growth	100[1]%		400		96
Klamath River	Survival	2%	6%	2,050	2	
	Growth	100[1]%		2,050		123
Lower Deschutes River	Survival	2%	4%	1,750	1	
	Growth	100[1]%		1,750		70
Malheur River	Survival	N/A	0%	150	0	
	Growth			150		0
Odell Lake	Survival	N/A	0%	50	0	
	Growth			50		0
Powder River	Survival	2%	44%	300	3	
	Growth	100[1]%		300		132
Umatilla River	Survival	2%	28%	50	0	
	Growth	100[1]%		50		14
Upper Willamette River	Survival	2%	40%	300	2	
	Growth	100[1]%		300		120
Walla Walla River	Survival	2%	7%	1,750	2	
	Growth	100[1]%		1,750		123
Totals:				**9,150**	**25**	**1,370**

[1]*Represents a 13% reduction in growth to 100% of the exposed individuals.*

Summary of Effects to Bull Trout from Exposure to Cadmium at the Proposed Acute and Chronic Criteria

The following summary is based on a reasonable worst case scenario:

Acute Cadmium Exposure

Implementation of the proposed action is likely to cause the mortality of up to 165 adult bull trout along 820.6 miles of bull trout habitat within the action area during each 3-year period the proposed action.

Chronic Cadmium Exposure

Implementation of the proposed action is likely to kill up to 25 adult bull trout, and reduce the fitness of up to 1,370 adult bull trout in 820.6 miles of bull trout habitat within the action area during each 3-year period of the proposed action.

4.4.1.7 Effects of Chromium on the Bull Trout

Proposed Acute Criterion for Trivalent Chromium (Chromium III)

The proposed action establishes an acute chromium III criterion of 570 µg/L at 100 mg/L $CaCO_3$. The LC10 developed using direct data for bull trout exposure to chromium III is 1,844 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. The LC10 equates to a potential lethality of up to 10% of exposed individual bull trout. This means that the LC10 is nearly four times higher than the standard under review. Although the proposed criterion is much lower than the LC10 for the acute exposure of bull trout to chromium III, there could be a slight effect of even low levels of exposure that are likely to adversely affect the bull trout. However, biological effects are not expected to significantly disrupt bull trout breeding, feeding, or sheltering behavior or cause injury or death of exposed individuals.

Proposed Chronic Criterion for Chromium III

The proposed chronic criterion for chromium III is 74 µg/L (at 100 mg/L $CaCO_3$). No information is available on the effects of this criterion on bull trout reproduction, nor are any surrogate reproduction data available for this criterion. The rainbow trout was used as a surrogate for examining the effects of this proposed criterion on bull trout growth and survival because data on the effects of chronic chromium exposure to the rainbow trout are available in a form that can be readily modeled. The rainbow trout and the bull trout are similar taxonomically (both salmonids) and their life histories are more similar than other possible surrogates that we modeled. These surrogate models using rainbow trout response data were used to evaluate the direct effects of chromium III on bull trout growth and survival.

Effects to Bull Trout Growth

Based on our modeling results for rainbow trout, no reduction in the weight of bull trout eyed embryos is expected with chronic exposure to chromium III at the proposed criterion. On the basis of these results, chronic exposure of bull trout to chromium III at the numeric criterion level is considered to have insignificant effects to the growth of all bull trout potentially exposed within the action area.

Effects to Bull Trout Survival

Modeling results indicate a small potential for a reduction in bull trout survival from the eyed-embryo to egg-hatching stage (95% confidence limits 0.00% – 3.03%) every 3 years during the 25-year term of the proposed action based on exposure to the chronic chromium III criterion concentration. However, based on the 95% confidence limits, we conclude these effects are discountable.

Proposed Acute Criterion for Hexavalent Chromium (Chromium VI)

The proposed action establishes an acute chromium VI criterion of 16 µg/L at 100 mg/L $CaCO_3$. The LC10 developed using direct data for bull trout exposure to chromium is 5,963 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. The LC10 equates to a potential lethality of up to 10% of exposed individual bull trout. This means that the LC10 is nearly 373 times higher than the proposed criterion. Because the proposed standard is significantly lower than the LC10, the effects of acute exposure of bull trout to chromium VI are considered insignificant (i.e., effects that cannot be meaningfully measured, detected, or evaluated), as any effects would be indistinguishable from background effects.

The acute exposure of bull trout to chromium VI could occur in a very limited portion of the action area. For the reason expressed above, the effects of this exposure on the bull trout are likely to be insignificant (as define above).

Proposed Chronic Criterion for Chromium VI

The proposed chronic criterion for chromium VI is 11 µg/L (at 100 mg/L $CaCO_3$). No information is available on the effects of this criterion on bull trout reproduction nor is any surrogate data on reproduction available for this criterion. The rainbow trout was used as a surrogate for examining the effects of this proposed criterion on bull trout growth and survival because data on the effects of chronic chromium exposure to the rainbow trout are available in a form that can be readily modeled. The rainbow trout and the bull trout are similar taxonomically (both salmonids) and their life histories are more similar than other possible surrogates that we modeled. These surrogate models using rainbow trout response data were used to evaluate the direct effects of chromium VI on bull trout growth and survival.

Effects to Bull Trout Growth

Our modeling results indicate the likelihood for a 6.63% (95% confidence limits 3.28% –9.86%) reduction in cumulative final length (mm) of rainbow trout after a 7-month exposure to chromium VI from egg to fingerling size. The effects of chronic exposure to chromium VI exposure at the proposed criterion concentration are considered adverse to the growth of all exposed bull trout within the action area. This reduction in growth would be considered a direct injury to the affected individuals and could result in a disruption of the breeding, feeding, or sheltering behaviors of exposed individuals

Effects to Bull Trout Survival

Our modeling results indicate a likely reduction in the survival of affected bull trout by about 3.88% (95% confidence limits 0.00% –12.0%) based on a 7-month exposure to chromium VI at the chronic criterion concentration from the egg to fingerling stage. These results support a

finding that approximately 4% of the bull trout exposed at the chronic criterion concentration for chromium VI could be subject to effects that cause injury or death of those individuals every 3 years during the 25-year term of the proposed action.

However, while the modeling presents possible adverse effects to growth (~7%), and reduced survival (~4%) in bull trout at the criterion level, the likelihood of exposure at the proposed chronic criterion is so low that the potential to affect bull trout is discountable. The only location where exposure to chromium VI is possible would be the mainstem lower Columbia River from an unnamed creek. This potential for exposure would only be during low flows (less dilution) when it would be unlikely for bull trout to be present because low flows most often occur during late summer and early fall when they would be in spawning and rearing habitats further upstream in the various river basins.

Summary of Effects to the Bull Trout from Chromium Exposure at the Proposed Acute and Chronic Criteria

Acute Chromium III Exposure

The effects on the bull trout caused by acute exposure to chromium III are likely to be adverse, but are not likely to cause injury or death of affected individuals based on model results for rainbow trout exposed to the proposed acute criterion for chromium III.

Chronic Chromium III Exposure

The effects on the bull trout caused by chronic exposure to chromium III are likely to be insignificant and are not likely to cause injury or death of affected individuals based on model results for rainbow trout exposed to the proposed chronic criterion for chromium III.

Acute Chromium VI Exposure

It is our opinion that the effects of acute exposure to chromium VI are insignificant.

Chronic Chromium VI Exposure

It is our opinion that the effects of chronic exposure to chromium VI are discountable.

4.4.1.8 Effects of Copper on Bull Trout

Although there are limited data available on the direct effects of copper exposure on bull trout, there is considerable information that indicates that copper can have adverse impacts on other fish species. The presence of copper may alter fish behavior. Hansen et al. (1999) found that both chinook salmon and rainbow trout avoided water with low concentrations of copper. Although there are conflicting data concerning the precise effects of copper exposure on fish, there is sufficient research to suggest that it may cause reduced reproduction, reduced growth and vigor, and lessen the rate of survival.

Proposed Acute Criterion for Copper

The proposed acute criterion for copper is 13 µg/L (at 100 mg/L as CaCO3). The LC10 developed for juvenile bull trout exposure to copper is 15.4 µg/L (at 100 mg/L $CaCO_3$). On that basis, we have determined that the proposed copper criterion is likely to adversely affect and kill

about 10% of the individuals exposed to copper at the proposed acute criterion concentration every 3 years during the 25-year term of the proposed action (Table 4-10, Column B).

Adverse effects, resulting in bull trout mortality from acute exposure to copper are reasonably certain to occur. The extent of bull trout mortality was determined using the following formula: 10% (LC10) x % Exposed Habitat x Estimated Adult Population (Table 4-10).

Table 4-10. Estimates of bull trout mortality caused by exposure to copper at the proposed acute criterion.

A	B	C	D	E
Basin	LC10	Percent Habitat Exposed	Estimated Average Adult Population Size	Maximum Mortality of Adult Bull Trout per 3 Year Period
Grande Ronde River	10%	16%	150	2
Hells Canyon Complex	10%	22%	300	7
Hood River	10%	16%	150	2
Imnaha River	10%	33%	1,750	58
John Day River	10%	24%	400	10
Klamath River	10%	6%	2,050	12
Lower Deschutes River	10%	4.3%	1,750	8
Malheur River	N/A	0%	150	0
Odell Lake	N/A	0%	50	0
Powder River	10%	44%	300	13
Umatilla River	10%	28%	50	2
Upper Willamette River	10%	40%	300	12
Walla Walla River	10%	7%	1,750	12
Totals:			9,150	138

Proposed Chronic Criterion for Copper

The proposed chronic criterion for copper is 9 µg/L (at 100 mg/L as $CaCO_3$). Bull trout exposure to this concentration of copper was modeled for its effects on the reproduction, growth and survival of individual bull trout. We used direct bull trout data (Hansen et al. 2002b) in this analysis and also compared our results with data from McKim and Benoit (1971) for fish in the same genus as the bull trout (i.e., brook trout).

Effects to Bull Trout Reproduction

Our modeling results, based on data from Sauter et al. (1976), indicate that bull trout exposure to copper at the proposed chronic criterion is likely to reduce reproduction (reduction in egg hatchability, number of eggs per spawn, or number of eggs per female) by 76.5% (95% confidence limits = 43.7%- 92.2%) every 3 years during the 25-year term of the proposed action. However, the data reported by Sauter et al. (1976) may not realistically represent actual conditions for char. USEPA (2007, p. 187) determined that some of the data relied upon by Sauer et al. (1976) were not valid due, in part, to the extremely low viability of eggs (even within control groups) for some of the species considered. The USEPA (2007) also discounted large portions of the data reported by Sauter et al. (1976) due to an inability to fit portions of the data into models. For these reasons, we are not relying upon the data from Sauter et al. (1976) in this analysis.

We used a model (with the lowest mean square error and largest F-ratio) of fathead minnow (a bony fish that is distantly related to the bull trout) response to a copper concentration at the proposed chronic criterion as a surrogate for informing an assessment of effects to the bull trout. . The results indicate that at the proposed chronic criterion level, bull trout could experience up to a 61% reduction in eggs/spawn (56.4-65.3 95% confidence interval) every 3 years during the 25-year term of the proposed action. Although we were not able to model data from closer related species, there are direct data from brook trout, in the same genus as the bull trout, which we considered (see below).

McKim and Benoit (1971, pp. 658-659) found that only the highest concentrations of copper they tested (65.66 µg/L at 100 mg/L $CaCO_3$), affected hatchability of brook trout eggs. Concentrations at the proposed chronic criterion level (6.87 and 11.51 µg/L at 100 mg/L $CaCO_3$) do not affect the eggs differently from controls. Even though findings by McKim and Benoit (1971) indicate that copper exposure at the proposed chronic criterion level would not affect brook trout eggs, we find that it is possible that there could be some level of adverse effects to bull trout eggs at the proposed chronic criterion because copper levels at the chronic criterion will cause some reduction hatching of bull trout eggs. However, we are unable to quantify these effects since we lack meaningful data to translate these effects to eggs and subsequently to the effects to bull trout populations.

Effects to Bull Trout Growth

Our modeling results indicate a reduction in growth (reduction in mass, length, or biomass) in juvenile bull trout by 20.4% per individual (95% confidence limits = 9.46%-30%) for fish exposed to copper at the proposed chronic criterion concentration every 3 years during the 25-year term of the proposed action. However, Hansen et al. (2002b, p. 693) found that growth of

bull trout was about 10% less than that of the control after 60 days of exposure at dissolved copper concentrations between 11.69 and 81.75 µg/L (100 mg/L CaCO3), although there was no clear conclusion as to the effects of copper exposure on growth (length or weight).

We assessed the effects of bull trout exposure to copper at the proposed chronic concentration using bull trout data from Hansen et al. (2002b). Our modeling results indicate that there is the likelihood that such a concentration could reduce bull trout growth by 10% (Table 4-11, column C) every 3 years during the 25-year term of the proposed action.

Effects to Bull Trout Survival

Our modeling results indicate a reduction in survival of bull trout juveniles by 2.83% (95% confidence limits = 0-6.67%) at the proposed chronic criterion of 9 µg/L (at 100 mg/L as $CaCO_3$) every 3 years during the 25-year term of the proposed action. Hansen et al. (2002b, p. 693) found that significant mortality of bull trout caused by chronic exposure to dissolved copper was observed only in fish exposed to 81.75 µg/L (100 mg/L CaCO3). This is nine times the proposed chronic criterion concentration. On that basis, we conclude that the effects of copper exposure on the bull trout at the proposed chronic criterion are not likely to adversely affect the survival of individual bull trout.

Table 4-11. Bull trout injury caused by exposure to the proposed chronic criterion for copper.

A	B	C	D	E	G
Basin	**Type of Effect**	**Effects per Modeling**	**Percent Habitat Exposed**	**Estimated Average Adult Population Size**	**Injury due to Reduced Growth of Adult Bull Trout**
Grande Ronde River	Growth	100%	16%	150	24
Hells Canyon Complex	Growth	100%	22%	300	66
Hood River	Growth	100%	16%	150	24
Imnaha River	Growth	100%	33%	1,750	578
John Day River	Growth	100%	24%	400	96
Klamath River	Growth	100%	6%	2,050	123
Lower Deschutes River	Growth	100%	4%	1,750	70
Malheur River	Growth	N/A	0%	150	0
Odell Lake	Growth	N/A	0%	50	0
Powder River	Growth	100%	44%	300	132
Umatilla River	Growth	100%	28%	50	14
Upper Willamette River	Growth	100%	40%	300	120
Walla Walla River	Growth	100%	7%	1,750	123
Totals:				**9,150**	**1,370**

Although Columbia River and Snake River bull trout foraging, migratory, and overwintering habitats have substantial areas of potential exposure to copper at the proposed chronic criterion, many bull trout utilizing these rivers are also resident in the tributary streams listed in Table 4-11. Chronic exposure to copper at the proposed standard may occur within portions of these foraging, migratory, and overwintering habitats that could adversely impact bull trout. Therefore, chronic exposure within the Columbia River and Snake River foraging, migratory, and overwintering habitats may cause adverse effects to bull trout. Because of the high flows within these streams, and the ability of fish to move through or avoid the highest concentrations of copper they encounter within large rivers, the adverse effects associated with copper exposure in these habitats is not likely to rise to a level of additional mortality or injury.

Other Potential Effects to the Bull Trout caused by the Proposed Acute and Chronic Criteria for Copper Exposure

Olfaction

Olfactory Sensory Neuron (OSN) receptors serve an important function in fish. These structures are important in feeding, homing, mating and predator response behaviors (Tierney et al. 2010, pp. 3-5). Copper is known to have impacts on OSN receptors in relatively small concentrations. Copper concentrations of 10 µg/L reduced responses in coho salmon. A 30-minute exposure to copper at this concentration found a reduction of 70% in the response value from the pre-exposure value for the same fish. The same study found that recovery from this exposure was not rapid with the response value improving to only ~60% of the pre-exposure level after 5 hours exposure to flowing clean water (Tierney et al. 2010, p. 11).

DeForest et al. (2011a, pp. 340-341, and 346) found that the actual effects related to the olfaction of copper in nature vary widely depending on many features, including water hardness and dissolved oxygen content. They further demonstrate through modeling that adverse effects from olfaction may actually only occur at much higher concentrations than those presented by this action, and state that additional study may be needed to confirm what actual criteria would adequately protect all aquatic resources.

Because of the lack of available scientific information, and the wide variability of site-specific conditions found within the action area, we are unable to effectively analyze the potential olfaction-related effects of bull trout exposure to copper at the proposed acute and chronic criteria concentrations. We acknowledge that there may be some effects related to olfaction, but believe that it would difficult to distinguish those effects from naturally occurring processes at this time.

Acid Mine Drainage

Water exposed to trace metal contamination (including copper) can result from acid mine drainage (AMD) and other industrial and non-industrial discharge, and often has a lowered pH (Tay and Momade 2008, pp., 1, 6 and 9) (Tierney et al. 2010, P. 11). While a greater problem associated with coal mining in the eastern U.S., mineral and metal mining also may produce AMD. In addition to the acid contribution to surface waters, AMD may cause metals such as arsenic, cadmium, copper, silver, and zinc to leach from mine wastes. The increased metal load

causes environmental damage, and is of greater concern than the acidity in environmental terms (USGS 1994, p. 1).

The combination of the presence of copper and lowered pH greatly reduced the swimming speed in brown trout (*S. trutta*) (Beaumont et al. 2002 P. 1). This could be relevant to bull trout as many areas occupied by the bull trout have extensive histories of mining activity. Waiwood and Beamish (1978), found that levels of copper and low pH that cause "considerable" mortality to rainbow trout (*O. mykiss*) also reduce the swimming performance of the survivors. This, they concluded, was due to the effect of the pollutants upon both oxygen demand and maximum uptake, leading to a greater requirement for oxygen at any given speed ("loading" effect) but a lower maximum rate of uptake ("limiting" effect).

While there is some AMD occurring in Oregon over 95% of the AMD of environmental concern occurs in Pennsylvania, West Virginia and Western Maryland (The Energy Library 2011). AMD within bull trout habitat most likely exists in isolated cases associated with individual historic hard rock mines. Thus it seems likely that effects associated with these sites will be limited (intensity) to small areas, and that very few bull trout would be directly affected. Those individuals would probably be capable of moving away from these localized effects so that their exposure would be of short duration. Therefore, the probability that adverse effects to bull trout will occur from AMD is considered discountable.

Summary of Effects to the Bull Trout from Copper Exposure at the Proposed Acute and Chronic Criteria

The following summary is based on a reasonable worst case scenario:

Acute Copper Exposure

Acute exposure of the bull trout to copper at the proposed criterion concentration is likely to cause the death of up to 138 adult bull trout during each 3-year period during the projected 25-year term of the proposed action; these effects are likely to occur in surface waters along 820.6 miles of bull trout spawning, rearing, foraging, migratory, and overwintering habitats that are likely to contain a concentration of copper at the proposed acute criterion level (Table 4-1).

Chronic Copper Exposure

Chronic exposure of the bull trout to copper at the proposed criterion concentration is likely to cause injury/ reduced fitness of up to 1,371 adult bull trout (Table 4-11) during each 3-year period during the projected 25-year term of the proposed action; these effects are likely to occur in surface waters along 820.6 miles of bull trout spawning, rearing, foraging, migratory, and overwintering habitats that are likely to contain a concentration of copper at the proposed chronic criterion level (Table 4-2).

We are unable to quantify the exact number of bull trout eggs that may be affected as it is not possible to accurately inventory this life stage within the action area at this time. However, we assume that some small portion of eggs will be adversely affected along 260.8 miles of bull trout spawning and rearing habitat exposed to copper concentrations at the proposed chronic criterion

(Table 4-2) because data indicates that copper presence at this level will cause some reduction hatching of bull trout eggs.

4.4.1.9 Effects of Lead on Bull Trout

Proposed Acute Criterion for Lead

Implementation of the proposed action would establish an acute standard of 65µg/L for lead. The LC10 developed for bull trout exposure to lead is 953 µg/L for juvenile fish. Based on the comparison of the magnitude of these two values, we find that far fewer than 10% of the individuals of the bull trout exposed to the proposed acute criterion concentration for lead are likely to be killed, since the LC10 concentration is nearly 15 times and an order of magnitude higher than the proposed standard. For those reasons, we conclude that the effects of the proposed acute criterion for lead on the bull trout are likely to be insignificant (i.e., effects that cannot be meaningfully measured, detected, or evaluated).

Effects of Chronic Lead Exposure

The proposed chronic criterion for lead is 2.5 µg/L. Brook trout data were used as a surrogate for assessing the effects of this proposed criterion on bull trout reproduction, and rainbow trout data were used as a surrogate for assessing the effects of the proposed criterion on the survival of individual bull trout. Since the brook trout is in the same genus as the bull trout (*Salvelinus*) and the rainbow trout is in the same family as the bull trout (*Salmonidae*), they serve as suitable surrogates. We found no specific data on the effects of lead on the growth of the bull trout or any other closely related species.

Effects to Bull Trout Reproduction

Modeling results of brook trout response to lead exposure at the proposed chronic concentration indicate a 3.17% reduction in egg hatching success (95% confidence limits = 0% - 21.7%, brook trout surrogate) every 3 years during the 25-year term of the proposed action. A possible reduction in the egg hatching rate by 3% is difficult to separate from natural background processes. However, another model suggests that chronic exposure to lead at the proposed criterion could cause deformities in 1.15% of individual bull trout exposed, that affect reproduction in hard water conditions (95% confidence limits: 0.03% – 34.0%; rainbow trout surrogate). Thus, adverse effects are likely to result in reduced fitness to adult bull trout (1.15% of exposed individuals) because of possible deformities that affect reproduction in areas where hard water conditions exist; such effects would occur every 3 years during the 25-year term of the proposed action. The risk of this effect occurring may be small but is not discountable. Therefore, chronic exposure of bull trout to lead at the proposed chronic criterion is likely to cause adverse effects to the bull trout (Table 4-12) every 3 years during the 25-year term of the proposed action.

Effects to Bull Trout Survival

Modeling results of brook trout response to the proposed chronic criterion concentration did not reveal a reduction in the survival of juvenile fish. The modeling results indicate no definitive correlation between chronic lead exposure at the proposed criterion level and fish survival.

Thus, effects to the survival of bull trout are considered insignificant (i.e., effects that cannot be meaningfully measured, detected, or evaluated) at the proposed chronic criterion.

Table 4-12. Effects to bull trout reproduction from chronic lead exposure.

A	B	C	D	E	F
Basin	Type of Effect	Effects per Modeling	Percent Habitat Exposed	Estimated Average Adult Population Size	Reduction In Reproduction of Adult Bull Trout per 3-Year Period
Grande Ronde River	Reproduction	1.2%	16%	150	0
Hells Canyon Complex	Reproduction	1.2%	22%	300	1
Hood River	Reproduction	1.2%	16%	150	0
Imnaha River	Reproduction	1.2%	33%	1,750	7
John Day River	Reproduction	1.2%	24%	400	1
Klamath River	Reproduction	1.2%	6%	2,050	1
Lower Deschutes River	Reproduction	1.2%	4%	1,750	1
Malheur River	Reproduction	N/A	0%	150	0
Odell Lake	Reproduction	N/A	0%	50	0
Powder River	Reproduction	1.2%	44%	300	2
Umatilla River	Reproduction	1.2%	28%	50	0
Upper Willamette River	Reproduction	1.2%	40%	300	1
Walla Walla River	Reproduction	1.2%	7%	1,750	1
Totals:				9,150	15

Although Columbia River and Snake River foraging, migratory, and overwintering habitats for the bull trout have substantial areas of potential exposure of bull trout to lead at the proposed chronic criterion, many of the bull trout utilizing these two rivers also reside in the tributary river basins listed in Table 4-12. Chronic exposure to lead at the proposed standard may occur within portions of these foraging, migratory, and overwintering habitats that could adversely affect bull trout. Because of the high flows within these two rivers, and the ability of fish to move through or avoid the highest concentrations they encounter within large streams, the adverse effects to the bull trout associated with lead exposure in these habitats are not likely to be additive to the effects to the bull trout from lead exposure that are likely to occur in the tributary streams listed in Table 4-12.

Summary of Effects to the Bull Trout from Lead Exposure

The following summary is based on a reasonable worst case scenario:

Acute Lead Exposure

Because the LC10 concentration is nearly 15 times and an order of magnitude higher than the proposed standard, we conclude that the effects of the proposed acute criterion for lead on the bull trout are likely to be insignificant (i.e., effects that cannot be meaningfully measured, detected, or evaluated).

Chronic Lead Exposure

Chronic exposure of bull trout to lead at the proposed criterion concentration is likely to cause injury and reduced fitness (deformities that adversely affect reproduction) of up to 15 adult bull trout per 3-year period over the 25-year term of the proposed action in surface waters in 820.6 miles of bull trout habitat within the action area.

4.4.1.10 Effects of Nickel on Bull Trout

In aqueous solution, nickel occurs mostly as the green hexa-aquanickel (II) ion, $Ni(H20)_6^{2+}$. The nickel ion content of ground water may increase as a result of the oxidation of natural nickel containing ferrosulphate deposits, from mining and milling processes, and from plating and manufacturing (Tay and Mamade 2008, p. 12; Svobodova et al. 1993, p. 27; Ntengwe and Maseka 2006, pp. 5-6).

Exposure to nickel can adversely affect fish. Sub-lethal concentrations of nickel were found to modify behavior and increase oxygen demand in Nile tilapia (Hadeed et al. 2008, p. 1388). Nickel exposure has the potential to create changes to red blood cells, and damage to internal organs and gills (Al-Attar 2007, pp. 80-82).

Proposed Acute Criterion for Nickel

The proposed acute standard for nickel is 468 µg/L (at 100 mg/L as CaCO3). The LC10 developed for bull trout exposure to nickel is 4,017 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. The LC10 is nearly 9 times higher and an order of magnitude higher than the proposed standard of nickel. On that basis, we conclude that the proposed standard would affect much less than 10% of the exposed bull trout population every 3 years during the 25-year term of the proposed action and to such a small extent that those effects are likely to be insignificant (i.e., not meaningfully measured, detected, or evaluated).

Effects of Chronic Nickel Exposure

The proposed chronic nickel criterion is 52 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. Data on flathead minnow response to nickel exposure were used as a surrogate to assess the effects of this proposed criterion on bull trout reproduction. Although the flathead minnow is not closely related to the bull trout, no data are available for species that are closely related to the bull trout to assess the effects of nickel exposure on reproduction. The rainbow trout (same family - *Salmonidae*) was selected as a surrogate for assessing the effects of nickel exposure at the proposed chronic criterion on both bull trout growth and survival because exposure response data are available and can be readily modeled. Since the rainbow trout and the bull trout are similar taxonomically and have similar life history and habitat preferences, the rainbow trout can serve as an adequate surrogate for this analysis.

Effects to Bull Trout Reproduction

Modeling results indicate that bull trout exposure to this chronic criterion concentration is likely to reduce fecundity by about 7.9% (95% confidence limits = 0% - 36%, flathead minnow surrogate) every 3 years during the 25-year term of the proposed action. A possible reduction of 7.9% in the egg production per spawn would be difficult to separate from such effects due to natural background processes.

We have little reliable data concerning the number of bull trout eggs produced annually within the action area. Egg production varies widely by region, season, local population, life form, and size of the female (Pratt 2003, p.50). We also possess little data as to the natural ratio of egg survival to number of eggs produced, which may also vary by the same factors listed above. Therefore, although it is likely that chronic exposure of bull trout to lead at the proposed criterion concentration will have some adverse effects on bull trout reproduction due to reduced fecundity, we are unable to more specifically quantify these effects at the population level.

Effects to Bull Trout Growth

Our modeling results for rainbow trout indicate that nickel exposure at the proposed chronic criterion concentration is likely to cause a 7.62% reduction in the growth/body weight (95% confidence limits 0% – 37.7 %) of individual bull trout at an early life stage (Nebecker et al. 1984) every 3 years during the 25-year term of the proposed action. Although the test results reported by Nebecker et al. (1984) involved very young fish, we assume these data are applicable to adult fish for purposes of this analysis.

Chronic exposure to nickel at the proposed criterion is likely to cause a measurable reduction (by 7.62%) in the body weight of all bull trout that would be exposed within the action area (Table 4-19, column G) every 3 years during the 25-year term of the proposed action.

Effects to Bull Trout Survival

Modeling results for rainbow trout exposure response data indicate a likely reduction in bull trout survival of 11.2% (95% confidence limits 0% – 37.76 %). This model is based on the reduction in survival of rainbow trout embryos in their early life stage (Test 1) with newly fertilized eggs exposed to lead for 85 days (Nebecker et al. 1984). Based on this modeling, we assume approximately 11% of the bull trout exposed at the chronic criterion concentration for nickel

could be killed (Table 4-13, column F) every 3 years during the 25-year term of the proposed action.

Although Columbia River and Snake River foraging, migratory, and overwintering habitats for the bull trout have substantial areas of potential exposure of bull trout to nickel at the proposed chronic criterion, many of the bull trout utilizing these two rivers also reside in the tributary river basins listed in Table 4-13. Chronic exposure to nickel at the proposed standard may occur within portions of these foraging, migratory, and overwintering habitats that could adversely affect bull trout. Because of the high flows within these two rivers, and the ability of fish to move through or avoid the highest concentrations they encounter within large streams, the adverse effects to the bull trout associated with nickel exposure in these habitats are not likely to be additive to the effects to the bull trout from nickel exposure that are likely to occur in the tributary streams listed in Table 4-13.

Table 4-13. Mortality and reduced growth of bull trout caused by chronic nickel exposure.

A	B	C	D	E	F	G
Basin	Type of Effect	Effects per Modeling	Percent Habitat Exposed	Estimated Average Adult Population Size	Maximum Mortality of Adult Bull Trout per 3-Year Period	Injury due to Reduced Growth of Adult Bull Trout per 3-Year Period
Grande Ronde River	Survival	11%	16%	150	3	
	Growth	100%[1]		150		24
Hells Canyon Complex	Survival	11%	22%	300	7	
	Growth	100%[1]		300		66
Hood River	Survival	11%	16%	150	3	
	Growth	100%[1]		150		24
Imnaha River	Survival	11%	33%	1,750	64	
	Growth	100%[1]		1,750		578
John Day River	Survival	11%	24%	400	11	
	Growth	100%[1]		400		96
Klamath River	Survival	11%	6%	2,050	14	
	Growth	100%[1]		2,050		123
Lower Deschutes River	Survival	11%	4%	1,750	8	
	Growth	100%[1]		1,750		70
Malheur River	Survival	N/A	0%	150	0	
	Growth			150		
Odell Lake	Survival	N/A	0%	50	0	
	Growth			50		0
Powder River	Survival	11%	44%	300	15	
	Growth	100%[1]		300		132
Umatilla River	Survival	11%	28%	50	0	
	Growth	100%[1]		50		14
Upper Willamette River	Survival	11%	40%	300	13	
	Growth	100%[1]		300		120
Walla Walla River	Survival	11%	7%	1,750	13	
	Growth	100%[1]		1,750		123
Totals:				9,150	151	1,370

[1] *~8% reduction of growth to all exposed individuals*

Summary of Effects to the Bull Trout from Nickel Exposure

The following summary is based on a reasonable worst case scenario:

Acute Nickel Exposure

The LC10 for the bull trout is nearly 9 times higher and an order of magnitude higher than the proposed acute standard for nickel. On that basis, we conclude that the proposed standard would affect much less than 10% of the exposed bull trout population every 3 years during the 25-year term of the proposed action and to such a small extent that those effects are likely to be insignificant (i.e., not meaningfully measured, detected, or evaluated).

Chronic Nickel Exposure

Based on model results relying upon rainbow trout response data for exposure to nickel at the proposed chronic criterion concentration, we conclude that chronic exposure of bull trout to nickel at the proposed chronic standard is likely to kill up to 151 adult bull trout, and injure/reduce the fitness (via reduced growth) of up to 1,370 individual adult bull trout per 3-year period over the 25-year term of the proposed action in surface waters along 820.6 miles of bull trout habitat within the action area (Table 4-2).

We are unable to quantify the exact number of bull trout eggs that may be affected as it is not possible to accurately inventory this life stage within the action area at this time. However, we assume that some small portion of eggs will be adversely affected every 3 years during the 25-year term of the proposed action along 260.8 miles of bull trout spawning and rearing habitat exposed to nickel concentrations at the proposed chronic criterion (Table 4-2) because modeling indicates a probable 7.9% of fecundity in bull trout when exposed at the proposed criterion.

4.4.1.11 Effects of Pentachlorophenol on Bull Trout

As discussed in the "Exposure" section of this document, it is highly unlikely that bull trout could be exposed to pentachlorophenol. The only area that could be potentially exposed would be in the mainstem lower Columbia River where bull trout forage, migrate or overwinter in late winter and early spring. This potential to be exposed is further reduced because a very low number of bull trout use the lower Columbia River and then, only on occasion. In addition, the low concentrations of pentachlorophenol likely to be present in the lower Columbia River combined with high flows when bull trout are likely to be present, make the likelihood of exposure and adverse effects to the bull trout caused by exposure to pentachlorophenol very remote. For those reasons, we conclude that the effects of pentachlorophenol on the bull trout are discountable (i.e., extremely unlikely to occur).

4.4.1.12 Effects of Silver on Bull Trout

Proposed Acute Criterion for Silver

The proposed acute standard for silver is 3.2 µg/L (at 100 mg/L as CaCO3). The LC10 developed for bull trout exposure to silver is 11.35 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. Since the proposed acute standard is 72% less than the LC10 acute concentration for silver, we conclude that while some adverse effects may occur to the bull trout they are likely to be sub-

lethal and not cause a significant disruption of breeding, feeding, migrating, or sheltering behavior.

Effects of Chronic Silver Exposure

The proposed chronic silver criterion is 0.1 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. Data on flathead minnow response to exposure at the proposed standard were used as a surrogate to assess effects on bull trout reproduction. No other fish species data were available to inform that analysis. Data on rainbow trout response to exposure at the proposed standard were used as a surrogate to assess effects on bull trout growth and survival because no direct bull trout data are available.. The rainbow trout and the bull trout are both in the *Salmonidae* family and share a similar life history and habitat use. .

Effects to Bull Trout Reproduction

Modeling results indicate that bull trout exposure to the proposed chronic criterion for silver is likely to reduce the fecundity of affected individual females by about 1.36% (95% confidence limits = 0% – 6.33%, rainbow trout surrogate) every 3 years during the 25-year term of the proposed action. However, such a small reduction in the hatching rate of eggs would be difficult to separate from the effects of natural background processes on bull trout egg hatching rates.

Effects to Bull Trout Growth

Modeling results indicate that bull trout exposure to the proposed chronic criterion for silver is likely to cause a 99% (95% confidence limits 98.9% – 99.1 %) reduction in the growth (reduction in length) of affected bull trout every 3 years during the 25-year term of the proposed action, based on the response of an early life stage (embryo-larval test) of steelhead trout that were exposed to the proposed chronic concentration for silver for 60 days (Nebecker et al. 1983). The steelhead trout is a suitable surrogate for assessing effects to the bull trout because it is in the same family (*Salmonidae*) and uses similar habitats as the bull trout. Although the test referenced above was done on very young fish; we are giving the benefit of doubt to the bull trout by assuming for purposes of this analysis that these response data and modeling results apply to adult fish as well.

Individual bull trout subject to a 99% reduction in growth, especially at early life stages, are likely to injured (Table 4-14, column G).

Effects to the Survival of Individual Bull Trout

Modeling results based on response data for the rainbow trout [embryo-larval test; fish exposed to silver for 18 months (Davies et al. 1978)] indicate that bull trout exposed to the proposed chronic silver criterion are likely to have a reduced survival of 18.9% (95% confidence limits 4.18% – 52.2) every 3 years during the 25-year term of the proposed action(Table 4-14, column F).

Although Columbia River and Snake River foraging, migratory, and overwintering habitats for the bull trout have substantial areas of potential exposure of bull trout to silver at the proposed chronic criterion, many of the bull trout utilizing these two rivers also reside in the tributary river basins listed in Table 4-14. Chronic exposure to silver at the proposed standard may occur

within portions of these foraging, migratory, and overwintering habitats that could adversely affect bull trout. Because of the high flows within these two rivers, and the ability of fish to move through or avoid the highest concentrations they encounter within large streams, the adverse effects to the bull trout associated with silver exposure in these habitats are not likely to be additive to the effects to the bull trout from silver exposure that are likely to occur in the tributary streams listed in Table 4-14.

Summary of Effects to the Bull Trout from Silver Exposure

The following is based on a reasonable worst case scenario:

Acute Silver Exposure

Since the proposed acute standard is 72% less than the LC10 acute concentration for silver, we conclude that while some adverse effects may occur to the bull trout, these effects are likely to be sub-lethal and not cause a significant disruption of breeding, feeding, migrating, or sheltering behavior during each 3-year period during the 25-year term of the proposed action.

Chronic Silver Exposure

We conclude that bull trout exposure to the proposed chronic criterion concentration of silver is likely to cause mortality of 263 adult bull trout during each 3-year period over the 25-year term of the proposed action, and injure another 1,371 individual adult bull due to reduced growth and fitness each 3-year period over the 25-year term of the proposed action in surface waters along 820.6 miles of bull trout habitat within the action area.

We are unable to quantify the exact number of bull trout eggs that may be affected by exposure to the proposed acute or chronic concentrations of silver because it is not possible to accurately inventory this life stage within the action area at this time. However, we assume that some small portion of bull trout eggs will be exposed and adversely affected along 260.8 miles of bull trout spawning and rearing habitat exposed to silver concentrations at the proposed chronic criterion because our modeling indicates a reduction in the hatching rate of bull trout eggs exposed at the chronic criterion.

Table 4-14. Mortality and injury of bull trout caused by chronic silver exposure for each 3-year period over the term of the proposed action.

A	B	C	D	E	F	G
Basin	Type of Effect	Effects per Modeling	Percent Habitat Exposed	Estimated Average Adult Population Size	Maximum Mortality of Adult Bull Trout	Reduced Fitness (Injury from reduced growth) of Adult Bull Trout
Grande Ronde River	Survival	19%	16%	150	5	
	Growth	100%		150		24
Hells Canyon Complex	Survival	19%	22%	300	13	
	Growth	100%		300		66
Hood River	Survival	19%	16%	150	5	
	Growth	100%		150		24
Imnaha River	Survival	19%	33%	1,750	110	
	Growth	100%		1,750		578
John Day River	Survival	19%	24%	400	18	
	Growth	100%		400		96
Klamath River	Survival	19%	6%	2,050	23	
	Growth	100%		2,050		123
Lower Deschutes River	Survival	19%	4%	1,750	13	
	Growth	100%		1,750		70
Malheur River	Survival	N/A	0%	150	0	
	Growth			150		0
Odell Lake	Survival	N/A	0%	50	0	
	Growth			50		0
Powder River	Survival	19%	44%	300	25	
	Growth	100%		300		132
Umatilla River	Survival	19%	28%	50	3	
	Growth	100%		50		14
Upper Willamette River	Survival	19%	40%	300	23	
	Growth	100%		300		120
Walla Walla River	Survival	19%	7%	1,750	23	
	Growth	100%		1,750		123
Totals:				9,150	261	1,370

4.4.1.13 Effects of Zinc on Bull Trout

Fish can be exposed to zinc from many sources. Effluent from mining is a common source of zinc contamination and is a danger to a variety of species that inhabit streams. Zinc is a metal used in galvanizing and alloying and is also used in manufacturing electric goods, dying, insecticides and cosmetics. It is listed as one of the 129 priority pollutants (Ntengwe and Maseka 2006, pp. 1- 6).

Proposed Acute Criterion for Zinc

The proposed acute standard concentration for zinc is 120µg/L (at 100 mg/L as CaCO3). The LC10 concentration for zinc (based on direct data on the bull trout's response to zinc exposure) is 100 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. Since the proposed acute standard exceeds the LC10, more than 10% of the exposed bull trout individuals are likely to be killed every 3 years during the 25-year term of the proposed action. Because we are unable to determine the exact portion of the exposed bull trout population that will be affected by the proposed acute standard, we have estimated the affected portion of the exposed bull trout population by adding the lowest and highest percent mortality values for LC10 test results and dividing by 2. For example, if four exposure test results yield 1%, 3%, 24%, and 73% mortality of exposed bull trout individuals, the calculation of the acute effect level in terms of bull trout mortality is 1% + 73% = 74%/2 = 37% (Table 4-15, Column B). The number of bull trout likely to be killed was determined using the following formula: 37% x % Exposed Habitat (Table 4-15) x Estimated Adult Population (Table 4-15).

Table 4-15. Mortality of bull trout caused by their exposure to the proposed acute criterion concentration for zinc.

A	B	C	D	E
Basin	% Reduction in Survival at the Acute Criterion	Percent Habitat Exposed	Estimated Average Adult Population Size	Maximum Mortality of Adult Bull Trout per 3-year period
Grande Ronde River	37%	16%	150	9
Hells Canyon Complex	37%	22%	300	24
Hood River	37%	16%	150	9
Imnaha River	37%	33%	1,750	214
John Day River	37%	24%	400	36
Klamath River	37%	6%	2,050	46
Lower Deschutes River	37%	4%	1,750	26
Malheur River	N/A	0%	150	0
Odell Lake	N/A	0%	50	0
Powder River	37%	44%	300	49
Umatilla River	37%	28%	50	5
Upper Willamette River	37%	40%	300	44
Walla Walla River	37%	7%	1,750	45
Totals:			9,150	507

Effects of Chronic Zinc Exposure

The proposed chronic criterion for zinc is 120 µg/L (at 100 mg/L $CaCO_3$) for juvenile fish. Exposure response data for the brook trout were used for assessing the effects of this proposed standard on bull trout reproduction because no exposure response data are available for the bull trout. The brook trout is a suitable surrogate because it's in the same genus (*Salvelinus*) as the bull trout. Exposure response data for the rainbow trout were used for the same reasons to evaluate the effects of this proposed criterion on bull trout growth and survival.

Effects to Bull Trout Reproduction

Modeling results indicate that exposure of the bull trout to the proposed chronic criterion concentration for zinc is likely to reduce the bull trout egg hatching rate by approximately 25.2% using a midpoint calculation based on the brook trout surrogate data (Holcombe et al. 1979). This effect would occur every 3 years during the 25-year term of the proposed action. We are unable to quantify the exact number of bull trout eggs that may be affected as it is not possible to accurately inventory this life stage within the action area at this time. However, we assume that some small portion of eggs will be adversely affected and killed every 3 years during the 25-year term of the proposed action along 260.8 miles of bull trout spawning and rearing habitat during each 3-year period that bull trout eggs are exposed to zinc concentrations at the proposed chronic criterion over the 25-year term of the proposed action (Table 4-2).

Effects to Bull Trout Growth

Modeling results indicate that bull trout exposure to the proposed zinc criterion is likely to cause a 26.6% reduction in growth (biomass) of exposed individuals (all life stages) every 3 years during the 25-year term of the proposed action, based on the response of rainbow trout eggs exposed to zinc at the proposed standard for up to 63 days at 30 mg/L hardness (Brinkman and Hansen 2004). Although these tests were done on very young fish, in the face of uncertainty, we are giving the benefit of doubt to the bull trout by assuming these data apply to adult fish as well (Table 4-16, column G).

Effects to Bull Trout Survival

Modeling results of rainbow trout exposure response data (Mebane et al. 2008) indicate that bull trout exposure to the proposed zinc criterion is likely to cause a reduction in individual bull trout survival of 17.7% every 3 years during the 25-year term of the proposed action., Based on this information, approximately 18% of the bull trout exposed at the proposed chronic zinc criterion are likely to be killed every 3 years during the 25-year term of the proposed action (Table 4-16, column F).

Although Columbia River and Snake River foraging, migratory, and overwintering habitats for the bull trout have substantial areas of potential exposure of bull trout to zinc at the proposed chronic criterion, many of the bull trout utilizing these two rivers also reside in the tributary river basins listed in Table 4-16. Chronic exposure to zinc at the proposed standard may occur within portions of these foraging, migratory, and overwintering habitats that could adversely affect bull trout. Because of the high flows within these two rivers, and the ability of fish to move through or avoid the highest concentrations they encounter within large streams, the adverse effects to the bull trout associated with zinc exposure in these habitats are not likely to be additive to the effects from zinc exposure that is likely to occur in the tributary streams listed in Table 4-16.

Table 4-16. Mortality and injury of bull trout caused by chronic zinc exposure during each 3-year period of the proposed action.

A	B	C	D	E	F	G
Basin	Type of Effect	Effects per Modeling	Percent Habitat Exposed	Estimated Average Adult Population Size	Maximum Mortality of Adult Bull Trout	Injury/ Reduced Fitness of Adult Bull Trout
Grande Ronde River	Survival	18%	16%	150	4	
	Growth	100%[1]		150		24
Hells Canyon Complex	Survival	18%	22%	300	12	
	Growth	100%[1]		300		66
Hood River	Survival	18%	16%	150	4	
	Growth	100%[1]		150		24
Imnaha River	Survival	18%	33%	1,750	104	
	Growth	100%[1]		1,750		578
John Day River	Survival	18%	24%	400	17	
	Growth	100%[1]		400		96
Klamath River	Survival	18%	6%	2,050	22	
	Growth	100%[1]		2,050		123
Lower Deschutes River	Survival	18%	4%	1,750	13	
	Growth	100%[1]		1,750		70
Malheur River	Survival	N/A	0%	150	0	
	Growth			150		0
Odell Lake	Survival	N/A	0%	50	0	
	Growth			50		0
Powder River	Survival	18%	44%	300	24	
	Growth	100%[1]		300		132
Umatilla River	Survival	18%	28%	50	3	
	Growth	100%[1]		50		14
Upper Willamette River	Survival	18%	40%	300	22	
	Growth	100%[1]		300		120
Walla Walla River	Survival	18%	7%	1,750	22	
	Growth	100%[1]		1,750		123
Totals:				**9,150**	**266**	**1,370**

[1]*~27% reduction in growth to all exposed individuals*

Summary of effects to bull trout from zinc exposure

The following is based on a reasonable worst case scenario:

Acute Zinc Exposure

Bull trout exposure to zinc at the proposed acute criterion is likely to result in the mortality of up to 507 adult bull trout in surface waters along 820.6 miles of habitat (Table 4-16) within the action area over each 3-year period during the 25-year term of the proposed action.

Chronic Zinc Exposure

Bull trout exposure to zinc at the proposed chronic criterion is likely to kill up to 266 adult bull trout, and injure (via reduced fitness) up to another 1,370 individual adult bull trout during each-3 year period over the 25-year term of the proposed action in 820.6 miles of bull trout habitat (Table 4-2) within the action area.

We are unable to quantify the exact number of bull trout eggs that may be affected by exposure to the proposed chronic concentration of zinc that reduces egg hatching rates by 25.2% because it is not possible to accurately inventory this life stage within the action area at this time. However, we assume that some small portion of bull trout eggs will be exposed and adversely affected along 260.8 miles of bull trout spawning and rearing habitat exposed to zinc concentrations at the proposed chronic criterion (Table 4-16) based on this modeling.

4.4.1.14 Aggregate Effects of the Proposed Acute and Chronic Criterion on the Bull Trout

Table 4-17 and Table 4-18 summarize the anticipated effects of the proposed acute and chronic standards for each toxic substance on the bull trout. The aggregate effects of the proposed action include the likely mortality of up to 1,929 adult bull trout and the injury (as a result of reduced fitness) of another 6,953 adult bull trout during each 3-year period of the 25-year term of the proposed action.

The above estimates are based on modeling and assumptions that, in the face of uncertainty, give the benefit of doubt to the bull trout. It is highly likely that the reduction in bull trout numbers is largely overestimated. Since the details of all potential sources of exposure are not available, for this consultation we assumed that any potential source would expose bull trout to the toxic substances at issue, and we assumed that for all areas of exposure that all bull trout would be exposed to all toxic substances (i.e., all bull trout within the action area exposed to all of the toxic substances at proposed criterion levels); see Exposure Analysis on p. 132. In addition, the models considered the effects of the action on very young juvenile fish, which are more sensitive to exposure than adults, but for purposes of this analysis we assumed that the effects to adult fish were the same as the effects to juveniles. We used adult abundance because it is the most reliable data that we have to base the analysis of the effects.

Because the analysis is based on potential exposure, we looked at the effects to bull trout in portions of their habitat with the potential to be exposed. Thus, we found that bull trout within approximately 820.6 miles of approximately 3,400 miles of habitat within the action area have the potential for exposure to the chemicals in this proposed action at the proposed acute and

chronic concentrations. Conversely, this means that more than 75% of bull trout habitat within the action area has little or no real potential for exposure to the chemicals in this proposed action.

The aggregate effect of reduced bull trout fitness caused by implementation of the proposed action is 6,953 bull trout with reduced fitness every 3 years during the 25-year term of the proposed action within 820.6 miles of bull trout habitat.

Summary of combined effects to bull trout

Acute exposure combined effects

Our analysis indicates that there is a maximum likelihood that 813 adult bull trout will be killed from acute exposure to Cadmium, Copper and Zinc across the action area (Table 4-17). Additionally, there will likely be some effect from acute exposure to Aluminum, Ammonia, Chromium III, Selenium, and Silver. However, we do not believe that these effects will significantly disrupt breeding, feeding or sheltering behavior or cause injury or death of affected individual bull trout.

Chronic exposure combined effects

Our analysis indicates that 1,116 bull trout will be killed and 6,953 bull trout will be injured through reduced fitness. Additional adverse effects to reproduction are also likely to occur from chronic exposure to Ammonia, Nickel and Zinc. We are unable to quantify these effects to eggs.

Table 4-17. Summary of adverse effects of the proposed action on the bull trout due to acute exposure to 8 toxic elements.

Basin	Aluminum	Ammonia	Cadmium	Chromium III	Copper	Selenium	Silver	Zinc	Bull Trout killed per 3 Year Period
Grande Ronde River	A	A	3	A	2	A	A	9	14
Hells Canyon Complex	A	A	8	A	7	A	A	24	39
Hood River	A	A	3	A	2	A	A	9	14
Imnaha River	A	A	69	A	58	A	A	214	341
John Day River	A	A	12	A	10	A	A	36	58
Klamath River	A	A	15	A	12	A	A	46	73
Lower Deschutes River	A	A	8	A	8	A	A	26	45
Powder River	A	A	19	A	13	A	A	49	81
Umatilla River	A	A	2	A	2	A	A	5	9
Upper Willamette River	A	A	14	A	12	A	A	44	70
Walla Walla River	A	A	15	A	12	A	A	45	72
Totals by Chemical			**168**		**138**			**507**	**813**
A = Adverse effects that do not rise to the level of significantly disrupting breeding, feeding, or sheltering behavior or cause injury or death of affected individuals									

Table 4-18. Summary of adverse effects of the proposed action on the bull trout due to chronic exposure to 9 toxic elements.

Basin	Type of Effect (L= Lethal) (RF = Reduced Fitness)	Ammonia	Arsenic	Cadmium	Copper	Lead	Nickel	Selenium	Silver	Zinc	Total Bull trout killed per 3 Year Period	Reduced Fitness Total per 3 Year Period
Grande Ronde River	L	2	1	0	NA	0	3	5	5	4	20	
	RF	NA	NA	24	24	0	24	NA	24	24		120
Hells Canyon Complex	L	3	4	1	NA	0	7	13	13	12	53	
	RF	NA	NA	66	66	1	66	NA	66	66		331
Hood River	L	4	4	0	NA	0	3	5	5	4	25	
	RF	NA	NA	24	24	0	24	NA	24	24		120
Imnaha River	L	0	29	12	NA	0	64	116	110	104	435	
	RF	NA	NA	578	578	7	578	NA	578	578		2,897
John Day River	L	2	5	2	NA	0	11	19	18	17	74	
	RF	NA	NA	96	96	1	96	NA	96	96		481
Klamath River	L	14	7	2	NA	0	14	25	23	22	107	
	RF	NA	NA	123	123	1	123	NA	123	123		616
Lower Deschutes River	L	12	4	1	NA	0	8	14	13	13	65	
	RF	NA	NA	70	70	1	70	NA	70	70		351
Powder River	L	5	7	3	NA	0	15	26	25	24	105	
	RF	NA	NA	132	132	2	132	NA	132	132		660
Umatilla River	L	3	6	0	NA	0	0	3	3	3	18	
	RF	NA	NA	14	14	0	14	NA	14	14		70
Upper Willamette River	L	18	6	2	NA	0	13	24	23	22	108	
	RF	NA	NA	120	120	1	120	NA	120	120		601
Walla Walla River	L	15	6	2	NA	0	13	25	23	22	106	
	RF	NA	NA	123	123	1	123	NA	123	123		616
Totals	L	78	79	25	0	0	151	256	261	266		6,953
	RF	0	0	1,370	1,370	15	1,370	NA	1,370	1,370	1,116	

4.4.2 Effects of the Proposed Action on Bull Trout Critical Habitat

Critical habitat (CH) was designated for the bull trout in the action area on October 18, 2010. The rule identifies 32 Critical Habitat Units (CHUs) that are essential to the conservation of the bull trout and nine primary constituent elements (PCEs) for bull trout that are essential for the primary biological needs of foraging, reproducing, rearing of young, dispersal, genetic exchange, or sheltering. These PCEs are described below:

1. Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia.

2. Migratory 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 with features such as large wood, side channels, pools, undercut banks and 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 at the upper end of this range. Specific temperatures within this range will vary depending on bull trout life-history stage and form; geography; elevation; diurnal and seasonal variation; shade, such as that provided by riparian habitat; and local groundwater influence.

6. In spawning and rearing areas, substrates 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 (e.g., less than 12%) of fine substrate less than 0.85 mm (0.03 in.) in diameter and minimal embeddedness of these fines in larger substrates are characteristic of these conditions.

7. A natural hydrograph, including peak, high, low, and base flows within historic and seasonal ranges or, if flows are controlled, they minimize departures from a natural hydrograph.

8. Sufficient water quality and quantity such that normal reproduction, growth, and survival are not inhibited.

9. Few or no nonnative predatory (e.g., lake trout, walleye, northern pike, smallmouth bass); interbreeding (e.g., brook trout); or competitive (e.g., brown trout) species present, or such species are temporally or spatially isolated from bull trout.

Table 4-19 summarizes the effects of the proposed standards on the PCEs of bull trout CH.

Table 4-19. Effects of the proposed standards on the PCEs of bull trout CH.

Toxic	PCE 1	PCE 2	PCE 3	PCE 4	PCE 5	PCE 6	PCE 7	PCE 8	PCE 9
Aluminum	N	N	I	N	N	N	N	I	N
Ammonia	N	N	A	N	N	N	N	A	N
Arsenic	N	N	A	N	N	N	N	A	N
Cadmium	N	N	A	N	N	N	N	A	N
Chromium III	N	N	I	N	N	N	N	N	N
Chromium VI	N	N	I	N	N	N	N	N	N
Copper	N	N	A	N	N	N	N	A	N
Dieldrin	N	N	D	N	N	N	N	N	N
Lead	N	N	I	N	N	N	N	A	N
Nickel	N	N	A	N	N	N	N	A	N
Pentachloro-phenol	N	N	D	N	N	N	N	D	N
Selenium	N	N	A	N	N	N	N	A	N
Silver	N	N	A	N	N	N	N	A	N
Zinc	N	N	A	N	N	N	N	A	N
N = No effect: A = adverse effect: I = Insignificant effect: D = Discountable effect									

PCEs 3 (abundant food base) and 8 (water quality) Are likely to be adversely affected by the proposed action. PCE 3 may be affected because many of the proposed standards may have adverse effects on prey species (mostly fish, but also macroinvertebrates) of the bull trout. Any action that may reduce the quantity or quality of food for bull trout would reduce the quality of PCE number 3. PCE 8 deals directly with water quality. Any action that may significantly reduce water quality would reduce the quality of PCE 8.This adverse effect could reduce the ability of the CH to support normal reproduction, growth, and survival of bull trout. Since effects caused by the proposed action are not likely to occur more than once every 3 years, these PCEs are likely to be temporarily affected in an adverse manner at that frequency for the 25-year term of the proposed action.

4.4.2.1 Effects by CHU

See Table 4-1 and Table 4-2 for miles of bull trout CH that are likely to be affected by the proposed action.

Grande Ronde River CHU

The conservation/recovery purpose of the Grande Ronde River CHU is to support the persistence of the bull trout in two core areas: (1) the Grande Ronde River; and (2) the Little Minam River core areas. This CHU encompasses about a 658 miles of bull trout CH.

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 within approximately 102 miles of the Grande Ronde River Core Area, which may experience degradation of PCE number 3 and 8. Of these 102 miles of CH, 64.6 miles are within bull trout spawning and rearing habitat and the remaining 37.5 miles of CH lie within bull trout foraging, migratory, and overwintering habitat (Table 4-1 and Table 4-2).

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 within an approximately a 5 mile reach of bull trout CH supporting spawning and rearing activities in the Little Minam Core Area

Overall, implementation of the proposed action is likely to adversely affect about 107 miles (about 16%) of the total critical habitat in this CHU through a reduction in prey base (both aquatic macroinvertebrates and fish species) PCE 3, and a reduction the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

These effects may occur from temporary increases in toxicity of the water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Hells Canyon Complex CHU

The conservation/recovery purpose of the Hells Canyon Complex CHU is to support the persistence of the bull trout in the Pine, Indian, and Wildhorse Creeks Core Area. There are approximately 234.6 miles of CH within this CHU. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 within an approximately a 52 mile reach (22%) of the total critical habitat in this CHU: a 35.6-mile reach of bull trout CH supporting spawning and rearing activities and a 16.3-mile reach of bull trout CH supporting foraging, migratory, and overwintering habitat. The proposed action is likely to affect these two PCEs through a reduction in prey base (both aquatic macroinvertebrates and fish species) PCE 3, and a reduction the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Hood River CHU

The conservation/recovery purpose of the Hood River CHU is to support the persistence of the bull trout in the Hood River Core Area. There are approximately 79.6 miles of CH within this CHU. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 within an approximately a 13 mile reach (16%) of the total critical habitat in this CHU. The proposed action is likely to affect these two PCEs through a reduction in prey base (both aquatic macroinvertebrates and fish species) PCE 3, and a reduction the CHU's capacity to have sufficient water quality to support normal growth, and survival (PCE 8). Reproduction is likely to remain unaffected by the proposed action as no known spawning habitat will be exposed within this CHU.

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not likely be exposed. Because the majority of CH within the CHU will be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Imnaha River CHU

The conservation/recovery purpose of the Imnaha River CHU is to support the persistence of the bull trout in the Imnaha River Core Area. There are approximately 178 miles of CH within this CHU. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 within an approximately 59 mile reach (33%) of the total critical habitat in this CHU supporting about 47 miles of spawning and rearing and 12 miles of foraging, migratory, and overwintering habitat for the bull trout.

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

John Day River CHU

The conservation/recovery purpose of the John Day River CHU is to support the persistence of the bull trout in three core areas: (1) the Middle Fork John Day River; (2) the North Fork John Day River; and (3) the Upper Mainstem John Day River core areas. Collectively these core areas contain approximately 649 miles of CH.

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in 5 miles of bull trout spawning and rearing habitat, and in 17.4 miles of bull trout foraging, migratory, and overwintering habitat within the Middle Fork John Day River Core Area through a reduction in prey base (both aquatic macroinvertebrates and fish species) PCE 3, and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in 43.2 miles of bull trout spawning and rearing habitat, and in 57.8 miles of bull trout foraging, migratory, and overwintering habitat within the North Fork John Day River Core Area through a reduction in prey base (both aquatic macroinvertebrates and fish species) PCE 3, and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in a 0.5 mile reach of bull trout spawning and rearing habitat and in 33 miles of bull trout foraging, migratory, and overwintering habitat within the Mainstem John Day River Core Area through a reduction in prey base (both aquatic macroinvertebrates and fish species) PCE 3, and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

Overall, implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 160 miles (24%) of the total critical habitat in this CHU including about 48.7 miles of spawning and rearing and 108.2 miles of foraging, migratory, and overwintering habitat for the bull trout.

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Klamath River CHU

The conservation and recovery purpose of the Klamath River CHU is to support the persistence of the bull trout in three core areas: (1) the Sycan River; (2) the Upper Klamath Lake; and (3) the

Upper Sprague River core areas. Collectively these core areas contain approximately 277 miles of CH.

Implementation of the proposed action is likely to adversely affect PCEs in bull trout CH within the Sycan River Core Area because it will cause a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction the CHU's ability to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in 1 mile of bull trout spawning and rearing habitat, and in 4.5 miles of bull trout foraging, migratory, and overwintering habitat within the Upper Klamath Lake Core through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in 5.5 miles of bull trout spawning and rearing habitat and in 8 miles of bull trout foraging, migratory, and overwintering habitat within the Upper Sprague River Core Area through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

Overall, implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 19 miles (24%) of the total critical habitat in this CHU supporting about 6.5 miles of spawning and rearing and 12.5 miles of foraging, migratory, and overwintering habitat for the bull trout through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Lower Deschutes River CHU

The conservation/recovery purpose of the Lower Deschutes River CHU is to support the persistence of the bull trout in the Lower Deschutes River Core Area. There are approximately 287.8 miles of CH within this CHU. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 13 miles (4%) of the total critical habitat in this CHU: 4.4 miles of spawning and rearing, and 8.2 miles of foraging, migratory, and overwintering habitat for the bull trout. The proposed action is likely to affect these two PCEs

through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not likely be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Malheur River CHU

Implementation of the proposed action is not likely to adversely affect PCEs in bull trout CH within this CHU because there is little potential for exposure.

Odell Lake

Implementation of the proposed action is not likely to adversely affect PCEs in bull trout CH within this CHU because there is little potential for exposure.

Powder River CHU

The conservation/recovery purpose of the Powder River CHU is to support the persistence of the bull trout in the Powder River Core Area. There are approximately 184.3 miles of CH within this CHU. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 80 miles (44%) of the total critical habitat in this CHU: 38 miles of spawning and rearing, and 42 miles of foraging, migratory, and overwintering habitat for the bull trout. The proposed action is likely to affect these two PCEs through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not likely be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Umatilla River CHU

The conservation and recovery purpose of the Umatilla River CHU is to support the persistence of the bull trout in the Umatilla River Core Area. There are approximately 132 miles of CH within this CHU. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 37.5 miles (28%) of the total critical habitat in this CHU. All 37.5 miles are

foraging, migratory, and overwintering habitat for the bull trout. The proposed action is likely to affect these two PCEs trout through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal growth, and survival (PCE 8). No effect to the ability of this CHU to support reproduction is anticipated as no known spawning habitat is affected by the proposed action.

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Upper Willamette River CHU

The conservation and recovery purpose of the Upper Willamette River CHU is to support the persistence of the bull trout in the Upper Willamette River Core Area. There are approximately 191 miles of CH within this CHU. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 78 miles (40%) of the total critical habitat in this CHU: 5.1 miles of spawning and rearing habitat and 73 miles of foraging, migratory, and overwintering habitat for bull trout. The proposed action is likely to affect these two PCEs through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

Walla Walla River

The conservation and recovery purpose of the Walla Walla River CHU is to support the persistence of the bull trout in two core areas: (1) the Walla Walla River; and (2) the Touchet River core areas. This CHU encompasses about a 281 miles of bull trout CH.

Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in 5.7 miles of spawning and rearing habitat and 14.8 miles of foraging, migratory, and overwintering habitat for bull trout within the Walla Walla River core area through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the CHU's capacity to have sufficient water quality to support normal reproduction, growth, and survival (PCE 8).

These effects may occur from increased toxicity of water within affected portions of the CHU that could cause conditions that result in a loss of fecundity (lower hatching rates), loss of recruitment (mortality of early life stages), decreased growth (both in length and total body weight) that could result in reduced fitness (lower competitive ability, reduced vigor), and direct mortality to bull trout in any life stage through exposure. While any or all of these may occur within portions of this CHU, the greater portion of the CHU will not be exposed. Because the majority of CH within the CHU will likely be unaffected by the proposed action it is likely that the CHU will continue to function and meet its goals for the conservation of bull trout.

Columbia River CH

The conservation and recovery purpose of the mainstem Columbia River CH is to provide essential bull trout foraging, migratory, and overwintering habitat in 301.2 miles of CH of the Columbia River. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 120 miles (40%) of the total critical habitat in this CHU. The proposed action is likely to affect these two PCEs in this area trout through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the capacity to have sufficient water quality to support normal growth and survival (PCE 8).

While implementation of the proposed action may reduce the quality of this CH it is not likely to preclude the CH from functioning in its intended conservation or recovery goal for bull trout. The majority of the CH will remain unaffected by the proposed action. This is a very large river which supports high flows and large quantities of water. It is likely that adverse effects on PCEs 3 and 8 will be reduced under such conditions because of increased dilution of any toxic discharges and increased passage area that will allow bull trout to avoid the highest concentrations of toxic substances.

Snake River CH

The conservation and recovery purpose of the Snake River CH is to provide essential bull trout foraging, migratory, and overwintering habitat in 107.7 miles of CH on the Snake River. Implementation of the proposed action is likely to adversely affect PCEs 3 and 8 in approximately 65 miles (60%) of the total critical habitat in this CHU. The proposed action is likely to affect these two PCEs in this area trout through a reduction in prey base (both aquatic macroinvertebrates and fish species) (PCE 3), and a reduction in the capacity to have sufficient water quality to support normal growth, and survival (PCE 8).

While implementation of the proposed action may reduce the quality of this CH it is not likely to preclude the CH from functioning in its intended conservation or recovery goal for bull trout. The majority of the CH will remain unaffected by the proposed action. This is a very large river which supports high flows and large quantities of water. It is likely that adverse effects on PCEs 3 and 8 will be reduced under such conditions because of increased dilution of any toxic discharges and increased passage area that will allow bull trout to avoid the highest concentrations of toxic substances.

Summary of Effects of the Proposed Action on Bull Trout Critical Habitat

Exposure to the toxic materials analyzed within this Opinion is likely to adversely affect both PCEs 3 and 8. PCE 3 expresses the need for an abundant food bases needed for bull trout to survive and recover. The body of literature we used in developing our models indicates that both non-metal (ammonia) and many metals (arsenic, cadmium, nickel, selenium, silver, zinc, and to a lesser degree copper) have the potential to adversely affect the food chain by reducing the number or size of forage fish available to bull trout. Additionally, this literature indicates that ammonia, copper, and zinc have high toxicity to macroinvertebrates, which are an important food source for bull trout during early life stages.

PCE 8 expresses the need for bull trout to have water quality conditions that enable normal reproduction, growth and survival. Based on the body of literature we used in developing our models, the presence of ammonia, arsenic, cadmium, copper, lead, nickel, silver and zinc at their proposed criteria is likely to affect water quality, and thereby cause adverse effects to one or more of the three essential needs of bull trout (reproduction, growth and survival) as stated within the PCE.

Nonmetals

Exposure of critical habitat to ammonia at the proposed standard has the potential to reduce the function of PCE number 3 and 8 in approximately 390 miles of streams within the action area. Of these miles 30.7 are spawning and rearing habitat the remaining 359.5 miles are foraging, migratory, and overwintering habitat (these miles include 113 miles within the Columbia River and 7.8 miles within the Snake River foraging, migratory, and overwintering habitats). These areas are a subset of the total miles potentially exposed by the proposed action and are not additional, or separate miles from those discussed below. These effects will occur in portions of 11 CHUs.

Metals

Arsenic, cadmium, copper, nickel, selenium, silver and zinc will all adversely affect PCE number 3 over approximately 820.6 miles of critical habitat in portions of 11 CHUs, and the Columbia and Snake Rivers as described above. Arsenic, cadmium, copper, lead, nickel, selenium, silver and zinc will all have adverse effects (as described above) on PCE number 8 over approximately 820.6 miles of critical habitat in portions of 11 CHUs. These include about 260.8 miles of spawning and rearing habitat the remaining

Summary

In all, 11 of the 13 CHUs in Oregon, and portions of both the Columbia and Snake Rivers have potential for exposure to the criteria analyzed in this Biological Opinion. PCE 3 and PCE 8 will be adversely affected in areas of exposure within these CHUs. Implementation of the proposed action will reduce the abundance of prey species for bull trout, and will reduce the ability of individual CHUs to provide water quality conducive to normal reproduction, growth and survival of bull trout.

The degree of these adverse effects varies tremendously by location across the action area (Table 4-2). Two CHUs (Malheur River and Odell Lake) have little to no potential to be exposed. Two CHUs (Powder River and Upper Willamette River) have 40% or more of their bull trout habitat that may be exposed to contaminants. While the area of exposure may be high in both the Powder River and Upper Willamette River CHUs, it is anticipated that both of these CHUs will continue to function and provide adequate contributions towards both survival and recovery of the species. This is because while the proposed action will adversely affect PCEs 3 and 8, it will have no effect on the remaining 7 PCEs for bull trout and almost 60% of the bull trout habitat within these two CHUs would remain unaffected, allowing for continued recruitment and dispersal of bull trout and functional habitat throughout the CHUs. In addition, it is anticipated that events that would trigger these effects would only occur once in a 3-year period and then for limited duration, and likely not throughout the 40% of the area that could potentially be exposed.

We anticipate that all 11 affected CHUs will continue to function in their intended conservation and recovery roles.

4.4.3 Effects of the Proposed Action on Lost River and Shortnose Suckers

4.4.3.1 Effects of Ammonia on Lost River and Shortnose Suckers

Oregon's acute total ammonia nitrogen criterion varies with pH, and at pH = 8.0 the total ammonia nitrogen criterion is 5.6 mg/L when salmonids are present and 8.4 mg/L as N when salmonids are absent. Oregon's chronic total ammonia nitrogen criterion = 1.7 mg/L as N (at pH = 8.0, 20 °C), salmonids present or absent. The total ammonia nitrogen criteria for the Klamath River Basin are calculated assuming salmonids are present (Kirk et al. 2010, p. 2-9). Table 4-20 below presents the acute total ammonia nitrogen criterion at different pH values and the equivalent un-ionized ammonia concentrations that would occur at 20 °C. At a pH of 8 and at 20 °C, approximately 4.6% of the total ammonia nitrogen is un-ionized ammonia, thus a total ammonia nitrogen concentration of 5.6 mg/L would equal approximately 0.26 mg/L un-ionized ammonia.

Table 4-20. Oregon's acute total ammonia nitrogen criterion at a range of pH values and the associated un-ionized ammonia concentrations at 20 °C.

pH	Acute Ammonia Criterion (mg/L total ammonia nitrogen)	Un-ionized ammonia concentrations at 20 °C (mg/L un-ionized ammonia)
7.5	13	0.19
8.0	5.6	0.26
8.5	2.2	0.30
9.0	0.89	0.30
9.5	0.47	0.32

Exposure Analysis

In the exposure analyses that follow for ammonia and other chemicals, the narratives for the Lost River sucker and shortnose sucker are combined. The reason for this is that throughout most of their range, the species are sympatric, meaning they occur together, and additionally they have similar habitat requirements, and thus both species are likely exposed to the same chemicals. Therefore, to avoid unnecessary duplication, the analyses are combined. However, any species-specific differences in distributions that could affect their exposure are identified.

Lost River suckers and shortnose suckers are exposed to ammonia throughout their range in Oregon from both natural and anthropogenic sources. Natural ammonia sources affecting the suckers are primarily the result of bacterial decomposition of algae, as evidenced that ammonia concentrations in Upper Klamath Lake usually peak following algae die-off events occurring from late July through October (Boyd et al. 2002, Wood et al. 2006, Holiman et al. 2008, Lindenberg and Wood 2009, Jassby and Kann 2010, Kannarr et al. 2010, Sullivan et al. 2011). Chlorophyll-a concentrations, which are a measure of algae biomass, can reach over 1 mg/L in Upper Klamath Lake, which is near the maximum that can occur in natural waters because of

shelf-shading limiting photosynthesis (Jassby and Kann 2010). The high biomass of algae in the lake is due to naturally-high levels of phosphorus in the sediment as well as from input from tributaries and areas surrounding the lake, especially pumping of drained and diked wetlands (Boyd et al. 2002). Nitrogen in Upper Klamath Lake is not limiting primary production in summer because it is fixed from the atmosphere by the dominant blue-green alga (cyanobacterium), *Aphanizomenon flos-aquae* (AFA; Kann 1997; Boyd et al. 2002). Anthropogenic sources of nutrients and ammonia entering sucker habitat could include pumping from diked and drained wetlands, runoff containing soil, animal wastes, or fertilizers, storm water runoff from buildings, streets and roads, and discharges from STPs.

The causal relationships among external nutrient loads, ammonia, and sucker mortality are:

Nutrients → algae blooms → algae bloom crash → decomposing organic matter → increased un-ionized ammonia →sucker mortality.

Because of the seasonally high algae biomass in Upper Klamath Lake it is classified as being hypereutrophic, the highest state of productivity in freshwater. Eutrophication is a problem throughout most of the Oregon portion of the range of the Lost River sucker and shortnose sucker and in order for these waters to meet state water quality standards, the TMDL for Upper Klamath Lake subbasin is targeted at a 40% reduction in external phosphorus loading (Boyd et al. 2002). The TMDL for the Oregon portion of the Klamath Basin downstream of Upper Klamath Lake (i.e., Keno Reservoir) and the Lost River are focused on nitrogen reduction (Kirk et al. 2010). The differences in the limiting nutrient, either phosphorus or nitrogen, in these two areas likely results from nitrogen fixation by AFA in Upper Klamath Lake and less nitrogen fixation occurring in the Keno Reservoir and Lost River.

Upper Klamath Lake In Situ Ammonia Concentrations

Ammonia levels are relatively high in Upper Klamath Lake, with measured total ammonia nitrogen concentrations reaching as high as 7 mg/L (Jassby and Kann 2010). However, on average total ammonia nitrogen concentrations are much less and during the early part of the bloom in June they are especially low, generally less than 100 µg/L. The un-ionized ammonia levels in the lake are seasonally elevated owing to pH values that are often above 9 and at times exceed 10 (Jassby and Kann 2010). However, the high total ammonia nitrogen concentrations and elevated pH are usually separated in time, with the highest pH and lowest total ammonia nitrogen occurring when the algae bloom is actively growing, usually in June (Holiman et al. 2008, Lindenberg et al. 2009). Nevertheless, pH often remains above 9 in the lake during most of the summer, as described below, which results in high concentrations of toxic un-ionized ammonia.

Morace (2007) reviewed water quality data, including pH and ammonia, for Upper Klamath Lake from a 17-year data set (1990-2006) representing >2000 samples. One focus of the analysis was to understand how various water quality parameters, including pH, temperature, dissolved oxygen, and un-ionized ammonia, likely affected the Lost River sucker and shortnose sucker. For the un-ionized ammonia analysis, a value of 530 µg/L was used as a reference value

with which to identify adverse conditions because it was the median LC50 value for Lost River sucker larvae as determined by Saiki et al. (1999)[11].

Some conclusions relevant to un-ionized ammonia and related pH in Upper Klamath Lake noted by Morace (2007) included the following:

- pH values >9.7 occurred at all stations at mean monthly frequencies of: May = <1; June = 20, July = 25; August = 7; September = 9; October = <1
- When pH was high it was often high throughout the lake and when low was more irregularly distributed
- Un-ionized ammonia concentrations >530 µg/L occurred at 3% of the 2020 samples and at least one exceedance occurred at each of the 10 stations
- Un-ionized ammonia concentrations >530 µg/L averaged 6.5% of samples taken from 1993 to 2003; peaked at 14% in 1997, which was the year of a large die-off of adult suckers in Upper Klamath Lake (Perkins et al. 2000a), and were <1% in other years
- One-third of the un-ionized ammonia concentrations >530 µg/L occurred at stations located in bays

Morace (2007, p. 15) also compared Upper Klamath Lake total ammonia nitrogen concentrations to USEPA (1999) acute total ammonia nitrogen criterion guidelines of 2.2 mg/L at pH 8.5 (consistent with criterion in this consultation) when adult salmonids were present and a 30-day chronic criterion of 0.8 mg/L at pH 8.5 when early life-stage salmonids were present. Morace found that less than 1% of the 2020 samples taken between 1990 and 2006 exceeded USEPA acute guideline criterion, but one-half of these were in one site - Shoalwater Bay. Additionally, approximately 10% of the 2020 samples exceeded USEPA (1999) chronic total ammonia nitrogen guideline criterion and one-half of these were in bays such as Coon Point, Shoalwater Bay, and Wocus Bay. An example of how variable ammonia concentrations can be in bays is seen in the 2005 data for Howard Bay (Lindenberg et al. 2009). In 2005, median total ammonia nitrogen concentrations in Howard Bay were relatively low being ~40 µg/L, but the upper 90 and 95 percentile values were ~1.1 mg/L, demonstrating that a few samples showed much higher total ammonia nitrogen concentrations. Another example is the comparison between un-ionized ammonia concentrations at two stations in the lake in 1997. In July 1997, un-ionized ammonia reached 2 mg/L in Shoalwater Bay, but at the mid-lake station at the same time was ~0.5 mg/L (Perkins et al. 2000a).

Because the toxic effects of ammonia are highly dependent on pH, knowing the seasonality of pH in Upper Klamath Lake is important to understanding ammonia toxicity. Morace (2007, p. 24) shows a seasonal plot of pH for Wocus Bay from 1990 to 2006. In nearly every year, pH was at 8 or above from June to October, and was above pH 9 from July to October in about two-thirds of the years, and in some years pH was above 10, indicating that un-ionized ammonia

[11] Note: at 22 °C, a typical summer water temperature in the lake, and a pH of 9 which is typical, approximately 31% of the total ammonia nitrogen is un-ionized ammonia, and consequently a value of 0.5 mg/L un-ionized ammonia is approximately equal to a total ammonia nitrogen concentration of 1.6 mg/L.

concentrations would have frequently represented 30% of the total ammonia nitrogen concentrations at 20 °C.

In 2007 and 2008, which are the most recent years in which there are published data for the lake, un-ionized ammonia concentrations in Upper Klamath Lake reached 284 µg/L and 271 µg/L (total ammonia nitrogen ≈ 1.4-1.6 mg/L, based on measurements of temperature and pH), respectively (Kannarr et al. 2010), and these concentrations were both short-lived and confined to a few stations (Figure 4-3). Based on the patterns seen in the 17-year data set for ammonia, and data for 2007 and 2008, ammonia levels known to be toxic to suckers are most likely confined to a few areas of the lake.

The seasonality of ammonia concentrations in Upper Klamath Lake is well known, at least during the May – October period when water quality has been extensively monitored. During the period 2002-2006, total ammonia nitrogen peaks above 0.3 mg/L occurred primarily in July, but have also extended into August in some years (Lindenberg et al. 2009), and peaked above 0.8 mg/L near the end of July in 2006 and was above 0.2 mg/L at three sites from approximately mid-July to mid-August that year. In 2007, peaks of ~1.4-1.6 mg/L total ammonia nitrogen occurred in September, October and November (Kannarr et al. 2010). In 2008, total ammonia nitrogen peaks above 0.5 mg/L occurred from late July through October (Kannarr et al. 2010). In both 2007 and 2008, the pH was ~8 in early July, increased to >9.5 in early July, and remained >9.5 through mid-August, and declined slightly to 9-9.5 into October (Kannarr et al. 2010). Based on these data, total ammonia nitrogen peaks of 0.2 mg/L and higher can occur from July through October in Upper Klamath Lake and because pH values are often between 9 and 9.5, un-ionized ammonia can represent 30% to 50% of the total ammonia nitrogen concentrations.

In regards to the seasonal relationship between ammonia levels and the presence of Lost River sucker and shortnose sucker early life stages in Upper Klamath Lake, most larvae transform into juveniles in July, so juveniles rather than larvae are most likely to experience high un-ionized ammonia concentrations because they are present from late July through October. Adult suckers would likely be exposed to high un-ionized ammonia concentrations at any time of year because they are present all year.

The above mentioned studies, including Wood et al. (2006) and Perkins et al. (2000), all suggest that in general, stressful to lethal water quality conditions for suckers and other fish in Upper Klamath Lake are likely the result of a combination of water quality factors related to the mid-summer crash of the algae bloom. At that time, water temperatures are usually at their highest (often being >20 °C), causing elevated respiratory oxygen demands, and the death of the algae reduces concentrations of dissolved oxygen because photosynthesis is halted, dissolved oxygen is consumed during decomposition and ammonia is produced as a byproduct of decomposition. Additionally, if wind speeds are low at the same time, conditions can be made worse because of a lack of wind-driven reaeration. Bays and shallow areas may experience wider variations in water quality because wind and currents might concentrate algae in those locations and bloom dynamics in bays can be independent of those occurring in open water of the lake (Wood et al. 2006, p. 11).

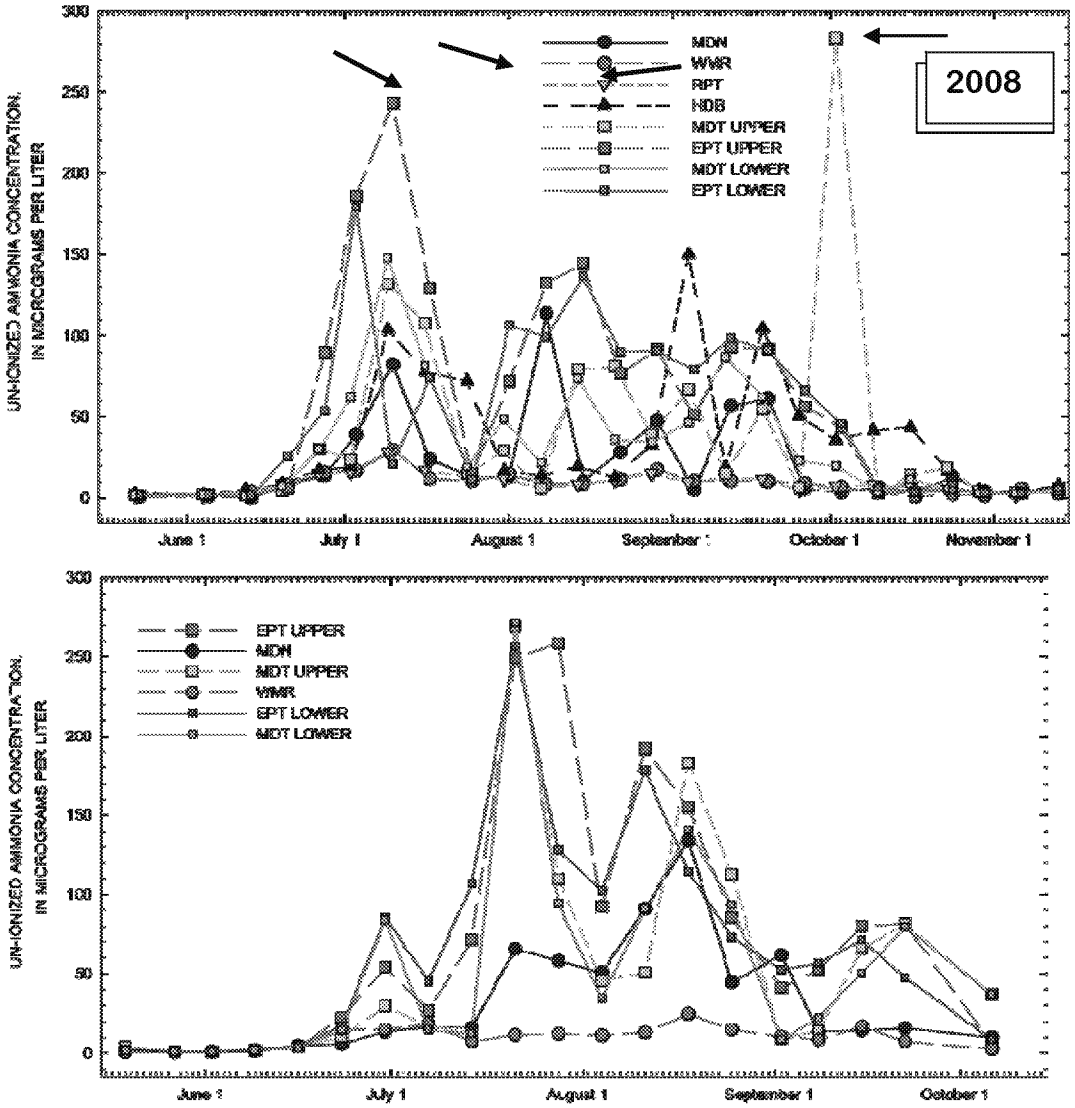


Figure 4-3. Un-ionized ammonia concentrations (µg/L) in Upper Klamath Lake, 2007 (upper panel) and 2008 (lower panel). Source: Kannarr (2010, p.21). Un-ionized ammonia concentrations reached 250 µg/L at EPT (Eagle Point site) upper in July 2007 and MDT upper (middle trench site) in October 2007 (upper panel), and EPT upper and lower and MDT upper and lower in July 2008 (lower panel).

Keno Reservoir In Situ Ammonia Concentrations

Ammonia concentrations are also high in the Keno Reservoir (Kirk et al. 2010; Sullivan et al. 2011), as described below, where there is both a large input of algae from Upper Klamath Lake and anthropogenic ammonia sources, including NPDES permitted discharges. Point-source discharges permitted under the NPDES program of the Clean Water Act are most likely to affect Lost River sucker and shortnose sucker in the Keno Reservoir which receives effluent from two waste-water treatment facilities serving Klamath Falls (i.e., the City of Klamath Falls and South Suburban Sanitary District together have a daily treated discharge of about 5 million gallons

(Kirk et al. 2010), and where there is also a high degree of organic loading coming from algal production upstream in Upper Klamath Lake (Kirk et al. 2010). Additionally there are 34 permitted stormwater discharges into Keno Reservoir; however, these have not been identified as significant contributors to pollution loading (Kirk et al. 2010).

The Keno Reservoir (Klamath River RM 231.5 to 253, including Lake Ewauna) is currently listed as impaired year-round for ammonia toxicity under section 303(d) of the Clean Water Act during all seasons (Kirk et al. 2010). In the 2010 TMDL for the Oregon portion of the Klamath River that includes the Keno Reservoir, Kirk et al. (2010) described ammonia concentrations, which peak at Miller Island (RM 245) in July and August (Figure 4-4 and Figure 4-5). Total ammonia nitrogen concentrations in the Keno Reservoir frequently exceed Oregon's chronic criteria from June–September and can exceed the acute criteria in both June and July (Kirk et al. 2010).

Sullivan et al. (2011) analyzed water quality at Lake Ewauna, which forms the upper reach of the Keno Reservoir, including ammonia concentrations from 2006-2009. Model fluxes suggested that the source of most ammonia was primarily from the decay of organic matter in the sediments and water column, with algal decomposition processes as an additional source.

Sullivan et al. (2011) found that pH at Miller Island was typically 8.5 in the spring, increasing to values at or near 9.5 in the summer, and decreasing to near 7.5 in the fall. Keno Reservoir is also listed as impaired under section 303(d) of the Clean Water Act for pH during the summer months. Temperatures at Miller Island have a 7-day maximum of about 25 °C in mid-July, the hottest month (Kirk et al. 2010).

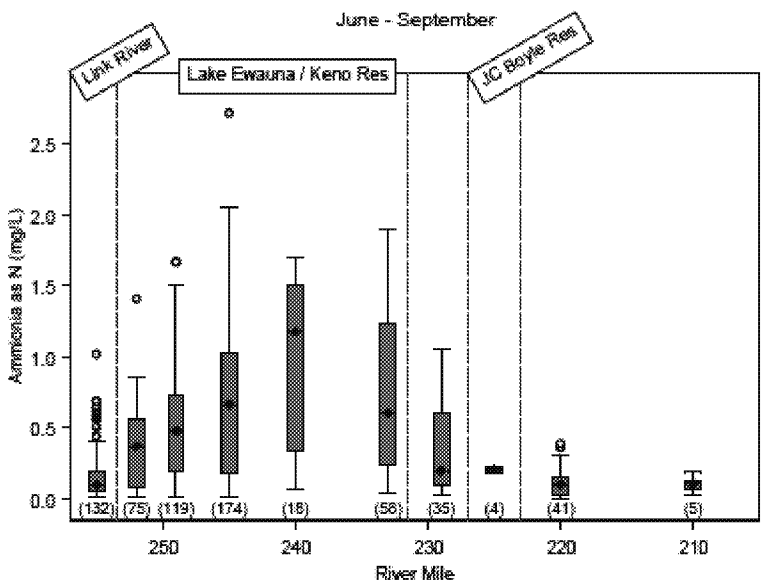


Figure 4-4. Box and whisker plot of total ammonia nitrogen (mg/L) longitudinal profile for the upper Klamath River, June – September, from Upper Klamath Lake to the Oregon/California Border. Source: Kirk et al. (2010). The box consists of 25% and 75% quartiles and the mean, and the whiskers represent extreme values.

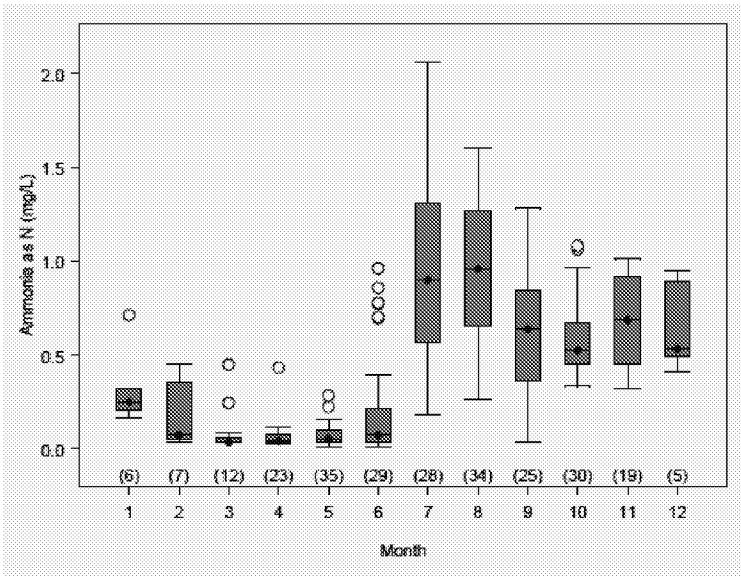


Figure 4-5. Box and whisker plots of seasonal total ammonia nitrogen (mg/L) concentrations at Miller Island (RM mile 250). The box consists of 25% and 75% quartiles and the mean, and the whiskers represent extreme values. Source: Kirk et al. (2010).

Because of the combination of high total ammonia nitrogen concentrations, a high pH, and high temperatures, the Keno Reservoir has high levels of un-ionized ammonia in summer, especially at Miller Island where they reach levels above 0.3 mg/L (Figure 4-6).

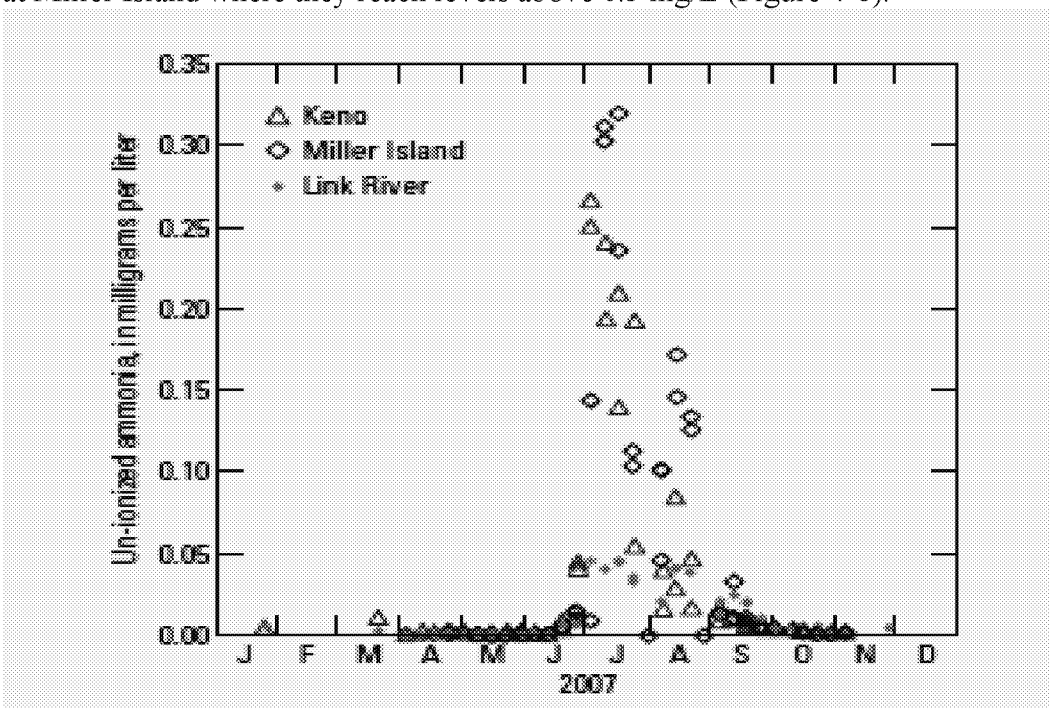


Figure 4-6. Un-ionized ammonia concentrations measured in 2007 at the Link River, Miller Island, and Keno. Link River is the most upstream site and Keno the most downstream site; Miller Island is located at RM 250. Source: Sullivan et al. (2011).

Lost River In Situ Ammonia Concentrations

In addition to the Keno Reservoir, the nearby Lost River, which is also in the Upper Klamath River Basin in Oregon (RM 4.8 to 65) is also 303(d)-listed for ammonia toxicity, as is the Straits Drain which discharges irrigation drain-water from the Lost River system into the lower end of the Keno Reservoir (Kirk et al. 2010). Total ammonia nitrogen concentrations are highest at the lower end of the Lost River near Anderson Rose Dam and in the Straits Drain (Kirk et al. 2010; Figure 4-7). Ammonia toxicity is likely to be highest in discharges from the Straits Drain because pH values are often between 8 and 9, or higher (Kirk et al. 2010). The Lost River has 19 permitted confined animal feeding operations (Kirk et al. 2010). Waste from these feeding operations that might contain ammonia include but are not limited to manure, silage pit drainage, wash-down waters, contaminated runoff, and bulk tank wastewater, and are operated under a general permit from the Oregon Department of Agriculture (Kirk et al. 2010).

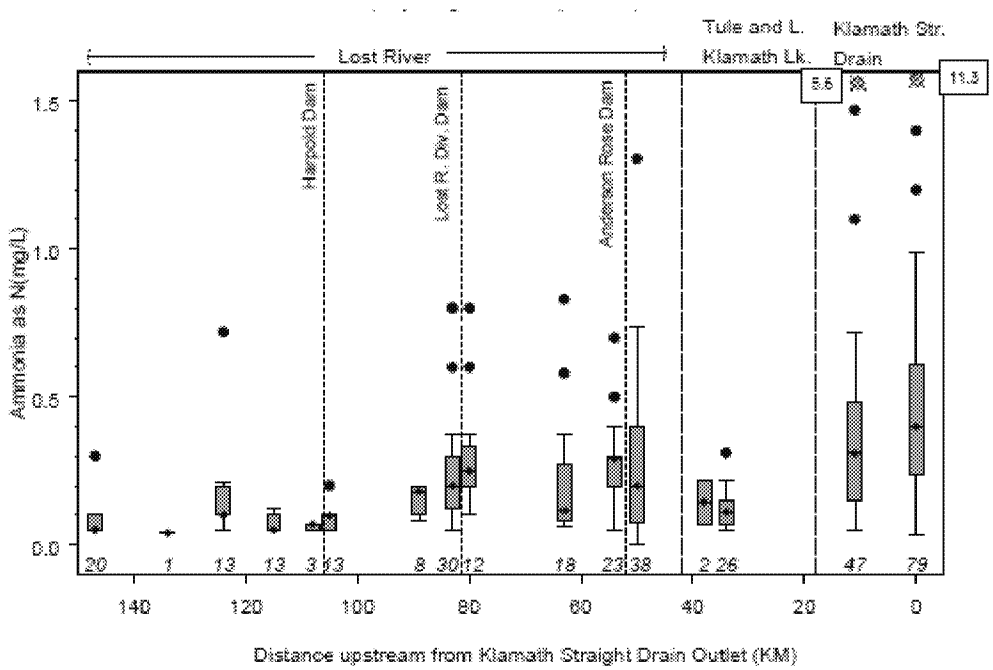


Figure 4-7. Total ammonia nitrogen concentrations at various sites along the Lost River in Oregon as well as in Tule Lake and Lower Klamath Lake, and the Straits Drain that empties into the Keno Reservoir. Source: Kirk et al. (2010).

Results of Laboratory Ammonia Toxicity Studies

Because of relatively high ammonia concentrations in Upper Klamath Lake and concerns for its effects on the endangered Lost River sucker and shortnose sucker, several laboratory studies were done to measure the sensitivity of suckers to ammonia (Monda and Saiki 1993, 1994; Bellerude and Saiki 1995; Meyer and Hansen 2002). Saiki et al. (1999) reviewed the results of a variety of tests using ammonia alone and in conjunction with pH, dissolved oxygen, and temperature to assess how ammonia affected survival of larval and juvenile suckers, and found that median LC50 values for un-ionized ammonia varied from 0.48-1.29 mg/L for 96-hour exposures for larval and juvenile suckers. In general, larvae were found to be more sensitive to

un-ionized ammonia than juveniles; however, due to the high level of variance among tests, the sensitivity to ammonia was not significantly different among life stages or species. At higher un-ionized ammonia concentrations, most of the mortality was in the first day or two, so the results for 24-hour and 96-hour LC50s did not vary substantially.

Table 4-21. Median lethal un-ionized ammonia concentrations (LC50) for Lost River sucker and shortnose sucker larvae and juveniles measured in the laboratory at 24, 48, or 96 hours. All tests were done at approximately 20 °C except those by Meyer and Hansen (2002) were done at 22 °C.

Species and Life Stage	Median LC50 NH_3 (mg/L)	95% Confidence Limits (mg/L)	pH	Hours	Reference
Lost River Sucker Larvae	0.54	0.43-0.85	7.7-8.0	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.48	0.31-0.58	9.5-9.6	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.51	0.35-0.63	9.3-9.6	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.46	0.21-0.66	9.1-9.5	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.50	0.42-0.67	9.2-9.6	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.89	0.78-1.01	9.8-9.9	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.71	0.62-0.81	9.8-9.9	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.67	0.41-0.90	9.7-9.9	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.70	0.46-1.08	9.7-9.9	96	Monda and Saiki 1994
Lost River Sucker Larvae	0.56	0.52-0.61	9.5	24	Meyer and Hansen 2002
Lost River Sucker Larvae	0.51	0.47-0.55	9.5	48	Meyer and Hansen 2002
Lost River Sucker Larvae[1]	0.79	0.78-0.70	9.5	48	Meyer and Hansen 2002
Lost River Sucker Juveniles	0.75	0.60-0.94	7.7-7.9	96	Bellerude and Saiki 1995
Lost River Sucker Juveniles	0.70	0.34-0.82	7.8-8.0	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.71	0.31-0.76	8.8-8.9	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.74	0.26-0.98	8.6-8.8	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.71	0.49-1.01	9.6-9.7	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.56	0.45-0.83	9.5-9.8	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.54	0.46-0.63	9.1-9.2	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.45	0.36-0.56	9.0-9.1	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.88	0.63-1.36	9.5-9.7	96	Monda and Saiki 1993
Lost River Sucker Juveniles	0.68	0.56-0.83	9.4-9.5	96	Monda and Saiki 1993
Lost River Sucker Juveniles	1.02	1.01-1.04	9.5	24	Meyer and Hansen 2002
Lost River Sucker Juveniles	0.92	0.82-1.04	9.5	48	Meyer and Hansen 2002
Shortnose Sucker Larvae	0.75	0.73-0.77	7.7-8.2	96	Bellerude and Saiki 1995
Shortnose Sucker Larvae	1.40	1.24-1.68	8.0-8.3	96	Bellerude and Saiki 1995
Shortnose Sucker Juveniles	0.96	0.32-2.46	7.5-8.2	96	Bellerude and Saiki 1995
Shortnose Sucker Juveniles	0.34	0.14-0.73	7.5-8.0	96	Monda and Saiki 1993

1. Larvae were identified as being "late larvae," suggesting they were probably metalarvae which is the stage just prior to metamorphosis.

Note that laboratory studies on the toxic effects of ammonia, like those mentioned here, that have been conducted at pH > 7.5 generally only state concentrations as un-ionized ammonia because toxicity is relatively constant as expressed as un-ionized ammonia (USEPA 1999).

Meyer and Hansen (2002) conducted both short-term (24-48 hours) and long-term (30 days) ammonia toxicity studies using Lost River sucker larvae and juveniles. They found that mortality thresholds for un-ionized ammonia was between 0.37 mg/L (Note: this equals the no observed effects concentration or NOEC) and 0.69 mg/L (Note: this equals the lowest observed effects concentration or LOEC). Calculated median LC50 values for Lost River suckers exposed to un-ionized ammonia for 24 to 48 hours ranged from 0.51 mg/L for larvae to >1.0 mg/L for late-stage larvae and juveniles (Table 4-21). Most deaths at 1.16 mg/L un-ionized ammonia occurred in the first 3 days of the test whereas deaths at 0.69 mg/L occurred gradually from days 3 to 24 (Figure 4-8). Sub-lethal effects (i.e., decreases in weight, length, whole-body ion content, and swimming performance) over 30 days were not obvious; however, there was evidence of increased gill thickness, as described below. Based on both the short-term and long-term un-ionized ammonia exposure tests, as well as the tests reported by Saiki et al. (1999), that showed an asymptote in mortality by 96 hours, Meyer and Hansen (2002) concluded that the LC50 for indefinite exposure of Lost River sucker early life stages to un-ionized ammonia is ~0.4-0.5 mg/L (Note: at pH = 9.5 and 22 °C, the ammonia criterion = 0.34 mg/L when expressed as un-ionized ammonia).

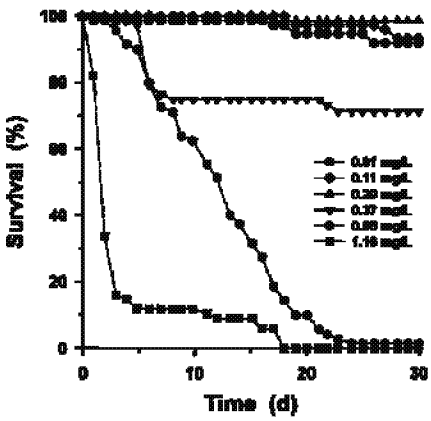


Figure 4-8. Mean survival of late-stage Lost River sucker larvae exposed to elevated ammonia. Mean un-ionized ammonia concentrations are shown in the legend. Source: Meyer and Hansen (2002).

There was one attempt to assess ammonia toxicity in situ. Martin and Saiki (1999) placed juvenile suckers in cages positioned in Upper Klamath Lake and monitored survival and water quality parameters. No mortality was observed even though un-ionized ammonia reached 0.61 mg/L and this concentration was shown to be lethal in the lab; however, because mean un-ionized ammonia concentrations never exceeded 0.21 mg/L, which is below the lethal level, Martin and Saiki (1999) suggested the lack of mortality was likely because toxic ammonia concentrations were not maintained sufficiently long.

In summary, the Lost River sucker and shortnose sucker in Oregon are likely to be exposed to seasonally high concentrations of ammonia in Upper Klamath Lake, the Keno Reservoir, and the Lost River, for at least a few days each year. Exposure to ammonia toxicity is primarily confined to juveniles and older suckers because ammonia concentrations, and high pH that

increase ammonia toxicity, mostly occur in late July through October after the larvae have transformed into juveniles.

Acute Effects Analysis

To determine how Lost River sucker and shortnose sucker populations could be affected by the implementation of the ammonia criteria, we assumed that if in situ ammonia concentrations were equal to the acute criterion it would lead to an equivalent mortality rate to that which was observed in the laboratory at those same concentrations. For example, if the total ammonia nitrogen criterion concentration is equal to an LC12 for a specific life stage, then we assumed that 12% of the sucker population represented by that life stage in the affected area would die as a result of ammonia toxicity.

Within the papers that reported the toxic effects of ammonia on Lost River sucker and shortnose sucker in the laboratory, we found several data sets amenable to running a "probit" analysis which could be used to produce a more refined estimate of the effect level for mortality associated with the acute criterion. A probit analysis is a specialized statistical regression analysis technique used to model binary responses, e.g., test organisms that either lived or died during the experiment. We found applicable data in single parameter tests for un-ionized ammonia as well as in multi-parameter tests with un-ionized ammonia, pH, temperature, and dissolved oxygen. Using those data, we used the linear regression equation from each test to obtain a percent effect value equivalent to the acute total ammonia nitrogen criterion of 5.6 mg/L at pH 8 and 20 °C. Results of the regression analysis are presented in Table 4-22.

Table 4-22. Estimated percent effect levels acute level of effect for suckers based on probit analysis.

Sucker Species and Lifestage	Test pH/ Temperature	Total Ammonia Criterion mg/L as N	Percent Level of Effect	Citation
Shortnose juvenile	8.0/20 °C	5.6	19%	Monda and Saiki 1993
Shortnose larvae	8.0/20 °C	5.6	>1%	Bellerud and Saiki 1995
Lost River juvenile	8.0/20 °C	5.6	>1%	Bellerud and Saiki 1995
Lost River juvenile	9.1/19 °C	0.76	7.1%	Monda and Saiki 1993
Lost River larvae	9.2/25 °C	0.66	22%	Monda and Saiki 1994
Lost River larvae	9.5/24.5 °C	0.46	9.3%	Monda and Saiki 1994
Lost River larvae	9.8/25 °C	0.37	3.5%	Monda and Saiki 1994

[1]*Tests were conducted with single variables or multivariables (ammonia, temperature, pH, and dissolved oxygen). Single variable tests were conducted at pH 8 and multivariable tests at pH 9 and above.*

We used the midrange of the values shown above in Table 4-22, as explained above, which for Lost River sucker was determined to be 13% for larvae and 4% for juveniles. Since there is only one reported value for shortnose sucker juvenile and larval stages, the percent level of effect are assumed to be 19% and 1%, respectively. We assumed, as described above, that this percentage of affected sucker populations would die from ammonia exposure at acute ammonia criterion concentrations.

Chronic Effects Analysis

In addition to the direct mortality that is likely to occur to suckers as a result of exposure to acutely-lethal concentrations of ammonia, suckers exposed to sub-lethal ammonia concentrations would likely experience physiological stress as a result of increased mucus production by the gills, gill hemorrhaging, increased gill lamellae thickness, and other adverse effects (Saiki et al. 1999, p 43; Foott et al. 2000, p. 3; Lease et al. 2003, p. 469). At an un-ionized ammonia concentration of 0.37 mg/L (at pH = 9.5 and 22 °C, the ammonia criterion = 0.34 mg/L when expressed as un-ionized ammonia), Lease et al. (2003, p. 469) found a 32% increase in mean gill lamellar thickness in comparison to controls in juvenile Lost River sucker. These changes have the potential to affect normal gill functioning and could reduce the rate that dissolved oxygen diffuses through the gills, and it could also adversely affect osmoregulation, which is another important gill function (Lease et al. 2003, p. 468). Additionally, suckers with damaged gills could be more vulnerable to bacterial pathogens, such as Columnaris (*Flavobacterium columnare*), *Aeromonas* species, and *Pseudomonas* species, which could use damaged gills as an entry route into the body, causing septicemia and death (Perkins et al. 2000a). These effects could occur to all sucker life stages, not just young suckers.

Although the sublethal effects of ammonia described above could adversely affect respiration and ion balance, or make suckers more vulnerable to pathogens, Meyer and Hansen (2002) found that a 14 day exposure to 0.3 mg/L un-ionized ammonia at pH 9.5 and 6 mg/L dissolved oxygen, followed by a 14 day exposure to 2 mg/L dissolved oxygen, pH 8, and no ammonia, showed no increase in mortality and there was no observable decrease in swimming performance or growth, thus suggesting that the gill thickening described by Lease et al. (2003) may not affect normal gill function substantially, at least in the laboratory at a concentration of 0.3 mg/L un-ionized ammonia.

In Table 4-23 below, there is an estimated 15% to 27% reduction in proportion of catostomid eggs hatching from 60 days to spawning, based on fathead minnows exposed to total ammonia. Chronic exposure to ammonia is unlikely to affect hatching of suckers because during the April – May period when the sucker embryos are developing, because ammonia levels and pH are generally quite low (see Figure 4-3 and Figure 4-6 above).

Table 4-23. Maximum likelihood estimate for percent level of effect on reproduction (reduction in egg hatchability, number of eggs per spawn, or number of eggs per female compared to control) and confidence limits based on early life stage fish exposure to total ammonia.

Listed species or family representative	Fathead minnow proportion eggs hatching from 60 days to Spawning (Thurston et al. 1986)	Fathead minnow proportion eggs hatching from 60 days to Spawning (Thurston et al. 1986)
	P = 0.009; F = 36.5; MSE = 0.05 Adj.R^2 = 0.90 n = 5 Input: 0.68-4.85 Test: 0.02 – 7.26	P = 0.003; F = 82.6 MSE = 0.02 Adj.R^2 = 0.95 n = 5 Input: 0.68-4.85 Test: 0.00 – 7.24
Catostomidae	27.0 11.6 – 41.2	15.6 4.62 – 26.0

The relevant question regarding chronic effects of ammonia to Lost River and shortnose suckers is at what point does chronic exposure actually cause adverse effects? The Act defines harm as an action that "…results in death or injury to listed species by significantly impairing behavioral patterns such as breeding, feeding, or sheltering." The sub-lethal, chronic effects of ammonia on gills of suckers described above do not appear to meet the Act's definition of harm until mortality actually results. Meyer and Hansen (2002) suggest that Lost River suckers have evolved physiological mechanisms enabling them to tolerate the cyclic adverse water quality conditions in Upper Klamath Lake, and that apparently results in a lack of serious physiological adverse effects until a lethal threshold is reached. Based on this, it appears that acute and chronic effects of ammonia that likely harm Lost River suckers and shortnose suckers occur over an overlapping range of concentrations making it impossible to separate them using existing data. Therefore at this time, we are unable to identify adverse chronic effects that meet the Act's definition of harm.

Effects Summary

In order to quantify the extent of adverse effects that could occur as a result of the proposed action, it is necessary to estimate what percentage of the lake would be affected by concentrations of ammonia sufficiently high to cause effects. In Upper Klamath Lake, ammonia toxicity is most likely to be a problem in bays where winds and currents concentrate *A. flos-aquae* algae and when the algae died it would result in ammonia formation. The major Upper Klamath Lake bays are Ball, Howard, and Shoalwater. These three bays have a combined surface area of approximately 15% of the lake.

We cannot exactly determine what the population level effect of the proposed action would be on suckers because of uncertainty in predicting future conditions; however, in what is likely to be the worst case that is most reasonably foreseeable, ammonia concentrations could reach criterion levels in the three principle bays in Upper Klamath Lake and as a result up to 19% of the young

suckers in these three bays could die from ammonia toxicity. If this occurred, it would represent up to a 3% (100 x 0.15 x 0.19 = 2.8) loss of the total lake-wide sucker population for these life stages. This percentage assumes an equal abundance of young sucker throughout the lake; however, in some years Ball Bay and Shoalwater Bay have greater abundances of young suckers than other areas of the lake (Simon et al. 2011), so based on this, it might be more accurate to assume a slightly higher mortality rate, such as 5%.

The periodicity of future toxic ammonia events in Upper Klamath Lake is unknown, but based on the data reviewed above, elevated un-ionized ammonia events are seasonal and will only be reached infrequently (here we assume their occurrence would be once every 3 years, but this is likely an over estimate based on the historical data described above). We would also expect some mortality of adult suckers to occur in these events, but fewer adults are likely to be killed because in general adults are considered less sensitive to toxicants than younger life stages (Sprague 1985), and because adult suckers have higher mobility and perhaps some ability to learn what areas of the lake have higher water quality, some adults would probably avoid such events, as has been documented in past die-off events where adult suckers have moved into Pelican Bay where water quality is better because of groundwater inputs (Perkins et al. 2000a; Banish et al. 2009). Thus, under this scenario, we assume <1% of the adults suckers would be killed every 3 years.

Above we determined that it is reasonably foreseeable that the proposed action would result in up to a 5% loss of young suckers and <1 % loss of adult suckers every 3 years in Upper Klamath Lake as a result of ammonia toxicity. In Keno Reservoir and the Lost River up to 19% of the young suckers are likely to be adversely affected every 3 years over the 25-year life of the proposed action.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect proposed critical habitat Unit 1 (Upper Klamath Lake, including Agency Lake and the Keno Reservoir) (USFWS 2011d). Effects of the proposed action are likely to degrade the quality of habitat through effects to PCE 1 (water). Our analysis determined that the most extreme effect of the proposed action that is reasonably foreseeable will be to allow concentrations of ammonia in the three major Upper Klamath Lake bays to reach levels sufficient to kill suckers once every 3 years. The area of these bays is equal to approximately 15% of the total area of Upper Klamath Lake habitat, or about 10% of proposed critical habitat for each species. The events creating toxic ammonia concentrations would likely be temporary, perhaps lasting several weeks or less, based on the historic ammonia data discussed above. Some adverse effects are also anticipated for PCE 3 (food) due to acute toxicity of ammonia on invertebrates, such as *Daphnia* and chironomid larvae, on which sucker feed. However, these water bodies are highly productive (Sullivan et al. 2011; NRC 2004), which reduces the likelihood of a food shortage. The anticipated adverse effects would be limited in area relative to the total area of proposed critical habitat designation, and thus the effect is unlikely to impair the ability of proposed critical habitat to support the conservation needs of the species. Furthermore, ammonia toxicity will be temporary, lasting for only a few days or weeks, and we assume would occur once every 3 years, and thus over most of that time

would not adversely affect the ability of proposed critical habitat to satisfy the essential requirements of the Lost River sucker and shortnose sucker.

4.4.3.2 Effects of Arsenic on Lost River and Shortnose Suckers

Exposure Analysis

Mining, smelting, manufacturing, electric power plants, pesticides, agricultural defoliants, and battery manufacturing and reclamation plants are all significant anthropogenic sources of arsenic (USEPA 2008). Of these, mining, specifically where arsenic is used to extract gold, may be the most important source affecting listed freshwater species in Oregon.

Presently, about 90% of all arsenic produced is used as a preservative for wood to make it resistant to rotting and decay. The preservative is copper chromated arsenate (CCA) and the treated wood is referred to as "pressure-treated." In 2003, U.S. manufacturers of wood preservatives containing arsenic began a voluntary transition from CCA to other wood preservatives that do not contain arsenic in wood products for certain residential uses, such as play structures, picnic tables, decks, fencing, and boardwalks. This phase out was completed in 2003; however, wood treated prior to this date could still be used and existing structures made with CCA-treated wood would not be affected. CCA-treated wood products continue to be used in industrial applications. Arsenic is also present in highway runoff and has been reported at levels up to 2.6 µg/L in Washington State (WADT 2007).

Within the range of the Lost River sucker and shortnose sucker, the presence of elevated arsenic levels in surface water is likely the result of groundwater moving through rock containing arsenic (Sturdevant 2011). In a well-water assessment in the Klamath Falls area by ODEQ, arsenic levels up to 15 µg/L were found in four well (Cole 2006). Two wells in Klamath Falls have arsenic concentrations of 29-35 µg/L (Chen et al. 2011). We believe that it is likely that a small volume of arsenic contaminated water from geothermal wells in Klamath Falls is discharged into the City of Klamath Falls wastewater treatment system. Arsenic was found in surface waters at Lower Klamath National Wildlife Refuge at a concentration of up to 62µg/L (Sorenson and Schwartzbach 1991). Although low levels of arsenic is present in surface water in the upper Klamath Basin there currently are no areas that are 303(d) listed for arsenic toxicity.

Because arsenic could enter the Klamath Falls wastewater system and additional arsenic could come from treated wood runoff from urban areas of Klamath Falls and from stormwater runoff from streets and highways and be discharged into the Keno Reservoir, we assume Lost River sucker and shortnose sucker are likely to be exposed to arsenic by the proposed action in Keno Reservoir because it receives discharges from two wastewater treatment facilities as well as stormwater runoff that could be contaminated with arsenic.

Acute Effects Analysis

The acute arsenic criterion under review is 340 µg/L. Table 4-24 shows the estimated population level of effect based on the ICE model. Because the acute criterion is approximately 60% of the LC10 we assume the population level of effect will up to 10% of the suckers in Keno Reservoir.

Table 4-24. Estimated level of effect to Lost River and shortnose suckers as a result of exposure to arsenic at the acute criterion of 340 µg/L.

Species	Data Source	LC50 (µg/L)	LC10 (µg/L)	Acute Criterion as Percent of LC10	Population Effect
Lost River sucker	ICE	1285	566	60	10
Shortnose sucker	ICE	1285	566	60	10

Chronic Effects Analysis

The chronic arsenic criterion under review is 150 µg/L. Table 4-25 provides data on the level of effect for the fathead minnow, flagfish, and bluegill exposed to arsenic. Because minnows and suckers belong to the same order of bony fishes, the Cypriniformes, the fathead minnow data are likely to be the best surrogate for suckers. The fathead minnow data presented in Table 4-25 suggests that there would likely be a reduction in embryo growth of up to 13% and a reduction of larval growth of up to 4% as a result of exposure to arsenic at the chronic criterion concentration. The broad confidence limits on most of the estimates in Table 4-25 indicate that the actual effects to Lost River sucker and shortnose sucker could be higher or lower.

Effects of chronic arsenic exposure to invertebrate prey were also analyzed. Based on studies by Lima et al. (1984), a reduction in survival of *Daphnia* spp. of up to 27% could occur (Table 4-26). *Daphnia* are a major food species of shortnose sucker suckers (USFWS 2008g).

Table 4-25. Maximum likelihood estimate for percent level of effect (reduction in survival, reproduction, or growth compared to control) and confidence limits based on early life stage fish exposure to arsenic.

Listed species or family representative	Fathead minnow embryos (<24 h old) reduction in growth (mean wet mass g) after 29-day exposure (Lima et al. 1984).	Fathead minnow embryos (<24 h old) reduction in growth (mean length mm) after 29-day exposure (Lima et al. 1984).	Fathead minnow reduction in growth (mean mass mg) after 32-day exposure (Spehar and Fiandt 1985).	Flagfish embryos (<24 h old) reduction in growth (mean wet mass g) after 31-day exposure (Lima et al. 1984).
	P = 0.02; F = 25.4 MSE = 0.01 Adj.R^2 = 0.86 n = 5 Input: 184 – 1,865 Test: 1,060 – 16,500	P = 0.01; F = 32.8 MSE = 0.0006 Adj.R^2 = 0.89 n = 5 Input: 184 – 1,865 Test: 1,060 – 16,500	P = 0.03; F = 35.1 MSE = 0.0007 Adj.R^2 = 0.92 n = 4 Input: 184 – 1,865 Test: 1,340 – 8,340	P = 0.04; F = 11.2 MSE = 0.02 Adj.R^2 = 0.72 n = 5 Input: 1,240 – 16,300 Test: 232 – 2349
Catostomidae	13.3 0.00 – 44.0	7.40 0.00 – 22.1	4.28 0.00 – 19.6	17.5 0.00 – 52.5

Listed species or family representative	Flagfish embryos (<24 h old) reduction in growth (mean length mm) after 31-day exposure (Lima et al. 1984).	Survival of immature (mean mass = 0.5 g) bluegill exposed to sodium arsenite pesticide introduced into ponds for 16 weeks (Lima et al. 1984).	Growth (g mass) of immature (mean mass = 0.5 g) bluegill exposed to sodium arsenite pesticide introduced into ponds for 16 weeks (Lima et al. 1984).	Growth (g mass) of adult (mean mass = 15.8 g) bluegill exposed to sodium arsenite pesticide introduced into ponds for 16 weeks (Lima et al. 1984).
	P = 0.04; F = 11.4 MSE = 0.002 Adj.R^2 = 0.72 n = 5 Input: 232 – 2,349 Test: 1,240 – 16,300	P = 0.002; F = 35.7 MSE = 0.04 Adj.R^2 = 0.85 n = 7 Input: 225 – 2,275 Test: 400 – 6,400	P = 0.004; F = 26.3 MSE = 0.003 Adj.R^2 = 0.81 n = 7 Input: 225 – 2,275 Test: 400 – 6,400	P = 0.004; F = 24.2 MSE = 0.0006 Adj.R^2 = 0.80 n = 7 Input: 225 – 2,275 Test: 400 – 6,400
Catostomidae	7.40 0.00 – 22.1	29.0 23.7 – 34.6	20.0 8.50 – 30.0	3.81 0.00 – 9.23

Table 4-26. Maximum likelihood estimate for percent level of effect (reduction in survival compared to control) and 95% confidence limits based on early life stage invertebrate exposure to arsenic.

Listed species or family representative	*Daphnia magna* reduction in survival, static renewal test (Lima et al. 1984).
	P = 0.008;F = 24.3; MSE = 0.22; Adj.R^2 = 0.82; n = 6 Log input: 2.943; Log test: 1.862 – 3.427
Daphnidae	26.7 (12.3 – 44.7)

Effects Summary

Based on the above-mentioned toxicological studies and modeling, Lost River sucker and shortnose sucker, if exposed to dissolved arsenic at the acute criterion would be expected to show up to a 10% reduction in survival. At chronic criteria concentrations there would likely be lower reproductive success in the form of reduced embryo and larval growth, thus the effect would be adverse; however, because there is no known reproduction in the Keno Reservoir, there would be no population effect. Effects to adults are unknown, but are likely less than for the more-sensitive early life stages (Sprague 1985). A reduction in invertebrate food supply is also possible but because Keno Reservoir is highly productive, a substantial reduction in food supply is unlikely to occur.

Analysis of Effects to Proposed Critical Habitat for Lost River and Shortnose Suckers

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality in the Keno Reservoir, and thus adversely affect the ability of PCE 1 (water quality) to support the species. These adverse effects to proposed critical habitat would be limited in extent because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), and thus the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker species. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

There could also be adverse effects to PCE 3 (food) from dissolved arsenic through reductions in food supply because invertebrates, such as planktonic crustaceans (*Daphnia*) and chironomid larvae on which suckers feed could be adversely affected by dissolved arsenic. The BE (USEPA 2008, page 5-107) states that the LC10 for cladocerans could be as low as 358 µg/L, which is less than the acute criterion, so some mortality of invertebrate prey is anticipated. Nevertheless, the effect of a reduced food supply, if there is an adverse effect, would likely be so small that its effect on the Lost River sucker or shortnose sucker would be insignificant because the reservoir is highly productive (Sullivan et al. 2011) and prey organisms would likely be entering the Keno Reservoir from upstream in Upper Klamath Lake, which is also highly productive (NRC 2004).

4.4.3.3 Effects of Cadmium on Lost River and Shortnose Suckers

Exposure Analysis

Cadmium enters the environment from various anthropogenic sources including: zinc refining, coal combustion, mine wastes, electroplating processes, iron and steel production, production of paint pigments, fertilizers, and pesticides, as a stabilizer for PVC plastics, in production and disposal of electrical batteries (e.g., nickel-cadmium rechargeable batteries), and cadmium concentrations also may be elevated in wastewater discharges due to corrosion of galvanized pipes (Eisler 1985a, USEPA 2001, Mebane 2006).

Total cadmium concentrations in highway runoff in Oregon and Washington State have been reported at levels of up to 2.8 µg/L (WADT 2007, Bloomquist 2009). Pitt et al. (1995) examined cadmium concentrations in stormwater runoff from a variety of sites and found the highest levels (up to 0.2 mg/L) were in street runoff; other areas of high concentrations were parking areas (up to 70 µg/L), roofs, vehicle service areas, and urban creeks (up to 30 µg/L). In a recent review of 10 years of data in USEPA's National Stormwater Quality Database comprising data from more than 200 municipalities, Pitt et al. (2004) concluded that median cadmium concentrations (dissolved) were in the range of 0.3-0.6 µg/L.

Within the range of the Lost River sucker and shortnose sucker, the presence of elevated cadmium levels in surface water is most likely to occur in the Keno Reservoir as a result of cadmium entering the Klamath Falls wastewater system and from discharge of untreated stormwater runoff from urbanized areas of Klamath Falls, and from other nonpoint-sources discharges into the reservoir.

Acute Effects Analysis

The acute cadmium criterion under review is 2 µg/L. Below in Table 4-27 are estimates of population level effects to Lost River sucker and shortnose sucker based on the ICE model. Because the acute criterion level is only 5% of the estimated LC10 value, we assume the effect will be either insignificant or discountable (i.e., NLAA).

Table 4-27. Estimated level of effect range to Lost River sucker and shortnose sucker from exposure to acute cadmium criterion of 2.0 µg/L at 100 mg/L $CaCO_3$.

Species [1]	Source of data	LC50 µg/L at 100 mg/L $CaCO_3$	LC10 µg/L at 100 mg/L $CaCO_3$	Population Effect
Lost River sucker	ICE	90.4	39.8	NLAA
Shortnose sucker	ICE	90.4	39.8	NLAA

Chronic Effects Analysis

The chronic cadmium criterion under review is 0.25 mg/L. Table 4-28 provides data on the level of effect for Catostomidae, based on the surrogate species fathead minnow. The data presented in Table 4-28 suggests that there is unlikely to be a reduction in growth of early life-stage suckers as a result of exposure to cadmium at the chronic criterion concentration.

Table 4-28. Maximum likelihood estimate for percent level of effect (reduction in reproduction compared to control) and confidence limits based on early life stage fish exposure to cadmium.

Listed species or family representative	Reduction in mean number of eggs per female of fathead minnow exposed to cadmium (Pickering and Gast 1972)	Reduction in hatching success of fathead minnow exposed to cadmium (Birge et al. 1985)
	P = 0.02; F = 45.0 MSE = 10.1 Adj.R^2 = 0.94 n = 4 Log input: -1.350 – 1.370 Log test: -1.099 – 1.995	P = 0.002; F = 51.1 MSE = 0.05 Adj.R^2 = 0.91 n = 6 Log input: -1.350 – 1.370 Log test: 0.652 – 2.907
Catostomidae	0.00 0.00 – 0.00	0.00 0.00 – 0.00

Table 4-29 provides data on the level of effect for Catostomidae based on the surrogate species brook trout. The data presented in Table 4-29 suggests that if a reduction in survival of early life-stage suckers occurs as a result of exposure to cadmium at the chronic criterion concentration, it will be small and is unlikely to result in adverse effects.

Table 4-29. Maximum likelihood estimate for percent level of effect (reduction in survival compared to control) and confidence limits based on early life stage fish exposure to cadmium.

Listed species or family representative	Reduction in survival of brook trout, early life stage, exposed to cadmium in soft water (37 mg/L CaCO3) from 31 to 60 days (Sauter et al. 1976).	Reduction in survival of brook trout, early life stage, exposed to cadmium in hard water (188 mg/L CaCO3) from 31 to 60 days (Sauter et al. 1976).
	P = 0.000; F = 106 MSE = 0.41 Adj.R^2 = 0.95 n = 7 Log input: -1.548 – 1.173 Log test: -0.702 – 1.970	P = 0.04; F = 9.4 MSE = 1.46 Adj.R^2 = 0.63 n = 6 Log input: -1.548 – 1.173 Log test: -0.780 – 1.442
Catostomidae	2.98 1.04 – 8.26	0.00 0.00 – 55.7

Listed species or family representative	Reduction in fathead minnow survival in early life stage test for 32 days at 43.9 mg/L CaCO3 (Spehar and Fiandt 1985).	Reduction in fathead minnow survival in early life stage test for 32 days at 55 mg/L CaCO3 (Spehar and Carlson 1984).	Reduction in fathead minnow survival in early life stage test for 8 days in test 1 (Birge et al. 1985).
	P = 0.01; F = 32.4 MSE = 1.15 Adj.R^2 = 0.89 n = 5 Log input: -1.350 – 1.370 Log test: 0.495 – 1.709	P = 0.002; F = 413 MSE = 0.03 Adj.R^2 = 0.99 n = 4 Log input: -1.350 – 1.370 Log test: 0.969 – 1.875	P = 0.009; F = 35.9 MSE = 1.04 Adj.R^2 = 0.90 n = 5 Log input: : -1.350 – 1.370 Log test: 0.652 – 2.466
Catostomidae	0.00 0.00 – 0.84	0.00 0.00 – 0.00	0.00 0.00 – 0.00

Effects Summary

Based on the data presented above, which summarize effects to both fathead minnow and brook trout as surrogates, exposure of Lost River sucker and shortnose sucker in the Keno Reservoir to cadmium at the acute and chronic criterion concentrations will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality in the Keno Reservoir slightly, and thus would have a

small adverse effect on the ability of PCE1 (water quality). Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

4.4.3.4 Effects of Chromium III on Lost River and Shortnose Suckers

Exposure Analysis

Chromium is used widely in domestic and industrial products and that some of its chemical forms, primarily hexavalent chromium (Cr +6) and trivalent chromium (Cr +3), are toxic (Eisler 1986). In aquatic environments, the major sources of chromium are discharges from electroplating and metal finishing industries and publicly-owned treatment plants; relatively minor sources (other than localized contamination) are iron and steel foundries, inorganic chemical plants, tanneries, textile manufacturing, and runoff from urban and residential areas (Eisler 1986). Sources of chromium in runoff include metal cleaners, caulking compounds, paint (Yost et al.1981). Chromium levels in wastewater effluent vary considerably depending on the types of industries present that discharge wastes into the wastewater system and can vary between at least 1-50 µg/L (Yost et al. 1981).

Total chromium concentrations in highway runoff in Washington State have been reported at levels of up to 18 µg/L (Herrara Environmental Consultants 2007, WADT 2007); however, chromium was not detected in an assessment of highway runoff in Oregon (Bloomquist 2009). Pitt et al. (1995) examined total chromium in stormwater runoff from a variety of sites across the U.S. and found the highest levels (up to 0.7 mg/L were in urban creeks; other areas of high concentrations were roofs (up to 0.5 mg/L), parking and storage areas (>0.3 mg/L) and landscape areas and detention ponds (>0.2 mg/L).

Within the range of the Lost River sucker and shortnose sucker, the presence of elevated chromium III levels in surface water is most likely to occur in the Keno Reservoir as a result of chromium entering the Klamath Falls wastewater system and from discharge of untreated stormwater runoff from urbanized areas of Klamath Falls.

Acute Effects Analysis

The acute chromium III criterion under review is 570µg/L at 100 mg/L $CaCO_3$. Below in Table 4-30 are estimates of population level effects of chromium III to the Lost River sucker and shortnose sucker based on the ICE model. Because the estimated level of effect from the ICE model is between the LC10 and LC50, we estimate the population level of effect to exposed populations of the Lost River sucker and shortnose sucker will be as high as 50%

Table 4-30. Estimated level of effect to the Lost River sucker and shortnose sucker from exposure to acute chromium III criterion of 570 µg/L at 100 mg/L $CaCO_3$, based on the ICE model.

Species	Data Source	LC50 µg/L at 100 mg/L $CaCO_3$	LC10 µg/L at 100 mg/L $CaCO_3$	Estimated Population Effect
Lost River sucker	ICE	863	380	Up to 50%
Shortnose sucker	ICE	863	380	Up to 50%

Chronic Effects Analysis

The chronic chromium III criterion under review is 74 µg/L at 100 mg/L $CaCO_3$. Table 4-31 provides the estimated level of effect for Catostomidae (sucker family) as a result of exposure to chromium III at the chronic criterion concentration, based on the surrogate species rainbow (steelhead) trout and fathead minnow. The data presented in Table 4-31 suggests that there could be a reduction in reproductive success based on reduced growth (both weight and length) of first and second generation early life-stage suckers of up to 15%, as a result of exposure to chromium III at the chronic criterion concentration. Because minnows and suckers belong to the same order of bony fishes, the Cypriniformes, the fathead minnow data are likely to be the best surrogate for suckers.

Table 4-31. Maximum likelihood estimate of percent level of effect on growth (reduction in mass, length, or biomass compared to control) and confidence limits based on early life stage fish exposure to trivalent chromium.

Listed species or family representative	Reduction in weight of rainbow trout, eyed embryos (Stevens and Chapman 1984)	Reduction in final length (mm) of first generation female fathead minnow exposed for 3 months (Pickering 1980b).	Reduction in final weight (g) of first generation female fathead minnow exposed for 3 months (Pickering 1980b).
	P = 0.01; F = 69.3 MSE = 0.00 Adj.R^2 = 0.96 n = 4 Log input: 1.981 – 2.667 Log test: 2.109 – 2.861	P = 0.02; F = 14.3 MSE = 0.00 Adj.R^2 = 0.73 n = 6 Log input: 2.007 – 2.693 Log test: 0.586 – 2.829	P = 0.02; F = 23.4 MSE = 0.00 Adj.R^2 = 0.85 n = 5 Log input: 2.007 – 2.693 Log test: 1.586 – 2.829
Catostomidae	4.02 (1.60 – 6.37)	5.92 (3.37 – 8.38)	15.4 (8.84 – 21.4)

Listed species or family representative	Reduction in 2nd generation mean length (mm) of fathead minnow exposed for 30 days (Pickering 1980b).
	P = 0.03; F = 16.2 MSE = 0.00 Adj.R^2 = 0.79 n = 5 Log input: 2.007 – 2.693 Log test: 1.586 – 2.829
Catostomidae	12.6 (5.37 – 19.4)

Table 4-32 provides estimated chronic level of effect for the survival of Catostomidae, based on the surrogate species rainbow (steelhead) trout and fathead minnow exposed to chromium III. Based on these data, we would expect a reduction in early life-stage sucker survival exposed to chromium III at the chronic criterion of up to 83% using the rainbow trout data, or 25% using the fathead minnow data. The fathead minnow data are likely to be the best for suckers, because minnows and suckers are in the same order of bony fishes, the Cypriniformes, therefore we assume there will be up to a 25% reduction in survival of young suckers exposed to chromium III at the chronic criterion.

Table 4-32. Maximum likelihood estimate for percent level of effect (reduction in survival compared to control) and confidence limits based on early life stage fish exposure to trivalent chromium.

Listed species or family representative	Reduction in survival of rainbow trout, fertilization to hatching (Stevens and Chapman 1984)	Reduction in survival of rainbow trout, hatching to end of test (Stevens and Chapman 1984)	Reduction in survival of rainbow trout, overall (Stevens and Chapman 1984)
	P = 0.04; F = 9.23 MSE = 3.54 Adj.R^2 = 0.62 n = 6 Log input: 1.981 – 2.667 Log test: 1.905 – 3.122	P = 0.01; F = 18.8 MSE = 0.52 Adj.R^2 = 0.78 n = 6 Log input: 1.981 – 2.667 Log test: 1.905 – 3.122	P = 0.04; F = 11.9 MSE = 1.79 Adj.R^2 = 0.73 n = 5 Log input: 1.981 – 2.667 Log test: 1.905 – 2.861
Catostomidae	82.7 (24.1 – 98.6)	24.6 (9.50 – 46.3)	79.8 (21.8 – 98.0)

Listed species or family representative	Reduction in survival of rainbow trout, eyed embryo to hatch (Stevens and Chapman 1984)	Reduction in survival of fathead minnow to 3 months (Pickering 1980b).	Reduction in survival of 2nd generation fathead minnow to 30 days (Pickering 1980b).
	P = 0.04; F = 27.6 MSE = 1.92 Adj.R^2 = 0.87 n = 5 Log input: 1.981 – 2.667 Log test: 2.109 – 3.122	P = 0.0004; F = 114 MSE = 0.25 Adj.R^2 = 0.96 n = 6 Log input: 2.007 – 2.693 Log test: 0.586 – 2.829	P = 0.02; F = 18.7 MSE = 0.02 Adj.R^2 = 0.82 n = 5 Log input: 2.007 – 2.693 Log test: 1.586 – 2.829
Catostomidae	31.8 (4.40 – 78.0)	25.3 (13.2 – 42.8)	12.5 (6.07 – 20.0)

Effects Summary

Based on the data presented above, exposure of Lost River sucker and shortnose sucker in the Keno Reservoir to chromium III at the acute criterion concentration is likely to result in both injury and mortality. A reduction in reproductive success is anticipated based on reduced growth (both weight and length) of first and second generation early life-stage suckers of up to 15% per 3 years over the 25-year life of the proposed action. Furthermore, as a result of exposure to chromium III at the chronic criterion concentration we found that there the likelihood for up to a 25% reduction in survival for young suckers every 3 years over the 25-year life of the proposed action.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality in the Keno Reservoir, and thus adversely affect the ability of PCE 1 (water quality) to support the species. This effect will be confined to the Keno Reservoir where the point-source discharges occur, as described above. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

4.4.3.5 Effects of Chromium IV on Lost River and Shortnose Suckers

Exposure Analysis

See summary under Chromium III, above.

Acute Effects Analysis

The acute chromium VI criterion under review is 16 µg/L at 100 mg/L $CaCO_3$. Below in Table 4-33 are estimates of population level effects to Lost River sucker and shortnose sucker based on the ICE model. Because the level of effects is less substantially less than 10%, we assume the effect will be too low to be measured and therefore is either discountable or insignificant (i.e., not likely to adversely affect).

Table 4-33. Estimated level of effect to Lost River sucker and shortnose sucker from exposure to acute chromium VI criterion of 16 µg/L based on the ICE model.

Species	Data Source	LC50 µg/L at 100 mg/L $CaCO_3$	LC10 µg/L at 100 mg/L $CaCO_3$	Population Effect
Lost River sucker	ICE	20728	9131	NLAA
Shortnose sucker	ICE	20728	9131	NLAA

Chronic Effects Analysis

The chronic chromium VI criterion under review is 11 µg/L at 100 mg/L $CaCO_3$. Table 4-34 provides data on the level of effect for Catostomidae, based on the surrogate species chinook salmon and rainbow trout. The data presented in Table 4-34 suggest that there could be a reduction in growth of early life-stage suckers of up to 25%, as a result of exposure to chromium VI at the chronic criterion concentration. Salmonids and suckers belong to different orders of fish (Salmoniformes and Cyprinidiformes, respectively); however, these are the best available data with which to evaluate the effects of chronic exposure to chromium VI on the Lost River and shortnose suckers. Furthermore, there is substantial difference in the results for chinook salmon and rainbow trout in Table 4-34, which shows that effects of chronic chromium VI

exposure to fish, even among closely-related species, can be quite variable, so the extrapolation of salmonid effects to suckers may be less accurate.

Table 4-34. Maximum likelihood estimate of percent level of effect on growth (reduction in mass, length, or biomass compared to control) and confidence limits based on early life stage fish exposure to hexavalent chromium.

Listed species or family representative	Reduction in cumulative final length (mm) of chinook salmon after 7-month exposure (egg to fingerling) (Olson and Foster).	Reduction in cumulative final length (mm) of rainbow trout after 7-month exposure (egg to fingerling) (Olson and Foster 1956).
	P = 0.007; F = 20.1 MSE = 0.002 Adj.R^2 = 0.76 n = 7 Log input: 1.735 – 2.012 Log test: 0.761 – 2.248	P = 0.003; F = 44.9 MSE = 0.0001 Adj.R^2 = 0.90 n = 6 Log input: 0.966 – 1.243 Log test: 0.886 – 2.208
Catostomidae	24.9 (14.5 –34.0)	3.64 (0.00 – 7.66)

Table 4-35 provides data on the chronic level of effect for the survival of Catostomidae, based on the surrogate species chinook salmon, rainbow trout, and fathead minnow exposed to chromium VI. The fathead minnow data are likely to be the best for suckers, because minnows are most closely related to suckers being in the same order of bony fishes, the Cypriniformes. Based on this, we would expect a low level of effect on sucker survival at the chronic criterion concentration. Similar to the growth effects mentioned above, the salmonid data for chinook salmon and rainbow trout show that effects, even among closely-related species are quite variable, so the extrapolation of fathead minnow survival to suckers should be viewed with caution.

Table 4-35. Maximum likelihood estimate for percent level of effect (reduction in survival compared to control) and confidence limits based on early life stage fish exposure to hexavalent chromium.

Listed species or family representative	Reduction in cumulative survival of chinook salmon after 7-month exposure (egg to fingerling) (Olson and Foster 1956).	Reduction in cumulative survival of rainbow trout after 7-month exposure (egg to fingerling) (Olson and Foster 1956).	Reduction in survival of fathead minnow surviving to 9 weeks in early life stage test (Pickering 1980a).
	P = 0.004; F = 26.2 MSE = 0.19 Adj.R^2 = 0.81 n = 7 Log input: 1.735 – 2.012 Log test: 0.761 – 2.248	P = 0.003; F = 40.1 MSE = 0.19 Adj.R^2 = 0.89 n = 6 Log input: 0.966 – 1.243 Log test: 0.886 – 2.208	P = 0.0006; F = 94.2 MSE = 0.14 Adj.R^2 = 0.90 n = 6 Log input: 1.226 – 1.502 Log test: 0.238 – 3.580
Catostomidae	36.3 (20.2 –52.9)	0.00 (0.00 – 6.42)	2.12 (0.90 – 4.09)

Summary of Effects

Based on the data presented above, acute exposure of Lost River sucker and shortnose sucker to chromium VI at the criterion concentration is unlikely to have a measureable or substantial effect, but chronic exposure to chromium VI could result in reduced growth and lowered survival. However, because chronic effects are quite variable, even among closely-related species, it is uncertain if the surrogate results are applicable to suckers. However, assuming a reasonable worst case, there is likely to be a reduction in survival of early life stage suckers of up to 36% in Keno Reservoir every 3 years over the 25-year life of the proposed action, based on the reduction in cumulative survival of chinook salmon after 7-month exposure.

Effects of acute and chronic chromium VI exposure to invertebrate prey were also analyzed. Based on the study by Munzinger and Monicelli (1991) on *Daphnia dubia*, the level of effect in the number of young produced per adult could be reduced up to 92%. *Daphnia* are a major food species of shortnose sucker (USFWS 2008g); however, a reduced food supply is unlikely to adversely affect suckers because of the high productivity in the reservoir (Sullivan et al. 2011) and input of *Daphnia* from upstream in Upper Klamath Lake.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality in the Keno Reservoir. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker.

Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

4.4.3.6 Effects of Copper on Lost River and Shortnose Suckers

Exposure Analysis

Lost River suckers and shortnose suckers in the Keno Reservoir are considered to be exposed to dissolved copper at criterion concentrations (acute = 13 µg/L and chronic = 9 µg/L, both at 100 mg/L $CaCO_3$) owing to the presence of two STPs that discharge into the Keno Reservoir, as well as discharges of untreated stormwater. Copper concentrations in wastewater effluent vary considerably based on the types of industries discharging into the wastewater system. Yost et al. (1981) found levels of copper in wastewater effluent ranged from 16 to 400 µg/L. Copper is also found in highway stormwater runoff as a result of vehicle brake pad wear, residual engine oil, and atmospheric deposition (Bloomquist 2009). Dissolved copper was found in Oregon highway runoff at concentrations ranging from 2-60 µg/L (Bloomquist 2009). In Washington State, dissolved copper concentrations in highway runoff were found to have with a median concentration of 5µg/L and a maximum concentration of 18 µg/L (WADT 2007). Pitt et al. (1995) reviewed available stormwater data from USEPA and found that concentrations of dissolved copper varied considerably based on the source area and were highest in parking and storage areas. Dissolved copper can also be found in agricultural runoff because copper salts are used to control some agricultural pests (Rice et al. 2004).

According to the 2008, 303(d) list submitted to USEPA by ODEQ, eleven Oregon surface water sites are currently listed as impaired with respect to dissolved copper, but none are in the Klamath River Basin.

Acute Effects Analysis

The acute copper criterion under review is 13 µg/L at 100 mg/L as $CaCO_3$ and Table 4-36 provides information on the acute level of effect for the Lost River sucker and shortnose sucker. The ICE model estimate of the LC50 for copper for the family Catostomidae is 75 µg/L. Dividing this value by 2.27 produces an LC10 estimate of 33.2 µg/L. Because the acute criterion concentration is equal to 39% of the LC10, we assume exposure to acute levels will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death.

Table 4-36. Estimated level of effect for acute copper criterion of 13 µg/L at 100 mg/L as $CaCO_3$ for endangered sucker species.

Species	Source of data	LC50 µg/L @ 100 mg/L $CaCO_3$	LC10 µg/L @ 100 mg/L $CaCO_3$	Acute Criterion as Percent of LC10	Population Effect
Lost River sucker	ICE	75.3	33.2	39	LAA
Shortnose sucker	ICE	75.3	33.2	39	LAA

Chronic Effects Analysis

The chronic copper criterion under review is 9 µg/L at 100 mg/L as $CaCO_3$. Table 4-37 provides information for using the level of effect, with fathead minnow embryos as surrogates for listed suckers. We estimated exposure to the chronic criterion dissolved copper concentration would reduce survival in catostomids by 9.9% (95% confidence limits = 6.7% to 14.2%). Because the 9.9 value is so close to 10 and the confidence limits clearly overlap 10, we assume there will be up to 10% population level effect. Fathead minnows are used in this analysis as a surrogate for suckers because there are no chronic toxicity data for suckers, and minnows (family Cyprinidae) are the most closely related group of fishes because they both belong to the bony fish order Cypriniformes. Test data for the spotfin chub could also have been used but test reliability was highest for the fathead minnow, so we used those data.

Table 4-37. Maximum likelihood estimate (and range) for percent level of effect (reduction in survival compared to control) and confidence limits for copper.

Listed family representative	Fathead minnow embryo reduction in survival (Lind 1978)	Spotfin chub reduction in survival (Besser et al. 2001)
Catostomidae	9.88% (6.67 – 14.2)	1.65% (0.60-3.35)

Other chronic toxicity studies have examined effects of copper on fathead minnow growth (Table 4-38). Using these studies to develop regression models and estimates of level of effect for catostomids, results predict that exposure of catostomid larvae to the chronic copper criterion concentration would have little or no effect on length, but reduction in biomass of embryo and larvae (measured as either wet or dry weight) would be much more likely to occur.

Table 4-38. Catostomidae – predicted maximum likelihood estimate of percent level of effect on growth (reduction in mass, length, or biomass compared to control) and confidence limits based on fathead minnow early life stage fish exposure to copper.

Fry reduction in growth based on female mass after 11-week exposure (Mount 1968)	6-week old fry reduction in growth based on male length (Mount and Stephen 1969)	6-week old fry reduction in growth based on female length (Mount and Stephen 1969)	Embryo reduction in growth based on wet mass (Lind 1978)	Reduction in larval growth based on dry biomass (Besser et al. 2001)
8.2 (5.44–10.9)	9.8 (5.33-14.0)	7.6 (0.00-14.0)	81 (51.9-92.8)	64 (48.9-74.2)

Other studies have examined the effects of copper on egg production by fathead minnows to assess possible impacts to reproduction. All these tests predict that catostomids exposed to copper at the criterion concentration would experience some reductions in egg production with the effect ranging from 34% to 75%, depending on exposure (Table 4-38Table 4-39).

Table 4-39. Catostomidae – predicted maximum likelihood estimate for percent level of effect on reduction in number of eggs per spawn, or number of eggs per female compared to control, and confidence limits, based on fathead minnow exposure to copper . Pickering (1977).

Reduction in number of eggs per spawning with no previous copper exposure	Reduction in number of eggs per female with no previous copper exposure	Reduction in number of eggs per spawning with 3 months previous copper exposure	Reduction in number of eggs per female with 3 months previous copper exposure	Reduction in number of eggs per spawning with 6 months previous copper exposure	Reduction in number of eggs per female with 6 months previous copper exposure
34% (28.5%-38.8%)	66% (54.4%-75.9%)	38% (23.4%-50.2%)	60% (31.8%-80.0%)	49% (45.4%-53.0%)	75% (63.2%-84.0%)

Effects Summary

Based on the above discussion, we anticipate adverse effects to Lost River sucker and shortnose sucker from dissolved copper to only occur in the Keno Reservoir where acute and chronic exposure to dissolved copper would result in mortality of up to 20% of the adults, juveniles, and larvae every 3 years over the 25-year life of the proposed action. Additionally there could be up to a 75% reduction in the numbers of eggs produced per female as a result of the proposed action; however, because there is no reproduction occurring in the Keno Reservoir owing to a lack of suitable spawning habitat, we do not anticipate this to effect the suckers.

In addition to the effects described above dissolved copper also can adversely affect olfaction in fish by competing with natural odorants for binding sites, by affecting activation of the olfactory receptor neurons or by affecting intracellular signaling in the neurons (McIntyre et al. 2008a,b). Even brief exposure to dissolved copper at environmentally realistic concentrations (i.e., 2-5 µg/L) can impair the function of olfactory receptor neurons in salmon, but longer exposures can kill these neurons making them non-functional and thus disabling olfaction. DeForest et al. (2011a) have recently analyzed copper criteria and its effects on salmon olfaction for all western states that have salmon runs, including Oregon, and found that the criteria were protective. Assuming that suckers are similar to salmon or even more likely to be less sensitive to olfactory effects of copper, the criteria should also not adversely affect suckers through olfaction.

There could also be indirect effects of dissolved copper on suckers, such as reductions in food supply because invertebrates (planktonic crustaceans and midge larvae on which suckers feed) are highly sensitive to copper. Both the acute and chronic criteria levels of copper are known to be lethal to *Daphnia magna* at criteria concentrations (Brix 2001, Bossuyt and Janssen 2003) however, a reduced food supply is unlikely to affect suckers because of the high productivity in the reservoir (Sullivan et al. 2011) and input of *Daphnia* from upstream in Upper Klamath Lake. Therefore, we do not anticipate that dissolved copper will limit the food supply available to suckers.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality (PCE 1) in the Keno Reservoir. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

There could also be adverse effects to PCE 3 (food) from dissolved copper through reductions in food supply because invertebrates, such as planktonic crustaceans (*Daphnia*) and chironomid larvae on which suckers feed could be adversely affected by dissolved copper. Nevertheless, the effect of a reduced food supply, if there is an adverse effect, would likely be so small that its effect on the Lost River sucker or shortnose sucker would be insignificant because the reservoir is highly productive (Sullivan et al. 2011) and prey organisms would likely be entering the Keno Reservoir from upstream in Upper Klamath Lake, which is also highly productive (NRC 2004).

4.4.3.7 Effects of Lead on Lost River and Shortnose Suckers

Exposure Analysis

Within their range in Oregon, we assume that Lost River sucker and shortnose sucker will only be affected by dissolved lead as result of the proposed action in the Keno Reservoir because of

the presence of two STPs that discharge into the reservoir. Lead is often present in STP discharges and concentrations in wastewater effluents are highly variable among cities and over time at the same STP, and can range from 11 to 167 µg/L, according to one USEPA assessment (Yost et al. 1981). Lead was found in runoff from highways in Oregon at concentrations ranging from below detection to 0.7 µg/L (Bloomquist 2009). In a study of highway runoff in Washington State dissolved lead concentrations had a median concentration of 3.2 µg/L (WADT 2007).

Acute Effects Analysis

The acute lead criterion under review is 65 µg/L at 100 mg/L as $CaCO_3$. Table 4-40 provides information on the acute level of effect for suckers based on the ICE model. The ICE model estimate of the LC50 for dissolved lead for the family Catostomidae is 579 µg/L (Table 4-40). Dividing this value by 2.27 produces an LC10 estimate of 255 µg/L. Because the criterion value of 65 µg/L is 25% of the LC10, we assume exposure to acute levels will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death of Lost River sucker and shortnose sucker individuals.

Table 4-40. Level of effect range for acute lead criteria for catostomids.

Catostomid Species	Source of data	LC50 µg/L @ 100 mg/L $CaCO_3$	LC10 µg/L @ 100 mg/L $CaCO_3$	Acute Criterion as Percent of LC10	Population Effect
Lost River sucker	ICE	579	255	25	LAA
Shortnose sucker	ICE	579	255	25	LAA

Chronic Effects Analysis

The chronic dissolved lead criterion under review is 2.5 µg/L (at 100 mg/L as $CaCO_3$). Table 4-41 provides information for using the level of effect methodology for the common carp (*Cyprinus carpio*), rainbow trout (*O. mykiss*) and brook trout, (*Salvelinus fontinalis*). Of the fishes where there are data on chronic effects of lead, the common carp is the best surrogate fish species for suckers because the Lost River sucker and shortnose sucker and common carp all belong to the order Cypriniformes. Nevertheless, the tests with the two trout species provide additional information on chronic effects. The data presented in Table 4-41 suggest that larval deformities and reduced hatching are likely results of chronic exposure to lead at the criterion concentration and that the level of effect ranges from 59% to 90% reduction for hatching success and for proportion of deformed hatchlings, respectively. The trout data suggest that there is a high likelihood of embryo deformities that would reduce reproduction, thus corroborating the carp data by Kapur and Yadav (1982) regarding deformities in hatchlings.

Table 4-41. Maximum likelihood estimate for percent level of effect (reduction in survival, reproduction, or growth compared to control) and confidence limits based on early life stage fish exposure to dissolved lead.

Listed species or family representative	Rainbow trout embryo reproduction (deformities that impact reproduction) after 19 month exposure, hard water, eggs not previously exposed (Davies et al. 1976)	Brook trout hatching success of eggs produced from parents previously exposed from yearling to spawning (2nd generation test) (Holcombe et al. 1976)	Common carp hatching success (Kapur and Yadav 1982)	Common carp proportion hatchlings deformed (Kapur and Yadav 1982)	Rainbow trout fry survival exposed for 30 days (Sauter et al. 1976)
	P = 0.03; F = 14.1 MSE = 3.53 Adj.R^2 = 0.77 n = 5 Input: 2.9 – 10.9 Test: 0.0 - 282	P = 0.03; F = 10.3 MSE = 0.42 Adj.R^2 = 0.65 n = 6 Input: 12.3-14.8 Test: 2.18 – 1,215	P = 0.001; F = 63.8 MSE = 0.17 Adj.R^2 = 0.93 n = 6 Input: 3.0 - 11 Test: 0.30 – 11.8	P = 0.0001; F = 63.8 MSE = 0.02 Adj.R^2 = 0.98 n = 6 Input: 3.0 - 11 Test: 0.30 – 11.8	P = 0.03; F = 8.4 MSE = 1.7 Adj.R^2 = 0.98 n = 6 Input: 2.9 – 10.9 Test: 3.74 – 2,516
Catostomidae	88.6 13.8 – 99.7	10.9 0.22 – 28.7	90.4 80.2 – 95.5	59.4 51.4 – 66.8	0.00 0.00 – 30.6

Effects Summary

Based on the above-mentioned toxicological studies and modeling, Lost River sucker and shortnose sucker if exposed to dissolved lead at the criteria concentrations would be predicted to show up to 10% mortality at acute concentrations of all life history stages in the Keno Reservoir. If exposed to lead at chronic criterion concentrations sucker populations could experience up to a 90% reduction in hatching success, up to a 59% in larval deformities, and up to an 80% adverse effects in reproduction as a result of embryo deformities. However because no known sucker spawning occurs in the Keno Reservoir, there will be no chronic effects to reproduction.

There could also be indirect effects of dissolved lead on suckers, such as reductions in food supply because invertebrates (e.g., planktonic crustaceans such as *Daphnia*) are sensitive to dissolved lead and show up to a 16% reduction in survival (Copper et al. 2009); however, a reduced food supply is unlikely to substantially affect suckers because of the high productivity in the reservoir (Sullivan et al. 2011) and input of *Daphnia* from upstream in Upper Klamath Lake.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the

proposed action will reduce water quality (PCE 1) in the Keno Reservoir. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

There could also be adverse effects to PCE 3 (food) from dissolved lead through reductions in food supply because invertebrates, such as planktonic crustaceans (*Daphnia*) and chironomid larvae on which suckers feed could be adversely affected by dissolved lead. Nevertheless, the effect of a reduced food supply, if there is an adverse effect, would likely be so small that its effect on the Lost River sucker or shortnose sucker would be insignificant because the reservoir is highly productive (Sullivan et al. 2011) and prey organisms would likely be entering the Keno Reservoir from upstream in Upper Klamath Lake, which is also highly productive (NRC 2004).

4.4.3.8 Effects of Nickel on Lost River and Shortnose Suckers

Exposure Analysis

Within their range in Oregon, we assume that Lost River sucker and shortnose sucker will only be affected by dissolved nickel as result of the proposed action in the Keno Reservoir because of the presence of two STPs that discharge into the reservoir. Nickel is often present in STP discharges and concentrations in wastewater effluents are highly variable among cities and over time at the same STP, and can range from 32 to 885 µg/L, according to one USEPA assessment (Yost et al. 1981, p. 12). Pitt et al. (1995) examined total nickel in stormwater runoff from a variety of sites and found the highest levels (up to 0.7 mg/L were in urban creeks); other areas of high concentrations were roofs (up to 0.5 mg/L), parking and storage areas (>0.3 mg/L) and landscape areas and detention ponds (>0.2 mg/L). Total nickel concentrations in highway runoff in Washington State have been reported at levels of up to 12.9 µg/L (Herrara Environmental Consultants 2007, WADT 2007). In Oregon, dissolved nickel concentrations in highway runoff were found to range from below detection limits of 9.6 µg/L (Bloomquist 2009).

Acute Effects Analysis

The acute nickel criterion under review is 468 µg/L at 100 mg/L $CaCO_3$. Table 4-42 provides data on the acute level of effect for suckers based on the ICE model. The ICE model estimate of the LC50 for dissolved nickel for the fish family Catostomidae is 693 µg/L. Dividing 693 µg/L by 2.27 produces an LC10 estimate of 305 µg/L. Because the nickel criterion value of 468 µg/L is between the LC10 and LC50, we assume that the proposed acute nickel criterion would lead to mortality of up to 50% of the Lost River sucker and shortnose sucker populations in the Keno Reservoir.

Table 4-42. Level of effect range to Lost River sucker and shortnose sucker from exposure to acute nickel criterion of 468 µg/L (at 100 mg/L $CaCO_3$) determined from the ICE model.

Species	Source of data	LC50 µg/L at 100 mg/L $CaCO_3$	LC10 µg/L at 100 mg/L $CaCO_3$	Population Effect %
Lost River sucker	ICE	693	305	Up to 50
Shortnose sucker	ICE	693	305	Up to 50

Chronic Effects Analysis

The chronic dissolved nickel criterion under review is 52 µg/L at 100 mg/L as $CaCO_3$. Table 4-43 provides data on the level of effect for the rainbow trout exposed to dissolved nickel. Although the rainbow trout is not the best surrogate fish species for suckers, because suckers and trout are not closely related since they belong to different fish orders, these are the best available scientific data to assess effects of nickel on early life stage growth of suckers. The data presented in Table 4-43 suggests that suckers would likely show a reduction in larval biomass (wet weight) of up to 44% and a reduction in larval length of up to 15%, as a result of exposure to dissolved nickel at the chronic criterion concentration.

Table 4-43. Maximum likelihood estimate for percent level of effect (reduction in growth compared to control) and confidence limits based on early life stage fish exposure to dissolved nickel.

Listed species or family representative	Reduction in growth (g wet mass) of rainbow trout early life stage (Test 4 starting with pre-swim-up larvae 5 days after hatching) and exposed for 38 days (Nebecker et al. 1984)	Reduction in length (mm) of rainbow trout early life stage (Test 4 starting with pre-swim-up larvae 5 days after hatching) and exposed for 38 days (Nebecker et al. 1985)
	P = 0.02; F = 22.6 MSE = 0.007 Adj.R^2 = 0.84 n = 5 Input: 73.4 – 965 Test: 24.0 – 6,691	P = 0.03; F = 16.6 MSE = 0.0007 Adj.R^2 = 0.80 n = 5 Input: 73.4 – 965 Test: 24.0 – 6,691
Catostomidae	44.3 27.3 – 57.4	15.2 7.43 – 22.3

In Table 4-44 below are presented data showing the estimated level of effect of dissolved nickel on fish reproduction (i.e., female fecundity, which is defined in the test as the number of eggs produced per spawning event). The surrogate species is fathead minnow, which is related to the suckers at the order level, and thus provides a more accurate estimate for the suckers. The data suggest that there is likely to be up to a 43% reduction in female fecundity for the Lost River sucker and shortnose sucker as a result of exposure to dissolved nickel at the chronic criterion concentration.

Table 4-44. Maximum likelihood estimate for percent level of effect (reduction in reproduction compared to control) and confidence limits based on early life stage fish exposure to nickel.

Listed species or family representative	Reduction in fecundity (no. eggs per spawning) of fathead minnow previously exposed (Pickering 1974)
	P = 0.04; F = 9.64 MSE = 1.92 Adj.R^2 = 0.63 n = 6 Input: 23.1 - 304 Test: 4.42 – 863
Catostomidae	42.6 19.4 – 62.0

In Table 4-45 below are shown test results from exposure of rainbow trout and Atlantic salmon eggs/embryos to dissolved nickel. Although these species are not in the same order of fishes as the suckers, these are the best available with which to evaluate effects to suckers; the salmonid data shows strong evidence for a reduction in survival of developing embryos, ranging from an estimated 72% to 99% for suckers.

Table 4-45. Maximum likelihood estimate for percent level of effect (reduction in survival compared to control) and confidence limits based on early life stage fish exposure to nickel.

Listed species or family representative	Reduction in survival of rainbow trout embryos in early life stage Test 1 with newly fertilized eggs exposed for 85 days (Nebecker et al. 1985)	Reduction in survival of rainbow trout early life stage from egg to end of test (Test 1 with newly fertilized eggs) exposed for 85 days (Nebecker et al. 1985)	Reduction in survival of Atlantic salmon eggs exposed until hatching (Grande and Andersen 1983)
	P = 0.006; F = 17.1 MSE = 1.38 Adj.R^2 = 0.70 n = 8 Input: 73.4 – 965 Test: 5.98 – 1,878	P = 0.008; F = 15.1 MSE = 1.60 Adj.R^2 = 0.67 n = 8 Input: 73.4 – 965 Test: 5.98 – 1,878	P = 0.02; F = 18.1 MSE = 5.01 Adj.R^2 = 0.81 n = 5 Input: 921 – 12,115 Test: 32.3 – 32,310
Catostomidae	71.5 37.7 – 90.6	84.7 54.5 – 96.0	99.2 98.3 – 100

Effects Summary

Based on the above-mentioned toxicological studies and modeling, Lost River sucker and shortnose sucker, if exposed to dissolved nickel at the acute criterion concentration of dissolved nickel is likely to kill up to 50% of all life stages of Lost River sucker and shortnose sucker in the Keno Reservoir every 3 years for the 25-year life of the proposed action. At the chronic criteria concentrations Lost River sucker and shortnose sucker would be predicted to show lower

reproductive success in the form of reduced in embryo survival, hatching success, and embryonic, as well as increased larval deformities. However because no known sucker spawning occurs in the Keno Reservoir, there would be no chronic effects to reproduction.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality (PCE 1) in the Keno Reservoir. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

4.4.3.9 Effects of Pentachlorophenol on Lost River and Shortnose Suckers

Exposure Analysis

The chemical Pentachlorophenol (PCP) is used as an industrial wood preservative and in production of herbicides (Eisler 1989a). As a wood preservative, PCP acts as both a fungicide and insecticide. In 1989, PCP was the third most heavily used pesticide, its use exceeded only by the herbicides atrazine and alachlor (Eisler 1989a). The chemical PCP has been used as a bactericide and fungicide to protect many products, such as adhesives, paper and paperboard, cable coverings, leather, paints, textiles, rope, ink, rubber, and petroleum drilling muds; PCP has also been used as an herbicide on ornamental lawns, golf courses, aquatic areas, and rights-of-way; for control of subterranean termites; as an anti-microbial agent in cooling towers, adhesives, latex paints, paper coatings, cements used with food can ends and seals, coatings in reusable bulk food storage containers, photographic solutions, leather tanneries, and pulp and paper mills; and, as a disinfectant (Fisher 1991). Commercial samples of technical grade PCP were heavily contaminated with many compounds, including chlorophenols, dioxins, dibenzofurans, hexachlorobenzene, and phenoxyphenols; the relative toxicities and accumulation potentials of some of these contaminants may exceed those of PCP by several orders of magnitude (Eisler 1989a). Because of the long-term use of PCP, it is a significant contaminant of soil, surface water, and groundwater, especially around sawmills and wood preserving facilities, and because most of the PCP in the environment will end up in soil, it can be transported to surface water by runoff (Fisher 1991).

Within the range of the Lost River sucker and shortnose sucker, the presence of PCP contamination of surface water is most likely to occur in the Keno Reservoir as a result of untreated stormwater runoff from urbanized areas of Klamath Falls, or from old mill sites present along the shoreline of the reservoir. Contamination from PCP has been detected at low levels of 2 µg/L in Klamath Basin groundwater by ODEQ (Cole 2006). PCP could enter sucker habitat in other areas of their range, but the low density of potential PCP sources would mean that PCP concentrations would be so low as to have no effect.

Acute Effects Analysis

The acute PCP criterion under review is 19 mg/L at pH 7.8. Below in Table 4-46 are estimates of population level effects to Lost River sucker and shortnose sucker from PCP at the acute criterion based on the ICE model. Because the acute criterion concentration is approximately 30% of the LC10, we assume exposure to acute levels will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death of Lost River sucker and shortnose sucker.

Table 4-46. Estimated level of effect to Lost River sucker and shortnose sucker from exposure to PCP at the acute criterion of 19 mg/L at pH 7.8.

Species	Source of data	LC50 µg/L at pH 7.8	LC10 µg/L at pH 7.8	Acute Criterion as Percent of LC10	Population Effect
Lost River sucker	ICE	142.33	62.70	30	LAA
Shortnose sucker	ICE	142.33	62.70	30	LAA

Chronic Effects Analysis

Reduction in Growth

The PCP chronic criterion under review is 15 mg/L at pH 7.8. Table 4-47 below provides the estimated level of effect for Catostomidae as a result of exposure to PCP at the chronic criterion concentration based on the surrogate species sockeye salmon, rainbow (steelhead) trout, and fathead minnow. The data presented in Table 4-47 suggests that there could be a reduction in growth for suckers and a lower gross conversion efficiency as a result of exposure to PCP at the chronic criterion concentration, based on the trout and sockeye salmon data. However, the fathead minnow data are likely to be the best with which to assess effects to suckers because minnows are more closely related to suckers because they are in the same order of bony fishes, the Cypriniformes. Based on this, all of the effects of PCP exposure at the chronic criterion concentration shown in Table 4-47 on fathead minnow growth were less than 10% and most results were substantially less than 10% and the confidence limits of most overlapped with zero. The results from the experiment conducted by Spehar et al. (1985) using fathead minnows exposed to PCP at pH 7.5; however, did result in an estimated effect to suckers of almost 10%. This suggests that suckers exposed to PPT and at chronic criterion level could experience a reduction in weight, but that this is likely to be too small to have a substantial effect, and therefore the effects are likely to be insignificant or discountable (i.e., NLAA).

Table 4-47. Maximum likelihood estimate for percent level of effect (reduction in growth compared to control) and confidence limits based on early life stage fish exposure to PCP at the chronic criterion of 15 mg/L at pH 7.8.

Listed species or family representative	Reduction in growth rate (% dry wt per day) of subyearling sockeye salmon (Webb and Brett 1973)	Reduction in gross conversion efficiency of subyearling sockeye salmon (Webb and Brett 1973)	Reduction in mean weight (mg) of steelhead trout exposed for 72 days (Dominquez and Chapman 1984)
	P = 0.000; F = 1781 MSE = 0.002 Adj.R^2 = 0.96 n = 8 Log input:0.938 – 1.018 Log test: 0.117 – 1.734	P = 0.004; F = 20.2 MSE = 0.05 Adj.R^2 = 0.73 n = 8 Log input:0.938 – 1.018 Log test: 0.117 – 1.734	P = 0.008; F = 131 MSE = 0.0001 Adj.R^2 = 0.98 n = 4 Log input:1.092 – 1.172 Log test: 0.614 – 1.302
Catostomidae	25.8 (22.7 – 28.7)	32.4 (20.1 – 43.0)	20.7 (16.3 – 24.7)
Listed species or family representative	Reduction in mean wet weight (mg) of fathead minnow exposed for 32 days at pH 7.5 (Spehar et al. 1985).	Reduction in mean wet weight (mg) of fathead minnow exposed for 32 days at pH 8.0 (Spehar et al. 1985).	Reduction in mean wet weight (mg) of fathead minnow exposed for 32 days at pH 8.5 (Spehar et al. 1985).
	P = 0.03; F = 36.8 MSE = 0.0001 Adj.R^2 = 0.92 n = 4 Log input: 1.391 – 1.472 Log test: 0.765 – 1.782	P = 0.02; F = 44.2 MSE = 0.0001 Adj.R^2 = 0.94 n = 8 Log input: 1.391 – 1.472 Log test: 1.093 – 2.196	P = 0.048; F = 19.5 MSE = 0.0001 Adj.R^2 = 0.86 n = 4 Log input: 1.391 – 1.472 Log test: 1.429 – 2.337
Catostomidae	9.45 (4.64 – 14.0)	3.57 (0.00 – 8.36)	0.00 (0.00 – 5.92)
Listed species or family representative	Reduction in mean wet weight (mg) of fathead minnow exposed for 28 days, 1 fish per cage (Arthur and Dixon 1994).	Reduction in mean wet weight (mg) of fathead minnow exposed for 28 days, 5 fish per cage (Arthur and Dixon 1994).	Reduction in mean wet weight (mg) of fathead minnow exposed for 28 days, 10 fish per cage (Arthur and Dixon 1994).
	P = 0.003; F = 23.9 MSE = 0.001 Adj.R^2 = 0.76 n = 8 Log input: 1.391 – 1.472 Log test: 1.146 – 2.124	P = 0.007; F = 15.8 MSE = 0.0002 Adj.R^2 = 0.68 n = 8 Log input: 1.391 – 1.472 Log test: 1.146 – 2.124	P = 0.0004; F = 50.9 MSE = 0.0002 Adj.R^2 = 0.88 n = 8 Log input: 1.391 – 1.472 Log test: 1.429 – 2.337
Catostomidae	1.44 (0.00 – 10.2)	2.72 (0.00 – 7.03)	3.62 (0.00 – 7.94)

Reduction in Survival

Table 4-48 below provides estimated chronic criterion level of effect for the survival of Catostomidae, based on the surrogate species rainbow (steelhead) trout and fathead minnow exposed to PCP. Once again the values obtained for fathead minnows are likely the best with which to assess effects to suckers. Because all of the effects to fathead minnows listed in Table 4-48 were substantially less than 10, we assume the effects of PCP at the chronic criterion on sucker survival will be too small to have a substantial effect and therefore are insignificant or discountable (i.e., NLAA).

Table 4-48. Maximum likelihood estimate for percent level of effect (reduction in survival compared to control) and confidence limits based on early life stage fish exposure to PCP.

Listed species or family representative	Reduction in pre-hatch steelhead embryo survival in 72-day early life stage test (Dominquez and Chapman 1984).	Reduction in post-hatch steelhead survival in 72-day early life stage test (Dominquez and Chapman 1984).	Reduction in overall steelhead survival in 72-day early life stage test (Dominquez and Chapman 1984).
	P = 0.02; F = 19.0 MSE = 0.35 Adj.R^2 = 0.82 n = 5 Log input: 1.092 – 1.172 Log test: 0.786 – 1.801	P = 0.02; F = 25.5 MSE = 1.52 Adj.R^2 = 0.86 n = 5 Log input: 1.092 – 1.172 Log test: 0.786 – 1.801	P = 0.02; F = 18.0 MSE = 1.80 Adj.R^2 = 0.81 n = 5 Log input: 1.092 – 1.172 Log test: 0.786 – 1.801
Catostomidae	5.21 (0.00 – 17.4)	18.7 (0.00 – 68.3)	21.4 (0.00 – 77.6)
Listed species or family representative	Reduction in pre-eyed steelhead embryo survival in 30-day early life stage static test (Chapman 1969).	Reduction in steelhead embryo survival at hatch in 30-day early life stage static test (Chapman 1969).	Reduction in steelhead embryo survival at 10 mg O2/L (Chapman and Shumway 1978).
	P = 0.03; F = 10.6 MSE = 0.12 Adj.R^2 = 0.66 n = 6 Log input: 1.092 – 1.172 Log test: 0.000 – 2.255	P = 0.01; F = 30.1 MSE = 0.31 Adj.R^2 = 0.88 n = 5 Log input: 1.092 – 1.172 Log test: 1.000 – 2.255	P = 0.04; F = 26.8 MSE = 0.003 Adj.R^2 = 0.90 n = 4 Log input: 1.092 – 1.172 Log test: 0.000 – 1.602
Catostomidae	6.59 (0.00 – 15.5)	0.58 (0.15 – 2.20)	2.67 (1.45 – 4.01)

Listed species or family representative	Reduction in survival of early life stage fathead minnow exposed for 32 days at pH 7.5 (Spehar et al. 1985).	Reduction in survival of early life stage fathead minnow exposed for 32 days at pH 8.0 (Spehar et al. 1985).	Reduction in survival of early life stage fathead minnow exposed for 32 days at pH 8.5 (Spehar et al. 1985).
	P = 0.03; F = 29.2 MSE = 0.25 Adj.R^2 = 0.90 n = 4 Log input: 1.391 – 1.472 Log test: 1.121 – 2.111	P = 0.008 F = 131 MSE = 0.08 Adj.R^2 = 0.98 n = 4 Log input: 1.391 – 1.472 Log test: 1.494 – 2.503	P = 0.03; F = 30.8 MSE = 0.09 Adj.R^2 = 0.91 n = 4 Log input: 1.391 – 1.472 Log test: 1.429 – 2.337
Catostomidae	5.78 (0.00 – 29.4)	0.00 (0.00 – 6.89)	0.00 (0.00 – 10.7)

Listed species or family representative	Reduction in survival of early life stage fathead minnow exposed for 28 days (Arthur and Dixon 1994).
	P = 0.0000; F = 130 MSE = 0.05 Adj.R^2 = 0.95 n = 8 Log input: 1.391 – 1.472 Log test: 1.446 – 2.124
Catostomidae	0.96 (0.49 – 1.57)

Effects Summary

Based on the data presented above, exposure of Lost River sucker and shortnose sucker to PCP at the acute and chronic criterion concentrations will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality (PCE 1) in the Keno Reservoir. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur

once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

There could also be adverse effects to PCE 3 (food) from PCP through reductions in food supply because invertebrates, such as planktonic crustaceans (*Daphnia*) and chironomid larvae on which suckers feed could be adversely affected by PCP. Nevertheless, the effect of a reduced food supply, if there is an adverse effect, would likely be so small that its effect on the Lost River sucker or shortnose sucker would be insignificant because the reservoir is highly productive (Sullivan et al. 2011) and prey organisms would likely be entering the Keno Reservoir from upstream in Upper Klamath Lake, which is also highly productive (NRC 2004).

4.4.3.10 Effects of Silver on Lost River and Shortnose Suckers

Exposure Analysis

Silver enters the environment from various anthropogenic sources including: mining, industrial and smelting wastes, electroplating plants hazardous waste sites new scrap generated in the manufacture of silver-containing products; coin and bullion; and old scrap from electrical products, old film and photo processing wastes, batteries, jewelry, silverware, and bearings (Eisler 1996). Additionally, emissions from smelting operations, manufacture and disposal of certain photographic and electrical supplies, coal combustion, and cloud seeding are some of the anthropogenic sources of silver in the biosphere (Eisler 1996). In terms of estimated release of silver into the aquatic environment in the US, Eisler (1996) lists these sources in order of total amounts: soil erosion, urban runoff, sewage treatment facilities, and the photographic industry.

Silver concentrations in stormwater runoff are apparently generally below detection limits (~1 µg/L) because none was reported in a 1995 review of urban stormwater pollutants by Pitt et al. (1995) and also in a recent review of 10 years of data in USEPA's National Stormwater Quality Database comprising data from more than 200 municipalities by Pitt et al. (2004). Also, in a study of highway runoff by the Washington Department of Transportation, silver was not among the heavy metals detected (Herrera Environmental Consultants 2007; WADT 2007), nor was silver detected in runoff from highways in Oregon (Bloomquist 2009).

Within the range of the Lost River sucker and shortnose sucker, the presence of elevated silver levels in surface water is most likely to occur in the Keno Reservoir as a result of silver entering the Klamath Falls wastewater system.

Acute Effects Analysis

The acute silver criterion under review is 3.2 µg/L at 100 mg/L $CaCO_3$. Below in Table 4-49 are estimates of population level effects to Lost River sucker and shortnose sucker, based on the ICE model. Because the silver criterion of 3.2 µg/L is above the estimated LC50 concentration of 2.47 µg/L (both at 100 mg/L $CaCO_3$), we assume the population level of effect to Lost River sucker and shortnose sucker is between 50% and 100%, with the worst case being 100% loss, which we assumed would occur in the Keno Reservoir because of NPDES-permitted discharges into the reservoir.

Table 4-49. Level of effect range to endangered species from exposure to the acute silver criterion of 3.2 µg/L at 100 mg/L $CaCO_3$.

Species	Source of Data	LC50 µg/L at 100 mg/L $CaCO_3$	LC10 µg/L at 100 mg/L $CaCO_3$	Percent Population Effect
Lost River sucker	ICE	2.47[1]	1.09	Up to 100
Shortnose sucker	ICE	2.47[1]	1.09	Up to 100

Chronic Effects Analysis

The chronic silver criterion under review is 0.1 µg/L at 100 mg/L $CaCO_3$. Table 4-50 provides data on the level of effect for Catostomidae exposed to silver at the chronic criterion concentration, based on the surrogate species rainbow (steelhead) trout and fathead minnow. The data presented in Table 4-50 suggests that there could be a reduction in growth as a result of exposure to silver at the chronic criterion concentration. The results for rainbow (steelhead) trout may not provide the most accurate estimates, because salmonids and suckers are not closely related. Because minnows and suckers belong to the same order of bony fishes, the Cypriniformes, the fathead minnow data are likely to be the best surrogate for suckers. Based on this, we would anticipate that exposure to silver at the chronic criteria concentration will result in up to a 19% reduction in mass of suckers.

Table 4-50. Maximum likelihood estimate for percent level of effect (reduction in growth compared to control) and confidence limits based on early life stage fish exposure to silver.

Listed species or family representative	Reduction in mass (g) of steelhead trout, embryo-larval test, exposed to silver for 60 days (Nebecker et al. 1983).	Reduction in length (mm) of steelhead trout, embryo-larval test, exposed to silver for 60 days (Nebecker et al. 1983).	Reduction in fathead minnow mass (mg) in short term chronic test under high flow conditions for 7 days (Naddy et al. 2007)
	P = 0.02; F = 12.8 MSE = 0.00 Adj.R^2 = 0.70 n = 6 Log input: -0.612 – -0.407 Log test: -0.699 – 0.025	P = 0.01; F = 23.0 MSE = 0.00 Adj.R^2 = 0.82 n = 6 Log input: -0.612 – -0.407 Log test: -0.699 – 0.025	P = 0.05; F = 10.1 MSE = 0.001 Adj.R^2 = 0.70 n = 5 Log input: -1.275 – -0.256 Log test: -2.097 – 0.092
Catostomidae	21.1 8.82 – 31.8	99.3 99.2 – 99.4	19.2 8.09 – 29.0

Effects Summary

Based on the data presented above, acute and chronic exposure of Lost River sucker and shortnose sucker to silver at the criterion concentrations is likely to have pronounced adverse effects on suckers; our models suggest up to 100% mortality in Keno Reservoir every 3 years

over the 25-year life of the proposed action. This worst case scenario is likely an overestimate of the actual effects given that suckers are currently exposed to some level of silver in Keno Reservoir, and both Lost River and shortnose suckers persist in the reservoir.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality (PCE 1) in the Keno Reservoir. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

4.4.3.11 Effects of Zinc on Lost River and Shortnose Suckers

Effects of zinc toxicity to freshwater organisms include reduced growth, reduced populations, and reduced survival in algae; reduced growth, activity, larval settlement, and reproduction, osmoregulatory impairment and reduced survival in freshwater invertebrates (including molluscs); and reduced growth, behavioral alteration (avoidance), reproduction impairment, increased respiration, decreased swimming ability, increased jaw and branchial abnormalities, hyperactivity, hyperglycemia, and reduced survival in freshwater fish (Eisler 1993). Zinc bioconcentrates in freshwater fish up to 1,000 times (USEPA 2005b). Zinc may be mutagenic and teratogenic (Eisler 1993). Toxicity of zinc to aquatic organisms is dependent on water hardness, pH, DO, presence of mixtures, and trophic level (Sorensen 1991, Eisler 1993). Zinc interacts with many chemicals to produce altered patterns of accumulation, metabolism, and toxicity; some interactions reduce toxicity and others increase toxicity (Eisler 1993).

Exposure Analysis

Within their range in Oregon, we assume that Lost River suckers and shortnose suckers will only be affected by dissolved zinc as result of the proposed action in the Keno Reservoir because of the presence of two STPs and stormwater runoff that discharges into the reservoir. Zinc is often present in STP discharges but concentrations vary greatly and are mostly dependent on the concentrations in the water supply. According to one USEPA assessment (Yost et al. 1981), zinc concentrations in wastewater were found to be highly variable among cities and over time at the same STP, and can range from 0.2 to 2.4 mg/L. In a study done of metal concentrations in New York City wastewater, the majority of the zinc was from residential sources and surface runoff, representing 42% and 31% of the total, respectively (Klein et al. 1974). Zinc was detected in highway runoff by the Washington Department of Transportation with median dissolved concentrations equaling 39 µg/L (WADT 2007). In an assessment of runoff from highways in Oregon, dissolved zinc concentrations were found to range from 10 to 190 µg/L (Bloomquist 2009).

Acute Effects Analysis

The acute zinc criterion under review is 120 µg/L at 100 mg/L $CaCO_3$. Table 4-51 provides data on the acute level of effect for suckers based on the ICE model. The ICE model estimate of the LC50 for dissolved zinc for the Lost River sucker and shortnose sucker is 1.72 mg/L at 100 mg/L $CaCO_3$. Dividing 1.72 mg/L by 2.27 produces an LC10 estimate of 758 µg/L. Because the zinc criterion value of 120 µg/L is much lower than the LC10, being 16% of 758 µg/L, we assume that the proposed acute zinc criterion is not likely to adversely affect Lost River sucker and shortnose sucker (i.e., NLAA).

Table 4-51. Level of effect range to Lost River sucker and shortnose sucker from exposure to acute zinc criterion of 120 µg/L (at 100 mg/L $CaCO_3$) determined from the ICE model.

Species	Source of Data	LC50 (µg/L at 100 mg/L $CaCO_3$)	LC10 (µg/L at 100 mg/L $CaCO_3$)	Acute Criterion as Percent of LC10	Population Effect
Lost River sucker	ICE	1721	758	16	NLAA
Shortnose sucker	ICE	1721	758	16	NLAA

Chronic Effects Analysis

Reduction in Biomass

The chronic dissolved zinc criterion under review is 120 µg/L (at 100 mg/L as $CaCO_3$). Table 4-52 provides data on the level of effect for the rainbow trout (*Oncorhynchus mykiss*) and guppy (*Poecilia reticulata*) exposed to dissolved zinc. Neither the rainbow trout nor the guppy is closely related to suckers, since they belong to different fish orders; however, these are the best scientific data available to assess effects of zinc on early life stage growth of suckers. Because the rainbow trout results overlap zero and the R^2 value for the regression analysis is low, we would assume the effect, if any, is small if we relied on these data. However the data for the guppy showing the reduction in the second generation biomass and the reduction in biomass of female guppy in a 134-day test both show a strong relationship with zinc concentrations, thus, we assume that there is likely to be up to a 54% reduction in Lost River sucker and shortnose sucker larval/juvenile biomass in Keno Reservoir as a result of exposure to dissolved zinc at the chronic criterion concentration.

Table 4-52. Maximum likelihood estimate for percent level of effect (reduction in growth compared to control) and confidence limits based on early life stage fish exposure to zinc.

Listed species or family representative	Reduction in biomass (g) of rainbow trout (early life stage) exposed up to 63 days at 30 mg/L hardness (Brinkman and Hansen 2004)	Reduction in wet weight (mg) of second generation guppy *Poecilia reticulata* in 134-day test (Pierson 1981)	Reduction in wet weight (mg) of female guppy *Poecilia reticulata* in 134-day test (Pierson 1981)
	P = 0.03; F = 10.1 MSE = 0.008 Adj.R^2 = 0.65 n = 6 Input: 39.6 – 455 Test: 27.7 – 2,505	P = 0.005; F = 54.1 MSE = 0.001 Adj.R^2 = 0.93 n = 5 Log input: 2.688 – 3.749 Log test: 2.530 – 3.091	P = 0.04; F = 13.1 MSE = 0.003 Adj.R^2 = 0.75 n = 5 Log input: 2.688 – 3.749 Log test: 2.530 – 3.091
Catostomidae	10.3 0.00 – 32.4	54.0 40.1 – 64.6	48.4 22.0 – 65.9

Reduction in Survival

In Table 4-53 A thru D, below are shown a number of survival test results from exposure of various fish species to dissolved zinc. The results shown in Table 4-53A indicates a small level of effect (6.9% to 7.3%) for reduction in early life stage rainbow trout exposed to dissolved zinc. Based on this, we assume Lost River sucker and shortnose sucker in Keno Reservoir will experience up to a 10% reduction in survival. The results shown in Table 4-53B indicate a large level of effect for reduction in flagfish (*Jordanella floridae*) embryos that were previously exposed to dissolved zinc as well as those not previously exposed to dissolved zinc. Both of the tests show 54% to 97% reduction in embryo survival. The results shown in Table 4-53C below show level of effects of early life stage guppy and fathead minnow exposed to dissolved zinc for 5 to 7 days and up to 135 days. The mean levels of effect range from 16% to 90%. The results shown in Table 4-53D indicates a small level of effect (7.5%) for reduction in early life stage rainbow trout exposed to dissolved zinc in 30 mg/L hardness water starting at day 42 and lasting for 30 days. The other tests in Table 4-53D were done in 150 mg/L hardness water which is considerably higher than water in the Keno Reservoir, so those results are not applicable.

Table 4-53. Maximum likelihood estimate for percent level of effect (reduction in survival compared to control) and confidence limits based on early life stage fish exposure to zinc.
(Table A)

Listed species or family representative	Reduction in survival of rainbow trout, early life stage, exposed from egg to 69 days (Mebane et al. 2008)	SAC FRY TEST - Reduction in survival of rainbow trout sac fry (early life stage test starting with newly-eyed eggs, remained as sac fry for 26 days before swim up) exposed up to 63 days at 30 mg/L hardness (Brinkman and Hansen 2004)	SWIM UP FRY TEST - Reduction in survival of rainbow trout swim up fry (early life stage test starting with newly-eyed eggs, remained as sac fry for 26 days before swim up) exposed up to 63 days at 30 mg/L hardness (Brinkman and Hansen 2004)
	P = 0.02; F = 53.3 MSE = 0.02 Adj.R^2 = 0.95 N = 4 Input: 39.6 – 455 Test: 39.6 – 2,887	P = 0.002; F = 47.1 MSE = 0.13 Adj.R^2 = 0.90 N = 6 Input: 39.6 – 455 Test: 27.7 – 2.505	P = 0.007; F = 25.2 MSE = 0.14 Adj.R^2 = 0.83 N = 6 Input: 39.6 – 455 Test: 27.7 – 2.505
Catostomidae	7.51 0.00 – 19.1	6.92 2.14 – 13.9	7.31 0.00 – 17.8

(Table B)

Listed species or family representative	Reduction in 8-week survival of brook trout transferred to zinc 4 weeks after hatch (Holcombe et al. 1979).	Reduction in 100-day survival of flagfish, embryos previously exposed to zinc (Spehar 1976)	Reduction in 100-day survival of flagfish, embryos not previously exposed to zinc (Spehar 1976)
	P = 0.004; F = 28.6 MSE = 0.95 Adj.R^2 = 0.85 n = 6 Input: 167 – 1,923 Test: 513 – 3,962	P = 0.03; F = 16.8 MSE = 0.47 Adj.R^2 = 0.80 n = 5 Input: 135 – 1,548 Test: 55.4 – 528	P = 0.004; F = 34.9 MSE = 0.98 Adj.R^2 = 0.87 n = 6 Input: 135 – 1,548 Test: 19.8 – 528
Catostomidae	0.00 0.00 – 2.02	53.6 14.2 – 85.6	96.7 82.0 – 99.5

(Table C)

Listed species or family representative	Reduction in survival of early stage guppy exposed to zinc for 135 days (Pierson 1981).	Reduction in 5-day survival (renewal Test 4) of fathead minnows exposed to zinc (Norberg-King 1987)	Reduction in 7-day survival (renewal Test 5) of fathead minnows exposed to zinc (Norberg-King 1987)
	P = 0.02; F = 16.8 MSE = 0.08 Adj.R^2 = 0.84 n = 5 Log input: 2.688 – 3.749 Log test: 2.530 – 3.091	P = 0.02; F = 49.4 MSE = 0.96 Adj.R^2 = 0.94 n = 4 Log input: 2.270 – 3.330 Log test: 2.047 – 3.000	P = 0.009; F = 105 MSE = 0.43 Adj.R^2 = 0.97 n = 4 Log input: 2.270 – 3.330 Log test: 2.047 – 3.000
Catostomidae	16.4 3.32 – 37.8	86.5 31.7 – 98.9	90.1 61.3 – 98.1

(Table D)

Listed species or family representative	TROUT FRY TEST - Reduction in survival of rainbow trout fry (starting test at 42 days) exposed for 30 days at 30 mg/L hardness (Brinkman and Hansen 2004)	SWIM UP FRY TEST - Reduction in survival of rainbow trout swim up fry (early life stage test starting with newly-eyed eggs, remained as sac fry for 26 days before swim up) exposed up to 63 days at 150 mg/L hardness (Brinkman and Hansen 2004)	TROUT FRY TEST - Reduction in survival of rainbow trout fry (starting test at 42 days) exposed for 30 days at 150 mg/L hardness (Brinkman and Hansen 2004)
	P = 0.005; F = 32.1 MSE = 0.89 Adj.R^2 = 0.86 n = 6 Input: 39.6 – 455 Test: 7.09 – 1,120	P = 0.0009; F = 22.6 MSE = 0.04 Adj.R^2 = 0.81 n = 6 Input: 39.6 – 455 Test: 7.09 – 1,120	P = 0.03; F = 10.6 MSE = 0.67 Adj.R^2 = 0.66 n = 6 Input: 39.6 – 455 Test: 7.09 – 1,120
Catostomidae	7.51 0.00 – 19.1	7.13 3.89 – 10.9	24.6 4.58 – 50.9

Effects Summary

Based on the data presented above, we determined that acute exposure to dissolved zinc at the criterion concentration is not likely to adversely affect the adult life stages of Lost River sucker and shortnose sucker found in the Keno Reservoir. Chronic exposure to dissolved zinc would lead to death of embryos and larvae as described above, but because no known spawning occurs in the Keno Reservoir, there would be no effects to reproduction.

There could also be indirect effects of dissolved zinc on suckers, such as reductions in food supply because invertebrates, such as planktonic crustaceans on which suckers feed, could be adversely affected by dissolved zinc. The BE (USEPA 2008, p. 5-687) states that the LC10 for aquatic invertebrates could be as low as 10 µg/L, which is less than the acute criterion, so some mortality would be expected. Nevertheless, the effect of a reduced food supply, if there is an adverse effect, would probably be small and thus unlikely to adversely affect suckers because of the high productivity in the reservoir (Sullivan et al. 2011) and input of *Daphnia* from upstream in Upper Klamath Lake.

4.4.4 Effects of the Proposed Action on Proposed Critical Habitat for Lost River and Shortnose Suckers

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality (PCE 1) in the Keno Reservoir. Because Keno Reservoir (approximately 2,500 acres in area) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for both the Lost River sucker and shortnose sucker (USFWS 2011d), the effects are unlikely to substantially reduce the ability of proposed critical habitat to support the conservation needs of the Lost River sucker and the shortnose sucker. Furthermore, the adverse effects will be of short duration and we assume would occur once every 3 years, and thus over most of that time proposed critical habitat should continue to satisfy the essential requirements of the Lost River sucker and the shortnose sucker.

There could also be adverse effects to PCE 3 (food) from dissolved zinc through reductions in food supply because invertebrates, such as planktonic crustaceans (*Daphnia*) and chironomid larvae on which suckers feed could be adversely affected by dissolved zinc. The BE (USEPA 2008) states that the zinc LC10 for aquatic invertebrates could be as low as 10 µg/L, which is less than the acute criterion, so some mortality would be expected. Nevertheless, the effect of a reduced food supply, if there is an adverse effect, would likely be so small that its effect on the Lost River sucker or shortnose sucker would be insignificant because the reservoir is highly productive (Sullivan et al. 2011) and prey organisms would likely be entering the Keno Reservoir from upstream in Upper Klamath Lake, which is also highly productive (NRC 2004).

4.4.5 Effects of the Proposed Action on Oregon Chub

4.4.5.1 Potential Exposure of Oregon Chub to Toxics

The number of chub populations, the percent of critical habitat, and the percent of the rangewide population of the chub exposed per toxic are shown in Table 4-54 and the Oregon chub populations with potential exposure to the toxics at issue herein are listed in Table 4-55. These populations represent 73% of the rangewide Oregon chub population. These populations may be exposed to different toxics depending on their distance from NPDES permitted discharges, major roads, and agricultural fields. Chub populations located within 2 miles downstream of a NPDES-permitted discharge may be exposed to ammonia, arsenic, cadmium, chromium III, chromium VI, copper, lead, nickel, selenium, silver, and zinc. Chub populations located within 500 feet of major roads may be exposed to copper, lead, and zinc. Chub populations located adjacent to agricultural fields may be exposed to ammonia. For a more detailed analysis of routes of exposure see the Exposure Analysis in Section 4.1.

Table 4-54. Oregon chub-occupied sites subject to exposure to the toxics considered under the proposed action.

Toxic	No. of Exposed Sites	% of Critical Habitat Area Exposed	% of the Rangewide Chub Population Exposed
Ammonia	28	79	70.3
Arsenic	7	15	2.3
Cadmium	7	15	2.3
Chromium (III, VI)	7	15	2.3
Copper	13	23	4.4
Lead	13	23	4.4
Nickel	7	15	2.3
Selenium	7	15	2.3
Silver	7	15	2.3
Zinc	13	23	4.4

Table 4-55. Oregon chub-occupied sites that are subject to potential exposure to the toxics considered under the proposed action.

Sites	2011 Chub Abundance[1]	Acres of Habitat[2]	NPDES Permit w/in 2 mi. upstream	Adjacent Land Use	Toxics Exposure
SANTIAM RIVER SUBBASIN					
Cold Creek Slough	59	Unknown	No	Agriculture	NH^3
Gray Slough (2008 data)	660	**6.2**	No	Agriculture	NH^3
Green's Bridge Slough	370	3.1	No	Agriculture	NH^3
Harris Slough	18	Unknown	No	Agriculture	NH^3
Koenig Slough	443	Unknown	No	Agriculture	NH^3
Logan Slough	1	Unknown	No	Agriculture	NH^3
Santiam Easement	54	10	No	Agriculture	NH^3
Santiam I-5 Side Channels	280	**3.3**	No	Agriculture/Road	NH^3, Cu, Pb, Zn
MID-WILLAMETTE RIVER SUBBASIN					
Ankeny Willow Marsh	96810	**34.5**	No	Agriculture	NH^3
Dry Muddy Creek	20	Unknown	No	Agriculture	NH^3
Dunn Wetland	47350	**15.2**	No	Agriculture	NH^3
Ezell Slough	29	Unknown	No	Agriculture	NH^3
Finley Beaver Pond	310	Unknown	No	Agriculture	NH^3
Finley Cheadle Pond	30	**2.3**	No	Agriculture	NH^3
Finley Display Pond	487	**2.4**	No	Agriculture	NH^3
Finley Gray Creek Swamp	2150	**7.4**	No	Agriculture	NH^3
Hendrick's Bridge Slough	2	Unknown	No	Road	Cu, Pb, Zn
Hunsaker Slough	640	Unknown	No	Agriculture	NH^3
Muddy Creek - Wilkens Road	33	Unknown	Yes	Agriculture	Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn
Russell Pond	340	**0.1**	No	Agriculture	NH^3
Shetzline Pond	5750	**0.3**	No	Agriculture	NH^3
MIDDLE FORK WILLAMETTE RIVER SUBBASIN					
Buckhead Creek	1900	**9.3**	Yes	Forest/Road	Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn
Dexter Dam Slough	820	5.2	Yes	State Park	Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn
Dexter Alcove- PIT1	350	**0.3**	No	Road	Cu, Pb, Zn
Dexter RV Alcove- DEX3	940	**0.9**	No	Road	Cu, Pb, Zn
E.F. Minnow Creek Pond	2170	**3.3**	No	Road	Cu, Pb, Zn
Elijah Bristow Berry Slough	1040	**12.7**	No	Park/Agriculture	NH^3
Elijah Bristow NE Slough	670	**5.4**	Yes	State Park	Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn
Elijah Bristow N. Gravel Pit	3	Unknown	Yes	State Park	Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn
Elijah Bristow Island Pond	1400	**5.2**	Yes	State Park	Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn
Elijah Bristow South Slough	380	1	Yes	State Park	Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn
Shady Dell Pond	1760	**2.8**	No	Road	Cu, Pb, Zn
COAST FORK WILLAMETTE RIVER SUBBASIN					
Coast Fork Side Channels	80	8.8	No	Agriculture	NH^3

[1] *Bangs et al. 2011*
[2] *Acreages in bold represent designated Critical Habitat*

4.4.5.2 Quantifying Effects To Individuals

Oregon chub populations are sampled on an annual basis by ODFW. Table 4-56 shows the annual abundance per subbasin and rangewide over the last 5 years. Oregon chub populations naturally experience wide annual fluctuations in abundance. Between 2004 and 2010, annual rangewide chub abundance varied by up to 30% from the previous year. However, the rangewide abundance of Oregon chub in 2011 was the highest abundance we have observed rangewide and was an increase of 123% from 2010 chub abundance. Although several new chub populations were discovered in 2011, these populations only account for 1.2% of this increase from 2010 chub abundance.

We consider the magnitude of the 2011 increase in chub abundance as unusually high and anticipate that rangewide variation in chub abundance will return to the range we have seen in recent years. However, because this is the best available data for Oregon chub abundance, we have relied on the 2011 Oregon chub abundance data to estimate the effects of this proposed action on individuals (Bangs et al. 2011).

Table 4-56. Annual estimates of Oregon chub abundance in terms of numbers of individuals.

Subbasin	2007	2008	2009	2010	2011
Santiam	3,390	5,622	10,194	14,232	16,445
Mid-Willamette	66,263	90,297	89,398	62,728	162,555
Middle Fork Willamette	32,524	32,483	30,888	26,038	51,172
Coast Fork	262	145	126	445	304
Totals	102,439	128,547	130,606	103,443	230,476

For most of the toxics considered herein, multiple tests were available to estimate the reduction in Oregon chub survival, growth, and reproduction due to exposure to the proposed acute or chronic criterion concentration levels. The results of these tests are included in Appendix 5. Tests using surrogate species from the same family as the listed species were selected over tests using surrogate species from other families. If no tests were available using a surrogate species from the same family, we followed the data rules outlined in the Methods Section (Appendix 1) to select the best representative test for each species. In a few instances we made exceptions to following the data rules. These exceptions were based on professional judgment using the best available scientific and commercial data for the species. Any exceptions we made are explained below in the effects analysis for each toxic substance.

4.4.5.3 Effects of Ammonia on Oregon Chub

Proposed Acute Criterion for Ammonia

The acute ammonia criterion under review is 5.6 mg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 concentration for ammonia at 9.43 mg/L. Because the

acute criterion concentration is 59% of the LC10 concentration, we expect that fewer than 10% of Oregon chub juveniles exposed to the proposed acute criterion for ammonia are likely to be killed. However, we are unable to determine the exact portion of the Oregon chub population that will be affected. Therefore, we are giving the species the benefit of the doubt and assuming that an acute criterion that is 50% or more of the LC10 will result in a mortality rate of 10%. The combined 2011 estimate of adult abundance for the 28 exposed Oregon chub populations is 162,447 chub (Table 4-56), so we estimate that 16,245 adult Oregon chub are likely to die as a result of acute exposure to ammonia per 3-year period over the 25-year life of the proposed action. However, Oregon chub populations have the reproductive capacity to overcome a 10% decrease in abundance every 3 years. Additionally, a 10% decrease in abundance is not likely to be detectible due to the wide annual fluctuations in abundance that naturally occur in Oregon chub populations.

Proposed Chronic Criterion for Ammonia

The proposed chronic criterion concentration for ammonia is 10.8 mg/L (at 100 mg/L as $CaCO_3$).

Effects on Juvenile Survival

The best test to estimate reduction in survival of individual chub determined that chub exposure to the proposed chronic criterion concentration of ammonia is likely to cause the death of 10.8% (95% confidence limits = 0.12% to 22.3%) of exposed juvenile chub (Mayes et al. 1986). Data are not available to estimate the ratio of juveniles to adults in Oregon chub populations. Accurate juvenile Oregon chub data are difficult to collect because of the individual's small size, secretive nature, and ability to hide within pond flora and substrate. The best and most reliable data that we have access to are estimates of adult abundance per population. Although our effects models were done to gauge effects to juvenile Oregon chub, it is widely believed that juveniles are far more sensitive and susceptible to the influences of toxins than are adults. As we have no reliable data for juvenile abundance we will apply our effects data to the adult Oregon chub populations. Although this approach may overestimate the effects of the proposed action on the adult Oregon chub population, we do so in keeping with the Act's mandate to "give the species the benefit of the doubt" when the best available scientific and commercial data are not adequate to address an effect (USFWS and NMFS 1998).

Oregon chub are likely to be exposed to chronic levels of ammonia in 28 populations. The combined 2011 estimate of adult abundance for these populations is 162,447 chub. Thus, based on 2011 abundance estimates, 17,544 adult Oregon chub are likely to be killed from exposure to chronic levels of ammonia per 3-year period over the 25-year life of the proposed action. However, due to the wide annual fluctuation in abundance that naturally occurs in Oregon chub populations, a 10.8% reduction in survival every 3 years would not be detectible and is unlikely to cause a decline in the adult chub population over the 25-year term of the proposed action.

Effects on Juvenile Growth

We made an exception to the data rules in selecting the best test for reduction in growth. Tests using cyprinids were not available to estimate the reduction in growth. Among the several tests available, the most reliable tested smallmouth bass (family Centrarchidae) reduction in growth measured as mean length versus wet mass. Measures of length are appropriate for the Oregon

chub because growth rate studies are available for the Oregon chub based on length; we have no data for growth rates measured as mass. The growth reduction test was available at a range of pH levels (6.6 and 7.25); since both fall within the range of pH found in chub habitats, we selected the test with the highest F-value per our data rules.

The best test to estimate reduction in juvenile growth determined that exposure to the proposed chronic criterion concentration for ammonia would result in a reduction in growth (measured as mean total length) of 1.57% of the exposed individuals (95% confidence limits = 0.63% to 2.51%) (Broderius et al. 1985). A reduction in growth could increase vulnerability of young fish to a host of potential stressors including inter- and intraspecific competition for food which could lead to injury or death due to reduced food intake. Reduced growth rates in individuals could also cause delayed maturity resulting in reduced fitness and a reduced population growth rate.

Among Oregon chub populations, age and growth can vary substantially (Scheerer and McDonald 2003). For instance, juveniles collected in July at Hospital Pond averaged a 0.30 mm increase in length per day and at East Fork Minnow Pond averaged a 0.26 mm increase in length per day; thus, a 14% difference in growth rates was observed. Given the naturally high variation in growth rates among Oregon chub populations, a 1.57% reduction in growth of exposed juveniles is unlikely to cause death or injury or other detectable impairment of affected individuals and is therefore considered insignificant.

Reduced Embryo Hatching Success

The best test to estimate the reduction in Oregon chub embryo hatching success determined that Oregon chub embryo exposure to the chronic criterion concentration of ammonia would result in a 17.9% reduction in embryo hatchability (95% confidence limits = 7.22% to 28.0%) (Thurston et al. 1986). In other words, about 18% of exposed chub embryos are not likely to hatch every 3 years over the 25-year term of the proposed action.

As stated in the exposure analysis for the Oregon chub, we anticipate that exposure to chronic levels of ammonia will not occur due to agricultural or forestry runoff during the spawning and hatching season. Therefore, Oregon chub embryos at sites with runoff potential will not be exposed to chronic levels of ammonia. However, embryos could be exposed to acute and chronic levels of ammonia at seven sites within 2 miles downstream of permitted point source discharges (Table 4-56).

Data are not available to estimate the number of embryos or embryo hatch rates in Oregon chub populations. However, hatch rates for other cyprinids have been estimated. In a study of the effects of electrofishing on embryos of three cyprinid species, Bohl et al. (2009) found embryo survival to hatching in the control groups ranged from 74% to 93%. In a study of the effects of copper on reproduction in fathead minnows, Pickering et al. (1977) found hatchability of eggs exposed to various levels of copper ranged from 87% to 97% and reported this was within the normal range of hatchability for unexposed eggs. It is reasonable to assume that the range of Oregon chub embryo hatching rates is similar to other cyprinids (Doug Markle, Fisheries Professor, Oregon State University, Corvallis, Oregon, pers. comm., 2011). Given these ranges, a 17.9% reduction in hatching success would not be detectible because it is likely to be within

the range of the naturally occurring variation in hatching success for Oregon chub. Thus, although we know seven populations with a total abundance of 5,206 Oregon chub are at risk for exposure to chronic levels of ammonia and that embryo hatching success is likely to be reduced every 3 years as a result, the magnitude of that effect cannot be estimated or detected.

Other Effects

There may be reductions in food supply if invertebrates are affected by exposure to ammonia. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Ammonia

Acute Ammonia Exposure

Acute exposure to ammonia is likely to result in the death of 10% of the Oregon chub in exposed populations every 3 years over the 25-year term of the proposed action; based on 2011 abundance estimates this would equate to 16,245 adult Oregon chub every 3 years.

Chronic Ammonia Exposure

Chronic exposure to ammonia is likely to result in the death of 10.8% of the Oregon chub in exposed populations every 3 years over the 25-year term of the proposed action; based on 2011 abundance estimates this would equate to 17,544 adult Oregon chub every 3 years. Additionally, in seven populations located within 2 miles downstream from point source discharges, 17.9% of embryos are likely to be killed due to their exposure to ammonia at the chronic criterion concentration every 3 years for the 25-year term of the proposed action.

4.4.5.4 Effects of Arsenic on Oregon Chub

Proposed Acute Criterion for Arsenic

The acute arsenic criterion under review is 340 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for arsenic is 4,625 µg/L. Because the acute criterion is only 7.4% of the LC10, we expect that effects from exposure to arsenic at this concentration are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Proposed Chronic Criterion for Arsenic

The chronic arsenic criterion under review is 150 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Juvenile Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of arsenic would result in a 6.83% (95% confidence limits = 0.36% to 15.2%)

reduction in juvenile fish (Gilderhus 1966). As described on page 285, we will apply our effects data to the adult populations.

Seven Oregon chub populations could be exposed to chronic levels of arsenic (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Thus, based on 2011 abundance estimates, 356 adult Oregon chub are likely to be killed from exposure to chronic levels of arsenic per 3-year period over the 25-year life of the proposed action. However, Oregon chub populations have the reproductive capacity to overcome a 6.83% decrease in abundance every 3 years. Additionally, a 6.83% reduction in survival is not likely to be detectible due to the wide annual fluctuations in abundance that naturally occur in Oregon chub populations.

Effects on Embryo Growth

The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of arsenic is not likely to reduce embryo growth (mean length) in Oregon chub (Lima et al. 1984).

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by arsenic. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Arsenic

Acute Arsenic Exposure

Effects to Oregon chub from acute exposure to arsenic are extremely unlikely to occur are therefore discountable (i.e., NLAA).

Chronic Arsenic Exposure

Chronic exposure to arsenic is likely to result in the death of 6.83% of the Oregon chub in exposed populations per 3-year period over the 25-year life of the proposed action; based on 2011 abundance estimates this would equate to 356 adult Oregon chub every 3 years (i.e., LAA with take).

4.4.5.5 Effect of Cadmium on Oregon Chub

Proposed Acute Criterion for Cadmium

The acute cadmium criterion under review is 2.0 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for cadmium is 78.4 µg/L. Because the acute criterion is

only 2.6% of the LC10, we expect that effects from exposure to cadmium at this concentration are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Proposed Chronic Criterion for Cadmium

The chronic cadmium criterion under review is 0.25 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Embryo Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of cadmium would not result in a reduction in survival (Spehar and Carlson 1984).

Effects on Growth

The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of cadmium would not result in a reduction in growth (Sauter et al. 1976).

Reduced Fecundity and Hatching Success

Two tests using fathead minnows as a surrogate species were available to estimate reduction in reproduction; one measured reduced fecundity and the other measured reduction in hatching success. We selected both tests.

The best tests to estimate reduction in fecundity and hatching success determined that exposure to the chronic criterion concentration of cadmium would not result in a reduction in fecundity (Pickering and Gast 1972) and would not result in a reduction in hatching success (Birge et al. 1985).

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by cadmium. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Cadmium

Acute Cadmium Exposure

Effects to Oregon chub from acute exposure to cadmium are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Chronic Cadmium Exposure

Chronic exposure to arsenic will not result in effects to Oregon chub (i.e., No Effect).

4.4.5.6 Effects of Chromium III on Oregon Chub

Proposed Acute Criterion for Chromium III

The acute chromium III criterion under review is 570 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for chromium III is 1,644 µg/L. Because the acute criterion is only 35% of the LC10, we expect that exposure to acute levels of chromium III will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death.

Proposed Chronic Criterion for Chromium III

The chronic chromium III criterion under review is 74 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Embryo Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of chromium III would result in a 4.93% (95% confidence limits = 2.86% to 8.38%) reduction in survival of embryos and young chub (Pickering 1980b). As described on page 285 we will apply our effects data to the adult populations.

Seven Oregon chub populations could be exposed to chronic levels of chromium III (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Thus, based on 2011 abundance estimates, 257 adult Oregon chub are likely to be killed from exposure to chronic levels of chromium III per 3-year period over the 25-year life of the proposed action. Oregon chub populations have the reproductive capacity to overcome this decrease in abundance every 3 years; therefore, we do not expect exposure to chromium III at the chronic criterion concentration will result in a reduction in population growth over the 25-year term of the proposed action. Additionally, a 4.93% reduction in survival is not likely to be detectible due to the wide annual fluctuations in abundance that naturally occur in Oregon chub populations.

Effects on Growth

We made an exception to the data rules in selecting the best test for reduction in growth. Several tests using fathead minnow as a surrogate species were available to estimate the reduction in growth of Oregon chub due to exposure to the chronic criterion level of chromium III. Two of these tests measured reduction in growth as mean length versus wet weight. Measures of length are more appropriate for Oregon chub because growth rate studies are available for Oregon chub based on length. Among these two tests we selected the test which resulted in the lowest MSE value and the highest F-value per our data rules.

The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of chromium III would result in a 3.35% reduction in growth of age one or younger fish (95% confidence limits = 0.00% to 8.96%) (Pickering 1980b). Exposure to chronic levels of chromium III could increase vulnerability of young fish to a host of potential stressors including inter- and intraspecific competition for food which could lead to injury or death due to reduced food intake. Reduced growth rates in individuals could also cause delayed maturity resulting in reduced fitness and a reduced population growth rate.

Among Oregon chub populations, age and growth can vary substantially (Scheerer and McDonald 2003). For instance, juveniles collected in July at Hospital Pond averaged a 0.30 mm increase in length per day and at East Fork Minnow Pond averaged a 0.26 increase in length per day; thus, a 14% difference in growth rates was observed. Additionally, growth rates varied 27% to 30% between summer and fall. Although this data is based on measured length, we can assume that the natural variation in growth rates measured as mass would be similar. Given the naturally high variation in growth rates among Oregon chub populations, a 3.35% reduction in growth of juveniles is unlikely to cause death or injury or other detectable impairment of affected individuals and is therefore considered insignificant.

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by chromium III. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Chromium III

Acute Chromium III Exposure

Exposure to acute levels of chromium III every 3 years will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death (i.e., LAA with no take).

Chronic Chromium III Exposure

Chronic exposure to chromium III is likely to result in the death of 4.93% of the Oregon chub in exposed populations every 3 years; based on 2011 abundance estimates this would equate to 257 adult Oregon chub every 3 years (i.e., LAA with take) over the 25-year term of the proposed action.

4.4.5.7 Effects of Chromium VI on Oregon Chub

Proposed Acute Criterion for Chromium VI

The acute chromium VI criterion under review is 16 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for chromium VI is 11,273 µg/L. Because the acute criterion is only 0.1% of the LC10, we expect that effects from exposure to chromium VI at this concentration are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Proposed Chronic Criterion for Chromium VI

The chronic chromium VI criterion under review is 11 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Embryo Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of chromium VI would result in a 1.77% (95% confidence limits = 0.65% to 3.62%) reduction in survival of embryos and young chub (Pickering 1980a). As described on page 285 we will apply our effects data to the adult populations.

Seven Oregon chub populations could be exposed to chronic levels of chromium VI (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Thus, based on 2011 abundance estimates, 92 adult Oregon chub are likely to be killed from exposure to chronic levels of chromium VI per 3-year period over the 25-year life of the proposed action. Oregon chub populations have the reproductive capacity to overcome this decrease in abundance every 3 years; therefore, we do not expect exposure to chromium VI at the chronic criterion concentration will result in a reduction in population growth over the 25-year term of the proposed action. Additionally, a 1.77% decrease in abundance is not likely to be detectible due to the wide annual fluctuations in abundance that naturally occur in Oregon chub populations.

Effects on Growth

The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of chromium VI would result in a 2.12% reduction in growth of age one or younger fish (95% confidence limits = 0.00% to 6.61%) (Olson and Foster 1956). Exposure to chronic levels of chromium VI could increase vulnerability of young fish to a host of potential stressors including inter- and intraspecific competition for food which could lead to injury or death due to reduced food intake. Reduced growth rates in individuals could also cause delayed maturity resulting in reduced fitness and a reduced population growth rate.

Among Oregon chub populations, age and growth can vary substantially (Scheerer and McDonald 2003). For instance, juveniles collected in July at Hospital Pond averaged a 0.30 mm increase in length per day and at East Fork Minnow Pond averaged a 0.26 increase in length per day; thus, a 14% difference in growth rates was observed. Additionally, growth rates varied 27% to 30% between summer and fall. Although this data is based on measured length, we can assume that the natural variation in growth rates measured as mass would be similar. Given the naturally high variation in growth rates among Oregon chub populations, a 2.12% reduction in growth of juveniles is unlikely to cause death or injury or other detectable impairment of affected individuals and is therefore considered insignificant.

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by chromium VI. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Chromium VI

Acute Chromium VI Exposure

Effects to Oregon chub from acute exposure to chromium VI are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Chronic Chromium VI Exposure

Chronic exposure to chromium VI is likely to result in the death of 1.77% of the Oregon chub in exposed populations every 3 years; based on 2011 abundance estimates this would equate to 92 adult Oregon chub per 3-year period over the 25-year life of the proposed action (i.e., LAA with take).

4.4.5.8 Effects of Copper on Oregon Chub

Proposed Acute Criterion for Copper

The acute copper criterion under review is 13 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for copper is 52 µg/L. Because the acute criterion is only 25% of the LC10, we expect that exposure to acute levels of copper will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death (i.e., LAA with no take).

Proposed Chronic Criterion for Copper

The chronic copper criterion under review is 9 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Embryo Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of copper would result in a 2.47% (95% confidence limits = 1.21% to 4.42%) reduction in survival of embryos and young chub (Lind et al. 1978). As described on page 285, above, we will apply our effects data to the adult populations.

Thirteen Oregon chub populations could be exposed to chronic levels of copper (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 10,708 chub. Thus, based on 2011 abundance estimates, 264 adult Oregon chub are likely to be killed from exposure to chronic levels of copper per 3-year period over the 25-year life of the proposed action. Oregon chub populations have the reproductive capacity to overcome this decrease in abundance every 3 years; therefore, we do not expect exposure to copper at the chronic criterion concentration will result in a reduction in population growth over the 25-year term of the proposed action. Additionally, a 2.47% decrease in abundance is not likely to be detectible due to the wide annual fluctuations in abundance that naturally occur in Oregon chub populations.

Effects on Growth

We made an exception to the data rules in selecting the best test for reduction in growth. Several tests using fathead minnow as a surrogate species were available to estimate the reduction in growth of Oregon chub due to exposure to the chronic criterion level of copper. Among the

several tests available, two measured reduction in growth as mean length (one based on male length and the other on female length) versus wet mass. Measures of length are appropriate for Oregon chub because growth rate studies are available for Oregon chub based on length; we have no data for growth rates measured as mass. Since these tests measured length in opposite sexes, we have selected both of these tests to estimate reduction in growth.

The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of copper would result in a 0.75% to 2.93% reduction in growth of age one or younger fish (95% confidence limits = 0.00% to 8.06% and 0.00% to 7.77% respectively) (Mount and Stephen 1969). Exposure to chronic levels of copper could increase vulnerability of young fish to a host of potential stressors including inter- and intraspecific competition for food which could lead to injury or death due to reduced food intake. Reduced growth rates in individuals could also cause delayed maturity resulting in reduced fitness and a reduced population growth rate.

Among Oregon chub populations, age and growth can vary substantially (Scheerer and McDonald 2003). For instance, juveniles collected in July at Hospital Pond averaged a 0.30 mm increase in length per day and at East Fork Minnow Pond averaged a 0.26 increase in length per day; thus, a 14% difference in growth rates was observed. Additionally, growth rates varied 27% to 30% between summer and fall. Given the naturally high variation in growth rates among Oregon chub populations, a 2.93% reduction in growth of juveniles is unlikely to cause death or injury or other detectable impairment of affected individuals and is therefore considered insignificant.

Effects on Fecundity

Multiple tests using fathead minnow as a surrogate species were available to estimate the reduction in number of eggs per spawning event in Oregon chub. All of these tests are based on the same study that treated eggs to different durations of copper exposure. Therefore, we selected all of these tests to show a range of levels of effect from chronic exposure to copper.

The best test to estimate reduction in fecundity determined that exposure to the chronic criterion concentration of copper would result in a 20.9% to 46.2% reduction in fecundity in Oregon chub depending on previous exposure to copper (Pickering et al. 1977).

Pearsons (1989) estimated fecundity of Oregon chub ranging from 147 eggs (41 mm TL) to 671 eggs (64 mm TL), but fecundity rates are likely to vary widely by local conditions and annual climate variability. We lack data to estimate the population level effects to Oregon chub due to reduced fecundity. However, studies of other cyprinids suggest that population effects caused by reduced fecundity at the anticipated levels may result in extirpation of exposed Oregon chub populations. Banks et al. (2010) studied different levels of reductions in fecundity in fathead minnows to determine the threshold level above which populations become extinct. The threshold for fathead minnow was approximately 18%. Although we are unable to estimate the effect this will ultimately have on a chub population, we assume they will respond similarly to fathead minnows. Therefore, given a 20.9% to 46.2% reduction in fecundity, Oregon chub populations exposed at chronic levels of copper may experience significantly lower growth rates

and be at risk of extirpation. However, given that this level of exposure may occur only once every 3 years and the rapid increases in abundance we have observed in populations, we anticipate that Oregon chub populations exposed to chronic levels of copper will be able to recover from effects to reproduction.

Other Effects

Olfactory Sensory Neuron (OSN) receptors serve an important function in fish. These structures are important in feeding, mating and predator response behaviors (Tierney et al. 2010). Copper exposure may interfere with olfaction in Oregon chub leading to an inability to find mates or to gather environmental clues affecting predator avoidance and other movements including avoidance of adverse water quality. Copper is known to have impacts on OSN receptors in relatively small concentrations. Copper concentrations of 10 µg per liter reduced responses in coho salmon (*O. kisutch*). A 30-minute exposure to copper at this concentration found a reduction of approximately 70% in the response value from the pre-exposure value for the same fish. The same study found that recovery from this exposure was not rapid with the response value improving to only approximately 60% of the pre-exposure level after 5 hours exposure to flowing clean water (Tierney et al. 2010).

Based on recent estimates of chub abundance, we assume that approximately 10,708 mature Oregon chub may be exposed to chronic levels of copper. Although we do not know how many chub will respond negatively to exposure or the extent of injury, it is possible that some will experience interference with olfaction resulting in an increased likelihood of being killed by predators, reduced reproduction due to inability to find mates, reduced growth or an increase in deaths caused by reduced ability to find food.

Other potentially adverse effects include reductions in food supply if invertebrates are affected by copper. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Copper

Acute Copper Exposure

Exposure to acute levels of copper will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death (i.e., LAA with no take).

Chronic Copper Exposure

Chronic exposure to copper is likely to result in the death of 2.47% of the Oregon chub in exposed populations every 3 years; based on 2011 abundance estimates this would equate to 264 adult Oregon chub per 3-year period over the 25-year life of the proposed action. Additionally,

an unknown number of Oregon chub will experience a 20.9% to 46.2% reduction in fecundity every 3 years (i.e., LAA with take).

4.4.5.9 Effects of Lead on Oregon Chub

Proposed Acute Criterion for Lead

The acute lead criterion under review is 65 µg/L (at 100 mg/L $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for lead is 755 µg/L. Because the acute criterion is only 8.6% of the LC10, we expect that effects from exposure to lead at this concentration are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Proposed Chronic Criterion for Lead

The chronic lead criterion under review is 2.5 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of lead would not result in a reduction in survival (Sauter et al. 1976).

Reduced hatching success and increase in deformities

Multiple tests were available to estimate the reduction in reproductive success. According to our data rules for selecting the most representative test, we selected two tests which resulted in the lowest MSE value and the highest F-value. Both tests were based on the same study using common carp (*Cyprinus carpio*) as a surrogate species. One test measured reduced hatching success and the other measured the increase in hatchling deformities.

The best tests to estimate reduction in hatching success and increase in deformities determined that exposure to the chronic criterion concentration would result in a 73.9% reduction in hatching success (95% confidence limits = 60.6% to 83.4%) and a 38.2% increase in hatchling deformities, primarily in the spinal column (95% confidence limits = 32.7% to 43.7%) (Kapur and Yadav 1982).

Thirteen Oregon chub populations could be exposed to chronic levels of lead (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 10,708 chub. Data are not available to estimate the number of embryos or embryo hatch rates in Oregon chub populations. However, a 73.9% reduction in hatching success is likely to be well outside the range of the naturally occurring variation in hatch rates. Thus, although we cannot estimate the numbers of embryos affected, we expect that embryo hatching success and survival may be severely reduced as a result of chronic exposure to lead. However, given that this level of exposure may occur only once per 3-year period over the 25-year life of the proposed action and the rapid increases in abundance we have observed in populations, we anticipate that Oregon chub populations exposed to chronic levels of lead will be able to recover from effects to reproduction.

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by lead. Many invertebrates are able to tolerate toxicity levels higher than the proposed

standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Lead

Acute Lead Exposure

Effects to Oregon chub from acute exposure to lead are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Chronic Lead Exposure

Chronic exposure to lead is likely to cause 73.9% of embryos to be killed and 38.2% of embryos to experience an increase in deformities per 3-year period over the 25-year life of the proposed action (i.e., LAA with take).

4.4.5.10 Effects of Nickel on Oregon Chub

Proposed Acute Criterion for Nickel

The acute nickel criterion under review is 468 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for nickel is 1,151 µg/L. Because the acute criterion is only 40.7% of the LC10, we expect that exposure to acute levels of nickel will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death (i.e., LAA with no take).

Proposed Chronic Criterion for Nickel

The chronic nickel criterion under review is 52 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Embryo Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of nickel would result in a reduction in embryo survival in cyprinids by 38.4% (95% confidence limits = 14.8% to 65.1%) (Nebecker et al. 1984). As described on page 285, above, we will apply our effects data to the adult populations.

Seven Oregon chub populations could be exposed to chronic levels of nickel (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Thus, based on 2011 abundance estimates, 1,999 adult Oregon chub are likely to be killed from exposure to chronic levels of nickel per 3-year period over the 25-year life of the proposed action. Oregon chub populations have the reproductive capacity to overcome this decrease in abundance every 3 years; therefore, we do not expect exposure to nickel at the chronic criterion concentration will result in a reduction in population growth over the 25-year term of the proposed action.

Effects on Embryo and Juvenile Growth

We made an exception to the data rules in selecting the best test for reduction in growth. Two tests using rainbow trout as a surrogate species were available to estimate the reduction in growth of Oregon chub due to exposure to the chronic criterion level of nickel. We selected the test that measured reduction in growth as mean length versus the test that measured wet mass. Measures of length are appropriate for Oregon chub because growth rate studies are available for Oregon chub based on length; we have no data for growth rates measured as mass.
The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of nickel would result in a reduction in early life stage growth by 8.74% (95% confidence limits = 0.00% to 17.1%) (Nebeker et al. 1985). A reduction in growth could decrease embryo survival and increase vulnerability to a host of potential stressors including inter- and intraspecific competition for food which could lead to injury or death due to reduced food intake. Reduced growth rates in individuals could also cause delayed maturity resulting in reduced fitness and a reduced population growth rate.

Among Oregon chub populations, age and growth can vary substantially (Scheerer and McDonald 2003). For instance, juveniles collected in July at Hospital Pond averaged a 0.30 mm increase in length per day and at East Fork Minnow Pond averaged a 0.26 increase in length per day; thus, a 14% difference in growth rates was observed. Additionally, growth rates varied 27% to 30% between summer and fall. Given the naturally high variation in growth rates among populations, an 8.74% reduction in growth of Oregon chub embryos and juveniles could not be detected and is unlikely to have population level effects. However, we are giving the species the benefit of the doubt and assuming that a reduction in growth greater than 5% is likely to significantly disrupt breeding, feeding, or sheltering behavior or cause injury or death of affected individual Oregon chub.

Oregon chub embryos and juveniles could be exposed to chronic levels of nickel in seven populations. The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Although we cannot estimate the number of juveniles or detect the reduction in growth, we anticipate that all Oregon chub embryos and juveniles in the exposed populations will experience a small reduction in growth.

Effects on Fecundity

The tests we selected for reduction in fecundity determined that exposure to the chronic criterion concentration of nickel would result in a 25.6% (95% confidence limits = 3.42% to 45.2%) reduction in egg production (Pickering 1974).

Pearsons (1989) estimated fecundity of Oregon chub ranging from 147 eggs (41 mm TL) to 671 eggs (64 mm TL), but fecundity rates are likely to vary widely by local conditions and annual climate variability. We lack data to estimate the population level effects to Oregon chub due to reduced fecundity. However, studies of other cyprinids suggest that population effects caused by reduced fecundity at this level may be significant. Banks et al. (2010) studied different levels of reductions in fecundity in fathead minnows to determine the threshold level above which populations become extinct. The threshold for fathead minnow was approximately 18%. Although we are unable to estimate the effect this will ultimately have on a chub population, we

assume they will respond similarly to fathead minnows. Therefore, given a 25.6% reduction in fecundity, Oregon chub populations exposed at chronic levels of nickel are likely to experience significantly lower growth rates. However, given that this level of exposure may occur only once every 3 years and the rapid increases in abundance we have seen in translocated populations, we anticipate that Oregon chub populations will be able to recover from effects to reproduction.

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by nickel. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Nickel

Acute Nickel Exposure

Exposure to acute levels of nickel will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it may cause injury or death (i.e., LAA with no take).

Chronic Nickel Exposure

Chronic exposure to nickel is likely to result in the death of 38.4% of the Oregon chub every 3 years in exposed populations; based on 2011 abundance estimates this would equate to 1,999 adult Oregon chub per 3-year period over the 25-year life of the proposed action. Additionally, in exposed sites, all embryos and juveniles are likely to be affected by an 8.74% reduction in growth and an unknown number of Oregon chub will experience a 25.6% reduction in fecundity every 3 years (i.e., LAA with take).

4.4.5.11 Effects of Pentachlorophenol on Oregon Chub

Wood treatment facilities are a source of pentachlorophenol in waterways. However, none of the known Oregon chub populations occur within 2 miles downstream of a permitted discharge from a wood treatment facility. Thus, we expect that Oregon chub will not be exposed to pentachlorophenol (i.e., No Effect).

4.4.5.12 Effects of Silver on Oregon Chub

Proposed Acute Criterion for Silver

The acute silver criterion under review is 3.2 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for silver is 2.81 µg/L. Based on these values, we estimate that Oregon chub mortality due to exposure to acute concentrations of silver will be between 10% to 50%. Because we are unable to determine the exact portion of the Oregon chub

population that will be affected, we estimated the range of expected mortalities in exposed populations.

Seven Oregon chub populations may be exposed to silver (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Therefore, we estimate that between 520 and 2,603 adult Oregon chub are likely to die as a result of acute exposure to silver per 3-year period over the 25-year life of the proposed action (i.e., LAA with take). Oregon chub populations have the reproductive capacity to overcome this decrease in abundance every 3 years; therefore, we do not expect exposure to silver at the acute criterion concentration will result in a reduction in population growth over the 25-year term of the proposed action.

Proposed Chronic Criterion for Silver

The chronic silver criterion under review is 0.10 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Embryo Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of silver would result in a 20.6% (95% confidence limits = 2.96% to 48.9%) reduction in survival of young fish (Naddy et al. 2007). As described on page 285, above, we will apply our effects data to the adult populations.

Seven Oregon chub populations could be exposed to chronic levels of silver. The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Thus, based on 2011 abundance estimates, 1,072 Oregon chub are likely to be killed from exposure to chronic levels of silver per 3-year period over the 25-year life of the proposed action. Oregon chub populations have the reproductive capacity to overcome this decrease in abundance every 3 years; therefore, we do not expect exposure to silver at the chronic criterion concentration will result in a reduction in population growth over the 25-year term of the proposed action.

Effects on Embryo and Juvenile Growth

The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of silver would result in reduced growth (measured as mass) of 14.4% (95% confidence limits = 5.92% to 22.7%) (Naddy et al. 2007). A reduction in growth could increase vulnerability of young fish to a host of potential stressors including inter- and intraspecific competition for food which could lead to injury or death due to reduced food intake. Reduced growth rates in individuals could also cause delayed maturity resulting in reduced fitness and a reduced population growth rate.

The only data we have available on growth rates in Oregon chub measured growth as length. Among Oregon chub populations, age and growth can vary substantially (Scheerer and McDonald 2003). For instance, juveniles collected in July at Hospital Pond averaged a 0.30 mm increase in length per day and at East Fork Minnow Pond averaged a 0.26 increase in length per day; thus, a 14% difference in growth rates was observed. Additionally, growth rates varied 27% to 30% between summer and fall. Although this data is based on measured length, we can assume that the natural variation in growth rates measured as mass would be similar. Given the naturally high variation in growth rates among populations, a 14.4% reduction in growth of

juvenile Oregon chub could not be detected and is unlikely to have population level effects. However, we are giving the species the benefit of the doubt and assuming that a reduction in growth greater than 5% is likely to significantly disrupt breeding, feeding, or sheltering behavior or cause injury or death of affected individual Oregon chub.

Seven Oregon chub populations could be exposed to chronic levels of silver. The combined 2010 estimate of adult abundance for these populations is 5,206 chub. Although we cannot estimate the number of juveniles or detect the reduction in growth, we anticipate that all juvenile Oregon chub in these seven exposed sites will be affected by a 14.4% reduction in growth rates and may be more susceptible to injury or death.

Reduced Embryo Hatching Success

The best test to estimate reduction in reproductive success determined that cyprinids exposed to silver at the criterion concentration would experience a 4.34% reduction in embryo hatching per 3-year period over the 25-year life of the proposed action (95% confidence limits = 0.00% to 15.0%) (Naddy et al. 2007).

Data are not available to estimate the number of embryos or embryo hatch rates in Oregon chub populations. However, hatch rates for other cyprinids have been estimated. In a study of the effects of electrofishing on embryos of three cyprinid species, Bohl et al. (2009) found embryo survival to hatching in the control groups ranged from 74% to 93%. In a study of the effects of copper on reproduction in fathead minnows, Pickering et al. (1977) found hatchability of eggs exposed to various levels of copper ranged from 87% to 97% and reported this was within the normal range of hatchability found in their laboratory. It is reasonable to assume that the range of Oregon chub embryo hatching rates is similar to other cyprinids (D. Markle, pers. comm., 2011). Given these ranges, a 4.34% reduction in hatching success would not be detectible because it is within the range of the naturally occurring variation in hatching success for Oregon chub. Thus, although we cannot estimate the numbers of embryos affected, we expect that embryo hatching success and survival will be reduced as a result of chronic exposure to silver. However, given that this level of exposure may occur only once every 3 years and the rapid increases in abundance we have observed in translocated populations, we anticipate that Oregon chub populations will be able to recover from effects to reproduction.

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by silver. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Silver

Acute Silver Exposure

Acute exposure to silver is likely to result in the death of between 520 and 2,603 adult Oregon chub per 3-year period over the 25-year life of the proposed action (i.e., LAA with take).

Chronic Silver Exposure

Chronic exposure to silver is likely to result in the death of 20.6% of the Oregon chub every 3 years in exposed populations; based on 2011 abundance estimates this would equate to 1,072 adult Oregon chub per 3-year period over the 25-year life of the proposed action. Additionally, in exposed sites, all juveniles are likely to experience a 14.4% reduction in growth and 4.34% of embryos are likely to be killed per 3-year period over the 25-year life of the proposed action (i.e., LAA with take).

4.4.5.13 Effects of Zinc on Oregon Chub

Proposed Acute Criterion for Zinc

The acute zinc criterion under review is 120 µg/L (at 100 mg/L as $CaCO_3$). The ICE model (family Cyprinidae) estimates the LC10 for zinc is 1,154 µg/L. Because the acute criterion is only 10.4% of the LC10, we expect that effects from exposure to zinc at this concentration are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Proposed Chronic Criterion for Zinc

The chronic zinc criterion under review is 120 µg/L (at 100 mg/L as $CaCO_3$).

Effects on Embryo Survival

The best test to estimate reduction in survival determined that exposure to the chronic criterion concentration of zinc would result in a 5.95% (95% confidence limits = 0.30% to 28.7%) reduction in survival of young fish (Norberg-King 1987). As previously described, we will apply our effects data to the adult populations.

Thirteen Oregon chub populations could be exposed to chronic levels of zinc (Table 4-56). The combined 2011 estimate of adult abundance for these populations is 10,708 chub. Thus, based on 2011 abundance estimates, 637 Oregon chub are likely to be killed from exposure to chronic levels of zinc per 3-year period over the 25-year life of the proposed action. Oregon chub populations have the reproductive capacity to overcome this decrease in abundance every 3 years; therefore, we do not expect exposure to zinc at the chronic criterion concentration will result in a reduction in population growth over the 25-year term of the proposed action. Additionally, a 5.95% decrease in abundance is not likely to be detectible due to the wide annual fluctuations in abundance that naturally occur in Oregon chub populations.

Effects on Growth

We made an exception to the data rules in selecting the best test for reduction in growth. Two tests using a cyprinid, the guppy (*Poecilia reticulata*), as a surrogate species were available to

estimate the reduction in growth of Oregon chub due to exposure to the chronic criterion level of zinc. Although measures of length are more appropriate for Oregon chub because growth rate studies are available for Oregon chub based on length, not mass, both of these tests measured reductions in growth rates as mass. We selected the test with the lowest MSE value and the highest F-value per our data rules.

The best test to estimate reduction in growth determined that exposure to the chronic criterion concentration of zinc would result in reduced growth (measured as mass) of 11.8% (95% confidence limits = 0.76% to 21.6%) (Pierson 1981). A reduction in growth could increase vulnerability of young fish to a host of potential stressors including inter- and intraspecific competition for food which could lead to injury or death due to reduced food intake. Reduced growth rates in individuals could also cause delayed maturity resulting in reduced fitness and a reduced population growth rate.

Among Oregon chub populations, age and growth can vary substantially (Scheerer and McDonald 2003). For instance, juveniles collected in July at Hospital Pond averaged a 0.30 mm increase in length per day and at East Fork Minnow Pond averaged a 0.26 increase in length per day; thus, a 14% difference in growth rates was observed. Additionally, growth rates varied 27% to 30% between summer and fall. Although this data is based on measured length, we can assume that the natural variation in growth rates measured as mass would be similar. Given the naturally high variation in growth rates among populations, an 11.8% reduction in growth of juvenile Oregon chub could not be detected and is unlikely to have population level effects. However, we are giving the species the benefit of the doubt and assuming that a reduction in growth greater than 5% is likely to significantly disrupt breeding, feeding, or sheltering behavior or cause injury or death of affected individual Oregon chub.

Thirteen Oregon chub populations could be exposed to chronic levels of zinc. The combined 2010 estimate of adult abundance for these populations is 10,708 chub. Although we cannot estimate the number of juveniles or detect the reduction in growth, we anticipate that all juvenile Oregon chub in these seven exposed sites will be affected by an 11.8% reduction in growth rates and may be more susceptible to injury or death.

Reduced Embryo Hatching Success

The best test to estimate reduction in reproductive success determined that cyprinids exposed to zinc at the criterion concentration would experience a 0.03% reduction in embryo hatching (95% confidence limits = 0.00% to 1.66%) (Holcombe et al. 1979). We would not be able to measure or detect this level of effect and therefore consider it to be insignificant.

Other Effects

Other potentially adverse effects include reductions in food supply if invertebrates are affected by zinc. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly

recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

Summary of Effects to Oregon Chub caused by Exposure to Proposed Acute and Chronic Criteria for Zinc

Acute Zinc Exposure

Effects to Oregon chub from acute exposure to zinc are extremely unlikely to occur and are therefore discountable (i.e., NLAA).

Chronic Zinc Exposure

Chronic exposure to zinc is likely to result in the death of 5.95% of the Oregon chub every 3 years in exposed populations; based on 2011 abundance estimates this would equate to 637 adult Oregon chub per 3-year period over the 25-year life of the proposed action. Additionally, in exposed sites, all juveniles are likely to experience an 11.8% reduction in growth per 3-year period over the 25-year life of the proposed action (i.e., LAA with take).

4.4.5.14 Aggregated Effects of the Proposed Action on the Oregon Chub

Proposed Acute Criteria

If the Oregon chub were simultaneously exposed to all of the proposed acute standard concentrations for the toxic substances addressed in the proposed action, there is the likelihood that up to 18,849 adult Oregon chub will be killed per 3-year period over the 25-year term of the proposed action (Table 4-57). Additionally, there may be negative effects from acute exposure to chromium III, copper, and nickel, but as explained in our analysis for these toxic substances, we do not believe that they will cause reduced fitness or mortality of affected Oregon chub.

Proposed Chronic Criteria

If the Oregon chub were simultaneously exposed to all of the proposed chronic standard concentrations for the toxic substances addressed in the proposed action, there is the likelihood that up to 23,266 adult Oregon chub will be killed and 5,206 adult Oregon chub will have reduced fitness due to reductions in growth and impaired reproduction per 3-year period over the 25-year life of the proposed action (Table 4-58).

Aggregate Effects of the Proposed Action on the Oregon Chub

The above effects are likely to cause the extirpation of six Oregon chub populations in the Middle Fork Willamette River subbasin and one population in the Mid-Willamette River subbasin because these populations do not have the capacity to compensate for the combined levels of mortality, reduced fitness, and impaired reproduction caused by the proposed action over its 25-year term. However, the abundance, distribution, and viability of Oregon chub populations which are not at risk of exposure to the toxic substances addressed herein are adequate to allow the species to meet the delisting criteria and to achieve recovery (USFWS 1998c)

4.4.6 Effects of the Proposed Action on Oregon Chub Critical Habitat

The percentages of exposed critical habitat are shown in Table 4-55 and the Oregon chub critical habitat sites that may be exposed to the chemicals analyzed in this Biological Opinion are shown in Table 4-56. In evaluating project effects on critical habitat, we must determine whether the primary constituent elements (PCE) of the critical habitat are likely to be adversely affected. The PCEs for Oregon chub are described in the Status of the Species. The PCE's for Oregon chub do not include specific water quality criteria, but PCE #3 describes an abundance of minute organisms (such as rotifers, copepods, cladocerans, and chironomid larvae) as a characteristic of habitat necessary to sustain the essential life history functions of the species. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub critical habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term.

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Table 4-57. Summary of adverse effects to the Oregon chub caused by acute exposure to the toxic substances at issue in this consultation.

Subbasin	Ammonia	Arsenic	Cadmium	Chromium III	Chromium VI	Copper	Lead	Nickel	Selenium	Silver	Zinc	Mortalities per 3 Year Period	Percent of Population Affected
Santiam River	189	D	D	A	D	A	D	A	D	0	D	189	1.1%
Mid-Willamette River	15,435	D	D	A	D	A	D	A	D	17	D	15,452	9.5%
Middle Fork Willamette River	621	D	D	A	D	A	D	A	D	2,587	D	3,208	6.3%
Coast Fork Willamette	0	D	D	A	D	A	D	A	D	0	D	0	0
Totals by Chemical	16,245	0	0	0	0	0	0	0	0	2,604	0	18,849	8.2%

A = Adverse effects that do not biologically conform to the statutory and regulatory definitions of take.
D = Effects that are discountable.

Table 4-58. Summary of adverse effects to the Oregon chub caused by chronic exposure to the toxic substances at issue in this consultation.

Subbasin	No. of Oregon chub	Type of Effect (L= Lethal) (RF = Reduced Fitness)	Ammonia	Arsenic	Cadmium	Chromium III	Chromium VI	Copper	Lead	Nickel	Selenium	Silver	Zinc	Mortalities per 3 Year Period	Reduced Fitness Total per 3 Year Period	Percent of Population Affected
Santiam River	16,445	L	204	0	0	0	0	7	0	0	0	0	17	228		1.4%
		RF	0	0	0	0	0	0	0	0	0	0	0		0	0%
Mid-Willamette River	162,555	L	16,670	2	0	2	1	1	0	13	7	7	2	16,705		10.3%
		RF	33	0	0	0	0	33	33	33	0	33	33		33	0.02%
Middle Fork Willamette River	51,172	L	671	353	0	255	92	257	0	1,986	1,035	1,066	618	6,333		12.4%
		RF	5,173	0	0	0	0	5,173	05,173	5,173	0	5,173	5,173		5,173	10.1%
Coast Fork Willamette River	304	L	0	0	0	0	0	0	0	0	0	0	0	0		0%
		RF	0	0	0	0	0	0	0	0	0	0	0		0	0%
Totals Rangewide	230,476	L	17,545	355	0	257	93	258	0	1,999	1,042	1,073	637	23,266		10.1%
		RF	5,206	0	0	0	0	5,206	5,206	5,206	0	5,206	5,206		5,206	2.3%

4.4.7 Effects of the Proposed Action on Vernal pool fairy shrimp

4.4.7.1 Potential Exposure to Toxics

This section evaluates the potential for vernal pool fairy shrimp exposure to 15 toxics at the freshwater acute and chronic water quality criteria. The vernal pool fairy shrimp in Oregon may have limited exposure to chromium VI, copper, lead, PCP, and zinc. They are not likely to be exposed to alpha-endosulfan, aluminum, ammonia, arsenic, beta-endosulfan, cadmium, chromium III, dieldrin, endrin, gamma-BHC (Lindane), heptachlor epoxide, nickel, selenium, silver, and tributyltin.

We are aware of four sites within the range of vernal pool fairy shrimp in Oregon that may be exposed to toxic residues; these include a historic landfill, a historic motor pool, an off-road vehicle recreational area, and an area where log deck debris treated with PCP was buried. The four sites are adjacent to vernal pool habitat, but these pools are not necessarily occupied by vernal pool fairy shrimp. All of the sites have undergone development to some degree so that the expressions of vernal pool characteristics are weak. We estimate that 272 acres of the 7,574 total acres of vernal pool fairy shrimp habitat in Oregon may be exposed to toxics. This is approximately 3.6% of the known habitat in Oregon. We lack data to estimate the number of vernal pool fairy shrimp in populations, but expect that exposure to this amount of habitat would expose less than 1% of the population rangewide.

Table 4-59. Vernal pool fairy shrimp Sites with potential toxic exposure

Toxic	No. of Exposed Sites	% of habitat	Acres of habitat exposed	% of vernal pool fairy shrimp rangewide population exposed
Chromium (VI)	1	0.4	33	< 1
Copper	3	1.2	92	< 1
Lead	2	1.2	92	< 1
PCP	1	2.8	213	< 1
Zinc	2	0.8	59	< 1

4.4.7.2 Effects of Chromium VI on Vernal Pool Fairy Shrimp

Exposure to Chromium VI

Although it has not been documented in vernal pools, chromium VI contamination could occur within 33 acres or less than 1% of vernal pool habitat in Oregon. The area with potential exposure follows:

Former Landfill Site

A 33-acre historic solid waste landfill site occurs within degraded habitat. This area was used by Camp White and the City of Medford from 1930 until 1962 and was used for

disposal of automobiles, machinery equipment, wood mill ash, wood mill waste, and other miscellaneous garbage (ODEQ 2012a). More recently a large food processing plant was built on the much of the former landfill (David Monnin, Terra Science, Inc., Portland, Oregon, pers. comm., 2012). Although detections of lead were reported from site testing, no tests for chromium VI were conducted (ODEQ 2012a). Chromium VI could potentially occur at the site from batteries associated with buried machinery equipment, appliances, and electrical components. Vernal pools in the surrounding areas are very sparse, but none have been observed lacking aquatic invertebrates or having a lack of algal plant matter. Vernal pools adjacent to the historic landfill site did have vernal pool fairy shrimp presence (D. Monnin, pers. comm., 2012). If this site is experiencing exposure to chromium VI, the continued presence of vernal pool fairy shrimp may indicate the species' resilience to this toxic.

Proposed Acute Criterion for Chromium VI

The acute chromium VI criterion under review is 16 µg/L (at 100 mg/L as $CaCO_3$). Using an SSD model to predict vernal pool fairy shrimp mortality, we estimated an LC50 of 108 µg/L and an LC10 of 47.6 µg/L. Because the acute standard is only 33.6% of the LC10, we expect that exposure to acute levels of chromium VI will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it will cause injury or death (i.e., LAA with no take).

Proposed Chronic Criterion for Chromium VI

The chronic chromium VI criterion under review is 11 µg/L (at 100 mg/L as $CaCO_3$).

Reduction in Survival

The best modeling test that was available in our analysis indicates that the surrogate test subject, *Daphnia magna,* would have up to a 7.78% (95% confidence limits = 0.00% – 39.4%) reduction in 7-day survival when exposed to chronic criteria standards (Spehar and Fiandt 1985). We expect that vernal pool fairy shrimp exposed to chronic concentrations of chromium VI would experience a similar response.

Reduction in Fecundity

The best modeling test indicates that the surrogate test subject, *Ceriodaphnia dubia*, would have 92.3% (95% confidence limits = 71.6% – 100%) reduction in number of young per adult (Munzinger and Monicelli 1991). We expect that vernal pool fairy shrimp exposed to chronic concentrations of chromium VI would experience a similar response.

Summary of Effects to Vernal Pool Fairy Shrimp caused by Exposure to Proposed Acute and Chronic Criteria for Chromium VI

Less than 1% of the vernal pool fairy shrimp rangewide will be exposed to chromium VI.

Acute criteria

We expect that exposure to acute levels of chromium VI will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it will cause injury or death. (i.e., LAA with no take).

Chronic Exposure

Exposure to chromium VI at the chronic criterion levels will result in up to 7.78% mortality and up to a 92.3% reduction in number of young per adult per 3-year period during the 25-year term of the proposed action (i.e., LAA with take).

4.4.7.3 Effects of Copper on Vernal Pool Fairy Shrimp

Exposure to Copper

Although it has not been documented in vernal pools, copper contamination could occur within 92 acres in Oregon. The areas with potential exposure follow:

Off-road Vehicle Sports Park

The 56-acre Jackson County Sports Park is comprised of heavily used dirt-tracks and is accessed by ORVs for recreational purposes. Historic agricultural and vehicle use degraded the functionality of much of the habitat at the site, even prior to listing of vernal pool fairy shrimp. The majority of the site no longer functions as a vernal pool complex, but two large potholes on one of the entrance roads were occupied by fairy shrimp in 2010 (D. Monnin, pers. comm., 2012). Copper from off-road vehicle brake pads and leaking vehicle fluids has the potential to enter the remaining habitat at the site.

Camp White Motor Pool Site

A criteria amount of copper may be present in the soils on 3 acres adjacent to the former motor pool and dumping site at the former Army training base, Camp White, which was in use from 1942 to 1952 (USFWS 2002g, pp. 3 and 6) (Figure 4-9). The White Mountain Middle School was constructed on the former motor pool site in 2003. Copper could potentially have entered the adjacent vernal pool complex from runoff (TDC Environmental, LLC 2004) while Camp White was in operation in the 1940s and 1950s. For this reason, sources of copper could be connected to the historic motor pools and dumping associated with former Camp White. All observed pools in the area do support aquatic invertebrates and algal matter. Although vernal pool fairy shrimp presence at the site is confirmed, surveys have not been conducted since 2002.

Former Landfill Site

A 33-acre former solid waste landfill site occurs within a degraded vernal pool complex (Figure 4-9). This area was used by Camp White and the City of Medford from 1930 until 1962 and was used for disposal of automobiles, machinery equipment, wood mill ash, wood mill waste, and other miscellaneous garbage (ODEQ 2012a). More recently a large food processing plant was built on the much of the former landfill (D. Monnin, pers. comm., 2012). Although detections of lead were reported from site testing, no tests for

copper were conducted (ODEQ 2012a). Copper could potentially occur at the site from buried machinery equipment, vehicle parts, and electrical wire. Log-deck waste was also spread at the site in 1993 and covered with additional fill. Vernal pools in the surrounding areas are very sparse, but none have been observed lacking aquatic invertebrates or having a lack of algal plant matter.

Other potential copper sources

Copper sulfate was once used as a fungicide on agriculture in the Middle Rogue River Valley (Randy White, District Manager, Jackson Soil and Water Conservation District, Medford, Oregon, pers. comm., 2011). This practice was abandoned approximately 10 years ago in favor of more effective chemicals such as Prosaro and Stratego (Brian Wolf, Service Technician, Helena Chemical Company, Medford, Oregon, pers. comm., 2011; R. White, pers. comm., 2011). Ranching and hay production are the primary uses of agricultural lands in the Middle Rogue River Valley and these practices do not require use of copper sulfate. Therefore; no copper exposure is anticipated to affect vernal pool fairy shrimp in habitat adjacent to agricultural areas in the Middle Rogue River Valley (R. White, pers. comm., 2011).

The presence of log-deck debris has been reported as a potential contamination issue for vernal pool fairy shrimp (Borgias et al. 2009, pp. 13-14). We investigated whether buried pressure-treated wood products containing a copper formulation could have been buried at the log-deck debris sites within the Denman Wildlife Area. We determined that pressure treated wood products are not known to be disposed within vernal pool fairy shrimp habitat at sites of reported log-deck debris disposal areas (V. Oredson, pers. comm., 2011; R. White, pers. comm., 2011).

Proposed Acute Criterion for Copper

The acute copper criterion under review is 13 µg/L (at 100 mg/L as $CaCO_3$). Using an SSD model to predict vernal pool fairy shrimp mortality, we estimated an LC50 of 3.28 µg/L and an LC10 of 1.4 µg/L. Because the acute standard is almost four times the LC50, we expect that more than 50% of the exposed vernal pool fairy shrimp population will be killed by exposure to copper at the acute criterion (i.e., LAA with take).

Proposed Chronic Criterion for Copper

The chronic copper criterion under review is 9 µg/L (at 100 mg/L as $CaCO_3$).

Reduction in Survival

The best modeling test estimated within 95% confidence that Daphniidae would have up to a 3.63% (0.66% – 11.2%) reduction in survival when exposed to chronic levels of copper (Cooper et al. 2009). We expect that vernal pool fairy shrimp exposed to chronic concentrations of copper would experience a similar response.

Reduction in Fecundity

The best modeling test estimated within 95% confidence that Daphniidae would have up to a 12.7% (0% to 50.4%) reduction in fecundity when exposed to chronic levels of copper (Spehar and Fiandt 1985). We expect that vernal pool fairy shrimp exposed to chronic concentrations of copper would experience a similar response.

Summary of Effects to Vernal Pool Fairy Shrimp caused by Exposure to Proposed Acute and Chronic Criteria for Copper

Less than 1% of vernal pool fairy shrimp rangewide will be exposed to copper.

Acute copper Exposure

We estimate that more than 50% of the exposed populations of vernal pool fairy shrimp are likely to be killed per 3-year period during the 25-year term of the proposed action due to acute criterion levels of copper (i.e., LAA with take).

Chronic Copper Exposure

Exposure to copper at the chronic criterion levels will result in up to 3.63% mortality and up to a 12.7% reduction in number of young per adult per 3-year period during the 25-year term of the proposed action (i.e., LAA with take).

4.4.7.4 Effects of Lead on Vernal Pool Fairy Shrimp

Exposure to Lead

Due to the historical patterns of land use within vernal pool habitat in the Rogue River Valley, there may be transient lead exposure to 92 acres due to nonpoint sources. The sites with potential exposure include habitat adjacent to the historical landfill, the historical motor pool, and the off-road vehicle recreation sites previously discussed.

Proposed Acute Criterion for Lead

The acute lead water standard criterion under review is 65 µg/L (at 100 mg/L of $CaCO_3$). Using an SSD model to predict vernal pool fairy shrimp mortality, we estimated an LC50 of 94 µg/L and an LC10 of 41 µg/L. Because the acute criterion falls between the LC10 and LC50, we expect that exposure to lead at this concentration would result in 10% to 50% mortality in vernal pool fairy shrimp populations (i.e., LAA with take).

Proposed Chronic Criterion for Lead

The chronic lead criterion under review is 2.5 µg/L (at 100 mg/L of $CaCO_3$).

Reduction in Survival

The best modeling test estimated within 95% confidence that Daphniidae would have up to a 16.5% (6.31 to 35.5 %) reduction in survival when exposed to chronic lead criteria levels (Cooper et al. 2009, p. 1527). We expect that vernal pool fairy shrimp exposed to chronic concentrations of lead would experience a similar response.

Reduction in Fecundity

The best modeling test estimated within 95% confidence that Daphniidae would have up to a 70.5% (57.0% – 81.5%) reduction in fecundity when exposed to chronic levels of lead (Cooper et al. 2009, p. 1527). We expect that vernal pool fairy shrimp exposed to chronic concentrations of lead would experience a similar response.

Summary of Effects to Vernal Pool Fairy Shrimp caused by Exposure to Proposed Acute and Chronic Criteria for Lead

Less than 1% of vernal pool fairy shrimp rangewide will be exposed to lead.

Acute Lead Exposure

We estimate that between 10% and 50% of the exposed populations of vernal pool fairy shrimp are likely to be killed per 3-year period during the 25-year term of the proposed action due to acute criterion levels of lead (i.e., LAA with take).

Chronic Lead Exposure

Exposure to lead at the chronic criterion levels will result in up to 16.5% mortality and up to a 70.5% reduction in number of young per adult per 3-year period during the 25-year term of the proposed action (i.e., LAA with take).

4.4.7.5 Effects of Pentachlorophenol on Vernal Pool Fairy Shrimp

Exposure to Pentachlorophenol

Pentachlorophenol (PCP) is a chlorinated hydrocarbon that is one of the most heavily used pesticides in the U.S. In the Rogue Valley area it was used primarily as an algicide and fungicide for wood products (Chris Galpin, Galpin Properties, White City, Oregon, pers. comm., 2011; ODEQ 2012b). We have no historical or current references that document vernal pool fairy shrimp have been affected by PCP in any vernal pool system, but vernal pools appear to have a darker hue near historic wood debris disposal sites (Borgias et al. 2009).

The vernal pool fairy shrimp have limited exposure to PCP, in the form of wood preservatives that were buried in specific areas in White City (C. Galpin, pers. comm., 2011; ODEQ 2012b). We are aware that several known wood product disposal sites were treated with PCP in Oregon's Rogue Valley within the range of vernal pool fairy shrimp. The area includes former log deck sites, burial of wood scrap from historic Camp White and White City mills, an old landfill, and from various former wood mills. These known wood disposal sites occur within vernal pool habitat that may not support vernal pool fairy shrimp currently. Based on aerial signatures and GIS mapping, we estimate that there may be PCP exposure to 213 acres or 2.8% of vernal pool fairy shrimp habitat in the Rogue River Valley due to these nonpoint sources (Borgias et al. 2009). These areas are all in the vicinity of White City, Oregon. PCP contamination could occur within the 33-acre historic landfill site previously discussed and the following:

Bark and Lumber Disposal Sites

The presence of log-deck debris has been reported as a potential contamination issue for vernal pool fairy shrimp (Borgias et al. 2009, pp. 13-14). Fairly high concentrations of PCP were found within soils at a privately owned property adjacent to the Denman Wildlife Area (ODEQ 2012b). Much of the 180 acres within this historic vernal pool complex has been leveled (Borgias et al. 2009). We determined that treated wood products were known to be disposed in the leveled areas within former vernal pool habitat (V. Oredson, pers. comm., 2011; Borgias et al. 2009, pp. 13-14). Where vernal pools occur within the disposal sites, the pool water is often noticeably

darker. However, this is potentially attributed to leaching of wood tannins, which is common at many other vernal pool locations (Darren Borgias, Stewardship Ecologist, The Nature Conservancy, Medford, Oregon, pers. comm., 2012). All pools surveyed within the disposal site areas in 2003 contained both macro invertebrates and algae (Borgias 2006). The vernal pool fairy shrimp have been observed at these properties, but only in a limited area where pools are still intact.

Proposed Acute Criterion for PCP

The acute PCP criterion under review is 19 µg/L (at pH 7.8). Using an SSD model (family Daphniidae) to predict vernal pool fairy shrimp mortality, we estimated an LC50 of 120 µg/L and an LC10 of 52.86 µg/L. Because the proposed acute criterion is only 36% of the LC10 value, we expect that exposure to acute levels of PCP will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it will cause injury or death (i.e., LAA with no take).

Proposed Chronic Criterion for PCP

The chronic PCP water standard criterion under review is 150 µg/L (at pH 7.8).

Reduction in Survival

The best data modeling test indicated that chronic PCP concentrations up to 504 µg/L did not affect *Ceriodaphnia affinis/dubia* survival (Hedtke et al. 1986). We anticipate no reduction in survival to vernal pool fairy shrimp, where they are exposed to PCP.

Reduction in Fecundity

The best modeling test estimated within 95% confidence that Daphniidae would have up to a 54.7% (32.6% to 72.3%) reduction in fecundity when exposed to chronic levels of PCP (Hedtke et al. 1986). We expect that vernal pool fairy shrimp exposed to chronic concentrations of PCP would experience a similar response.

Summary of Effects to Vernal Pool Fairy Shrimp caused by Exposure to Proposed Acute and Chronic Criteria for PCP

Less than 1% of vernal pool fairy shrimp rangewide will be exposed to PCP.

Acute PCP Exposure

We expect that exposure to acute levels of PCP will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it will cause injury or death (i.e., LAA with no take).

Chronic PCP Exposure

Exposure to PCP at the chronic criterion levels will result in up to a 54.7% reduction in number of young per adult per 3-year period during the 25-year term of the proposed action (i.e., LAA with take).

4.4.7.6 Effects of Zinc on Vernal Pool Fairy Shrimp

Although zinc sources do occur in the Rogue River Valley area, none are anticipated to have runoff that could enter vernal pool habitat. We are aware of two sites that may have zinc residues from vehicle fuel combustion sources. These sites include the historic motor pool from Camp White and the off-road recreational park previously discussed. These two sites are adjacent to vernal pool habitat that has already experienced some disturbance and may not support vernal pool fairy shrimp currently. We estimate that there may be zinc exposure to 59 acres or less of vernal pool fairy shrimp habitat in Oregon due to nonpoint sources.

Proposed Acute Criterion for Zinc

The acute zinc criterion under review is 120 µg/L (at 100 mg/L as $CaCO_3$). Using an SSD model (family Daphniidae) to predict vernal pool fairy shrimp mortality, we estimated an LC50 of 61.8 µg/L and an LC10 of 27.2 µg/L. Because the acute standard is almost twice the LC50, we expect that between 50% and 100% of the exposed vernal pool fairy shrimp population will be killed by exposure to zinc at the acute criterion (i.e., LAA with take).

Proposed Chronic Criterion for Zinc

The chronic zinc criterion under review is 120 µg/L (at 100 mg/L as $CaCO_3$). Based on literature, we have not been able to confirm or deny that exposure to the chronic zinc criterion would affect vernal pool fairy shrimp. Consequently, we will give the benefit of the doubt to the species and assume that chronic levels of zinc will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it will cause injury or death (i.e., LAA with no take).

Summary of Effects to Vernal Pool Fairy Shrimp caused by Exposure to Proposed Acute and Chronic Criteria for Zinc

Less than 1% of vernal pool fairy shrimp rangewide will be exposed to zinc.

Acute Zinc Exposure

We estimate that between 50% and 100% of the exposed populations of vernal pool fairy shrimp are likely to be killed per 3-year period during the 25-year term of the proposed action due to acute criterion levels of zinc (i.e., LAA with take).

Chronic Zinc Exposure

We expect that exposure to acute levels of zinc will cause negative effects that may rise to the level of adverse effects, but will not rise to the level where it will cause injury or death (i.e., LAA with no take).

4.4.7.7 Summary of Effects to Vernal Pool Fairy Shrimp caused by Exposure to Proposed Acute and Chronic Criteria for All Toxics

If exposure to these five toxics were to occur simultaneously, the additive effect would be the loss of all vernal pool fairy shrimp populations associated with up to 2.8% of Vernal Pool habitat in Oregon (Table 4-60); this amounts to less than 1% of vernal pool fairy shrimp rangewide.

Therefore, effects from the proposed action are not likely to considerably reduce the survival and recovery of the species.

Table 4-60. Summary of adverse effects to vernal pool fairy shrimp

Toxin	Percent of habitat in Oregon exposed	Percent of Population Affected		
		Acute Exposure Mortalities	Chronic Exposure Mortalities	Chronic Exposure Reduced Fitness
Chromium VI	<1%	<10%	7.78%	92.3%
Copper	1.2%	>50%	3.63%	12.7%
Lead	1.2%	10% to 50%	16.5%	70.5%
PCP	2.8%	<10%	0%	54.7%
Zinc	<1%	50% to 100%	0%	0

4.4.8 Effects of the Proposed Action on Vernal Pool Fairy Shrimp Critical Habitat

All of the sites with potential exposure to toxics are within designated critical habitat for vernal pool fairy shrimp. One PCE (#3) for vernal pool fairy shrimp critical habitat could be affected by the five toxics. PCE #3 is described as:

> Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding.

According to the exposure analysis we predict that approximately 272 acres or 3.5% of vernal pool fairy shrimp critical habitat are likely to be exposed to toxics. Due to the permeable soils and isolated hydrology of the habitat, the sources of water in the pools that derive from precipitation, and no field evidence of pools lacking aquatic invertebrates and algae, we consider that there is a low potential for vernal pool fairy shrimp food sources to be reduced substantially by exposure to toxics. Therefore, we anticipate a very low level of adverse effects to vernal pool fairy shrimp critical habitat.

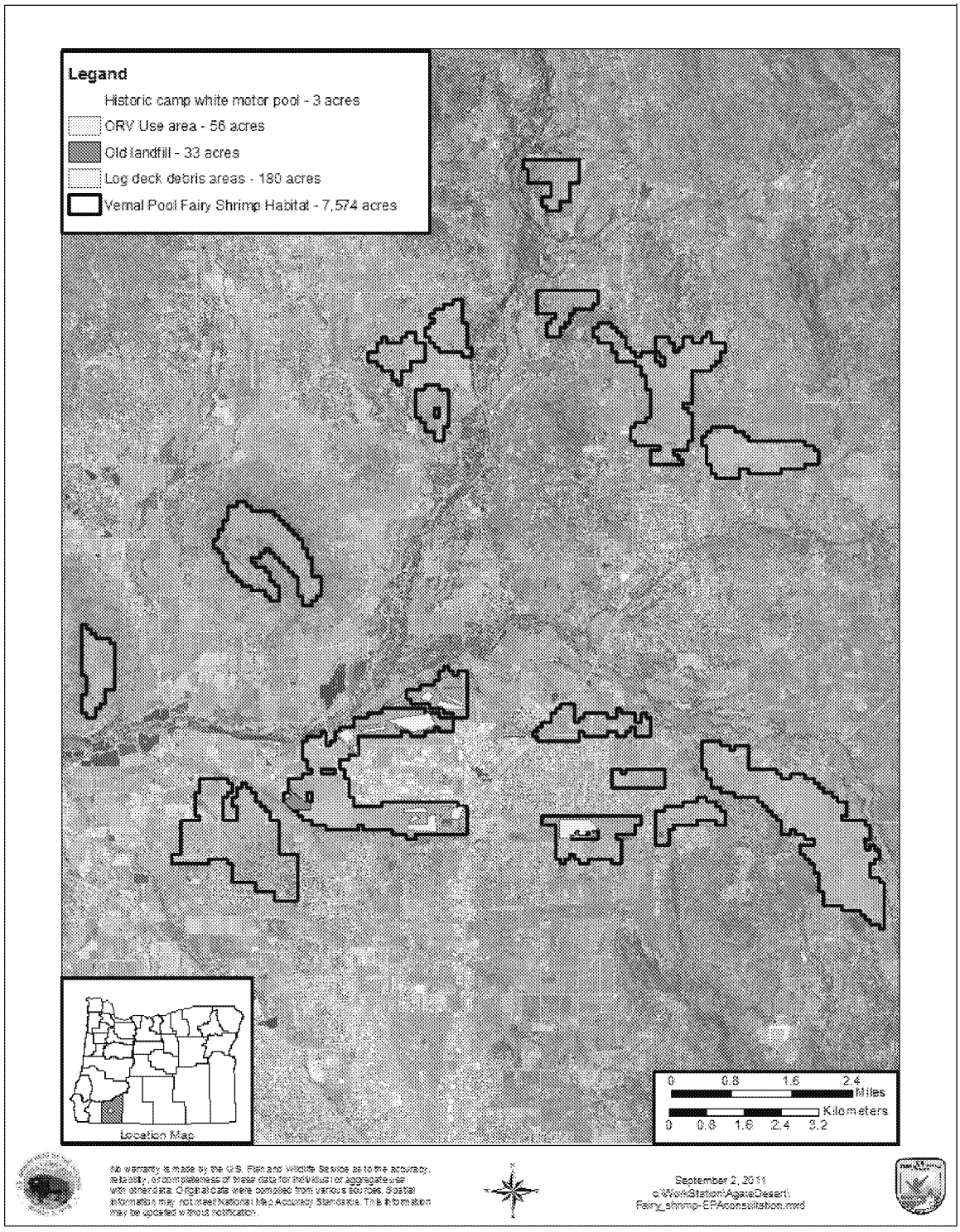

Figure 4-9. Locations of Potential Contamination in the Rogue River Valley

4.5 Effects of Selenium on Listed Species and Critical Habitats

The chronic criteria procedure explicitly set forth in the "Guidelines for Deriving Numerical National Water Quality Criteria for Protection of Aquatic Organisms and Their Uses" (Stephan et al. 1985) is not well suited for assessing bioaccumulative contaminants for which diet is the primary route of aquatic life exposure (USEPA 2004). This is exemplified in the fact that USEPA has drafted tissue-based criterion for selenium (USEPA 2004). In their draft updated selenium aquatic life criteria document, USEPA (2004) states that saltwater aquatic life should be protected from acute effects of selenium if the 24-hour average concentration of selenite seldom exceeds 127 µg/L. In addition, selenium might share similar freshwater and saltwater chronic toxicity values to freshwater fishes, as the saltwater tissue monitoring concentrations are the same as freshwater (USEPA 2004, p. 82).

According to the BE (USEPA 2008, p. 5-423), as long as water column or dietary concentrations are not so high as to overwhelm homeostatic mechanism, some aquatic species are able to regulate their internal body burden of selenium. However, selenium is distinguished from other hermetic nutrients by the very narrow range between the deficiency threshold and the toxicity threshold (Sorenson 1991). Hamilton (2004, p. 1) points out that although moderate concentrations can be stored and homeostatic functions maintained, toxic effects can occur at elevated concentrations.

Although uptake of selenium in biota includes both water and diet (Hamilton 2004, p. 2), the critical exposure pathway for fish is dietary exposure to organic selenium (DeForest and Adams 2011, p. 194). Selenium typically enters aquatic systems in an inorganic form of selenite or selenate, yet exposure to the organic selenium forms in the diet present the greatest threat to fish. Selenium concentrations in food webs tend to be much higher in standing water compared to flowing streams (DeForest and Adams 2011, p. 194). Primary producers and microorganisms appear to be the critical link for selenium to enter the aquatic food chain. Oviparous vertebrates are the most sensitive organisms to selenium as they efficiently transfer selenomethionine (the organic form) to their eggs (DeForest and Adams 2011, p. 196). Skorupa et al. (2004, p. 3) states that the long-term chronic criterion for selenium is the primary controlling standard from a risk management perspective, and USEPA's normal procedure for developing water quality criterion is much better suited for application to non-bioaccumulative pollutants (Skorupa et al. 2004, p. 5). Therefore, it is critical that the selenium exposure route in chronic toxicity tests include evaluation of the dietary exposure pathway (DeForest and Adams 2011, p. 197).

In 2000, USFWS completed a section 7 consultation on the freshwater selenium criteria in California (USFWS and NMFS 2000). Several of the listed species evaluated in the biological opinion for the California Toxics Rule (CTR) are also undergoing consultation in our Biological Opinion, and we consider that the adverse effects to these listed species would be the same as discussed in the CTR. For fish species not covered in the CTR, we consider the analysis as we would for a surrogate species. Therefore, we reviewed the CTR effects analysis and incorporated relevant sections into our analysis that pertain to our species. Some of this information is taken directly from the final biological opinion on the effects of the final promulgation of the CTR on listed species and critical habitats in California (USFWS and NMFS 2000, pp. 123-143).

4.5.1 Selenium Sources

Eisler (1985b, p. 1) states that the major source of environmental selenium is the weathering of natural rock. Sorenson (1991, p. 19) includes other sources such as coal and petroleum combustion, glass industry, metallurgical processes, mining, milling, xerography, and electronics. Abnormally high mass-loading of selenium into aquatic environments is most typically associated with the use of fossil fuels, with intensive irrigation and over-grazing of arid lands, and with mining of sulfide ores (Skorupa 1998, as cited in USFWS and NMFS 2000, p. 123). Confined livestock feeding operations and WWTPs may also contribute to loadings of selenium into surface water bodies (USFWS and NMFS 2000, pp. 123-124). The BE states that sources of selenium result from discharge of subsurface agricultural drainage water to surface waters, sulfide ore mining such as uranium, copper, bentonite, and phosphoria, commercial livestock feeding operations and dairies (USEPA 2008, p. 4-13). Municipal wastewater treatment systems are also identified as a source although it states that most systems process non-seleniferous wastewater to (USEPA 2008, pp. 4-13–4-14).

Lemly (2002, p. 3) states that selenium, a naturally occurring trace element, can be concentrated and released in waste materials from certain mining, agricultural, petrochemical, and industrial manufacturing operations. Two mining sources are identified as being responsible for mobilizing selenium in the environment: (1) the procurement, processing, and combustion of coal for electric power production, and (2) mining for gold, silver, nickel and phosphates (Lemly 2002, pp. 4, 7-9). Relatively new mining technologies, such as the cyanide heap-leach process used in gold mining, have contributed to selenium sources as have open-pit phosphate mining. Selenium is associated with the mineral matrix of these ores and hence in tailings and other surface residuals left behind from mining activity. The other sources of selenium listed by Lemly (2002, pp. 9-14) include metal smelting, municipal landfills, oil transport, refining, and utilization, and agricultural irrigation. Although municipal landfills are identified as a source, Lemly (2002, p. 10) states that they are not currently recognized as a widespread cause of selenium pollution. Source materials for selenium in examples of seleniferous materials disposed in landfills contributing to leachwater containing elevated concentrations of selenium include large amounts of photoelectronic components.

Lemly and Smith (1987, p. 3) recognize that the relative contribution of processes regulating selenium cycling may vary in different habitats. For example, fine organic sediments are often rare in fast-flowing streams, therefore limiting the surface sediment from which benthic-detrital components contribute to the food pathway. This differs from standing water habitats such as wetlands or reservoirs. The authors postulate that shallow, standing or slow-moving waters with low flush rates accumulate selenium most efficiently as biological productivity is often high; selenium load builds up in the sediments and can gradually be remobilized through lower trophic levels. However, aquatic species often use a variety of interconnected habitat types and their use of these habitats must be considered. Due to the differences in selenium cycling among aquatic habitats, various parties (States, USFWS, local municipalities) have pursued development of site specific criteria (Lemly 2002, p. 103) and brought into question the validity of national criteria (Lemly 2002, p. 110).

4.5.2 Selenium Toxicity

Much about the toxicity of selenium has been described previously. We include here excerpts from the CTR biological opinion (USFWS and NMFS 2000, pp. 123-143) that describe the toxicity through water contact and from bioaccumulation and are relevant to Oregon listed species.

4.5.2.1 Waterborne Toxicity

> "Selenium occurs in natural waters primarily in two oxidation states, selenate (+6) and selenite (+4). Waters associated with various fossil-fuel extraction, refining, and waste disposal pathways contain selenium predominantly in the selenite (+4) oxidation state. Waters associated with irrigated agriculture in the western U.S. contain selenium predominantly in the selenate (+6) oxidation state. Based on traditional bioassay measures of toxicity (24- to 96-hour contact exposure to contaminated water *without* concomitant dietary exposure), selenite is more toxic than selenate to most aquatic taxa (e.g., see review in Moore et al. 1990).
>
> Most aquatic organisms, however, are relatively insensitive to waterborne contact exposure to either dissolved selenate or dissolved selenite, with adverse-effects concentrations generally above 1,000 µg/L (ppb). By contrast, waterborne contact toxicity for selenium in the form of dissolved seleno-amino-acids (such as selenomethionine and selenocysteine) has been reported at concentrations as low as 3-4 µg/L for striped bass (*Morone saxitilis)* (ppb; Moore *et al.* 1990). It would be expected, however, that at a chronic standard of 5 µg/L (ppb) *total selenium* the concentration of dissolved seleno-amino-acids would be substantively below 3-4 µg/L (ppb) because seleno-amino-acids usually make up much less than 60%-80% of *total dissolved selenium* in natural waters. For example, it was estimated that organoselenium made up only 4.5% of the total dissolved selenium in highly contaminated drainage water from the San Joaquin Valley (Besser *et al.* 1989). Under most circumstances, a 5 µg/L chronic criterion should be protective of aquatic life *with regard to direct contact toxicity*. Selenium, however, is bioaccumulative and therefore direct contact exposure is only a minor exposure pathway for aquatic organisms (e.g., see review by Lemly 1996)."

4.5.2.2 Bioaccumulative Dietary Toxicity

> "Although typical concentrations of different chemical forms of selenium would be unlikely to cause direct contact toxicity at an aquatic life chronic standard of 5 µg/L (ppb), as little as 0.1 µg/L of dissolved selenomethionine has been found sufficient, via bioaccumulation, to cause an average concentration of 14.9 µg/g (ppm, dry weight) selenium in zooplankton (Besser *et al.* 1993), a concentration that would cause dietary toxicity to most species of fish (Lemly 1996). Based on Besser et al. (1993) bioaccumulation factors (BAFs) for low concentrations of selenomethionine, as little as 6 ng/L (ppt) of dissolved selenomethionine would be sufficient to cause foodchain bioaccumulation of selenium to concentrations exceeding toxic thresholds for dietary exposure of fish and wildlife. Thus, at a

chronic aquatic life standard of 5 µg/L (ppb) as *total selenium*, if more than 0.1% of the total dissolved selenium were in the form of selenomethionine, foodchain accumulation of selenium to levels sufficient to cause dietary toxicity in sensitive species of fish and birds would occur. For highly contaminated water (100-300 µg/L selenium) in the San Joaquin Valley about 4.5% of all dissolved selenium was in the form of organoselenium (Besser *et al.* 1989). Unfortunately, relative concentrations of selenoamino-acids have not been determined in the field in California for waters where total selenium is found in the critical 1-5 µg/L range. Further research is required to characterize typical proportions of seleno-amino-acids in waters containing 1-5 µg/L (ppb) *total selenium*.

Based on waters containing 1-5 µg/L (ppb) *total selenium*, composite bioaccumulation factors (defined as: the total bioaccumulation of selenium from exposure to a composite mixture of different selenium species measured only as *total selenium*) for aquatic foodchain items (algae, zooplankton, macroinvertebrates) are typically between 1,000 and 10,000 (on dry weight basis; Lillebo *et al.* 1988; Lemly 1996). Therefore, based on risk from bioaccumulative dietary toxicity, a generic aquatic life chronic criterion in the range of 0.2 to 2 µg/L (ppb) would be justified (where generic is defined as: the absence of site-specific and species-specific toxicological data). In fact, based on an analysis of bioaccumulative dietary risk and a literature database, Lillebo *et al.* (1988) concluded that a chronic criterion of 0.9 µg/L (ppb) for *total selenium* is required to protect fish from adverse toxic effects. Furthermore, Peterson and Nebeker (1992) applied a bioaccumulative risk analysis to semi-aquatic wildlife taxa and concluded that a chronic standard of 1 µg/L (ppb) for *total selenium* was warranted. Most recently, Skorupa (1998) has compiled a summary of field data that includes multiple examples of fish and wildlife toxicity in nature at waterborne selenium concentrations below 5 µg/L (ppb), supporting the criteria recommendations of Lillebo *et al.* (1988) and Peterson and Nebeker (1992). Furthermore, a recently concluded regional survey of irrigation related selenium mobilization in the western U.S., conducted jointly by several agencies of the U.S. Department of the Interior over a ten-year period, found that at 5 µg/L total Se in surface waters about 60% of associated sets of avian eggs exceeded the toxic threshold for selenium, i.e., that 5 µg/L Se was only about 40% protective against excessive bioaccumulation of selenium into the eggs of waterbirds (Seiler and Skorupa, In Press)."

4.5.3 Exposure Baseline

In response to concerns indicating irrigation drainage water contained high concentrations of selenium, the Department of Interior began the National Irrigation Water Quality Program (NIWQP) in 1985 (Henny et al. 2000, p. 3). One of the main goals of NIWQP is to evaluate DOI irrigation projects for drainwater contamination and related impacts to endangered species or migratory birds (Henny et al. 2000, p. 3). Three irrigation projects in Oregon have been evaluated under NIWQP and include a reconnaissance investigation at Malheur NWR (1988-89), a reconnaissance investigation at Owyhee and Vale Projects (1990-91), and a detailed study at

the Klamath Basin Refuge Complex (1990-1992) (Rinella and Schuler 1992, Rinella et al. 1994, Dileanis et al. 1996).

The reconnaissance studies of the Malheur Project concluded that agricultural runoff to the refuge in 1988 was not causing significant harmful effects to fish and wildlife resources (Rinella and Schuler 1992, p. 1) and there was no evidence that irrigation drainage was causing adverse effects (USBR 2008b). Henny et al. (2000, p. 56) reported that selenium concentrations in eggs of waterbirds nesting at Malheur NWR in 1986 and 1987 did not contain selenium concentrations above background.

In the Owyhee Project, although selenium was not elevated in aquatic-plant samples, concentrations in aquatic insects exceeded a dietary criterion for waterfowl in 60% of the Owyhee River samples and selenium concentrations in fish generally were small and not considered elevated in waterbird eggs (Rinella et al. 1994, p. 49). The median dissolved selenium concentration in the Owyhee and Snake River Basins were 2 µg/L and <1 µg/L, respectively and bottom sediment concentrations were within the expected 95% baseline range for soils from the western U.S. (Rinella et al. 1994, pp. 35-36). The conclusion from the reconnaissance investigation was that chemical and biological data did not prove conclusively that irrigation drainage from the Owyhee and Vale project areas has caused, or have the potential to cause, harmful effects on human health, fish, and wildlife (USBR 2008c).

Data from the study include concentrations of selenium in biological tissue samples from 1990-1992. Invertebrates (10 gram composites of backswimmer and waterboatman) had the highest selenium concentrations (12-18 ppm dry weight) while fish samples (20 gram composites of fathead minnow and tui chub) were below detection limits. In bird eggs, selenium residues were only reported in American avocet (0.92-2.85 mg/kg) and not detected in American coot, egret, mallard, or western grebe (MacCoy 1994, pp. 94-95). These concentrations in bird eggs are below selenium residues that are typical for uncontaminated aquatic systems (Lemly 2002, p. 119). Based on a detailed review from the studies conducted at the Klamath Basin Refuge Complex, it was determined that trace constituents dissolved from soils and mobilized by irrigation return flows do not occur at levels in water, sediment, and the biota that constitute threats to migratory birds and endangered species (USBR 2008a).

4.5.4 Analysis of Effects

4.5.4.1 Acute Criterion

The CTR biological opinion determined that the acute selenium criterion is not protective of endangered species due to the rapid bioaccumulation of selenium in aquatic organisms, and even a single pulse of selenium above the criteria into aquatic ecosystems could result in elevated selenium concentrations in aquatic food (USFWS and NMFS 2000, pp. 123-143). In addition, the criterion is speciation weighted according to the assumption that selenate is more toxic than selenite, which is the reverse of what has been found in most acute selenium toxicity studies. Specifically, the opinion stated the following:

> "The Services believe that the promulgation of the proposed speciation weighted acute criterion for selenium in the CTR would not afford adequate protection to listed species because: (1) selenium bioaccumulates rapidly in aquatic organisms and a single pulse of selenium (>10 µg/L) into aquatic ecosystems could have lasting ramifications, including elevated selenium concentrations in aquatic food webs (Maier et al. (in press); Hansen's Biological Consulting et al. 1997a, 1997b, 1998; Hanson et al. 1996; Tulare Lake Drainage District 1996); (2) USEPA's speciation weighted criterion assumes that selenate is more toxic than selenite, which is the reverse of what has been found in most acute selenium toxicity studies; (3) and the site-specific criterion of 20 µg/L promulgated in the CTR may fail to adequately protect aquatic-dependent fish and wildlife (Lemly 1997; Maier *et al.* 1998; Hansen's Biological Consulting et al. 1997a, 1997b, 1998; Hanson et al. 1996; Tulare Lake Drainage District 1996)."

Lemly (2002, p. 76) reports that a one-time or periodic waterborne inputs of high selenium concentrations can result in toxic impacts to fish and wildlife. Spikes in concentration can load sediments and be cycled into detrital food chains long after the waterborne spike is gone. Depending on the concentration of the spike, it may be enough to bioaccumulate and impact aquatic organisms. As the toxicity threshold for selenium is very steep, an increase of only a few µg/L can cause a much more serious impact to fish and aquatic-dependent birds. In addition, the 3-year period between exceedances is not appropriate for selenium given the present knowledge of the environmental dynamics and cycling (Lemly 2002, pp. 126-127).

Based on similar justification and reasoning used in the opinion for the CTR, we do not concur that the acute selenium criteria are NLAA freshwater fish or aquatic-dependent birds. We consider one level of effect for both acute and chronic criteria as they are both contributing to the selenium load.

Marbled Murrelet

Exposure to selenium was analyzed using the current level as expected levels into the future (Table 4-61 below). In contrast to other regions where problems with selenium has been reported, the Oregon Coast has no oil refining or metal smelting facilities and very little irrigation (selenium contamination from irrigation drainwater typically occurs in more arid and semi-arid regions). Therefore, these sources are ruled out for exposure of marbled murrelet. Selenium sources being discharged either directly or indirectly into the ocean within 2 miles of the coast, as identified through GIS, include the following:

- Twelve metal mines were identified in southern Oregon, four in Coos County and eight in Curry County. All mines along the coast are surface placer mines, one categorized as medium and all others classified as small. These mines are not expected to produce significant tailings (the primary source of selenium related to mining activity).

- Wastewater treatment plants and confined animal feeding operations. Table 4-61 lists sewage or wastewater treatment NPDES permits discharging directly into to the Pacific Ocean or within a waterway entering the ocean within 2 miles of the coastline. Point source discharges for confined animal feeding operations are regulated under the Oregon

Department of Agriculture, under a general permit, and no permits are identified within 2 miles of the Oregon coast (ODEQ undated [a]).

Unlike in California, marbled murrelets in Oregon only use the coastal marine environment and do not use bays or other inland water bodies. Based on aquatic habitat use by marbled murrelets in Oregon, the primary potential for exposure to any chemicals would be via discharges into the Pacific Ocean. Saltwater criteria restrictions are applicable to discharges into the ocean (acute saltwater criterion for selenium is 290 µg/L and the chronic criterion is 71 µg/L). Saltwater criteria concentrations are considerably higher than freshwater criteria concentrations. Current discharge directly into the ocean is limited to nine STPs or WWTPs (Table 4-61) which are not a major source of selenium. Freshwater discharge criteria restrictions are lower than the saltwater criteria restrictions, and are diluted before entering the ocean. Therefore, we conclude that selenium exposure to marbled murrelets in the marine environment is expected to be minimal given the (1) length of Oregon coastline, (2) dilution factor into a large body of water for both marine and freshwater discharges, and (3) limited point source discharges along the Oregon coast.

Table 4-61. All point source discharges along the Oregon Coast were reviewed via GIS mapping followed by a review of permits in the ODEQ wastewater permit document search (ODEQ undated [b]).

Facility	Location	Discharge Type
Clatsop		
City of Seaside	Necanicum River RM 0.5	Domestic major
City of Cannon Beach	Ecola Creek	Domestic minor
Arch Cape Service District	Arch Cape Creek RM 0.5	Domestic minor
Tillamook		
City of Rockaway Beach	Pacific Ocean	Domestic minor
Twin Rocks Sanitary District	Pacific Ocean	Domestic minor
City of Garibaldi	Tillamook Bay	Domestic minor
Netarts Oceanside Sanitary District	Pacific Ocean	Domestic minor
Pacific City Joint Water-Sanitary Authority	Nestucca River RM 1.5	Domestic minor
Neskowin Regional Sanitary Authority	Neskowin Creek RM 0.5	Domestic minor
Lincoln		
City of Depoe Bay	Pacific Ocean	Domestic minor
City of Lincoln City	Schooner Creek RM 1.1	Domestic major
City of Newport	Pacific Ocean	Domestic major
City of Waldport	Alsea River RM 0.6	Domestic minor
City of Yachats	Pacific Ocean	Domestic minor
Salishan Sanitary District	Siletz Bay	Domestic minor
Inn at Otter Crest	Pacific Ocean	Domestic minor
Lane		
Driftwood Shores Surfside Inn	Siuslaw River RM 0.2	Domestic minor
Douglas		
Winchester Bay Sanitary	Umpqua River RM 0.7	Domestic minor
Coos		
City of Coos Bay	Coos Bay	Domestic major
Curry		
City of Port Orford	Pacific Ocean	Domestic minor
City of Gold Beach	Riley Creek RM 0.06	Domestic minor
City of Brookings	Pacific Ocean	Domestic major
Rainbow Rock STP	Pacific Ocean	Domestic major

We believe current sources of exposure of selenium to marine prey species with a marine lifecycle do not present a significant exposure potential to marbled murrelets. Selenium exposure in the marine environment is expected to be minimal for the same reasons listed above for murrelets.

Some marbled murrelet prey (e.g., herring) spend a portion of their life cycle in Oregon bays and estuaries where there are more sources of discharge into a more concentrated area. Consistent with the CTR biological opinion, we assume marbled murrelet prey species are exposed to selenium from point sources discharging into Oregon bays and estuaries by consuming

contaminated prey that moves out of the estuaries into the marine environment. As prey from estuaries may also spend time at sea before being consumed, potentially lessening there selenium concentrations, we are unable to quantify what percent of a marbled murrelets diet would be from prey exposed to selenium. However, common murres (*Uria aalge*) may serve as a surrogate for marbled murrelets, as these two species of diving alcids coexist in the coastal waters of western North America and show considerable overlap in diet (Ronconi and Burger 2011, p. 357). Data are available on contaminants in murres from an investigation of common murre mortalities along the Oregon coast (Materna et al. 2011). We use data from this study on common murres as surrogate data to assess the exposure and potential level of effect that marbled murrelets in Oregon may have experienced.

In the murre study, liver tissue from live, dead, juvenile and adult birds were analyzed for inorganic compounds including selenium (Materna et al. 2011, pp. 4-6). Mean selenium liver concentrations from murres collected along the Oregon coast were below 5 ppm wet weight (Materna et al. 2011, pp. 50-51). The highest concentration in one individual slightly exceeded 5 ppm wet weight (Materna et al. 2011, p. 17). Selenium body burdens in birds collected along the Oregon Coast could have included sources outside Oregon, as birds do move or relocate (Manuwal and Carter 2001, p. 21-21). Heinz (1996, p. 452) recommends, based on laboratory and field data, that a concentration greater than 10 ppm wet weight selenium in the liver be considered potentially harmful to the health of young and adult birds via sublethal effects. In addition, he states that wet-weight concentrations exceeding 20 ppm may jeopardize the survival of young and adult birds. Concentrations above about 3 ppm in the liver of laying females may be associated with reproductive impairment, although measurements of selenium in the livers of females collected outside the breeding season are probably not very useful in predicting reproductive effects (Heinz 1996, p. 453). According to Heinz (1996, p. 453), if birds were collected in the wild or outside the egg laying period, then 4 to 6 ppm might be a more appropriate threshold. There is a very high risk of embryonic deformity within a population when the selenium concentration in the liver exceeds 9 ppm wet weight, while populations with means below about 3 ppm wet weight selenium in liver generally did not have many deformed embryos. In the murre population studied along the Oregon coast, the mean was above 4 ppm wet weight and a low risk was identified for adverse effects to occur from reproduction impairment. No murre was above the higher 6 ppm wet weight threshold suggested for wild birds. Likewise, no female murre was above 3ppm wet weight, but males were found to have concentrations above 3 ppm wet weight.

The murre study also investigated dry weight concentrations in the samples. Several investigators report tissue residues or threshold levels in dry weight rather than wet weight. Oregon murre liver concentrations presented as dry weight range from 2.7 to 21.1 ppm. Lemly (2002, p. 30) reports that the toxic effect threshold for reproductive failure in aquatic birds is 10 µg/g (ppm) dry weight in liver tissue. USFWS (1990, p. 23) report a very high risk of embryonic deformity when mean liver selenium exceeds 30 ppm dry weight during the egg-laying period and population means below 10 ppm for livers are not associated with biological risk. In the coastal Oregon murre samples, 16 of the 48 exceeded 10 ppm dry weight, and none were above 30 ppm dry weight. The data suggest a potential risk for adverse effects to murres and possibly murrelets from reproduction failure.

Between 10 and 30 ppm dry weight thresholds, the biological implications of mean concentrations cannot be interpreted with high confidence without a detailed field study of reproductive performance. Additionally, many of the studies used in developing the above thresholds were based on freshwater birds (Heinz 1996, pp. 452-453) or field collected data of freshwater or estuarine birds (USFWS 1990), but not marine species.

Ohlendorf and Heinz (2011) looked at both marine species and reproduction performance in the field. Ohlendorf and Heinz (2011, p. 673) report the same concentration of 10 to 30 ppm dry weight for various freshwater and terrestrial species, with no effects observed. Further, dry weight concentrations of 20-75 mg/kg in various marine species produced no effects. These authors also note that some species of marine birds can accumulate high concentrations of selenium in their livers without transferring correspondingly high concentrations in their eggs (embryos being the most sensitive to selenium poisoning)(Ohlendorf and Heinz 2011, p. 686). Only 2 murre samples are at the very bottom range of no effects reported for marine species (20-75 ppm). These results indicate that Oregon coast murres are below the recommended toxicity levels, and that a percentage of individuals may potentially be at risk for reproductive failure. There is only a potential for adverse effects to murres, as Ohlendorf and Heinz (2011, p. 673) points out that thresholds for marine birds are considerably higher than freshwater birds.

Figure 1 displays densities of marbled murrelet along the length of the Oregon Coast. The greatest densities of marbled murrelets are found associated with remaining old growth habitat on Federal land. Areas subject to greater human development, such as the north coast, maintain lower marbled murrelet densities. These areas of greater development also have more potential sources of selenium. Consequently, marbled murrelet exposure may be slightly less than murres.

Given the above analysis, we believe there is also a potential for adverse effects to marbled murrelet from reproductive failure. Low recruitment into the population is a major impairment to recovery of the species, and an effect to reproduction would not be insignificant to the ability of a marbled murrelet to recover. As the threshold guidelines to birds in general suggest there is a potential risk to a small percent of individual birds, we believe adverse effect cannot completely be discounted based on the existing data for marine birds other than murrelets.

Although there is the potential to cause biological effects to the marbled murrelet that conform to the regulatory definition of an adverse effect, the mere potential for an adverse effect does not mean it is a significant disruption to breeding. Therefore, neither harm nor harassment to the marbled murrelet is expected (i.e., LAA with no take). At the conservation zone and range wide scales, and at current levels of exposure, we believe the potential for effects will not be able to be measured and therefore is not likely to jeopardize the species.

Our effects call is different from the effects call within the BE. The BE (USEPA 2008, pp. 5-652) states that there are no facilities permitted for the discharge of selenium and there are no coastal waters listed on the 303(d) list for exceeding the saltwater selenium criteria. USEPA determined that due to the low to non-existent exposure potential of saltwater selenium criterion to the marbled murrelet, any adverse effects associated with Oregon's saltwater chronic selenium

criteria are insignificant and discountable. A brief search for selenium within the effluent limit database provided by USEPA found selenium as a parameter for several wastewater treatment facilities (not all on the coast) including the City of Coos Bay, the Roseburg Sanitary Authority, Clackamas County Service District #1, City of Medford, etc. Consequently, we expect all STPs and WWTPs to be sources of selenium and do not concur that effects associated with the saltwater criteria are not likely to adversely affect marbled murrelet.

The Oregon coast is similar to the California coast within their conservation zone 4. As such, it is not surprising that our findings are consistent with California findings for conservation zone 4. They found "Adverse impacts to prey species such as the Pacific sardine, herring, topsmelt, and northern anchovies, has the potential to significantly reduce long-term reproductive success of marbled murrelets (USFWS 1997a). Adverse effects to prey species spawning and nursery habitats have the potential to impair population size and reduce recruitment throughout their range in California. The vulnerability of marbled murrelet populations in conservation zones 5 and 6, coupled with elevated concentrations of contaminants in spawning and nursery areas for marbled murrelet prey species increase the risk of bioaccumulation of selenium. The synergistic effects of these contaminants pose a significant threat to marbled murrelet reproduction throughout conservation zones 5 and 6 and to a lesser degree in conservation zone 4." It is to this lesser degree of adverse effects to marbled murrelets that will occur in Oregon.

The PCEs of marbled murrelet critical habitat focused on individual trees with potential nesting platforms, and forested areas within 0.8 kilometers (0.5 miles) of individual trees with potential nesting platforms that had a canopy height of at least one-half the site potential tree height. This habitat will not be exposed to the toxics addressed in this Biological Opinion. Therefore, marbled murrelet critical habitat is not likely to be affected by this proposed action (NLAA).

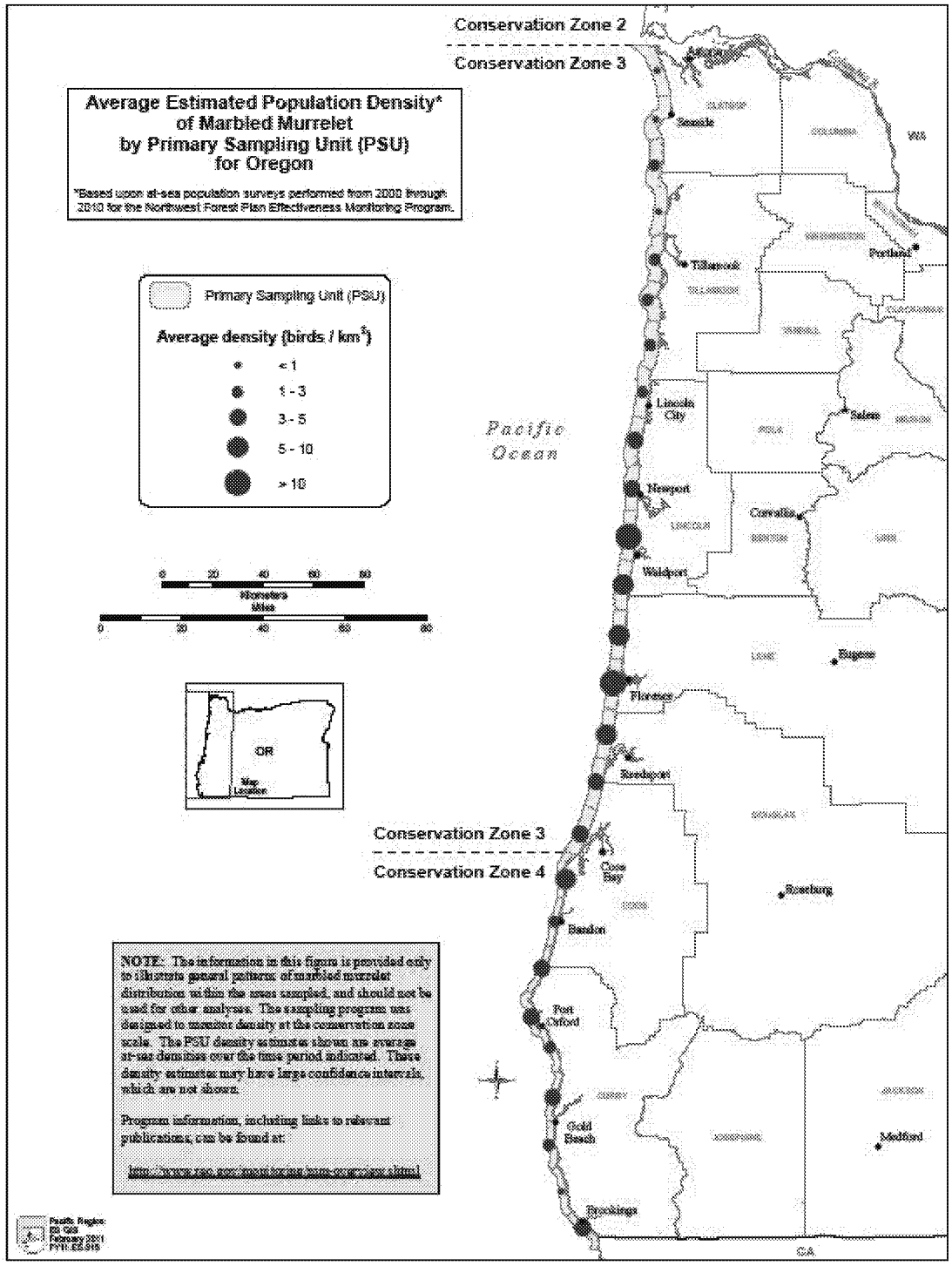

Figure 4-10. Map of marbled murrelet densities along the Oregon Coast.

Western Snowy Plover

Oregon coastal populations of snowy plover would not be exposed to significant selenium sources and results documented for populations of interior-nesting snowy plovers are expected to apply to the listed Pacific Coast populations of the snowy plover. The CTR biological opinion analysis for snowy plovers stated the following (USFWS and NMFS 2000):

> "Interior populations of the western snowy plover have been studied at breeding sites averaging about 5 µg/L (ppb) waterborne selenium in California's Tulare Lake Basin (Skorupa et al. unpubl. data). At those sites, eggs averaged about 9 µg/g selenium. That exceeds the 6 µg/g threshold for embryotoxicity among black-necked stilts, but species-specific data for snowy plover eggs containing a wide range of selenium concentrations (egg selenium from 2-50 µg/g) suggest that snowy plovers are less sensitive to selenium exposure than blacknecked stilts (Skorupa et al. unpubl. data; Page et al. 1995; Washington Dept. of Fish and Wildlife 1995). Western snowy plovers appear to be about as tolerant of selenium exposure as American avocets *(Recurvirostra americana)* (cf. Skorupa 1996; 1998) which suggests that they would not be at risk of reproductive impairment when nesting at sites with up to 5 µg/L (ppb) waterborne selenium . . . . The coastal populations of the snowy plover have a significant terrestrial component to their diet which likely provides dietary dilution of aquatic system selenium exposures . . . . Unless coastal populations would be exposed to significant selenium-mercury interaction effects (cf. Heinz and Hoffman 1998), the results documented for populations of interior-nesting snowy plovers are expected to apply to the listed Pacific Coast populations of the snowy plover. Therefore, the western snowy plover is considered not likely to be adversely affected by the CTR proposed selenium criterion of 5 µg/L (ppb)."

Based on the CTR analysis above, snowy plover uptake of selenium from aquatic sources will be diluted due to their consumption of terrestrial invertebrates. In addition, studies show that snowy plovers are not as sensitive to adverse effects of selenium as compared to other shorebirds. Consequently, the western snowy plover is considered not likely to be adversely affected by the approval of Oregon proposed selenium criterion of 5 µg/L (ppb) (i.e., NLAA).

The PCEs for the western snowy plover critical habitat are described in the final rule for critical habitat (USFWS 2012a). The PCE #2 describes habitat that supports small invertebrates, such as crabs, worms, flies, beetles, spiders, sand hoppers, clams, and ostracods, as a characteristic of habitat necessary to sustain the essential life history functions of the species. Given the variety of prey species consumed by western snowy plover we expect that there should not be any meaningful reduction in prey availability due to exposure to selenium. Therefore, exposure to selenium is NLAA to western snowy plover critical habitat (i.e., NLAA).

Fish

Information on toxics effects of selenium to fish have been reviewed in the biological opinion for the CTR (USFWS and NMFS 2000). Information from the opinion states the following:

"Lemly [(1996b)] reports that salmonids are very sensitive to selenium contamination and exhibit toxic symptoms even when tissue concentrations are quite low. Survival of juvenile rainbow trout (*Oncorhynchus mykiss*) was reduced when whole-body concentrations of selenium exceeded 5 µg/g (dry wt.). Smoltification and seawater migration among juvenile chinook salmon (*Oncorhynchus tshawytscha*) were impaired when whole-body tissue concentrations reached about 20 µg/g. However, mortality among larvae, a more sensitive life stage, occurred when concentrations exceeded 5 µg/g. Whole-body concentrations of selenium in juvenile striped bass (*Morone saxitilis*) collected from areas in California impacted by irrigation drainage ranged from 5 to 8 µg/g. . . . Summarizing studies of warm-water fish Lemly reports that growth was inhibited at whole-body tissue concentrations of 5 to 8 µg/g selenium or greater among juvenile and adult fathead minnows (*Pimephales promelas*). Several species of centrarchids (sunfish) exhibited physiologically important changes in blood parameters, tissue structure in major organs (ovary, kidney, liver, heart, gills), and organ weight-body weight relations when skeletal muscle tissue contained 8 to 36 µg/g selenium. . . . Based on Lemly's review of more than 100 papers, he recommended the following toxic effects thresholds for the overall health and reproductive vigor of freshwater and anadromous fish exposed to elevated concentrations of selenium: 4 µg/g whole body; 8 µg/g skinless fillets; 12 µg/g liver; and 10 µg/g ovary and eggs. He also recommended 3 µg/g as the toxic threshold for selenium in aquatic food-chain organisms consumed by fish. Lemly reported that when waterborne concentrations of inorganic selenium (the predominant form in aquatic environments) are in the 7- to 10-µg/L (ppb) range bioconcentration factors in phytoplankton are about 3,000. Consequently, he concluded that patterns and magnitudes of bioaccumulation are similar enough among various aquatic systems that a common number, 2 µg/L (ppb) (for filtered samples of water), could be given as a threshold for conditions "highly hazardous to the health and long-term survival of fish".

Depending on the form of selenium and the life-stage of fish considered, waterborne concentrations of selenium less than the CTR-proposed 5 µg/L (ppb) concentration can have direct toxic impacts on salmonids (Hodson et al. 1980; Moore et al. 1990). Hodson et al. reported that rainbow trout (*0. mykiss)* eggs respond physiologically (reduced median time to hatch) at selenium (as selenite) concentrations above 4.3 µg/L (ppb). However, the most dangerous exposure pathway for salmonids, as with other fish, is via dietary bioaccumulation of selenium. As little as 3.2 µg/g selenium in the diet was sufficient to adversely affect early life stages of chinook salmon under controlled conditions (Hamilton et al. 1989; 1990). Based on a bioaccumulation factor for dry weight concentrations of selenium in aquatic invertebrates (compared to water) of 1,800 (Pease et al. 1992), a concentration of as little as 1.8 µg/L (ppb) selenium could result in salmonid foods averaging more than 3.2 µg/g selenium . . . . The Services thus conclude that currently available data for salmonids do not support

the CTR-proposed selenium criterion of 5 µg/L (ppb) as adequately protective of salmonids."

Because toxic effects of selenium are primarily from the bioaccumulation pathway, it is not appropriate to apply our approach and model used for other chemicals to quantify level of effect for selenium. Therefore, our analysis for level of effect with selenium is modified and similar to the approach used by USEPA BE (USEPA 2008) and is described below.

In developing their freshwater selenium multiple routes of exposure assessment, USEPA used a BAF of 113 L/kg for all fish species (USEPA 2008, p. 5-426). However, in the freshwater selenium aquatic-dependent wildlife effects assessment, USEPA recognizes that BAFs differ for various trophic levels (USEPA 2008, p. 5-428); trophic level 1 and 2 are 113 L/kg, trophic level 3 is 483 L/kg, and trophic level 4 is 521 L/kg. The latter BAF is consistent with the bioconcentration factor (geometric mean) reported for chinook salmon of 531 (Saiki et al. 1992, p. 84), however, DeForest and Adams (2011, p. 196) report that bioaccumulation factors in fish are generally greater than 1,000. Although the specific classification of USEPA's trophic levels are not stated, we assume the general categories are as follows:

1. Level 1 consists of algae, plankton, and detritus;
2. Level 2 is made up of zooplankton and invertebrates;
3. Level 3 is fish with an invertebrate diet; and
4. Level 4 is piscivorous fish.

Vernal pool fairy shrimp, at trophic level 2, would be appropriate to model using the 113 BAF. Oregon chub is a trophic level 2, feeding primarily on water column fauna, especially invertebrates that live in dense aquatic vegetation. Markle et al. (1991, p. 288) found that the diet of Oregon chub adults consisted primarily of minute crustaceans including copepods, cladocerans, and chironomid larvae. Juveniles also are trophic level 3 with a diet consisting of minute organisms such as rotifers, copepods, and cladocerans (Pearsons 1989, pp. 41–42). Suckers are at trophic level 3 feeding mostly on small zooplankton (starting with rotifers and moving to cladocerans) and on larval and adult midges. Both of these invertebrates feed on algae, bacteria, and detritus. Detritus is nearly always found in the guts of suckers, although its nutritional value is uncertain. Bull trout is classified as trophic level 3 and 4. Throughout their first year, bull trout primarily eat small aquatic invertebrates (Bjornn 1991; USFWS 1998a), often ephemeropterans or dipterans (Bjornn 1991), which are readily available in the lower water column and interstitial spaces inhabited by juveniles. As the bull trout grow larger, while continuing to feed on aquatic invertebrates, they begin to feed more and more on fish species (Bjornn 1991). This increased piscivorous behavior increases as the fish grows. The adult diet of bull trout consists of a higher percentage of fish than most other salmonids. In certain parts of their range the mountain whitefish (*Prosopium williamsoni*) comprise a large part of the bull trout diet (Bjornn 1991). In addition to whitefish, bull trout will feed on sculpins, darters or other trout and where applicable, salmon fry (USFWS 1998a).

Using the BAF from these higher levels considerably changes the risk assessment. If the trophic level 3 BAF is used in the assessment effects concentration calculation, then the result is 2.4 mg/kg. A trophic level 4 BAF produces a result of 2.6 mg/kg. Comparing this to the whole body residues for surrogate species shows that there would be additional effects including

reductions in body weight and fork length after 90 days, and reduction in survival and growth after 90 days (USEPA 2008, p. 5-425). Table 4-62 below was modified from the Table 5.2.16.7 in the BE.

USEPA used wet weights in its Table 5.2.16.7 reporting the adverse effects associated with whole body selenium residues (USEPA 2008, p. 5-425). Most literature reports results in dry weight (Lemly 2002, pp. 24-25; DeForest and Adams 2011, pp. 202-203). Converting the assessment effects concentration to dry weight, assuming a 75% moisture level (Skorupa et al. 2004, p. 20), the three assessment effects concentrations for each trophic level are as follows: trophic level 1-2 = 2.3 mg/kg, trophic level 3 = 9.6 mg/kg, and trophic level 4 = 10.4 mg/kg. Comparing the level 3 and 4 trophic level assessment effects concentrations indicates that the potential effects beyond what USEPA found in their BE to also include reduction in survival in rainbow trout and razorback sucker in addition to significant reductions in body weight and fork length in rainbow trout after exposure for 90 days. Lemly's (2002, pp. 24-25) summary of tissue selenium concentrations associated with toxic effects also indicates that mortality and reduced growth could occur in trophic level 1 and 2 fish and mortality, reduced growth and reproductive failure could occur in trophic level 3 or 4 fish at the criterion concentration. Toxicity effects to embryos/larvae resulting from maternal transfer reported in DeForest and Adams (2011, pp. 202-203) indicate that fish in trophic level 3 and 4 would experience greater than 20% larval mortality based when exposed to chronic criteria concentrations of selenium in water based on estimated whole body weights. Furthermore, there would be approximately 20% of larval edema from exposure to a concentration at the chronic criterion.

Many of the reproductive adverse effects in fish are reported with associated selenium concentrations in either egg or ovary. DeForest and Adams (2011, p. 198) believe that the clearest approach to evaluating selenium bioaccumulation in order to overcome species-specific relationships is to focus on an egg or ovary-based selenium threshold. Skorupa et al. (2004) report a regression equation that is most applicable to this conversion as follows: whole body selenium concentration = 0.37 (ovary selenium) – 0.13. Converting the whole body concentrations to ovary concentrations yields the results presented in Table 4-62. According to the table presented by DeForest and Adams (2011, p. 202-203) for ovary concentrations in 4 fish species at the EC10 and EC20 selenium concentrations, it appears that at chronic criterion concentrations fish within trophic levels 1 or 2 would experience larval edema (fluid accumulation in tissues) or larval mortalities in less than 10% of the exposed population. However, fish in trophic level 3 or 4 could experience approximately a 20% effect for larval mortality or greater than 20% for larval edema based on these estimates. DeForest et al. (2011b, p. 8) reports an EC10 at 22 µg/g dw selenium in largemouth bass ovary in its summary of studies evaluating selenium toxicity to embryos and larvae resulting from maternal transfer. Therefore, based on our estimated concentrations in body tissues and the proximity to EC20 values presented by DeForest and Adams et al. (2011, pp. 202-203), our estimated level of effect for fish from exposure of fish to chronic selenium criterion concentrations is <10% for larval edema in trophic level 1 or 2 fish, and 20% for mortality in trophic level 3 and 4 fish. Therefore we will estimate a 20% effect level at the criterion concentration based on these reported test responses.

Table 4-62. Adverse effects associated with whole body selenium concentrations compiled from reviews by DeForest and Adams (2011, Table 5.2), Lemly (2002, Table 2.2) and USEPA (2008, Table 5.2.16.7).

Estimated Selenium Whole Body Residue Assessment Effects Concentration (mg/kg wet wt: dry wt)	Estimated selenium ovary concentration (mg/kg dry wt[1])	Species	Effect	Reference
0.23: 0.92	2.62	Fathead minnow *(Pimephales promelas)*	Inhibition of growth after 3 weeks exposure to waterborne selenium	Bertram and Brooks 1986
0.51: 2.04	NA[2]	Midge (*Chironomus decorus*)	Reduction in growth	Malchow et al. 1995
0.57 :2.28	***6.3***	***Estimated residue-based C_A for trophic levels 1 and 2***	***Derived from product of chronic criterion and bioconcentration factor***	***CCC = 5 µg/L BAF = 113 L/kg***
0.58 : 2.32	6.4	Rainbow trout (*Oncorhynchus mykiss*)	Significant reductions in body weight (33%) and fork length (10%) after 90 days	Vidal et al. 2005
0.98 : 3.92	10.7	Chinook salmon (*Oncorhynchus tshawytscha*)	Reduction in survival and growth after 90 days	Hamilton and Wiedmeyer 1990
1.22 : 4.88	13.3	Razorback sucker *(Xyrauchen texanus)*	80% reduction in survival after 10 days, no consistent effect on growth	Hamilton et al. 2002
2.4-2.6 : 9.6-10.4	***26.1-28.2***	***Estimated residue-based C_A for trophic levels 3 and 4***	***Derived from product of chronic criterion and bioconcentration factor***	***CCC = 5 µg/L BAF = 483 L/kg or 521 L/kg***
4.3 : 17	46.1	Bluegill (*Lepomis macrochirus*)	Reproductive failure	Coyle et al. 1993
4.6 :18.4	50	Bluegill (*Lepomis macrochirus*)	Reduced adult survival, growth, increased larval mortality	Hermanutz et al. 1992
6.34 : 25.4	NA	Cladoceran (*Daphnia magna*)	Reproductive ED47	Ingersoll et al. 1990
6.5 : 26	NA	Rotifer (*Brachionus calcyflorus*)	Reduction in population biomass	Dobbs et al. 1996
11.2 : 44.8	NA	Cladoceran (*Daphnia pulex*)	Decrease in number of neonates/brood	Reading and Buikema 1983

[1]Converted from prior table using a moisture content of 75% (Skorupa et al. 2004, p. 20)
[2]Not applicable

Summary of Effects

We do not anticipate the proposed action of approval of the acute selenium criteria will incidentally take any listed species. The section above includes a finding that implementation of the proposed action has the potential to cause biological effects to bull trout, Oregon chub, Lost River and shortnose suckers, and vernal pool fairy shrimp that conform to the regulatory definition of take. However, the mere potential for take is not a legitimate basis for a take exemption. Since no take is anticipated or exempted, no reasonable and prudent measures and terms and conditions are provided below. If take is detected during implementation of the proposed action, then reinitiation of formal consultation should be requested, and any operations causing such take must cease pending the outcome of the reinitiated consultation.

4.5.4.2 Chronic Criterion

Effects of Selenium on Bull Trout

Depending on the form of selenium and the life-stage of fish considered, waterborne concentrations of selenium less than 5 µg/L can have direct toxic impacts on salmonids. Potential effects include reduction in survival and growth, and reproductive failure in trophic level 3 or 4 fish at the criterion concentration. We estimate these adverse effects will occur to 20% of the population for larval edema and mortality based on exposure to chronic criterion concentrations.

According to DeForest and Adams (2011, p. 202-203), Dolly varden appear to be considerably less sensitive to selenium toxicity as compared to other fish species, including salmonids. Bull trout are rarely in waters more commonly associated with selenium contamination (ponds and wetlands) but inhabit streams or large reservoirs.

Effects of acute selenium exposure

While some adverse effects may occur from the acute exposure to selenium (see Selenium exposure discussion section) it in unlikely these effects will rise to the level of harm.

Effects of chronic selenium exposure

The chronic selenium criterion analyzed in this Biological Opinion is 5 µg/L. Because selenium bio-accumulates we believe that chronic exposure may well have adverse effects to bull trout survival. We estimate that as many as 20% of the juvenile population may be affected. As we have no data on the current number of juvenile bull trout within the action area we will apply this percentage to our estimated number of adults. We realize that this may greatly overestimate the effects, but will give the species the benefit of the doubt in accordance with USFWS policy (USFWS and NMFS 1998).

The possibility of reduced survival exists. The risk of this occurring is considered significant because individual bull trout may be killed (Table 4-63).

Table 4-63. Effects to bull trout from chronic selenium exposure

A	B	C	D	E	F
Basin	Type of Effect	Effects per Modeling	Percent Habitat Exposed	Estimated Average Adult Population Size	Mortalities of Adult Bull Trout per 3-year period
Grande Ronde River	Survival	20%	16%	150	5
Hells Canyon Complex	Survival	20%	22%	300	13
Hood River	Survival	20%	16%	150	5
Imnaha River	Survival	20%	33%	1,750	116
John Day River	Survival	20%	24%	400	19
Klamath River	Survival	20%	6%	2,050	25
Lower Deschutes River	Survival	20%	4%	1,750	14
Malheur River	Survival	N/A	0%	150	0
Odell Lake	Survival	N/A	0%	50	0
Powder River	Survival	20%	44%	300	26
Umatilla River	Survival	20%	28%	50	3
Upper Willamette River	Survival	20%	40%	300	24
Walla Walla River	Survival	20%	7%	1,750	25
Totals: (mortalities rounded up to the nearest individual)				9,150	256

Although the Columbia and Snake River foraging, migratory, and overwintering habitats have substantial areas of potential exposure, many bull trout utilizing these rivers would belong to basins already represented in Table 4-63. Chronic exposure to selenium at the proposed standard may occur within portions of these foraging, migratory, and overwintering habitats that could adversely affect bull trout. Therefore, chronic exposure within the Columbia River and Snake River foraging, migratory, and overwintering habitats "may affect and is likely to adversely affect" bull trout. Because of the high flows within these rivers and the ability of fish to move through or avoid the highest concentrations they encounter within large systems. Also, bull trout are more likely to be exposed during their early life stages. Very few juvenile bull trout are

likely to be present in these rivers. Any adverse effects associated with these habitats are not likely to rise to a level of additional harm or injury.

Summary of effects to bull trout from selenium exposure

The following summary is based on a worst case scenario:

Acute Selenium Exposure

It is our opinion that acute exposure to selenium at the criterion is significant, but will not result in harm to bull trout within the action area (i.e., LAA with no take).

Chronic Selenium Exposure

It is our opinion that chronic exposure to selenium at the criterion could result in lethal harm to an estimated 259 individual adult bull trout associated with 820.6 miles of bull trout habitat within the action area during every 3-year period covered under this opinion (i.e., LAA with take).

Effects of Selenium on Lost River and Shortnose Suckers

Suckers are considered a trophic level 3 fish. Likely effects including reduction in survival and growth, and reproductive failure could occur in trophic level 3 or 4 fish at the criterion concentration. We estimate these adverse effects will occur to 20% of the population for larval edema and mortality based on exposure to chronic criterion concentrations.

The Klamath Irrigation Drainwater Project determined that trace constituents dissolved from soils and mobilized by irrigation return flows do not occur at levels in water, sediment, and the biota that constitute threats to migratory birds and endangered species. Data from the study report concentrations of selenium in biological tissue samples from 1990-1992. Invertebrates (10 gram composites of backswimmer and waterboatman) had the highest selenium concentrations (12-18 ppm dry weight) while there no detections in fish samples (20 gram composites of fathead minnow and tui chub). In bird eggs, selenium residues were only reported in American avocet (0.92-2.85 mg/kg) and not detected in American coot, egret, mallard, or western grebe (MacCoy 1994, pp. 94-95).

Exposure Analysis

The BE states that sources of selenium result from discharge of subsurface agricultural drainage water to surface waters, sulfide ore mining such as uranium, copper, bentonite, and phosphoria, commercial livestock feeding operations and dairies (USEPA 2008). Municipal wastewater treatment systems are also identified as a source although it states that most systems process non-seleniferous wastewater (USEPA 2008).

Selenium sources listed by Lemly (2002, 2004) include metal smelting, municipal landfills, oil transport, refining, and utilization, and agricultural irrigation (Lemly 2002, pp. 9-12; Lemly 2004). Most often listed selenium sources in Lemly (2004) are coal wastes, gold, silver, and nickel mining, metal smelting, oil refinery, agricultural irrigation, but feedlot wastes are becoming more recognized as selenium sources owing to selenium supplements in livestock feed. Although municipal landfills are identified as a selenium source, Lemly (2002) states that

they are not currently recognized as a widespread cause of selenium pollution. Selenium is not among the heavy metals listed as contributing to water pollution from Washington State highways (Herrera Environmental Consultants 2007; WADT 2007), nor was it detected in a study of metals in runoff from Oregon highways (Bloomquist 2009).

In a study of environmental contaminants at the Tule Lake and Lower Klamath National Wildlife Refuges (Sorenson and Schwarzbach 1991), selenium concentrations in all water samples were determined to be less than reporting limits of 1 µg/L and the highest concentration in sediment was 0.7 µg/g at one site at Tule Lake. Maximum selenium concentrations in tissues were 1.2 µg/g dry weight in tui chub from Tule Lake and 1.7 µg/g dry weight in freshwater mussels from the Straits Drain, but none was detected from any biota from Upper Klamath Lake or the Link River. According to Sorenson and Schwarzbach (1991), the selenium concentrations in fish from the Klamath River Basin are in the range that would be considered uncontaminated according to data presented in reviews by Eisler (1985b) and Lemly and Smith (1987) and are less than the 85th percentile selenium concentrations from the National Contaminant Biomonitoring Program of 0.73 µg/g wet weight (about 2.9 µg/g dry weight). Based on a detailed review from the studies conducted at the Klamath Basin Refuge Complex, it was determined that trace constituents dissolved from soils and mobilized by irrigation return flows do not occur at levels in water, sediment, and the biota that constitute threats to migratory birds and endangered species (USBR 2008). Selenium was not detected in Klamath River Basin groundwater by ODEQ (Cole 2006).

Within the range of the Lost River sucker and shortnose sucker, the presence of selenium contamination of surface water is most likely to occur in the Lost River subbasin as a result of runoff from irrigated agriculture and confined animal feeding operations, and in the Keno Reservoir that receives winter runoff from the Lost River via the Lost River Diversion Channel. This assumption seems justified based on the occurrence of selenium in Tule Lake sediments and fish, as reported by Sorenson and Schwarzbach (1991). Tule Lake is a sump for some of the Lost River water and is downstream of most of the Klamath Irrigation Project.

Acute Effects Analysis

Because toxic effects of selenium are primarily from the bioaccumulation pathway, it is not appropriate to apply our approach and model used for other chemicals to quantify level of effect for selenium. Therefore, our analysis for level of effect with selenium is modified and similar to the approach used in the USEPA BE (USEPA 2008) and is described elsewhere. Based on our analysis presented elsewhere, we do not anticipate the proposed action of approval of the selenium acute criterion will incidentally take any Lost River sucker or shortnose sucker.

Chronic Effects Analysis

Suckers are considered a trophic level three fish because they primarily feed on small invertebrates, such as *Daphnia* and midge larvae. Likely effects of selenium exposure including reduction in survival and growth, and reproductive failure could occur in fish at trophic levels three or four at the chronic criterion concentration. We estimate these adverse effects will occur to 20% of the larval sucker population as a result of edema and mortality based on exposure to

chronic criterion concentrations. The effects will be limited to suckers in the Lost River and Keno Reservoir, as described above.

Estimates of Take

Based on the information presented above, exposure of Lost River sucker and shortnose sucker to selenium at the chronic criterion concentrations is likely to harm or kill 20% of the Lost River sucker and shortnose sucker larvae present in the Lost River and Keno Reservoir once every 3 years over the 25-year life of the proposed action.

Analysis of Effects to Proposed Critical Habitat

The proposed action will affect Keno Reservoir, which is part of proposed critical habitat Unit 1 (Upper Klamath Lake) (USFWS 2011d). Based on the information presented above, the proposed action will reduce water quality in the Keno Reservoir, and thus adversely affect the ability of PCE 1 (water quality) to support the species. These effects will be confined to the Keno Reservoir where the point-source discharges occur, as described above. We do not anticipate that this would lead to significant adverse effects to the proposed critical habitat because Keno Reservoir (approximately 2,500 acres) represents only about 2% of approximately 120,000 acres that were proposed for critical habitat for each species (USFWS 2011a), and the adverse effects will occur once every 3 years, allowing time for recovery to occur.

Effects of Selenium on Oregon chub

Depending on the form of selenium and the life-stage of fish considered, waterborne concentrations of selenium less than 5 µg/L can have direct toxic impacts on Oregon chub. Oregon chub fall within the trophic level 3 and 4 categories and likely effects could include reduction in survival and growth, and reproductive failure at the criterion concentration. We estimate these adverse effects will occur to 20% of the population for larval edema and mortality based on exposure to chronic criterion concentrations.

The most important factors associated with trace-element contaminants in irrigation drainwater are pervasive geological source, arid to semiarid climate, and topographically closed basin (Seiler et al. 1999, p. 2). This is inconsistent with the temperate marine climate of the Willamette River Basin (cool, wet winters and warm, dry summers) and the abundant flow of water which does not terminate in lakes and ponds. In summary statistics reported in the Willamette River Basin Water Quality Study 1992-1994, there were zero detections of selenium (minimum reporting level 1 µg/L) in 27 samples collected throughout the basin in targeted areas subject to agricultural chemicals (Anderson et al. 1996, p. 59).

Effects from Chronic Exposure

Larval Survival. Data are not available to estimate the ratio of recently hatched fish to adults in Oregon chub populations. Accurate data on young fish are difficult to collect because of the individual's small size, secretive nature, and ability to hide within pond flora and substrate. The best and most reliable data that we have access to are estimates of adult abundance per population. Although our effects models were done to gauge effects to young Oregon chub, it is widely believed that early life stages of fish are far more sensitive and susceptible to the influences of toxins than are adults. As we have no reliable data for the abundance of young fish

we will apply our effects data to the adult populations. Although this approach may overestimate harm to the population, we believe this is the most cautious approach and errs on the side of conservation of the species.
Seven Oregon chub populations could be exposed to chronic levels of selenium (Table 4-55). The combined 2011 estimate of adult abundance for these populations is 5,206 chub. Thus, based on 2011 abundance estimates, 1,041 Oregon chub may be killed from exposure to chronic levels of selenium. However, given the life history of Oregon chub and the wide annual fluctuations in abundance that naturally occur in populations, a 20.0% reduction in survival every 3 years would not be detectible.

Other Effects. There may also be other adverse effects including reductions in food supply if invertebrates are impacted by selenium. However, many invertebrates are able to tolerate toxicity levels higher than the criteria for selenium (USEPA 2008). Given the variety of prey species consumed by Oregon chub we expect that there should not be any meaningful reduction in prey availability due to exposure to selenium at the criteria levels.

Summary of Effects to Oregon chub from Selenium Exposure

Chronic Selenium Exposure

Chronic exposure to selenium will result in the death of approximately 1,041 adult Oregon chub every 3 years.

Effects of Selenium on Vernal pool fairy shrimp

Vernal pool fairy shrimp have limited exposure potential for selenium as there are no anthropogenic sources of selenium identified in vernal pool fairy shrimp habitat. Although landfills are listed as a potential source of selenium, the probability of exposure is remote since the landfills near vernal pool fairy shrimp habitat contain military waste which is not expected to harbor selenium. As invertebrates are lower on the food chain, within the trophic level 2, their bioaccumulation is less than the higher trophic level fish. Adverse effects noted for invertebrates in Lemly (2002, p. 25) and USEPA (2008, pp. 5-425) indicate invertebrates are much less sensitive than fish. Therefore, we determine that approval of the chronic selenium criterion is not likely to adversely affect vernal pool fairy shrimp.

4.6 Effects of Interrelated/ Interdependent Actions

Interdependent actions are defined as actions with no independent use apart from the proposed action. Interrelated actions include those that are part of a larger action and depend on the larger action for justification. USEPA's approval of TMDLs based upon these water quality standards is an interdependent action, but the specifics of such approvals are not known at this time, and will have to be consulted on individually in the future. We are not aware of any other significant interrelated or interdependent actions.

5 CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, tribal, local, or private actions on listed species or critical habitat that are reasonably certain to occur in the action area considered in this

Biological and Conference Opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Future anticipated non-Federal actions that are likely to occur in or near surface waters in the State of Oregon include water management, timber harvest, grazing, mining, agriculture, urban development, municipal and industrial wastewater discharges, road building, sand and gravel operations, introduction of nonnative fishes, off-road vehicle use, fishing, hiking, and camping. The effects of these types of non-Federal actions in the action area are largely responsible for the current condition of the listed species and critical habitats at issue in this opinion and are likely to continue having a variety of adverse effects on the endangered and threatened species addressed in this Opinion, and on their designated and proposed critical habitat.

Future non-Federal actions causing diffuse and concentrated urban growth are likely to increase the overall volume of contaminant loading from wastewater treatment plants and sediment runoff into riverine, estuarine, and marine habitats. Urban runoff from impervious surfaces and roadways may also contain oil, heavy metals, PAHs, and other chemical pollutants and flow into state surface waters. Inputs of these point and non-point pollution sources into numerous rivers and their tributaries will affect water quality in habitats occupied by the species and critical habitats considered in this opinion. Based on the likely increase in human population growth within the action area, an associated increase in the number of NPDES permits issued and a concomitant increase of pollutant loading are likely to occur that adversely affect listed species and critical habitats.

Mining has historically been a major component of western state economies. With national output for metals projected to increase by 4.3% annually, output of western mines should increase markedly (Figueroa and Woods 2007). Increases in mining activity will add to existing significant levels of mining contaminants entering river basins. Given this trend, we expect existing water degradation in Oregon streams that feed into or provide habitat for threatened and endangered species to be exacerbated.

As the western states have large tracts of irrigated agriculture, a 2.2% rise in agricultural output is anticipated, in part, during the 25-year term of the proposed action (Figueroa and Woods 2007). Impacts from heightened agricultural production will likely result in two negative impacts on listed species. The first impact is the greater use and application of pesticide, fertilizers, and herbicides and their increased concentrations and entry into freshwater systems. insecticides, and other pollutants from agricultural runoff may further degrade existing fish habitats. Second, increased output and water diversions for agriculture may also place greater demands upon limited water resources. Water diversions will reduce flow rates and alter habitat throughout freshwater systems. As water is drawn off, contaminants will become more concentrated in these systems, exacerbating contamination issues in habitats for protected species.

The above non-federal actions are likely to pose continuous unquantifiable negative effects on the listed species and critical habitats addressed in this opinion. These effects include increases in sedimentation, increased point and non-point pollution discharges, decreased infiltration of

rainwater (leading to decreases in shallow groundwater recharge, decreases in hyporheic flow, and decreases in summer low flows).

There are also non-Federal actions likely to occur in or near surface waters in the State of Oregon that are likely to have beneficial effects on listed species and critical habitats. These include implementation of riparian improvement measures, best management practices associated with timber harvest, grazing, agricultural activities, urban development, road building and abandonment, recreational activities, and other nonpoint source pollution controls.

Due to the mixed positive and negative effects of such a variety of actions across the State, we conclude that cumulative effects from non-Federal actions that have affected the listed species and critical habitats addressed in this opinion will continue during the 25-year term of the proposed action at similar scales and intensities as in recent years.

6 CONCLUSION

After reviewing the current status of the affected species and critical habitats, the environmental baseline for the action area relative to those species and critical habitats, the effects of USEPA's proposed approval of revised water quality standards for toxics on listed species and critical habitats, and cumulative effects relative to those species and critical habitats, it is the USFWS's biological opinion that the proposed action is not likely to jeopardize the continued existence of the bull trout, Oregon chub, Lost River sucker, shortnose sucker, vernal pool fairy shrimp, and the marbled murrelet, and is not likely to destroy or adversely modify critical habitat for the bull trout, Oregon chub, vernal pool fairy shrimp, or marbled murrelet.

Critical habitat has been proposed for the Lost River sucker and the shortnose sucker. After reviewing the current status of the critical habitat, the environmental baseline for the action area relative to that critical habitats, the effects of USEPA's proposed approval of revised water quality standards for toxics on the proposed critical habitats, and cumulative effects, it is USFWS's conference opinion that the proposed action is not likely to destroy or adversely modify the proposed critical habitat.

A summary of our rationale for these conclusions is presented below.

6.1 Bull Trout

As discussed previously in this document, the jeopardy determination for the bull trout, just like for all listed species, is made at the scale of the listed entity (in this case, the coterminous U.S. population of the bull trout). However, in recognition of (interim) recovery units (IRUs) being established for the bull trout and in accordance with USFWS policy, the jeopardy determination for the bull trout relies upon an assessment of how the effects of the proposed action, together with cumulative effects, are likely to affect the survival and recovery role of the Columbia River and Klamath River IRUs. This approach recognizes that if the effects of a proposed Federal action, taken together with cumulative effects, are compatible with the survival and recovery role assigned to each recovery unit, then that action is not likely to appreciably reduce the likelihood

of the survival and recovery of the bull trout at the coterminous listing scale (i.e., the action is not likely to jeopardize the bull trout).

In this document, we have used a hierarchical approach to inform the above assessment. The effects of the proposed action on the bull trout were characterized at the local population and core area scales. Maintenance or enhancement of bull trout core areas, which consist of one or more local populations, is fundamental to conserving the bull trout at the rangewide scale. Conserving core areas provides for persistence of the bull trout from a distribution, genetic, and demographic perspective. To the extent that the effects of the action, together with cumulative effects, are compatible with what is necessary for core area persistence means those effects are compatible with the survival and recovery role assigned to each recovery unit. That compatibility serves as the basis for concluding that the proposed action is not likely to jeopardize the continued existence of the bull trout.

There are 121 core areas identified in the coterminous U.S. population of the bull trout (USFWS 2008d). Of these, 97 core areas are identified in the Columbia River IRU and 3 core areas are identified within the Klamath River IRU. Of the 97 core areas in the Columbia River IRU, 16 core areas within 11 river basins will be adversely affected by the proposed action. Two of the three core areas in the Klamath River IRU will be adversely affected by the proposed action. A summary of how these river basins and bull trout core areas are likely to be affected is presented below.

Columbia River IRU Core Areas

Grande Ronde River

The Grande Ronde River Basin contains two core areas: the Grande Ronde and the Little Minam. These core areas contain 10 local populations: Upper Grande Ronde River, Catherine Creek, Indian Creek, Minam River/Deer Creek, Lostine River/Bear Creek, Upper Hurricane Creek, Wenaha River, South Fork Wenaha River, Butte and West Fork Butte creeks, and Lookingglass Creek. These core areas contain approximately 658 miles of bull trout habitat. About 107.1 miles or about 16% of the total bull trout habitat in this river basin are likely to be adversely affected by the proposed action. Some local populations of the bull trout in this river basin are at an increased risk of exposure and others will have little to no risk of exposure. It is likely that exposure to the proposed toxic concentrations may kill or injure some bull trout (see Table 4-17 and Table 4-18), a large number of bull trout within this river basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Hells Canyon Complex

This area contains one core area comprised of seven local populations: Wildhorse, Indian, Upper Pine Creek, Clear, East Pine, Lake Fork, and Elk Creeks, of which five may be affected by the

proposed action. Bull trout within Indian and Wildhorse creeks will not be affected by the proposed action. Overall, about 22% of existing bull trout habitat within the Hell's Canyon Complex may be affected by the proposed action. Some local populations of bull trout in this area are at an increased risk of exposure and others will have little to no risk of exposure. It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Hood River

The Hood River basin contains one core area comprised of two local populations: the Hood River and Clear Branch. There are approximately 79.6 miles of bull trout habitat within this basin, of which 16% may be adversely affected by the proposed action. There is no spawning and rearing habitat that could be exposed to contaminants in this core area, all exposure is expected to occur in foraging, migratory and overwintering habitat. Both local populations of the bull trout in this basin have some risk of exposure, but the vast majority of bull trout habitat within the basin will not be exposed. It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Imnaha River

The Imnaha River basin contains one core area comprised of seven local populations: Upper Imnaha River, Lower Imnaha River, Upper Big Sheep Creek, Lower Big Sheep Creek, Upper Little Sheep Creek, Lower Little Sheep Creek and McCully Creek. There are approximately 177.6 miles of bull trout habitat within this basin, of which 33% may be adversely affected by the proposed action. Some local populations of bull trout in this basin are at an increased risk of exposure and others will have little to no risk of exposure. It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

John Day River

This Basin contains three core areas: the Middle Fork John Day River, The North Fork John Day River and the Upper Mainstem John Day River. It also contains foraging, migratory, and overwintering habitat in the lower mainstem John Day River.

The Middle Fork John Day Core Area contains three local populations: Granite/Boulder Creeks, Clear Creek, and Big Creek. Bull trout within the Middle Fork John Day Core Area may experience adverse effects over approximately 22.4 miles of habitat; 5 miles are spawning and rearing habitat, and the remaining 17.4 miles are foraging, migratory, and overwintering habitat.

The North Fork John Day River contains seven local populations: Desolation Creek above the falls, Desolation Creek below the falls, Granite Creek, Clear Creek, Crane Creek, Onion Creek, and the upper mainstem North Fork John Day River. Bull trout within the North Fork John Day Core Area may experience adverse effects in approximately 43.2 miles of spawning and rearing habitat, and 57.8 miles of foraging, migratory, and overwintering habitat.

The Upper Mainstem John Day River core Area contains two local populations: Indian Creek and the Upper Mainstem John Day River. Bull trout within the Upper Mainstem John Day Core Area may experience adverse effects over 0.5 miles of spawning and rearing habitat, and 33 miles of foraging, migratory, and overwintering habitat.

Overall, this basin contains a total of approximately 648.6 miles of bull trout habitat, of which about 24% is likely to be affected by the proposed action.

It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Lower Deschutes River

The Deschutes River Basin below Big Falls contains one core area: The Lower Deschutes River Core Area. This core area contains six local populations: Warm Springs River, Shitike Creek, Whitewater River, Jefferson, Candle Complex, and Jack Creek/Canyon Creek/Heising Creek. There are approximately 287.8 miles of bull trout habitat in this basin. Bull trout may experience adverse effects over about 4.4 miles of spawning and rearing habitat and 8.2 miles of foraging, migratory, and overwintering habitat. A maximum area of 12.6 miles (about 4%) is likely to be affected by the proposed action.

It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for

our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Malheur River

Bull trout within the Malheur River basin are not likely to be adversely affected by the proposed action due to the low likelihood of exposure (see exposure analysis in Section 4.4.1.1).

Odell Lake

Bull trout in the Odell Lake core area are not likely to be adversely affected by the proposed action due to the low likelihood of exposure (see exposure analysis in Section 4.4.1.1).

Powder River

The Powder River basin contains one core area, including up to ten local populations: Lake Creek, Upper Powder River, Rock Creek, Big Muddy Creek, Salmon Creek, Pine Creek, North Powder River, Anthony Creek, Indian Creek (Oregon), and Wolf Creek. There are approximately 184.3 miles of bull trout habitat within this basin. Bull trout within this core area may experience adverse effects over about 38.4 miles of spawning and rearing and 42 miles of foraging, migratory, and overwintering habitat. Approximately 44% (80.4 miles) of bull trout habitat is likely to be exposed to the proposed action.

It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Umatilla River

The Umatilla River basin contains one core area, including two local populations: North Fork Meacham Creek and the Upper Umatilla River. There are approximately 131.6 miles of bull trout habitat within this basin. About 37.5 miles (28%) could be exposed to the proposed action. All affected habitat consists or foraging, migratory and overwintering habitat. Although the bull trout population in this basin is at an extremely low level, it is unlikely that the affected local populations would be extirpated as a result of the proposed action because the effects would likely not occur in spawning and rearing habitat.

It is likely that exposure to the proposed toxic concentrations may kill or injure some bull trout (see Table 4-17 and Table 4-18. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst

case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Upper Willamette River

The Upper Willamette River Basin contains one core area, including four local populations: Upper Middle Fork Willamette River, McKenzie River, Trail Bridge, and South Fork McKenzie River. There are approximately 191.4 miles of bull trout habitat, of which about 40% may be exposed by the proposed action.

It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Walla Walla River

The Walla Walla River Basin contains two core areas, the Walla Walla River and the Touchet River, which includes 3 local populations: South Fork Walla Walla River, Mill Creek, and Low Creek. Approximately 281.1 miles of bull trout habitat exist within this basin. In the Walla Walla River Core Area 20.5 miles (about 7%) of the bull trout habitat may be exposed to the proposed action. Of these affected miles, 5.7 miles are spawning and rearing habitat, and the remaining 14.8 miles are foraging, migratory, and overwintering habitat. No habitat within the Touchet River Core Area will be affected by the proposed action because it is located outside of the action area (in the State of Washington).

Some local populations of bull trout in this basin are at an increased risk of exposure and others will have little to no risk of exposure. It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Klamath River IRU

Klamath River

The Klamath River Basin contains three core areas: Upper Sycan River, Upper Klamath Lake, and the Upper Sprague River Core Areas. The Upper Sycan River Core Area contains one local

population in Long Creek. The Upper Klamath Lake contains three local populations: Threemile Creek, Sun Creek, and Lost Creek. The Upper Sprague River Core Area contains four local populations: Boulder/Dixon creeks, Brownsworth Creek, Deming Creek, and Leonard Creek. Overall, this basin contains approximately 276.6 miles of bull trout habitat. Bull trout within the Upper Sycan River Core Area will not experience adverse effects from the proposed action because there is no potential for exposure. Bull trout within the Upper Klamath Lake Core Area will likely experience adverse effects over 1 mile of spawning and rearing habitat, and 5.5 miles of foraging, migratory, and overwintering habitat. Bull trout within the Upper Sprague River Core Area will likely experience adverse effects over 5.5 miles of spawning and rearing habitat, and in 8 miles of foraging, migratory, and overwintering habitat. The maximum total of likely adverse effects should not exceed 19 miles (about 6% may be affected by the proposed action.

It is likely that exposure to the proposed toxic concentrations will kill or injure some bull trout (see Table 4-17 and Table 4-18), but a large number of bull trout within this basin would not be adversely affected by the proposed action. As the aforementioned tables elucidate, the basis for our number of bull trout affected is on the area exposed to the contaminant and estimate of adult bull trout utilizing that habitat. Sufficient numbers of bull trout are likely to survive (under the worst case scenario used in our analysis) to refound any affected areas, and support a persistent population of the bull trout in this basin. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

6.1.1 Evaluation of Adverse Effects caused by the Proposed Action at the Interim Recovery Unit Scale

Columbia River Interim Recovery Unit

In evaluating effects by core area and river basin level (as applicable), we have determined that bull trout in two core areas within the Columbia River Interim Recovery Unit have exposure risks where at least 40% of their current habitat could be affected through exposure. These two core areas display a higher degree of risk due to the effects of the proposed action than any of the other core areas in the action area. These include local populations in the Powder River and Upper Willamette River core areas.

The total miles of bull trout habitat exposed under the worst case scenario for implementation of the proposed action within the Powder River and Upper Willamette River core areas is approximately 158.5 miles. The Columbia River Interim Recovery Unit contains over 17,600 miles of bull trout habitat. The portion we are evaluating from these two core areas combined amounts to less than 1% of the total habitat available for bull trout. To take this a step further, if we include the total potentially exposed habitat for all core areas in this analysis that fall within the Columbia River Interim Recovery Unit, it amounts to about 800 miles, which is less than 5% of the total bull trout habitat within the Columbia River Interim Recovery Unit. In addition, we believe that sufficient numbers of bull trout are expected to persist with implementation of the proposed action such that no extirpation of the bull trout from any core area is likely to occur.

Thus, given the low percentage of habitat exposed at higher levels (relative to the overall occupied habitat) and low risk of extirpation from any one core area, the proposed action is not likely to appreciably impair, nor will it preclude the capacity of the Columbia River Interim Recovery Unit from providing both the survival and recovery function assigned it (i.e., maintaining or enhancing core area populations to provide distribution, genetic, and demographic support to insure persistence of the bull trout within this portion of its range, as listed).

The above determination is based on the following findings:

- While we are unable to quantify effects to juvenile bull trout and bull trout eggs, it is our best professional judgment that the use of estimated numbers of adult bull trout affected by the proposed action likely overestimates possible effects to local populations, and represents a reasonable worst case analysis of effects.
- All 12 river basins and 16 bull trout core areas affected by the proposed action, inclusive of effects that may or are likely to kill or injure bull trout and taken together with cumulative effects, are likely to continue to function in a manner that supports, in part, the survival and recovery of role of the Columbia River Interim Recovery Unit for the bull trout because sufficient amounts of suitable habitat supporting sufficient numbers of adult bull trout are likely to persist over the term of effects caused by the proposed action. The size, distribution, and reproductive capacity of these residual populations are likely to provide for the refounding of local populations that may or are likely to be diminished by the effects of the proposed action. In addition, it is unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously.

Klamath River Interim Recovery Unit

Two bull trout core areas in the Klamath River Basin may be adversely affected by the proposed action to an extent that some of the affected bull trout may be killed or injured. These effects may occur within 19 miles (6%) of bull trout habitat within this basin. Such an effect, taken together with cumulative effects, is not likely to appreciably impair or preclude the Klamath River Interim Recovery Unit from providing both its survival and recovery function as intended because sufficient amounts of suitable habitat supporting sufficient numbers of adult bull trout are likely to persist over the term of effects caused by the proposed action. It is our best professional judgment that the size, distribution, and reproductive capacity of these residual populations will be able to provide for the refounding local populations that may or are likely to be diminished by the effects of the proposed action.

After reviewing the current status of the bull trout, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is USFWS's biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the bull trout.

6.2 Bull Trout Critical Habitat

As stated in the Status of the Species section for bull trout, this Biological 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.

In all, portions of 11 of the 13 critical habitat units (CHUs) in Oregon will be affected by implementation of the proposed action. Additionally portions of critical habitat (CH) within the Columbia River and Snake River will also be affected. These effects are not evenly distributed between the CHUs, nor are they evenly distributed between core areas within individual CHUs. Some local bull trout core areas in a given CHU are at an increased risk of exposure and others will have little to no risk of exposure. Therefore, portions of an affected CHU may have reduced capacity to provide an abundant food base, including terrestrial organisms of riparian origin, aquatic macro invertebrates, and forage fish (PCE3), and sufficient water quality such that normal reproduction, growth, and survival are inhibited (PCE8).

The Malheur River CHU and Odell Lake CHU have no risk of exposure. The Klamath River (6%), Lower Deschutes River (4%) and Walla Walla River (7%) CHUs have less than 10% of their CH at risk of exposed by the proposed action: the Grande Ronde River (16%), Hells Canyon Complex (22%), Hood River (16%), Imnaha River (33%), John Day River (24%), Powder River (44%), Umatilla River (28%), and Upper Willamette River (40%) comprising the group of CHUs with greater than 10% of their CH at risk of exposure.

As discussed there will be adverse effects to bull trout critical habitat. Of the nine Primary Constituent Elements (PCEs) for bull trout critical habitat, two will be affected. The affected PCEs are PCE 3 (an abundant food base, including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish), and PCE 8 (sufficient water quality and quantity such that normal reproduction, growth, and survival are not inhibited).

In all CHUs the majority of CH within them are not at 50% or more risk of exposure. The largest portion of each individual CHU within the action area of the proposed action will likely be unaffected. We note that 2 CHUs: the Powder River and the Upper Willamette River have the highest risk of exposure with 40-44% of the CH exposed under the proposed action. Since the exposure even in these two areas is not widespread, would not be for more than 4 days in any 3 year period, nor in all spawning areas, it is anticipated that these CHUs will continue to function and provide adequate contributions towards both survival and recovery of the species. In addition, it is highly unlikely that the exposure of any of the contaminants would be widespread and occur simultaneously. Additionally, while the proposed action will adversely affect PCEs 3 and 8, it will have no effect on the remaining 7 PCEs for bull trout. PCEs 3 and 8 will be degraded in portions of these CHUs, but will not be totally absent. Almost 60% of the bull trout habitat within these two CHUs would remain unaffected, allowing for continued recruitment and dispersal of bull trout throughout the CHUs. We anticipate that all 11 affected CHUs will continue to function in their intended conservation and recovery roles. When we compare the total CH affected by this proposed action (820 miles) it is about 4% of the total stream miles contained in the bull trout CH rangewide.

Based on the factors described above, we conclude that all of the CHUs affected by the implementation of the proposed action will continue to function as intended and will continue to

contribute to the survival and recovery of bull trout both at the CHU and coterminus scales because two of the nine PCEs (3 and 8), will be adversely affected only in the Powder River CHU and the Upper Willamette CHU. Most of the habitat will remain unaffected within these two CHUs (56% and 60% respectively) and the remaining PCEs within these CHUs will be unaffected by the proposed action. The Klamath River CHU may also experience adverse effects to both PCE 3 and PCE 8. At the coterminous scale, designated bull trout critical habitat will continue to function for both survival and recovery of bull trout.

We therefore conclude that the proposed action will not adversely modify or destroy bull trout critical habitat.

After reviewing the current status of the bull trout, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is USFWS's biological opinion that the action, as proposed, is not likely to destroy or adversely modify designated critical habitat.

6.3 Oregon Chub

After reviewing the current status of Oregon chub, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is the USFWS's biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Oregon chub. This conclusion is warranted because:

- Although adverse effects to the Oregon chub are likely to occur and cause mortality and injury of exposed individuals due to reduced fitness in three of the four subbasins constituting the current range of the Oregon chub, these effects are likely to occur to only a small portion of the chub population within each subbasin and rangewide (Table 6-1). Loss of the affected populations will not preclude achievement of the delisting criteria for the species.

Table 6-1. Summary of adverse effects to Oregon chub (percent of total number of chub affected in each subbasin based on 2011 abundance estimates) likely to be caused by the proposed action.

Subbasin	Percent of Oregon Chub (per subbasin) Affected		
	Exposure to Acute Criteria: Mortalities	Exposure to Chronic Criteria: Mortalities	Exposure to Chronic Criteria: Sublethal Effects
Santiam River	1.1%	1.4%	0%
Mid-Willamette River (includes McKenzie River subbasin)	9.5%	10.3%	0.02%
Middle Fork Willamette River	6.3%	12.4%	10.1%

- While the aggregated effects of exposure to toxics may cause extirpation of six Oregon chub populations in the Middle Fork Willamette River subbasin and one population in the Mid-Willamette River subbasin, the abundance, distribution, and viability of Oregon chub populations which are not at risk of exposure to toxics with implementation of the proposed action are adequate to allow the species to meet the delisting criteria and to achieve recovery.

6.4 Oregon Chub Critical Habitat

After reviewing the current status of Oregon chub, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is the USFWS's biological opinion that the action, as proposed, is not likely to destroy or adversely modify critical habitat for Oregon chub. This conclusion is warranted because:

Oregon chub critical habitat PCE 3 describes an abundance of minute organisms (such as rotifers, copepods, cladocerans, and chironomid larvae) as a characteristic of habitat necessary to sustain the essential life history functions of the species. Many invertebrates are able to tolerate toxicity levels higher than the proposed standards considered herein (USEPA 2008); however, a reduction in the quantity of prey species available in Oregon chub critical habitat may occur during periods of acute or chronic exposure. If this does occur, it is unlikely to affect all prey species and it is likely that invertebrates will quickly recolonize the habitat. Given this and the variety of prey species consumed by Oregon chub, we expect any reduction in prey to be short-term. On that basis, the effects of the proposed action, taken together with cumulative effects, are not likely to appreciably impair the capability of Oregon chub critical habitat to support conservation of the Oregon chub, and thus is not likely to adversely modify or destroy Oregon chub critical habitat.

6.5 Lost River Sucker and Shortnose Sucker

After reviewing the current status of the Lost River sucker and shortnose sucker, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is USFWS's Biological Opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Lost River sucker or shortnose sucker.

As a result of the proposed action, we determined the following population-level effects are likely to occur: up to 100% of all life stages of the Lost River sucker and up to 100% of all life stages of shortnose sucker in the Keno Reservoir, up to 19% of young Lost River sucker and up to 19% of young shortnose sucker in the Lost River, and up to 5% of young Lost River sucker and up to 5% of young shortnose sucker and <1% of adult Lost River sucker and shortnose sucker in Upper Klamath Lake will likely be killed or harmed every 3 years over the life of the proposed action. These losses are not likely to jeopardize the continued existence of the Lost River sucker or shortnose sucker because:

- The Lost River sucker and shortnose sucker populations in the Keno Reservoir are non-reproducing, sink-populations, meaning suckers there originate elsewhere, and therefore they have only a small effect on survival and recovery of the species;

- Lost River sucker and shortnose sucker in the affected portions of the Lost River, are in low abundance because of a lack of spawning and rearing habitat and therefore they have only a small effect on survival and recovery of the species;

- The small level of effect to Lost River sucker and shortnose sucker in Upper Klamath Lake as a result of exposure to ammonia is unlikely to be substantial because the populations, although declining, are still relatively large, numbering in the tens of thousands, and our estimates of effects represent a worst case analysis;

- Our analyses of effects from exposure to the chemicals addressed in this consultation have been conservative to the species, and are based on a reasonable worst case analysis of the affected habitat being at criterion concentrations, which, in practice, is not likely to occur.

6.6 Lost River Sucker and Shortnose Sucker Proposed Critical Habitat

It is our conference opinion that the proposed action is not likely to destroy or adversely modify the proposed critical habitat for the Lost River sucker or shortnose sucker because the proposed action will have only minor affects to Unit #1-Upper Klamath Lake, Agency Lake and the upper Klamath River to Keno. Effects of the proposed action are likely to degrade the quality of habitat through effects to PCE 1 (water) and PCE 3 (food). Our analysis determined that the most extreme effect of the proposed action that is reasonably foreseeable will be to allow concentrations of ammonia in the three major Upper Klamath Lake bays to reach levels sufficient to affect the PCEs. The area of these bays is equal to approximately 15% of the total area of Upper Klamath Lake habitat, or about 12% of Unit #1. The events creating toxic ammonia concentrations would likely be temporary, perhaps lasting several weeks or less, based on the historic ammonia data discussed above.

Based on the information presented above, the proposed action will also likely adversely affect PCE 1(water) and PCE 3(food) in the Keno Reservoir. We do not anticipate that this would lead to adverse modification of the proposed critical habitat because Keno Reservoir (~2,500 acres) represents a small proportion, (only about 3% of the species total habitat of ~80,000 acres), and habitat there is highly degraded for a variety of reasons, but especially from the input of algae from upstream in Upper Klamath Lake which leads to severe water quality problems in most years. The adverse effect to PCE 3 (food) is not likely substantial because of the high productivity of the reservoir and input of food from upstream in Upper Klamath Lake.

6.7 Vernal pool fairy shrimp

After reviewing the current status of the listed species, the environmental baseline for the Action Area, effects of the Proposed Action and cumulative effects, it is USFWS's biological opinion that the Proposed Action is not likely to jeopardize the continued existence of the vernal pool fairy shrimp.

Our finding of no jeopardy is supported by the following:

- The overall condition of the four affected areas have been severely degraded due to past landfill operations, debris burial, development for recreational use, and development. The areas represent poorly functioning habitat with low occupancy of vernal pool fairy shrimp. In their present state, the four sites are of limited ecological and conservation value.
- While we are unable to quantify effects to individuals, we believe that less than 1% of vernal pool fairy shrimp rangewide will be exposed to toxics.

6.8 Vernal Pool Fairy Shrimp Critical Habitat

After reviewing the current status of the listed species, the environmental baseline for the Action Area, effects of the Proposed Action and cumulative effects, it is USFWS's biological opinion that the Proposed Action is not likely to destroy or adversely modify designated critical habitat.

Our finding of no adverse modification or destruction is supported by the following:

Up to 272 acres of vernal pool fairy shrimp critical habitat will be exposed to toxics; this is only 3.6% of the vernal pool fairy shrimp critical habitat in Oregon and less than 1% of the vernal pool fairy shrimp critical habitat rangewide.

6.9 Marbled Murrelet

After reviewing the current status of marbled murrelet, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is USFWS's biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of marbled murrelets.

This no jeopardy finding is supported by the following:

- Although exposure to selenium through bioaccumulation has the potential to cause biological effects to the marbled murrelet that conform to the regulatory definition of an adverse effect, the mere potential for an adverse effect does not mean it is a significant disruption to breeding. Therefore, neither harm nor harassment to the marbled murrelet is expected. At the conservation zone and range wide scales, and at current levels of exposure, we believe the potential for effects will not be able to be measured and therefore is not likely to jeopardize the species.

6.10 Marbled Murrelet Critical Habitat

After reviewing the current status of marbled murrelet, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, it is USFWS's biological opinion that the action, as proposed, is not likely to destroy or adversely modify designated critical habitat.

This no adverse modification finding is supported by the following:

The PCEs of marbled murrelet critical habitat focused on individual trees with potential nesting platforms, and forested areas within 0.8 kilometers (0.5 miles) of individual trees with potential nesting platforms that had a canopy height of at least one-half the site potential tree height. This habitat will not be exposed to the toxics addressed in this Biological Opinion.

7 INCIDENTAL TAKE STATEMENT

Section 9 of the Act and Federal regulation pursuant to section 4(d) of the Act prohibit the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct. Harm is defined by USFWS as an act which actually kills or injures wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavior patterns, including breeding, feeding, or sheltering (50 CFR 17.3). Harass is defined by USFWS as an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3). Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the Act provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

The measures described below are non-discretionary, and must be undertaken by USEPA so that they become binding conditions of any grant or permit issued to ODEQ, as appropriate, for the exemption in section 7(o)(2) to apply. The USEPA has a continuing duty to regulate the activity covered by this incidental take statement. If USEPA: (1) fails to assume and implement the terms and conditions, or (2) fails to require the State to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, USEPA must report the progress of the action and its impact on the species to the USFWS as specified in the incidental take statement (50 CFR 402.14[i][3]).

7.1 Amount of Take

The USFWS anticipates that take of bull trout, Oregon chub, Lost River sucker, shortnose sucker, and vernal pool fairy shrimp may occur in the form of harm or harassment. Our estimate of the specific amounts of incidental take associated with the proposed action is presented below.

7.1.1 Marbled Murrelet

The USFWS does not anticipate the proposed action of approving selenium criteria will incidentally take marbled murrelets. Although the "Effects of the Action" section above includes a finding that implementation of the proposed action has the potential to cause biological effects to the marbled murrelet that conform to the regulatory definition of take, the mere potential for take is not a legitimate basis for a take exemption. Since no take is anticipated or exempted, no

reasonable and prudent measures and terms and conditions are provided below. If take is detected during implementation of the proposed action, reinitiation of formal consultation should be requested, and any operations causing such take must cease pending the outcome of the reinitiated consultation.

7.1.2 Bull Trout

The USFWS anticipates the following incidental take of bull trout to occur as a result to exposure to toxic chemicals at the acute and chronic criteria:

Acute -- Our analysis indicates that if exposure to the acute criteria for all toxics were to occur concurrently, there is a maximum likelihood for lethal harm to occur to 813 adult bull trout every 3 years (Table 4-17). This amounts to 8.8% of the adult bull trout population within the action area, and associated with 820.6 miles of habitat.

Chronic -- Our analysis indicates that if exposure to the chronic criteria for all toxics were to occur concurrently there is a maximum likelihood that 1,116 bull trout may be killed (12.2% of the bull trout adult population within the action area) and 6,953 bull trout will experience reduced fitness due to reductions in growth, or reductions in reproduction every 3 years (Table 4-18). These effects are associated with 820.6 miles of bull trout habitat.

7.1.3 Oregon Chub

In order to estimate the effects of this proposed action on individuals we have relied on 2011 Oregon chub abundance data collected by ODFW (Bangs et al. 2011). With our continued recovery efforts, we expect to see Oregon chub abundance continue to increase rangewide. If the rangewide abundance of Oregon chub does increase above the 2011 abundance estimates, it is possible that the number of individuals taken could exceed the number of individuals we have anticipated in the Amount of Take section below. Since we expect that the same levels of effect we discussed in the Effects Section of the Biological Opinion (e.g., 5.95 % reduction in survival, 17.9% reduction in hatching success, etc.) will occur for each toxic, an increase in Oregon chub abundance would result in an increased number of harmed individuals. Thus, it is our intent that anticipated take per 3-year period for the 25-year life of the Biological Opinion is based on the levels of effect percentages for each chemical.

Our analysis indicates exposure of Oregon chub populations to the acute criteria for toxics will occur in the following four combinations: ammonia only; ammonia, copper, lead and zinc; copper, lead, and zinc; or ammonia, arsenic, cadmium, chromium III and VI, copper, lead, nickel, selenium, silver, and zinc (Table 4-55). The combined levels of effect percentages are as follows in Table 7-1.

Table 7-1. Summary of harm to Oregon chub from exposure to toxics.

Toxic Exposure	Percent of Exposed Population Harmed		
	Exposure to Acute Criteria:	Exposure to Chronic Criteria:	Exposure to Chronic Criteria:
	Lethal Harm	Lethal Harm	Sublethal Harm
NH^3	10%	11%	up to 100%
NH^3, Cu, Pb, Zn	10%	19%	up to 100%
Cu, Pb, Zn	Discountable	8%	up to 100%
Ag, As, Cd, CrIII, CrVI, Cu, NH^3, Ni, Pb, Se, Zn	60%	100%	up to 100%

By estimating take using these levels of effect percentages and the 2011 Oregon chub abundance estimates, USFWS anticipates the following incidental take of Oregon chub to occur as a result to exposure to toxic chemicals at the acute and chronic criteria:

Acute -- Up to 18,849 adult Oregon chub are likely to be killed per 3-year period over the 25-year life of this Biological Opinion due to exposure at acute criterion levels (Table 4-57). This amounts to 8.2% of the adult Oregon chub population rangewide.

Chronic – Up to 23,266 Oregon chub are likely to be killed (10.1% of the Oregon chub adult population rangewide) and 5,206 Oregon chub will experience harm in the form of reduced fitness due to reductions in growth and reproduction per 3-year period over the 25-year life of this Biological Opinion due to chronic criterion levels (Table 4-58).

7.1.4 Lost River Sucker and Shortnose Sucker

The USFWS anticipates the following incidental take of Lost River suckers and shortnose suckers to occur as a result to exposure to toxic chemicals at the acute and chronic criteria:

7.1.4.1 Lost River Sucker

Up to 100% of all life stages of Lost River suckers in the Keno Reservoir; 19% of young Lost River suckers in the Lost River; up to 5% of young Lost River suckers in Upper Klamath Lake; and <1% of adult Lost River suckers in Upper Klamath Lake, will be likely killed or harmed every 3 years over the 25-year life of the proposed action. The exact numbers of suckers incidentally taken as a result of the proposed action and total, range-wide populations sizes are unknown.

7.1.4.2 Shortnose Suckers

Up to 100% of all life stages of shortnose suckers in the Keno Reservoir; 19% of young shortnose suckers in the Lost River; up to 5% of young shortnose suckers in Upper Klamath Lake; and <1% of adult shortnose suckers in Upper Klamath Lake, will be likely killed or

harmed every 3 years over the 25-year life of the proposed action. The exact numbers of suckers incidentally taken as a result of the proposed action and total, range-wide populations sizes are unknown.

7.1.5 Vernal Pool Fairy Shrimp

The USFWS anticipates incidental take of vernal pool fairy shrimp from exposure to acute and chronic criteria levels of toxics to include lethal harm, injury, and harassment of individuals, for both cysts and adults. The best scientific and commercial data available are not sufficient to enable us to estimate a specific amount of fairy shrimp adults or cysts that will be taken associated with the exposed acreage. However, the extent of exposed habitat is less than 1% of the total habitat rangewide; therefore we anticipate that less than 1% of the entire vernal pool fairy shrimp population will be harmed by the proposed action every 3 years over the 25-year life of the proposed action.

7.2 Summary

In the analysis above, we have used the best available information to estimate the expected take associated with the proposed project. However, given the large geographical scale of the proposed project, we anticipate that incidental take of bull trout, Oregon chub, Lost River sucker, shortnose sucker and vernal pool fairy shrimp will be difficult to detect and measure as finding dead or impaired specimens is unlikely. Furthermore, we do not expect that it will be possible to detect chronically or sublethally affected individuals attributable to the proposed action. Not only are these effects difficult to detect in the complex natural environment, but take associated with the proposed project may be masked by seasonal fluctuations in numbers or other confounding effects. Therefore, we will use quantitative measurements of ambient concentrations of criteria chemicals within the action area as a surrogate for the extent of incidental take due to individual criteria.

7.3 Effect of the Take

In the accompanying Biological and Conference Opinion, USFWS determined that this level of anticipated take is not likely to result in jeopardy to the bull trout, Oregon chub, shortnose sucker, Lost River sucker, or vernal pool fairy shrimp or result in destruction or adverse modification of critical habitat or proposed critical habitat for any of these species. These reasonable and prudent measures, and their implementing terms and conditions, are designed to minimize the impact of incidental take that might otherwise result from the proposed action.

7.4 Reasonable and Prudent Measure

The USFWS believes the following reasonable and prudent measure is necessary and appropriate to minimize take of bull trout, Oregon chub, Lost River sucker, shortnose sucker, and vernal pool fairy shrimp:

1. USEPA will provide to USFWS a report every two years summarizing ODEQ's monitoring efforts included in the action under Conservation Measures. The USEPA's report will focus on the waterbodies and water quality criteria relevant

to the listed species addressed in this Biological Opinion. This reportage will tell us if the listed species is experiencing criteria concentration in their natural habitats.

2. USEPA will develop methods to estimate toxicity of contaminants to the vernal pool fairy shrimp. These methods will be used to test vernal pool fairy shrimp alongside two commonly used surrogate species to compare sensitivities to a series of five chemicals. Information gathered in this reasonable and prudent measure will validate the assumption that invertebrates in the family Daphnidae are appropriate surrogates for assessing the protectiveness of Oregon's ambient water quality criteria.

7.5 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, USEPA must comply with the following term and condition which implements the reasonable and prudent measure described above. Terms and conditions are non-discretionary.

A. This term and condition applies to Reasonable and Prudent Measure 1. USEPA's action includes monitoring required under the CWA. Oregon provides this information to USEPA for their 303(d) list and review of water quality standards (including criteria). Although the scope of the monitoring is broad, information collected for this effort can be applied to assess compliance with the exempted incidental take associated with this action. The USEPA will compile a report to USFWS for those waterbodies containing bull trout, Oregon chub, Lost River sucker, shortnose sucker, and vernal pool fairy shrimp. The report will specify which waterbodies are not in compliance with Clean Water Act (CWA) regulations and which the State has not added to the 303(d) list or developed a TMDL. The USEPA will provide a copy of the results and report from these monitoring efforts to USFWS within 3 months of ODEQ's submittal of its 303(d) report. This report will inform us that the water quality criteria that were analyzed in this Biological Opinion are being adhered to and the incidental take is not being exceeded.

B. This term and condition applies to Reasonable and Prudent Measure 2. USEPA will work with research partners at Gulf Ecology Division, Gulf Breeze, Florida and USGS, Columbia Environmental Research Center, Columbia, Missouri. Toxicity tests will be conducted with standard test organisms from Branchiopod taxa (i.e., *Dahpnia* and *Ceriodaphnia*) and fairy shrimp of the genus *Branchinecta*. Tests will be conducted on five chemicals with varying modes of action including candidates of two subjects of this consultation, copper and pentachlorophenol. Data gathered from these tests will be used to develop ICE models for the vernal pool fairy shrimp. Results from this study will provide feedback in assessing the validity of the assumptions inherent in our analyses. If our assumptions are found to be invalid, it would tell us that the surrogate analysis was inappropriate and additional analyses will be required.

8 CONSERVATIONS RECOMMENDATIONS

Section 7(a)(1) of the Act directs Federal agencies to use their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

1) Fund research with the appropriate agency to investigate assumptions and uncertainties (see Appendix 1) inherent in the assessment of toxic chemical water quality criteria to endangered species.

2) Develop with qualified experts, and in coordination with USFWS, an ecological risk population model(s) for water quality criteria for chemicals under consultation.

3) Work with ODEQ to assess the feasibility of implementing biotic ligand models for all applicable chemicals under consultation.

4) Monitor and improve techniques (e.g., bioswales, vegetated buffers) to control and reduce chemical exposure resulting from nonpoint source pollution to listed species.

5) Monitor levels of cadmium, copper, silver and zinc in the most affected bull trout river basins (e.g., Powder, Imnaha, Upper Willamette and John Day River basins).

6) Expand direct toxicity testing with juvenile and adult bull trout to include chemicals in this consultation lacking direct data, such as arsenic, chromium VI and silver.

7) Work with ODEQ to develop a monitoring program for toxic pollutants that is designed to measure particular point-source discharge compliance with aquatic life criteria.

In order for USFWS to be kept informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, USFWS requests notification of the implementation of any conservation recommendations.

9 REINITIATION STATEMENT

This concludes formal consultation on the action outlined in USEPA's BE. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this Biological Opinion, (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this Biological Opinion, or (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.

This also concludes the conference on proposed critical habitat designations for Lost River sucker and shortnose sucker for the action outlined in USEPA's BE. You may ask USFWS to confirm the Conference Opinion as a biological opinion issued through formal consultation if the critical habitat is designated. The request must be in writing. If USFWS reviews the proposed action and finds that there have been no significant changes in the action as planned or in the information used during the conference, USFWS will confirm the Conference Opinion as the biological opinion on the project for sucker critical habitat and no further section 7 consultation will be necessary. After designation of critical habitat for Lost River sucker and shortnose sucker and any subsequent adoption of this Conference Opinion, USEPA shall request reinitiation of consultation if: (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action that may affect the species or critical habitat in a manner or to an extent not considered in this Conference Opinion, (3) the agency action is subsequently modified in a manner that causes an effect to the species or critical habitat that was not considered in this Conference Opinion, or (4) a new species is listed or critical habitat designated that may be affected by the action.

10 REFERENCES CITED

Note: References in this Biological Opinion that appear in directly quoted or other material from either the USEPA's Biological Evaluation (USEPA 2008) or the California Toxics Rule (USFWS and NMFS 2000) are not included in this References Cited section. However, those references are listed in the USEPA's Biological Evaluation or the California Toxics Rule, both of which are provided as a courtesy at the following FTP site:
http://www.fws.gov/filedownloads/ftp_OFWO/EPA%20Toxics%20BO/

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Appendix 1

Detailed methodologies used to characterize exposure and ecological effects are described in this appendix. In addition, this appendix includes a discussion of the uncertainties within the risk assessment models used in the BE along with a description of our approach to refining them. Finally, a description of our approach used to estimate a level of effect from acute and chronic criteria is contained in this appendix. This information is essential to understanding our effects determinations contained in Section 4 of this Biological Opinion.

1 Evaluation of Contaminant Sources

To evaluate the potential for exposure from specific sources a variety of databases/sources of information were reviewed including: (1) ODEQ databases on discharge permits and water-quality limited streams, (2) the Oregon Department of Geology and Mineral Industries (DOGAMI) data on mine sites in Oregon (sites were mapped using GIS and reviewed for proximity to listed species habitats), (3) GIS data layers of adjacent land use activities, and (4) USFWS lead staff biologists for the affected listed species and various State agencies to gather additional information. Each of these is discussed in more detail below.

1.1 ODEQ Databases

1.1.1 Water-quality limited streams

Information from the 303(d) list is included in our baseline information and also considered here for exposure assessment. Every 2 years, ODEQ is required to assess water quality and report to USEPA on the condition of Oregon's waters. ODEQ prepares an Integrated Report that meets the requirements of the federal CWA for Section 305(b) and Section 303(d). Section 305(b) requires a report on the overall condition of Oregon's waters, while Section 303(d) requires identifying waters that do not meet water quality standards and thus where a TMDL needs to be developed. An Integrated Report includes an assessment of each water body where data are available, and the list of waters identified under Section 303(d) as water-quality limited and needing a TMDL.

ODEQ submitted Oregon's 2010 Integrated Report and 303(d) list to USEPA in May 2011 (ODEQ 2011b), and USEPA took action approving the submitted 303(d) listings and de-listings on March 15, 2012. The USEPA disapproved ODEQ's submittal for not including other waters and proposed adding other waters to Oregon's 303(d) list. Once the process for public comment is completed and USEPA finalizes its decisions, ODEQ will update Oregon's 2010 303(d) list with USEPA's additions. Because the 2010 Oregon Integrated Report contains the most up-to-date information on the condition of Oregon waters, we used this report in gathering information for our exposure evaluation although its approval is not official yet. Additional waters proposed by USEPA were reviewed and did not affect our analysis.

In the Integrated Report Database, ODEQ assigned an assessment status category to each water body where data were available to evaluate. Category 2 (attaining—specific water quality standards are met) and Category 3 (insufficient data to determine whether a standard is met) were not included in the information presented for each species. The specific regulatory categories from the 303(d) list which were used in our exposure evaluation include the

following: category 3B— potential concern; category 4A— water-quality limited with an approved TMDL; category 4B— water-quality limited with other control measures; and category 5— water-quality limited on the 303(d) list awaiting an approved TMDL. Any streams listed for flow or habitat modification or sedimentation were not included as these are physical limitations that cannot be evaluated properly without greater knowledge of the impairment and the aquatic system.

1.1.2 Permits

One of the primary regulatory mechanisms of States to enact the criteria is through their NPDES program. An ODEQ water quality permit is required whenever there is a discharge of pollutants into waters of the State or to the ground. Waters of the State include surface waters (wetlands, pond, lakes, streams, rivers, etc.) and groundwater. Permits are required for discharges of wastewater (sewage, processing water, etc.), wash water, and even for wastewater that may be relatively clean, such as non-contact cooling water. These discharges may occur through a variety of disposal systems including land irrigation, seep ponds, onsite sewage systems, dry wells, etc., or may discharge to surface water directly through a pipe or ditch or indirectly through a storm sewer system. Certain industries and activities may also be required to obtain permits for stormwater runoff from their properties.

There are two types of water quality permits regulated by ODEQ and issuance of the permit depends on what is being discharged and where the discharge occurs.

NPDES permit
The NPDES permit is a requirement of the Federal Water Pollution Control Act (Clean Water Act) and Oregon law. ODEQ has been given authority from USEPA to issue these permits. NPDES permits are required for "point source" discharges of pollutants to surface waters. The term "point source" refers to a natural or human-made conveyance such as pipes culverts, ditches, catch basins or any other type of channel. NPDES permits are issued for wastewater discharges from sewage treatment plants (STPs), pulp and paper mills, and other types of businesses. This also includes discharges to the storm sewer system or to drainage ditches that eventually reach surface waters. Certain industries and activities are also required to obtain NPDES permits for stormwater runoff.

Water Pollution Control Facilities (WPCFs)
The WPCF permit is a State requirement for the discharge of wastewater to the ground; discharge to surface water is not allowed. WPCF permits are issued for land irrigation of wastewater, wastewater lagoons, onsite sewage disposal systems, and underground injection control systems (i.e., dry wells, sumps, etc.). The primary purpose of a WPCF permit is to prevent discharges to surface waters and to protect groundwater from contamination. This permit is also used to prevent nuisance conditions such as odors and mosquitoes.

Both the NPDES and WPCF programs use "individual" and "general" permits. An "individual permit" is site-specific; it is developed to address discharges from a specific facility. Individual permits are usually issued for a period of 5 years. ODEQ currently has over 800 facilities under NPDES and WPCF individual permits. Approximately 70% of these permits are for the treatment and disposal of sewage. Individual permits often require more frequent monitoring by

the permittee to assure that permit limitations are being met, as well as monitoring for a greater variety of pollutants. A "general permit" is used to cover a category of similar discharges, rather than a specific site. ODEQ may issue a general permit when there are several minor sources or activities involved in similar operations that may be adequately regulated with a standard set of conditions. A general permit is issued once and expires within 5 years. ODEQ currently uses 29 different NPDES and WPCF general permits that regulate such discharges as boiler blowdown, non-contact cooling water, wash water from vehicle and equipment cleaning, seafood processing, petroleum hydrocarbon cleanups, small domestic onsite sewage systems, etc. As WPCF permits do not allow discharge into surface waters of the State, these facilities are not considered a source of exposure in this consultation and therefore not addressed further in this analysis.

Construction activities that disturb 1 acre or more, including clearing, grading and excavation, are required to have an NPDES general permit number 1200-C. Projects that disturb 1 acre or more over a period of time or are part of a common plan are also required to apply for a permit. This permit will require that erosion at the construction site be controlled to prevent sediment from entering waters of the State. Construction permits associated with State clean-up sites where cleanup was completed over 10 years ago are excluded from bull trout exposure analysis.

Searches were conducted for all NPDES permits within the ranges of listed aquatic species using both ArcGIS layers and ODEQ website (ODEQ 2011b).

USEPA provided the NMFS and USFWS with spreadsheets of NPDES permits in Oregon containing information on specific chemical effluent limits and monitoring requirements. These spreadsheets were queried for our exposure analysis and are provided in Appendix 2.

The majority of Oregon's discharge permits concern treatment and disposal of sewage and NPDES discharges of this type are identified as STPs or wastewater treatment plants (WWTPs). Permits for these facilities often have effluent limits and monitoring requirements for a standard set of parameters including biological oxygen demand (BOD), total suspended solids (TSS), *Escherichia coli*, fecal coliform, pH, temperature, and BOD and TSS removal efficiency. Sewage or wastewater permits may also include a limit for chlorine or chlorine residual if the compound is used for disinfecting. Ammonia can be included in NPDES permits if a reasonable potential analysis determines that there is potential for the discharge to cause toxicity. An ammonia limit may also be included in a permit if the receiving stream is water-quality limited for dissolved oxygen. A review of Oregon's monitoring requirements for municipal STP or WWTP NPDES permits indicates that several of the metals which are considered in this consultation are commonly listed as a monitoring requirement, including arsenic, cadmium, chromium, copper, lead, nickel, selenium, silver, and zinc. Consequently, we considered STP and WWTP discharges to be potential sources for all these metals.

Certain chemicals are commonly associated with stormwater discharges and typically are included in NPDES permits. Chemicals associated with stormwater discharges that are considered in this consultation include arsenic, cadmium, chromium, copper, lead, nickel, and zinc (USEPA 2005b, p. 22). Thus, for the purposes of this consultation, all stormwater permits are considered to contain these chemicals.

1.2 Mines

Contamination of surface water can occur from mining processes and mills can generate large amounts of tailings which can be toxic. Waste rock from mining often contains sulfide minerals associated with metals such as lead, zinc copper, silver, or cadmium (Hudson et al. 1999, p. 24). Oxidation of pyrite, another important sulfide mineral common in waste rock, produces acidic conditions that can dissolve metals such as copper, zinc, and silver resulting in the contamination of streams (Hudson et al. 1999, pp. 24-25).

The two primary types of mining are surface (removing surface vegetation, dirt, and possibly bedrock to reach buried ore deposits) and strip (stripping surface layers off to reveal buried ore). Most placer deposits are shallow in nature and mined by surface methods while sub-surface mining consists of digging tunnels or shafts into the earth to reach deeply buried ore deposits which then brought to the surface for processing.

The Oregon DOGAMI maintains data on mine sites within the State. The information can be found through the Mineral Information Layer for Oregon, Release 2 (MILO-2) at Oregon DOGAMI's website (http://www.oregongeology.org/sub/milo/index.htm). We used this information to ascertain which mine sites exist near listed species habitats. Mine sites are categorized as coal, aggregate, industrial minerals and metals. Metal mines were the only category of interest in this consultation. Mines are also categorized according to their status, and for the purposes of this consultation we selected only present or past producers and disregarded categories such as prospecting and exploration.

With the exception of hand panning, ODEQ requires a water quality permit for all metal mining operations which include prospecting and mining of placer or lode (hard rock) and processing of metal ores using hand and power equipment. ODEQ also requires water quality permits for best management practices to properly manage stormwater that may contact metal mining activities and runoff into Oregon waters. In addition, ORS 517.956(2)(c) provides that chemical process mining operations are required to meet the following standard: "No loss of existing critical habitat of any state or federally listed threatened or endangered species." This "no loss" standard appears to be more stringent than USFWS's standard for "adverse modification" of critical habitat.

1.3 GIS Layers

We used the GIS program ArcView to overlay the NPDES and mine data on range and distribution shapefiles of listed species to determine areas of potential co-occurrence of chemicals in this consultation. Species range shapefiles were developed by USFWS. These files exist for every listed species and consist of polygons encompassing the hydrologic units where that species can be found. In all cases, these polygons include areas that are currently occupied.

The data are largely presence/absence data collected by governmental agencies and university researchers. A 2-mile "buffer" was created on each side of the aquatic habitat when determining sources. Once co-occurrence is determined via GIS for each chemical, we evaluated the spatial and temporal extent of potential exposure for the species, given its life history. Some species may be in the system seasonally, while others continuously, thus, the spatial and biological overlap is important in analyzing this action.

1.4 USFWS Lead Biologists

Each species lead biologist in USFWS's offices Oregon has in-depth knowledge of land-use activities within their species' habitats, which was an important component of evaluating exposure. Lead biologists also contacted local State and Federal agencies to gather site-specific information for the analysis.

2 Effects Analysis Methods

The USEPA evaluation of potential adverse effects to listed species resulting from their proposed action to approve certain water quality criteria for toxics follows the standard ecological risk assessment paradigm (USEPA 1998). The first (preliminary) tier of their two-tiered approach adheres to the *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria Methods Manual* (USEPA 2005a) which USEPA views as incorporating conservative estimates of exposure and toxicity to identify species not likely to be adversely affected by criteria chemicals.

Any risk assessment has areas of uncertainty and uses unproven assumptions to make risk determinations, especially in evaluations of listed species where toxicity data are sparse. In the BE methodology, we found the level of uncertainty associated with some determinations of adverse effects from proposed approval of the toxics water quality criteria was too high to confidently err on the side of caution for protection of the species. Within our assessment, we use the same models as USEPA with slight refinements to either reduce uncertainty or provide for a more conservative decision when faced with such uncertainty. For example, counter to the USEPA approach, we determined that any affects that are not insignificant, discountable, or not entirely beneficial would lead to a likely to adversely affect (LAA) determination, and used a modified approach to improve confidence in our determinations. These refinements give us greater confidence in the risk determinations related to vulnerable threatened and endangered species populations, although uncertainty still remains.

In this section we identify uncertainties within the risk assessment models as used in the BE and describe our approach to refining them. Subsequently, we summarize our method for estimating a level of effect to listed species populations from acute and chronic criteria.

2.1 USFWS Methodology and Differences from the BE Approach

As identified in the BE, there are two components of uncertainty, variability and true uncertainty. Variability can be quantified with statistical analyses, yet with true uncertainty information is unknown. Within the risk assessment conducted for this consultation there are several areas of uncertainty. For example, intra- and inter-laboratory variation within the toxicity test data, use of surrogate test species data as an indicator of endangered species sensitivity (extrapolating from one species to another), use of the same adjustment factor for all species and chemicals, and estimating a chronic threshold level of concern from acute data. Specific sources of uncertainty are discussed in detail below, along with the any specific modifications applied to the BE methodology to address uncertainty.

2.1.1 Data Compilation Rules

The USEPA's (2008) BE used rules to sort and limit data for inclusion into the risk assessment. The BE excluded data if:

(1) one or more pieces of information regarding study methodology or calculation of results needed to assess the reliability of the study was missing,
(2) data quality of the study was less than acceptable (e.g., control mortality greater than 80%),
(3) the tested species did not have a reproducing population in North America,
(4) the test species was exposed to a chemical mixture or previously exposed to the test chemical,
(5) the study reported effects on an endpoint other than survival, reproduction, or growth, or
(6) non-standard time was used for the test (e.g., fish toxicity test reporting a 24-hour LC50 rather than the more standard 96-hour LC50).

Our approach generally followed USEPAs data rules but included the following supplemental data:

(1) Behavioral and neurophysiological impairments associated with survival, reproduction, or growth (in addition to endpoints directly linked to survival, reproduction, or growth).
(2) Test species routinely used in laboratory toxicity assessments that are not North American in their distribution.
(3) Data from nonstandard acute test duration (standard test durations 24, 48, or 96 hours) for direct endangered species toxicity data or potentially in the case of very few values in the database to run a species sensitivity distribution model.

We conducted a literature search to build upon or update the existing core data used by USEPA in their BE for each chemical that was considered LAA based on exposure (see Exposure section). USEPA core data served as the foundation for our database, and we supplemented with data that met our criteria (e.g., from current national USEPA criteria) and updated with newer literature (last year cited in USEPA database until current). We conducted literature searches via the USFWS National Conservation Training Center Library Literature Search Service using Aquatic Science and Fisheries Abstracts and BIOSIS databases. We used the primary search terms "acute toxicity" or "chronic toxicity" and "fish" or "invertebrates".

For our ammonia and copper databases, we used a slightly different approach compared to the other criteria chemicals. For ammonia, our dataset was largely reproduced directly from the Draft 2009 Update Aquatic Life Ambient Water Quality Criteria for Ammonia-Freshwater USEPA (2009), as updated with any more recent papers. For our copper database, during the period September 2008 through February 2009, USEPA's online database ECOTOX 4.0 (http://cfpub.epa.gov/ecotox/) was searched following methods consistent with those detailed in the BE, with a primary focus on peer-reviewed literature reporting (exclusively or in part) data for copper toxicity in aquatic and terrestrial biota. An additional literature search was conducted by USGS and used primary database providers Cambridge Scientific Abstracts (CSA) and Online Computer Library Center, Inc. (OCLC) FirstSearch (Greg Linder, Research Biologist,

U.S. Geological Survey, Columbia Environmental Research Center, Brooks, Oregon, pers. comm., 2009). Databases searched in CSA included Aquatic Sciences and Fisheries Abstracts, Biological Sciences, Environmental Sciences and Pollution Management; and to a lesser extent, AquaLine, Water Resources Abstracts, GeoRef, Biology Digest, Conference Papers Index, Medline, and Toxline. Databases in OCLC FirstSearch that were searched included Agricola, ArticleFirst, BasicBiosis, Dissertations, GeoBase, WorldCat, BioAgIndex, Electronic Collections Online, PapersFirst, Proceedings, and Ingenta database provider.

2.1.2 Selection of "Effects" Point Estimates

2.1.2.1 Acute LC50 Data

The standard measure of acute toxicity is the median lethal concentration. This is a statistically derived estimate of the concentration that kills 50% of test organisms over a given exposure period. These data are collected in a laboratory according to uniform, detailed methods allowing for clear interpretation of results. However, laboratory conditions are not usually reflective of the complexities of natural systems.

Test organisms used in standard laboratory toxicity tests are chosen for their suitability of culture under laboratory conditions and are not necessarily representative of aquatic endangered species. Mount et al. (2003, p. 82) state that the selection of species for toxicity testing is influenced by a number of practical and historical factors and there is no reason to assume that these species represent a typical subsampling of all organisms. Species routinely used in toxicity tests are chosen because they survived well in the laboratory and were convenient to test, indicating that they might not be representative of the most sensitive species in nature (Moore et al. 2003, p. 132). Given the notable lack of toxicity information for most listed species, interspecies extrapolation is a viable alternative. However, this extrapolation is generally considered the greatest source of uncertainty in ecological risk assessment (Hart et al. 2001). Peer reviewers of the USEPA 2005 methodology, which is very similar to the methodology used in the BE, note its reliance on extrapolation of data which comes with inherent uncertainty and the lack of a consistent approach to deal with this uncertainty (Versar, Inc. 2004). Consequently, we believe it is necessary to take a more conservative approach in the methodology to make accommodation for the interspecies extrapolation uncertainty and the specifics have been described throughout this section.

Laboratory tests are used to isolate test organisms from confounding factors to delineate species responses to the chemical or stressor under study. However, laboratory studies do not replicate complex environmental conditions, including species competition, varying physical conditions, and exposure to other stressors. Extrapolating the stress responses of individuals in a limited number of laboratory tests to organisms exposed to similar chemical concentrations while in highly complex natural environments provides for gross scale predictions. The uncertainty involved in applying toxicity test data to the prediction of environmental outcomes increases with the discrepancy in scale between the measurement endpoint and the assessment endpoint or management goal (Cairns and Niederlehner 2003). Although review of mesocosm or field studies could help provide additional information for risk assessment, no such studies were reviewed for this analysis due to time constraints.

2.1.2.2 Chronic point estimates

Standard measurements for chronic data typically are based on *post hoc* hypothesis testing (e.g., Dunnet's test), the results of which determine the no observed effect concentration (NOEC) or no observed effect level (NOEL), and lowest observed effect concentration (LOEC) or level (LOEL). The NOEC or NOEL is defined as the highest test concentration resulting in responses that are not statistically different from the control, whereas the LOEC or LOEL is the lowest test concentration that is statistically different from the control. The geometric mean of these values (NOEC and LOEC) is used to estimate the chronic value (ChV) or maximum acceptable threshold concentration (MATC), which is assumed to be the threshold of unacceptable chronic effects.

The use of post-ANOVA testing to identify NOEC, LOEC, and ChV values may not always be appropriate (Crane and Newman 2000; Hoekstra and Van Ewijk 1993; Laskowski 1995; and Moore and Caux 1997). Critics suggest the emphasis on Type I error and cursory consideration of Type II error are inconsistent with risk assessment, in which the probability of not identifying a problem is of greater concern than the possibility of falsely identifying a problem when one does not exist (Brosi and Biber 2009, p. 488; Crane and Newman 2000, p. 516). Thus, the NOEC may not be considered safe because a large, biologically significant effect may occur at a particular concentration yet not be detected by a hypothesis test (Crane and Newman 2000, p. 516). Furthermore, the ability to detect a statistically significant effect depends not only on biological response, but also on the experimental design (e.g., small sample size, improper spacing of treatment doses, and large intra-treatment variance) and the power ($1-\beta$) of the applied statistical test (Crane and Newman 2000, p. 516; Moore and Caux 1997, p. 794). Other criticisms include: (1) the use of an arbitrary number ($\alpha = 0.05$) to identify statistical significance rather than biological significance (Crane and Newman 2000, p. 516), (2) the use of test concentrations (NOEC and LOEC), which do not correspond to biologically relevant thresholds or specified effects levels (Crane and Newman 2000, pp. 516-517; Laskowski 1995, p. 1443; and Moore and Caux 1997, p. 174), and (3) the inherent loss of information associated with the dose, or response curve (e.g., the slope, confidence limits) resulting in an inability to evaluate test results and estimate risks of differing severity (Crane and Newman 2000, p. 517; Moore and Caux 1997, p. 794). Because the LOEC and ChV values are based on the NOEC, they also are considered inappropriate.

Moore and Caux (1997) investigated methods of regression in comparison to hypothesis testing. They reported that hypothesis testing techniques produced NOELs at levels that correspond to effective concentrations (EC_X; percent of test organisms with response) between 10 and 30%. Regression analysis is more robust, however, it is also data intensive and many of the traditional tests do not use the test concentration intervals necessary for this type of regression. With poor concentration-response data, the regression-based approach will not produce an adequate model fit. Hypothesis testing using data from currently used toxicity test protocols cannot detect effects at low concentrations. This is due in part to the lack of statistical power given the number of replicates and the intrinsic laboratory and organismal variability within the experiments. Use of EC_X, which is based on more robust statistical procedures, provides a greater degree of certainty in the prediction of effects concentrations. Mebane (2006, p. 15) notes that in hypothesis testing, biologically important adverse effects may not be statistically significant due to high variability,

small samples sizes, or statistical vagaries. LaPoint et al. (2003, p. 4) provide support that regression analysis rather than hypothesis testing is needed to properly express chronic effects.

Several alternative methods to calculating a "no-effect" to "low-effect" estimate have been suggested (Hoekstra and Van Ewijk 1993, USEPA 1995, Besser et al. 2005). Hoekstra and Van Ewijk (1993) discuss alternative methods of finding a dose with a negligible effect including a two-step procedure that involves finding the dose whose effect is at most 25%, followed by linear extrapolation to a dose whose effect is acceptably small. The procedure leads to higher values if efforts are made to increase the power of the experiment.

Besser et al. (2005, p. 156) used linear interpolation to estimate inhibition concentrations (IC*p*), which is an alternative way to analyze chronic toxicity data from either hypothesis testing or regression analysis which depends on effect concentrations far from the midpoint of the regression where there is a wider confidence interval. This linear interpolation method starts with a monotonic concentration-response curve, which may be smoothed using moving averages, and makes linear interpolations between treatments means to estimate the concentration associated with *p* percent inhibition. This method allows estimation of chemical concentration with low levels of effects rather than a statically significant level of effect that varies among tests with different statistical power. Confidence intervals for these estimates can be done by a nonparametric bootstrap technique. Besser et al. (2005, p. 161) were able to derive logistic regression models that adequately fit the low-effect portion of the concentration-response curves for over 20% of possible test endpoints (EC25), yet the IC25s were calculable for almost 80% of endpoints.

Another option for chronic data assessment is the benchmark dose approach. The USEPA (1995, pp. 1-2) defines a benchmark dose (BMD) or benchmark concentration (BMC) as a statistical lower confidence limit for a dose that produces a predetermined change in response rate of an adverse effect (called the benchmark response or BMR) compared to background. The BMC takes into account dose-response information by fitting mathematical models (e.g., regression equations) to dose-response data and is generally set near the lower limit of responses that can be measured directly in animal experiments of typical size. Thus, the BMC method does not extrapolate to doses far below the experimental range. BMC also takes into account the study sample size, thereby reducing the variability and resulting in a reduced range in the confidence limits and reduced uncertainty in the dose estimate. The method statistically fits dose-response to determine no observed adverse effect level (NOAEL) values. Hecht et al. (2007) used this BMC approach to estimate thresholds for sublethal effects of dissolved copper on the chemosensory physiology and predator avoidance behaviors of juvenile coho salmon.

EC20 values are increasingly used as chronic no-effect values in aquatic toxicology, including during the derivation of water quality criteria (USEPA 1999, 2003). However, Moore and Caux (1997) reviewed 198 toxicity datasets, finding that one out of every six LOECs occurred at a concentration below the reported EC20 value. Moore and Caux (1997) also found that calculated EC10 concentrations were lower than 100% of the reported LOECs in their dataset, and were also lower than 76.9% of the NOEC values in the same dataset. Besser et al. (2005, p. 156) caution that regression analysis depends on effect concentrations far from the midpoint of the regression where confidence intervals are wider.

Our preferred database values for chronic protection are those determined through regression modeling (EC10 or EC20). Ideally, treatments for the regression models are closely spaced and cover a broad range (Mount et al. 2003, p. 62); however, such dose-response experiments are more expensive and thus less common. Consequently, data from such studies are more limited as compared to availability of NOEC or LOEC.

Effect concentrations calculated through various regression techniques were obtained from the literature whenever possible (e.g., national USEPA criteria). Where series data were presented in the literature, we compute an ECx using regression techniques that best fit the data. In situations where an EC10 or EC20 value that was interpolated to be <x, the x value was used to ensure that the lowest measurable effect was considered in the endangered species analysis. If no source data were present for deriving an EC10 or EC20, the NOEC, ChV, or LOEC were used. The rank order for selection of the "effects" concentration was EC10 < EC20 < NOEC < ChV < LOEC. Use of this data was only necessary for the vernal pool fairy shrimp effects analysis when there was no data from which to estimate a level of effect estimate as described in the USFWS (2010e) draft biological opinion for the proposed approval of cyanide aquatic life criteria.

2.1.3 Acute to Chronic Ratio

In the absence of measured NOEC data, both our Biological Opinion and the BE use an estimated NOEC. The estimated NOEC is derived from the LC50 (real or estimated) divided by the acute to chronic ratio (ACR). Procedures that use effect concentrations divided by numerical factors (e.g., acute to chronic extrapolation) are subject to substantial uncertainties (Mount et al. 2003, p. 62 and 66). The majority of chronic assessment effects concentrations in the BE were estimated by this method due to lack of measured chronic data, which is often the case.

Mayer et al. (1994) point out several limitations with using ACRs. For example, biological endpoints and degrees of response are often not comparable between acute and chronic data. The acute LC50 is compared to a NOEC often derived from an endpoint other than lethality (often growth and reproduction). In addition, the ACR is for different degrees of response; 50% for acute toxicity and 0% for chronic toxicity. Ahlers et al. (2006) note that some chemicals show different modes of action under short- and long-term conditions and mechanisms of biological action will differ from species to species. Mount et al. (2003) point out that acute toxicity tests are conducted with varying life stages and along with the differing endpoints, leads to an inherent uncertainty that cannot be easily quantified or expressed. Grist et al. (2002, p. 1519) reported that the European Union is currently reluctant to use acute to chronic extrapolations with species sensitivity distributions.

The majority of final ACRs in the BE were derived from water quality criteria documents and may be based on relatively small datasets. In the case of copper, the BE uses one ACR for all aquatic species. According to the USEPA (1985, pp. 51-54) copper criteria document, ACRs can vary considerably even for the same species. In the criteria document, an ACR of 10.33 is presented for fathead minnow based on the geometric mean of four values ranging from 5.37 to 21.5. For *Daphnia magna*, the geometric mean of 2.42 results from three distinct values of 1.02, 1.91, and 7.24. The ACR used by the USEPA in their BE for copper analysis is 2.82, which is a combination of all aquatic ACRs from their 1984 criteria document.

ACRs can vary depending on the data used to compute them. Many ACRs are based on a relatively small number of tests and a variety of taxonomic groups including invertebrates, fish, and amphibians (Ahlers et al. 2006). The mechanisms of biological action for these species exposed to contaminants will differ from species to species. Ahlers et al. (2006) suggest that individual ACRs vary considerably and a safety factor of 100 is not protective for all chemicals and trophic levels. The authors state that life cycle tests are the only way to conservatively predict long-term toxicity. Duboudin et al. (2004b, p. 1774) have a similar opinion of ACRs, stating that the correct use of this approach would necessitate knowing the ACRs for virtually every species.

Given the uncertainty surrounding derivation of the ACRs, contrary to the USEPA BE (2008), we selected not to use them in the initial determinations as to whether or not an endangered species is likely to be adversely affected. ACRs were used to a very limited extent in our assessment, only for vernal pool fairy shrimp if there were no data for which to compute a chronic "level of effect." When an ACR was employed, values were recomputed based on taxonomic relatedness to the endangered species to reduce the uncertainty. Tables with detailed information regarding the computation of copper ACRs for the Biological Opinion are included as Appendix 7.

2.1.4 Adjustment Factor

The BE methodology includes an adjustment factor of 2.27 (USEPA 1978) to transform the concentration estimated to be acutely lethal to 50% of the exposed individuals (i.e., the LC50) to a concentration estimated to affect or kill 0% to 10% (LC0 to LC10) of the exposed individuals. The predicted LC50 data are divided by this adjustment factor of 2.27 to derive the assessment effects concentration. USEPA considers the assessment effects concentration to be the highest concentration of a contaminant where species are not likely to be adversely affected. Specifically, the USEPA BE (2008, p. 5-21) states

> "Once the LC50 values are obtained, each LC50 is divided by 2.27 to calculate the acute [assessment effects concentration] value for each species. The acute [assessment effects concentration] for each ESA listed species undergoing evaluation is compared to the acute water quality criterion . . . to determine if the acute [assessment effects concentration] is greater than the acute water quality criterion. The value of 2.27 is intended to convert the LC50 concentration to an 'LC_{LOW}' value that should result in little or no toxicity to the test species."

USEPA further states that, "The basis for this adjustment factor is an analysis of data from 219 acute toxicity tests showing that the mean concentration lethal to 0%-10% of the test population was 0.44 times the LC50 or the LC50 divided by 2.27. The analysis consisted of calculating the geometric mean of the ratios of the highest concentration affecting or killing 0%-10% of organisms divided by the LC50 or EC50 for the same organisms in the same acute test (i.e., the geometric mean of 219 highest concentration/LC50 ratios from toxicity tests with a variety of chemicals)." USEPA derived the adjustment factor from a combined dataset on freshwater and marine fish and invertebrates, a large portion of which comes from whole effluents of various sources rather than specific chemicals. Many values used in the derivation of the 2.27 are

supported only by personal communication, and many are based on whole effluent analysis and unknown toxicity test parameters. Several sources of variability exist within a universally pooled sample of multiple chemicals and multiple test organisms and applying an adjustment factor based on a geometric mean to all chemicals and aquatic species introduces a substantial uncertainty into estimates of assessment effects concentrations.

Results from work evaluating sensitivities of various chemical groups to listed species by Dwyer et al. (2005) provide support for the 2.27. Their work with five chemicals and 17 species, including some listed species, shows the average factor to calculate a no- or low-effect concentration varies among pollutants and species from 0.50 to 0.66 with a geometric mean factor for all species of 0.56 (inverse of 1.8). The USFWS (2010e) draft biological opinion for the proposed approval of cyanide aquatic life criteria includes a recalculation of the adjustment factor specific to cyanide for different lifestages of several fish species (USFWS 2010e). All computations of the mean LC50/LC10 are substantially lower than the value of 2.27, which suggests an adjustment factor of 2.27 can be protective.

Similarly, an evaluation of invertebrate data provides support for use of the adjustment factor for *Daphnia*. Using results provided by Shaw et al. (2007, p. 1534), for *Daphnia pulex* exposed to various arsenic compounds, we can compare the use of the 2.27 adjustment factor to actual LC10 values computed by probit analysis (USEPA Probit Ver. 1.5) as presented in Table 2-1 shows that the adjustment factor more often under predicts the LC10 compared to the probit estimate. The highest notable over prediction is for monomethylarsonic acid where the LC10 is less than half of that predicted by the adjustment factor estimate. Computation of a geometric mean based on results of the LC50/LC10 probit estimates equals 2.3, which is very close to 2.27.

Table 2-1.Comparison of adjustment factor to actual LC10 estimated by probit analysis in Shaw et al. (2007) for *Daphnia pulex* exposed to various arsenic compounds.

Arsenic Compound	LC50	LC10	LC50/2.27	LC50/LC10
Arsenite	2566	2043	1130	1.26
Arsenate	3933	1119	1733	3.51
Monomethylarsonous acid	123	66	54	1.86
Monomethylarsonic acid	15737	2925	6933	5.38
Dimethylarsonic acid	13836	6404	6095	2.16
3-Nitro-4-hydrophenylarsonic acid	14673	8902	6464	1.65
4-Amino-phenylarsonic acid	14596	6811	6430	2.14

The adjustment factor provides an endpoint of effect of up to 10% lethality. Ideally, an adjustment factor should be derived based on the LC50 divided by a LC0 or LC01 value (rather than an LC10). However, the LC10 is a typical point of standardization in toxicological studies and regression of toxicological data typically does not have enough statistical power to obtain reliable values lower than the LC10 or EC10. Also, the lower 95% confidence interval estimated for a chronic LC10 is zero or below, but is above zero for an LC20 value. In their study evaluating sensitivities of various chemical groups to listed species, Dwyer et al. (2005, p. 147) noted "A 10% level of mortality was used in this evaluation because it is considered acceptable control mortality and therefore could not be discerned as an effect of chemical exposure." The authors further support this contention by citing testing authorities including the Committee on Methods for Toxicity Tests with Aquatic Organisms (1975) and the American Society for Testing and Materials (ASTM 2003).

2.1.5 Selection of Surrogates

Because there are very limited empirical toxicity data for endangered species, the use of surrogate species can be an important tool in assessing risk to endangered species. However, there is uncertainty and variability associated with these extrapolations, which need to be understood. Dwyer et al. (1995) found that for 30% (8 of 27) of the possible surrogate/listed species comparisons, the standard 96-hour median lethal concentration (LC50) for the listed species was lower than the surrogate species. Furthermore, these authors found that warmwater listed species were more sensitive than fathead minnows, one of the most commonly used test species, 33% of the time. Dwyer et al. (1995) found that listed warmwater species were always less sensitive than rainbow trout (*Oncorhynchus mykiss*) and suggest that hazard assessment using rainbow trout would be appropriate for warmwater species tested in their study. Listed salmonids were more sensitive than rainbow trout for 25% of the comparisons after 96-hour exposures. Dwyer et al. (1995) state that an additional margin of safety may be necessary to protect listed salmonid species when toxicity assessments use data obtained from studies using rainbow trout. The authors advise that further testing should be conducted with other listed species and/or their surrogates before definitive policy decisions concerning the protection of endangered and threatened species are made.

Dwyer et al. (2005) tested 17 listed or closely-related species against several common toxicity test species in 96-hour exposures to five chemicals with a range of toxic modes of action

(carbaryl, copper, 4-nonylphenol, pentachlorophenol, and permethrin). They found that no single species was the most sensitive to these chemicals. However, they found rainbow trout was equal to or more sensitive than listed species 81% of the time (Dwyer et al. 2005, p. 143). Furthermore, their results indicate that acute aquatic assessments using rainbow trout data would often be protective of listed fish species (Dwyer et al. 2005, p. 151) and that rainbow trout data likely represent the response of sensitive warmwater species and not merely responses of coldwater species (Dwyer et al. 2005, p. 151). This is consistent with Stephan et al. (1985) who indicate that toxicity tests with representative species are necessary so that data available for tested species can be considered a useful indication of the sensitivities of untested species. Sappington et al. (2001) found that sensitivity of standard test species (rainbow trout, fathead minnow, and sheepshead minnow) were generally not greatly different from that of corresponding listed species in their static tests with several federally-listed fish species (Apache trout, Lahontan cutthroat trout, greenback cutthroat trout, bonytail chub, Colorado pikeminnow, razorback sucker, Leon Springs pupfish, and desert pupfish). For approximately 30% of the tested surrogate/listed species comparisons, the 96-hour LC50 for the listed species was 1.5 to 2.1 times lower than that for the corresponding surrogate (Sappington et al. 2001, p. 2875). However, in two cases a listed species had a 96-hour LC50 less than one-half its surrogate LC50. The authors state that additional margins of safety may be warranted for protection of individuals in certain circumstances. They also believe that unless specific data are generated for listed species, their data indicate that rainbow trout should not be eliminated from the database as it serves as a sensitive species.

Using information from previously published studies, Teather and Parrott (2006, p. 104) examined the relative sensitivity of common fish test species, including fathead minnow, rainbow trout, and coho salmon (*Oncorhynchus kisutch*), to a variety of chemicals. They found that of the nine species tested, coho salmon and rainbow trout were the most sensitive species and fathead minnow among the least sensitive to 190 chemicals. Fathead minnows and rainbow trout were not equally sensitive to 13 different classes of chemicals. Trout were significantly more sensitive to metals, yet fathead minnows were more sensitive to hyrdocarbons.

Fairchild et al. (2007, p. 628) tested bull trout (*Salvelinus confluentus*) alongside rainbow trout in acute 96-hour exposures to three herbicides (picloram, 2,4-D, and clopyralid) and found that bull trout was of similar sensitivity (within a factor of two) to rainbow trout, the standard salmonid test species.

In comparing the sensitivity of surrogate species to endangered species exposed to aquatic contaminants, Dwyer et al. (2005, pp. 150-151) state that their findings indicate a factor of 0.63 can be applied to the geometric mean LC50 of rainbow trout toxicity data for protection of a listed aquatic species. Safety or uncertainty factors are commonly used to address uncertainty in regulating for protective aquatic chemical concentrations, but deriving a safety factor should be based on science or logical arguments to avoid adding new sources of uncertainty to the process. With the variety of species and chemicals we are addressing in the consultation, the general safety factor used by USEPA (2005a) may introduce additional uncertainty to the consultation. However, we do not have the resources to develop species-specific or chemical-specific safety factors at this time.

2.1.6 Expression of Criteria

For most of the metals in this consultation (cadmium, chromium, copper, lead, nickel, silver, and zinc), the criteria are expressed as a function of water hardness based upon a regression equation specific to each chemical. The criterion presented in the consultation for each of these metals is normalized to hardness of 100 mg/L as $CaCO_3$. Accordingly, all values used in our risk assessment have been normalized to a hardness of 100 mg/L as $CaCO_3$.

It has long been recognized that several other water quality characteristics in addition to hardness affect toxicity to these metals. Erickson et al. (1996) and USEPA (2003) explain this in relation to copper toxicity. Niyogi and Wood (2004, p. 6177) state that the biotic ligand model integrates the concept of bioavailability into ambient water quality criteria. In simple terms, the biotic ligand model incorporates the competition of the free metal ion with other naturally occurring cations (e.g., $Ca2^+$, Na^+, $Mg2^+$, H^+) together with complexation by abiotic ligands (e.g., dissolved organic matter, chloride, carbonates, sulfide) for binding with the biotic ligand which is the site of toxic action on the organism (Niyogi and Wood 2004, p. 6177). Of all the biotic ligand models being generated, copper appears to be the most advanced in development (Niyogi and Wood 2004, p. 6180). In their 2007 Revision for the Aquatic Life Ambient Water Quality for Copper, USEPA recommends the biotic ligand model for aquatic life criteria. Input parameters to the model include: pH, dissolved organic carbon, percent humic acid; temperature, major cations (calcium, magnesium, sodium, and potassium), major anions (chloride and sulfate), and dissolved inorganic carbon (USEPA 2007, p. 12). According to Keithly et al. (2004, p. 693), cadmium, lead, zinc and nickel models are in various stages of development with copper and silver being the only well-developed models available.

Schubauer-Berigan et al. (1993) found that the acute toxicity of metals such as cadmium, copper, nickel, lead, and zinc in hard water can be greatly affected by test pH even within the range of pH 6 to 9. Furthermore, Hoang et al. (2004, p. 91) conducted tests with varying regimes of hardness, pH, alkalinity, and natural organic matter and concluded that all four characteristics influenced nickel toxicity in fathead minnows.

Although the current USEPA Aquatic Life Ambient Water Quality for copper uses the biotic ligand model, the criterion under this consultation is a previous USEPA criteria recommendation (hardness dependent). The other metals in this consultation are designated by USEPA as either hardness dependent, pH dependent or not influenced by water characteristics. Thus, the analysis for each metal in this consultation does not address the additional interacting variables that may influence toxicity to aquatic species. We believe this may add uncertainty to the analysis for several chemicals

2.1.7 Models Used to Estimate Risk Parameters

We used two models to estimate acute toxicity values for endangered species from surrogate species; the Interspecies Correlation Estimate (ICE) model and the species sensitivity distribution model. These models are described in the BE. Below we provide an overview of ICE and adjustments we made to the models for use in our analysis.

2.1.7.1 Interspecies Correlation Estimate (ICE) Model

The ICE model is a log-linear least-squares regression of the sensitivity relationship between surrogate and predicted taxon based on a database of acute toxicity values (Raimondo et al. 2010a, p. 4). ICE software was developed by the USEPA (Asfaw 2003, p. 3) to predict acute effects to aquatic and terrestrial species based on toxicity results from surrogate species, including threatened and endangered species. ICE models permit estimations of LC50s to be made for a species or higher taxa (up to family) having no measured acute toxicity information from a species for which four or more LC50 values have been measured (Raimondo et al. 2010a, p. 7).

ICE models have been used to populate aquatic species sensitivity distributions and have been recommended for generating reasonable estimates of hazard levels when toxicity data are limited (Dyer et al. 2006, Awkerman et al. 2008, p. 3447). Raimondo et al. (2007, p. 5893) state that ICE models provide an additional tool for estimating the acute toxicity of chemicals to a diversity of wildlife species with known uncertainty across chemical modes of action and taxonomic level. Previous versions of ICE lacked the validation and uncertainty of models for aquatic species (Raimondo et al. 2010b, p. 7711). The recently updated Web-based ICE (Web-ICE 3.1) includes a suite of aquatic models using strict standardization criteria for test methods and conditions (Raimondo et al. 2010b, p. 7715). We used Web-ICE 3.1 to estimate acute toxicity values for endangered species in this Biological Opinion to replace values listed in the BE obtained from the original ICE software from 2004, which lacked data standardization and statistical rigor compared to the newer version.

The accuracy of ICE estimates is influenced by the relative sensitivity of the listed species and surrogate species to the pollutant of interest as well as the relative sensitivity of species to other chemicals and chemical classes. Raimondo et al. (2010b, p. 7714) identified that the taxonomic relatedness of the surrogate and predicted species is a driving factor for predicting accuracy. As taxonomic distance increased there was a decline in the percentage of toxicity values predicted within five-fold of the actual LC50 value (Raimondo et al. 2010b, p. 7714). Model accuracy was determined as the percentage of values predicted within five-fold of the reported value and is based on the range of inter-laboratory variation estimated by Fairbrother (2008) for acute toxicity (Raimondo et al. 2009, p. 921). Although chemical mode of action is a factor in species sensitivity and is a critical component of toxicity estimation, Web-ICE does not contain mode of action-specific models at this time. Raimondo et al. (2010b, pp. 7714) stipulate that mode of action-specific models should generally improve toxicity estimation for unrelated species, yet their predictions for closely related species were not significantly better than all chemical data models.

To attain a low degree of uncertainty, the USEPA recommends selecting a model based on taxonomic distance, model parameters and cross-validation success rate (Raimondo et al. 2010a, p. 18). For best estimates, models should have the followings selection guidelines from Raimondo et al. (2010a, p. 18):

(1) at least eight degrees of freedom (n-2),
(2) probability or p-values <0.01,
(3) R^2 values >0.6,
(4) a cross validation success rate $>85\%$,

(5) taxonomic distance ≤3 (genus = 1; family = 2; order = 3; class = 4; phylum = 5; kingdom = 6),
(6) a low mean square error (MSE) <0.22, and
(7) narrow confidence bands.

These parameters reduce uncertainty by giving the model greater statistical power and reducing the chance of Type I errors in toxicity estimation (Raimondo et al. 2010a, pp. 18-19). Raimondo et al. (2010b, p. 7715) identified species taxonomic distance and model MSE as the two most important variables for selecting models followed by the relationship of the input value relative to the surrogate toxicity values used to develop the model.

Because of the combined uncertainties associated with different species sensitivities to a chemical and to help ensure the sensitivity of listed species is not underestimated, the USEPA (2005a, pp. 13-14) recommends using the lower bound of the 95% confidence interval for ICE model estimates to calculate the assessment effects concentration for listed species. The BE used a 90% confidence interval in all ICE derivations, rather than a 95% confidence interval as specified in the methodology. The reason for deviating from the methodology is not clearly described in the BE. The methodology indicates that the 95% confidence interval uses both lower and upper bounds. Additional information provided by the USEPA indicates they are using the lower 95% confidence interval as a one-tailed test (hence using the 90% selection of the ICE model). We believe the confidence interval around the estimate is an expression of the variability surrounding the regression model, not a measure of significance for a hypothesis test (one-tailed vs. two-tailed). Awkerman et al. (2008, p. 3447) state that confidence intervals calculated with ICE estimates are based on the linear relationship and mean model error and may be used to assess the robustness of model predictions. The confidence interval is a range of values used to estimate the true value of the population parameter. The degree of confidence tells us the percentage of times that the confidence interval actually does contain the population parameter, assuming the estimation process is repeated multiple times. Therefore, this 95% confidence interval for the ICE model indicates that the interval of lower 95% confidence interval to upper 95% confidence interval actually does contain the true value of the estimated LC50 for the listed species.

ICE Validation

Table 2-2 displays a comparison of copper ICE models (BE version and Web-ICE version) and actual measured LC50 values as an example of our model output. The table also includes outputs for different species and genus surrogates and presents the correlation coefficient from Web-ICE.

For most estimates, the ICE model provides a lower, more conservative estimate of the LC50 for the endangered species than the actual measured values (especially when an estimate is selected by following the ICE model selection guidelines listed in the previous section). In some cases where some selection conditions are not met, case specific judgments may be made based on estimates providing greater protection of the listed species.

Table 2-2. Comparison of ICE model results and the associated correlation coefficient with measured 96-hour LC50 results.

Species	ICE Model species, genus, family/ surrogate species	MSE	Correlation Coefficient (R^2)	Degrees of Freedom (N-2)	ICE Result Lower confidence interval* µg/L	96 hr LC50 µg/L @ 100 $CaCO_3$	LC50 Reference
Bull trout *Salvelinus confluentus*	Genus RT	0.16	**1.00**/0.89	37	**25.5**/62.2	58.3	Hansen et al. 2002a
	Genus CS	0.053	**0.75**/0.98	12	**39.2**/54.4		
Lahontan cutthroat trout *Oncorhynchus clarki henshawi*	Genus ChS	0.050	**0.93**/0.99	6	**48.7**/26.3	41	Sappington et al. 2001
	Genus RT	0.072	**1.12**/0.96	45	**62.9**/74.5		
Rainbow trout *Oncorhynchus mykiss*	Genus ChS	0.050	**0.93**/0.99	6	**48.7**/26.3	74.2	USEPA 2008
	Species ChS	0.070	0.98	6	21.5		
Coho salmon *Oncorhynchus kisutch*	Genus ChS	0.050	**0.93**/0.99	6	**48.7**/26.3	124.3	USEPA 2008
Sockeye salmon *Oncorhynchus nerka*	Genus ChS	0.050	**0.93**/0.99	6	**48.7**/26.3	282.6	USEPA 2008
	Genus RT	0.072	0.96	45	74.5		
Chinook salmon *Oncorhynchus tshawytscha*	Genus RT	0.072	0.96	45	74.5	59.3	USEPA 2008
	Species RT	0.064	0.98	6	64.4		
Brook trout *Salvelinus fontinalis*	Family ChS	0.062	0.98	6	38.6	201.9	USEPA 2008
	Family RT	0.071	0.95	55	75.0		
	Family CS	0.026	0.99	16	69.7		
Fathead minnow *Pimephales promelas*	Species CC	0.043	0.99	6	131.0	171.7	USEPA 2008
	Family CC	0.20	**0.94**/0.99	23	**133.1**/100.7		

- *The BE (USEPA 2008) used a lower 90% confidence interval and an earlier version of the ICE model. The ICE model was rerun by USFWS with a lower 95% confidence interval and the newer Web-ICE version of the ICE model. Results are presented as the BE (USEPA 2008) results in bold and USFWS recalculated results in regular font.*
- *RT-rainbow trout, CS-coho salmon, ChS-chinook salmon, CC-common carp*

Dyer et al. (2006, p. 3102) explored the potential of using the USEPA's ICE program to predict single species toxicity values for a diverse set of aquatic species from a single known toxicity value from one of three surrogate species; rainbow trout, fathead minnow, or *Daphnia magna*. In their study, ICE-based species sensitivity distributions were compared to published species sensitivity distributions employing acute toxicity data to evaluate the use of acute toxicity data in environmental risk assessments. By holding surrogate species toxicity values constant they evaluated the different levels within a robust dataset to examine the potential issues regarding surrogate choices on species sensitivity distributions. They tested across a range of chemicals with different modes of actions (dodecyl linear alkylbenzene sulfonate, nonylphenol, fenvalerate, atrazine, and copper) (Dyer et al. 2006, p. 3103). Results of their ranking clearly indicate that coldwater salmonids are the most sensitive (Dyer et al. 2006, p. 3108). They also report there is a general tendency that the addition of more species to the dataset results in a greater tendency to add more tolerant species to the distribution rather than more sensitive species, thus increasing the final toxicity value (Dyer et al. 2006, p. 3108). ICE-based predicted 5th percentile values were generally within a factor of 10 of the 5th percentile based on empirical data (Dyer et al. 2006, p. 3108).

More recent work by Dyer et al. (2008, p. 3076) had the following objectives: (1) comprehensively evaluating the accuracy of ICE-generated species sensitivity distributions and, (2) validating a procedure for generating ICE-based HC5s (hazard concentration for the 5th percentile, similar to the species sensitivity distribution 5th percentile). Species sensitivity distribution accuracy was assessed by comparing ICE-based HC5s to HC5s derived from measured acute aquatic toxicity values for 55 chemicals of diverse structure and ecotoxicological modes of action (Dyer et al. 2008, p. 3076). Results show that combined fish predicted values (fathead minnow) and invertebrate predicted values (*Daphnia magna*) used to create a species sensitivity distribution and HC5 compared with predicted HC5 values were within a factor of 3 (±6.7 standard deviation) of measured HC5s (Dyer et al. 2008, p. 3080). Using rainbow trout rather than fathead minnow yielded, on average, predicted HC5 values within a factor of 1.8 (±3.0 standard deviation) of measured HC5s (Dyer et al. 2008, p. 3080). The approach illustrates that a combination of fish and invertebrate surrogate species could typically predict HC5 values within an order of magnitude of measured toxicity HC5s (Dyer et al. 2008, p. 3080). Their study also showed that surrogate species selection in ICE is important for predicting accurate HC5s (Dyer et al. 2008, p. 3081). Factors important for selecting the appropriate surrogate include highly significant interspecies correlations, a minimum of 10 pairwise observations, and slopes greater than 0.65 (Dyer et al. 2008, p. 3081).

The BE acknowledges the uncertainty associated with regression models and states that they will use a lower 95% confidence interval of the estimate. However, the lower 90% confidence interval of the estimate is what is presented in the BE. We believe it is more appropriate to use the lower 95% confidence interval, especially given the differences noted in the comparison presented in Table 2-2. Therefore, we reevaluated the aquatic endangered species based on the updated Web-ICE model using the lower 95% confidence interval.

Differences in Parameters for Secondary Toxicity Assessment

The BE analysis departed from the methods manual (USEPA 2005a) through the addition of a secondary toxicity assessment. The intent of the secondary screen in the BE is to conduct an in-depth examination of data including endpoints, effect concentrations, test parameters, and other water quality parameters. The USEPA is using this more refined analysis in those instances where data suggest that the criteria are likely to adversely affect listed species in the preliminary toxicity assessment. Those chemicals and species were considered in a secondary toxicity assessment where the USEPA replaced the conservative assumptions of exposure and toxicity with what they believe are more realistic species-specific assumptions and information regarding exposure and toxicity.

In the USEPA's analysis, if the species is not screened out as a NLAA, they use a secondary assessment incorporating the central tendency or 50th percentile of the ICE estimate. For determinations of NLAA, assessment effects concentrations based on ICE models were recalculated using the mean LC50 rather than the lower 95% confidence interval. The rationale for using the central tendency or 50th percentile estimates from the ICE model is because those estimates are the most statistically meaningful and most objective estimates of toxicity.
The USFWS comments on the USEPA's final biological evaluation of aquatic life criteria for cyanide (USFWS, in litt., 2006) described an evaluation of ICE performance by comparing results from 70 ICE-modeled LC50 values with measured values. Results showed there was considerable variation in the ability of ICE models to accurately estimate cyanide LC50 values. Fifty-six percent of the measured LC50 values were less than the 50th percentile estimate with 37% being closer to the lower bound of the 95% confidence interval than to the 50th percentile estimates. Approximately one quarter of the models produced estimates where the measured LC50 fell outside the 95% confidence interval (lower and upper) with 17% of those having the measured LC50 values less that the lower bound of the 95% confidence interval. We also compared ICE-modeled results to measured LC50 values for species included in our consultation (results presented in Table 2-2). Although more limited than the comparison presented by USFWS (in litt., 2006), our results corroborate the theory that the ICE model's lower 95% confidence interval does not always capture the LC50. Consequently, we do not agree with the USEPA's use of the mean ICE output in their BE, and our assessment differed from the BE by incorporating the lower 95% confidence interval as a more appropriate measure to estimate the assessment effects concentrations.

2.1.7.2 Species Sensitivity Distribution

Species sensitivity distributions are a major component of acute and chronic risk assessment methodologies and have been adopted formally in both North America and Europe within their ecological risk assessment frameworks (European Chemicals Bureau 2003, pp. 102-104). The species sensitivity distribution is a probabilistic model of toxicity values for multiple species. Species sensitivity distributions are used to calculate the concentration at which a specified proportion of species will be affected: the 5th percentile for the purposes of this consultation. The species sensitivity distribution concept was proposed over two decades ago as an ecological tool and various methodologies have evolved, each with its own support and criticisms (Posthuma et al. 2002, p. 3). Several authors have illustrated the diversity in approaches that calculate the species sensitivity distribution and the variation in the results caused by these

methodologies and their data parameters (e.g., Duboudin et al. 2004a, Grist et al. 2002, Wheeler et al. 2002, and Newman et al. 2000). Species sensitivity distributions can be influenced by data quality, data quantity, data summary and model choice (Wheeler et al. 2002, p. 199), in the manner described below.

Data Quality

Wheeler et al. (2002, pp. 194-195) examined the effects of data quality on species sensitivity distribution outputs for acute freshwater copper and pentachlorophenol data. They demonstrated a change in the HC5 value between using data of all qualities to using data of only the best quality (e.g., well-documented procedures, satisfactory controls, measured concentrations); however, the more stringent data criteria only provided a higher concentration protection level for copper, while the opposite was observed for pentachlorophenol (Wheeler et al. 2002, pp. 194-195). Wheeler et al. (2002, p. 195) states that there is no *a priori* reason to expect a relationship between species sensitivity and data quality for test endpoints except for the no observed effect concentration (NOEC). Statistically, the NOEC is the highest concentration in a test with a mean response not significantly different from the mean response of the control (Crane and Newman 2000, p. 516). Crane and Newman (2000, p. 519) showed that poorly designed studies can yield higher NOEC estimates entirely due to high variability in measured endpoints leading to low statistical power in the significance test.

Data Quantity

Toxicity datasets are generally quite small and have to be regarded as a representative sample from a larger set of species (Aldenberg et al. 2002, p. 51). If the dataset is large ($n \geq 20$), statistical resampling techniques can be used to determine the uncertainty of quantile and proportion estimates; however, if the dataset is small ($n < 20$), parametric techniques must be used (Aldenberg et al. 2002, p. 51). Effects of data quantity on log-logistic and log-normal distribution were assessed using a resampling approach without replacement (Wheeler et al. 2002, p. 193). Wheeler et al. (2002, p. 194) determined that 10 to 15 data points was sufficient to obtain an effective estimation of the sub-population of the data; below 10 data points the parameters varied widely and would not yield a reliable estimate of the HC5. For resampling techniques, a sample size of 20 is the minimum value needed to determine the HC5 and its upper and lower confidence intervals because under 20 these values cannot be defined using the bootstrap statistic (minimum percentile = 100/n) (Grist et al. 2002, p. 1517). Unlike parametric methods there is no clear guidance on the minimum sample size required for bootstrapped species sensitivity distributions to have adequate precision (Grist et al. 2002, p. 1519-1520; Newman et al. 2002, p. 128). Newman et al. (2000, p. 514) using bootstrap procedures suggested the optimal sample sizes for good HC5 estimation ranged from 15 to 55 with a median of 30 data points. Grist et al. (2002, p. 1520) refutes this claim, suggesting that Newman et al. (2000) did not run their test far enough because convergence was seen to continue past were the plot was terminated. Newman et al. (2002, pp. 129-130) found that species subsets decreased the sample size necessary for precision as well as decreased the HC5 values as subsetting increased (from all species down to arthropods only).

Data Summary

Wheeler et al. (2002, pp. 195-196) found that the four possible options for calculating the HC5 using multiple species data in species sensitivity distribution analyses varied in sensitivity as follows (in order of most sensitive to least sensitive): (1) most sensitive value for each species, (2) species geometric mean, (3) species arithmetic mean, and (4) all reliable species data. However, analyses using only the most sensitive data instead of all available values are more likely to be biased by outliers (Wheeler et al. 2002, p. 196). The geometric or arithmetic mean is a summary statistic that yields a single corresponding data point for each species thus creating a new dataset of species mean values. This approach ignores intraspecies variation attributed to differences in experimental conditions, chosen criteria, strains of species, and developmental stage of species (Duboudin et al. 2004a, p. 490). Including all available data would allow for the capture of inter- and intraspecies variation in the analysis (Wheeler et al. 2002, p. 196). However, Duboudin et al. (2004a, p. 490) raise the concern that when including all available data there is a potential to bias the analysis towards one direction if there are more data for one species than the others. These authors favor a weighted method (Duboudin et al. 2004a, p. 498), using the entire dataset without favoring one species over another, to retain intraspecies variation while giving each species the same weight within a species sensitivity distribution. Duboudin et al. (2004a, p. 498) admits that there can be problems with each of the methodologies used when analyzing weighted datasets. Each methodology includes its own set of problems but two of major concerns are related to different bootstrap analyses: (1) the HC5 of small datasets can be influenced by a few largely weighted data points, and (2) a high redundancy of samples generated in the bootstrap for a few data points with high weights (Duboudin et al. 2004a, p. 498).

Using all data from all species available for a toxin can generate problems when analyzing the dataset. Newman et al. (2000, p. 514) found that half of their datasets failed the null hypothesis of a log-normal distribution, and plots suggested strong multimodal patterns to the residuals. Analysis of weighted data for all species suggested that the HC5 differed first in terms of taxonomic grouping and intraspecies variation followed to a lesser extent by distribution (Duboudin et al. 2004a, p. 497). These patterns are not surprising because data from separate taxons and ecological groups have separate underlying distributions; by combining these data the new dataset cannot be represented by a single regression line (Newman et al. 2000, p. 514; Wheeler et al. 2002, p. 197).

Model Choice

Although some species sensitivity distribution risk assessors used a log-normal model to describe the distribution (Maltby et al. 2005, p. 380), others believe the log-logistic distribution fits the data best (Awkerman et al. 2008, p. 3448; Dyer et al. 2008, p. 3079; Raimondo et al. 2008, p. 2600). Yet, Newman et al. (2000, pp. 509, 514) point out that the assumption of a specific distribution may not be justifiable for many datasets. Grist et al. (2002, p. 1516) identify the basic problem with a singular approach to species sensitivity distribution construction is that no single underlying statistical distribution has been found to provide a good model fit to all data samples. Consequently, if the model is a poor fit, the 5th percentile obtained will be prone to distortion through the use of an inappropriate curve to fit the data. Bootstrap techniques do not rely on distributional assumptions (Gist et al. 2002, p. 1515) and have been presented as an

alternative to those methods with distribution assumptions (Newman et al. 2000). Newman et al. (2000, p. 509) use a simple bootstrap method, which also provides confidence intervals for a 5th percentile, while Grist et al. (2002, p. 1515) used a more advanced bootstrap methodology (hybrid bootstrap regression approach) to estimate confidence intervals for species sensitivity distribution estimates. Although methods to derive confidence intervals can take into account this uncertainty, using the lower 95% confidence interval in the case of metals can lead to thresholds below normal background levels (Grist et al. 2002, p. 1516). The bootstrap method obtains a point estimate and a confidence interval of that estimate by resampling randomly from the original sample with replacement.

Differences in Use of USFWS and USEPA Species Sensitivity Distribution Models

There are at least four sources of variation that are either incompletely or not accounted for in the USEPA's methodology for constructing species sensitivity distributions: (1) the variability around individual LC50 estimates, (2) the use of the species mean acute value (SMAV), (3) the minimum number of species used, and (4) the derivation of the 5th percentile. There is little we can do to lessen the uncertainty surrounding individual LC50 estimates due to the lack of consistent reported variability measurements. To lessen the uncertainty associated with variability surrounding SMAVs, we used individual data points (published LC50s) in bootstrap estimates of the 5th percentile. Our minimum number of values for computation of the 5th percentile of the species sensitivity distribution is 20 from within the closest taxonomic relation. Finally, we use an alternative estimation of the 5th percentile (described below) to lessen the uncertainty surrounding the estimate.

Relative scarcity of data in the lower tail of the cumulative distribution means an inherent degree of uncertainty associated with the 5th percentile extrapolation (Grist et al. 2002, p. 1516), yet no measure of variability is included in the 5th percentile estimates used to calculate the acute or chronic assessment effects concentration in the BE risk assessment. Mount et al. (2003, pp. 83-84) present options for dealing with uncertainty in the development of water quality criteria, specifically with the species sensitivity distributions: (1) statistically calculate the uncertainty or variance associated with the estimation of each criterion, or (2) create an alternative class of criteria for those with less than the minimum data requirements and incorporate a safety factor to reflect expected inaccuracies. Duboudin et al. (2004a, p. 491) state that there are three ways to estimate the confidence interval of a species sensitivity distribution: bootstrap, Bayesian techniques, or asymptotic theory (see Grist et al. 2002). Wheeler et al. (2002, p. 199) suggests analyzing data with four approaches and using the HC5 generated from the model with the best fit, but as a general guideline using the standard bootstrap method when there is enough species data ($n \geq 20$) or using the bootstrap regression for small datasets ($n = 10\text{-}20$) for which a good fit is achieved by a specified regression model. Although a bootstrap method can be used to obtain confidence intervals for a 5th percentile to account for uncertainty, in some cases the lower 95% confidence interval can lead to thresholds below normal background levels (Grist et al. 2002, p. 1516) which was found to be the case in an invertebrate copper analysis. In addition, the species sensitivity distribution provides a 5th percentile as an LC50 estimate as compared to a 50th percentile estimate of ICE or direct toxicity data. We use the species sensitivity distribution for estimating a level of effect, as compared to the USEPA using the species sensitivity distribution for determining whether the criteria concentrations will adversely affect listed species.

Uncertainty in estimating the level of effect involves less risk to endangered species as compared to uncertainty in the initial determination of LAA or NLAA. Consequently, we use the median 5th percentile estimate for the estimate of LC50 rather than the lower confidence interval.

The USEPA found that for specific data contained in the BE, no one distribution fit all of the data the best. The USEPA used the PERCENTILE and PERCENTRANK functions in Microsoft Excel to generate the BE 5th percentile. The functions relied on the rank ordered acute and chronic assessment effects concentration values to estimate the 5th percentiles of the species sensitivity distributions generated from the compiled toxicity data.

Our method to estimate the 5th percentiles differed slightly from USEPA. We computed 5th percentile values using SYSTAT 12 statistical software and the application's drop down menus. Within SYSTAT, there are several ways to calculate percentiles depending on the underlying assumptions regarding the analysis of rank order statistics. Table 2-3 and Table 2-4 display results of the various methods computed in SYSTAT 12 for 5th percentile estimates. Data used for this comparison were taken from the BE copper tables for order Cypriniformes (n = 14) with no supplemental USFWS data. Table 2-3 of acute estimates shows that output varied by greater than 36%, while chronic estimate outputs varied by approximately 26%. Based on the limited comparison below it appears that the Excel method used in the BE produces a result similar to the weighted average method (WTDAVG3) of SYSTAT 12. Both the weighted average and the Excel results are at least 1.5 times the other 5th percentile ranking estimates and use of one method over the other would influence on the final analysis. SYSTAT 12 provides numerous algorithms for calculating percentiles, and uses the Cleveland method, which gives less emphasis on single extreme values, as the default option. We opted for the Cleveland method because it was also used in the bootstrap script developed collaboratively with SYSTAT for derivation of 5th percentiles (see Appendix 8). Appendix 9 includes a description of the various 5th percentile algorithms.

Table 2-3. Comparison of 5th percentile estimates from SYSTAT 12 for acute data (LC50) reported in the BE for copper.

Copper 5th percentile acute geometric mean[1]		
Method	SYSTAT	Excel[2]
CLEVELAND	23.54	41.89
WTDAVG1	15.38	
CLOSEST	15.38	
EMPCDF	15.38	
WTDAVG2	15.38	
EMPCDFAVG	15.38	
WTDAVG3	41.97	

[1]Based on acute ECA (LC50/2.27)
[2] Excel method of ranking unknown.

Table 2-4. Comparison of 5th percentile estimates from SYSTAT 12 for chronic data (NOEC) reported in the BE for copper. These are from 14 values for order Cypriniformes .

Copper 5th percentile acute geometric mean[1]		
Method	Systat	Excel[2]
CLEVELAND	4.29	8.84
WTDAVG1	2.27	
CLOSEST	2.27	
EMPCDF	2.27	
WTDAVG2	2.27	
EMPCDFAVG	2.27	
WTDAVG3	8.84	

[1]Based on acute ECA (LC50/2.27)
[2] Excel method of ranking unknown.

2.1.8 Variability of LC50 and Species Mean Acute Values

Although toxicity test data in the literature often include some measure of variability (e.g., 95% confidence interval, variance, standard error of the mean), there is no consistent measure of variability reported in the literature and hence included in the ECOTOX database on which the BE data are based. Consequently, the variability surrounding these individual LC50 values used in the analysis is poorly characterized with respect to measures of dispersion.

When more than one test result is available for a species, the USEPA computed a geometric mean to obtain a SMAV to represent the central tendency for all the individual LC50s values. However, even a measured LC50 value for the same species can show considerable variation. For example, the core dataset for copper lists 103 individual LC50 values for rainbow trout which range from 9.1 to 832 µg/L dissolved copper adjusted to a hardness of 100 mg/L as $CaCO_3$. This wide range of variation in LC50 results for the same species indicates the test results could be considered independent test results, and the SMAV approach may be less representative. Also, using a SMAV without variability measures, which are often used as model inputs, adds to the uncertainty of the risk assessment. The variability around the LC50 estimate and the variability of the SMAV are not considered in the BE analysis. Although incorporating data variability will greatly improve confidence in determinations made from endangered species risk assessments, these measurements are often inconsistent in the literature and including a standard variability measure around each LC50 is not possible. In addition, using individual measurements as data inputs is not suitable for obtaining results from the ICE model. Consequently, we used SMAVs as inputs for the ICE model similar to the approach in the BE, but used individual data points for our 5th percentile calculations in our species sensitivity distribution models to improve our understanding of variability and ability to assess in model output.

2.1.9 Representing Data Quality

Suter et al. (2007, p. 135) reports that numerical results in a risk assessment are often "plagued by false precision." Assessors often neglect the concept of significant digits in practice and results are often presented as more precise than the precision of the input parameters. The last digit of an approximate number should always be significant and imply a range for the true measurement. For example, Sokal and Rohlf (1981, p. 14) state "If a scale of measurement is so precise that a value of 12.32 would clearly have been recognized, then measurement should be recorded originally to four significant figures . . . and implies the true value lies between 12.315 and 12.325."

USEPA used ECOTOX as its source data and compiled a dataset for each chemical in this consultation, reporting from two to seven significant digits for data presented in their BE. Suter et al. (2007, p. 136) cautions when extracting numbers from databases rather than original sources, it is often unknown if the reported value actually represents the precision of measured value. Measurements can be reported to a greater precision than the equipment supports or spurious digits can be introduced, for example, when computer software retains eight or ten significant figures, which can be greater than the precision of the measurement. It is possible that the number of times these values have been translated for this document has misrepresented the precision of measurement in the original data and resulted in some rounding errors. To minimize rounding errors, we retained significant digits reported in the BE and supplemental information for any calculations but reported results to three significant digits.

During our review of the copper core dataset, numerous errors were found which we expect are the result of data quality issues similar to those described by Suter et al. (2007, p. 136), especially in regards to transcription errors for pH, LC50, and hardness values. This required us to go back to the source data for copper and confirm all numbers in the database by reference original papers and correcting errors. Source data for ammonia provided by USGS were not confirmed in the ammonia database due to time constraints. In addition, because the ammonia database served was used for the Draft 2009 Update Aquatic Life Ambient Water Quality Criteria for Ammonia-Freshwater (USEPA 2009), we expect that it went through more rigorous quality assurance checks. Due to time constraints, no other chemicals were double checked for transcription errors with this level of detail.

2.1.10 Summary of USFWS Methodology

2.1.10.1 Acute Toxicity

Method for Estimating LC50 Values for Listed Species

For selecting LC50 data, we followed a hierarchy that closely resembles that recommended in the BE: (1) direct data on a listed species, (2) ICE model presenting best fit for genus or family (lower 95% confidence interval), or (3) fifth percentile estimate from species sensitivity distribution of closest taxonomic grouping with at least twenty data points.

Direct Data

Direct data refers to results of toxicity tests from which the exposed test organisms are the listed species. Typically, we used the lowest reported mean LC50 value from duplicate tests (usually two) run under the same test conditions for comparison to the criterion. In some cases direct data for multiple tests run under different test conditions were available and a measure of central tendency was used in determining an acute level of effect (see bull trout and Lost River and shortnose suckers effects analysis in sections 4.3.1 and 4.3.3, respectively)

ICE

We used USEPA's Web-ICE 3.1 to estimate acute toxicity values for endangered species when direct data were not available. We normalized surrogate input values (geometric means) to USEPA's Web-ICE 3.1 specifications (e.g., pH and hardness). The Web-ICE models were derived using only juvenile data. For our copper analysis, the surrogate inputs were limited to early life stage fish (<12 months as best determined). For all other criteria chemicals, we used all life stages to calculate the surrogate inputs. For each endangered species LC50 estimate obtained from ICE, we ran a minimum of two surrogate models and ultimately selected the model with lowest degree of uncertainty by following the recommendations of Raimondo et al. (2010a, p. 18) indicating models should have:

(1) at least eight degrees of freedom (n-2),
(2) probability or p-values <0.01,
(3) R^2 values > 0.6,
(4) a cross validation success rate > 85%,
(5) taxonomic distance ≤3 (genus = 1; family = 2; order = 3; class = 4; phylum = 5; kingdom = 6),
(6) a low mean square error (MSE) <0.22, and
(7) narrow confidence bands.

To ensure adequate protection of listed species and account for uncertainty in chemical sensitivity differences among species, we used the lower bound of the 95% confidence interval for Web-ICE model estimates to calculate the assessment effects concentration (the estimated concentration where adverse effects would not occur) for the listed species as recommended by USEPA (2005a, pp. 13-14).

Estimated No Effect Concentrations (NOECs) and Chronic Species Sensitivity Distribution

Estimated NOECs are derived from the LC50 divided by the ACR for a variety of species, and these values are then used in a species sensitivity distribution to obtain a 5th percentile. Although estimated NOECs were used extensively in the BE (USEPA 2008), we only used them to a very limited extent when it was not possible to estimate a chronic level of effect for vernal pool fairy shrimp in accordance with the method described in the USFWS (2010e) draft biological opinion for the proposed approval of cyanide aquatic life criteria due to lack of data.

ACRs were calculated to the lowest taxonomic level for a given species. For example, if species chronic data were available, then the ACR would be calculated using data only for that species. If no data were available for a given species, then an ACR would be calculated using data from next lowest taxonomic level (i.e., genus, class, etc.). ACRs were calculated for that species

either using direct species data or data for the closest taxonomic surrogate (i.e., class Branchiopoda for vernal pool fairy shrimp) aiming for a minimum of 20 values. These estimated NOECs along with available actual chronic data were used to calculate the median 5th percentile through a species sensitivity distribution. The species sensitivity distribution followed the same SYSTAT methodology as described in the above acute species sensitivity distribution section. Estimated NOECs and ACRs were calculated for species in our copper analysis. Tables with detailed information about the computation of copper ACRs for the biological opinion are included in Appendix 7.

Estimating Acute Level of Effect

Our approach to evaluate the acute level of effect analysis is very basic and follows the risk assessment models presented in the BE. We first quantified level of effect into three categories (<10%, 10%-50%, and 50%-100%) for estimating reduced survival. These categories are all based on the actual or estimated LC50 for each endangered species. By dividing the LC50 by the adjustment factor of 2.27 we produce an assessment effects concentration or $LC_{low\ (0\text{-}10\%)}$, which we use to represent the LC10. The acute criterion value is then compared to the LC50 and LC10 values to see where it falls within our three categories for estimating what percentage mortality would occur to the population at the criterion concentration. If the criterion falls between an endangered species LC10 and LC50, then the effect to the population would be between 10% and 50% mortality. For instance, if the acute criterion is greater than the LC50, then the effect to a population would be greater than 50% mortality.

Using the $LC_{low\ (0\text{-}10\%)}$ to represent the assessment effects concentration introduces a high degree of uncertainty. Typically in toxicology studies, the $LC_{low\ (0\text{-}10\%)}$ value is considered to be within the experimental background noise and unlikely to cause measureable effects (Dwyer et al. 2005, p. 147), as it is well within the margin of error for the study. However, it is possible that the result could be lethal to up to 10% of the population and therefore would not be considered a concentration where effects are minimal. Lethality of up to 10% would not be considered an insignificant or discountable effect, and would not meet conditions for a not likely to adversely affect (NLAA) determination. Therefore, we added a more conservative element to the decision to help account for the uncertainty in using a one-size-fits-all adjustment factor of 2.27 and to address the possibility that in certain cases up to 10% lethality could be observed. We used an adjustment based on the magnitude of difference between the assessment effects concentration and the acute criterion, and derived the following decision rules.

(1) Estimated assessment effects concentrations over 4 times lower than the acute criterion are NLAA listed species.
(2) Estimated assessment effects concentrations between 2 and 4 times lower than the acute criterion are LAA listed species but incidental take would not be quantifiable.
(3) Estimated assessment effects concentrations up to 2 times lower than the acute criterion are LAA listed species and incidental take is estimated as 10% of the population.

We acknowledge that estimating effects and quantifying incidental take below the LC10 is associated with a high degree of uncertainty, mostly because we do not know the relationship between the dose and response below the LC10. We used a simple additive method to associate

the magnitude of difference between the assessment effects concentration and the acute criterion and to help quantify take as appropriate, but acknowledge that the slope of the dose-response curve is unlikely to be linear below the LC10 (based on the typical dose-response curve model). However, we believe our method addresses uncertainty by adding a degree of conservatism that will better protect listed species.

When our initial acute assessment results in a level of effect that might be significant enough to result in jeopardy (e.g., effect level between 50% and 100%), we performed additional analysis with direct data to refine the acute level of effect estimate. We used probit analysis with raw data to refine the estimate of level of effect from a category to a single percentage. Data sets must be amenable to prohibit analysis, which requires at least two test concentrations that result in partial mortalities in the test organisms (Stephan 1977). The resulting slope and intercept of the regression line are used to calculate the percent level of effect at the criterion concentration.

In the refined assessment with probit analysis, multiple datasets are available for each listed species. Tests must be conducted under good laboratory practices using water quality conditions similar to those experienced by the listed species, with no abnormalities in the test reported. For these reasons, there is no basis for censoring or weighting one result over another and both the low and high percentage results are equally representative numbers to use in calculating a measure of central tendency. Our method for determining the final acute level of effect is to use the midpoint of the minimum and maximum values (extremes), calculated by adding the extremes and dividing by 2. For example, if four test results yield 1%, 3%, 24%, and 73% mortality, the calculation of acute effect level is 1% + 73% = 74%/2 = 37%. We acknowledge that there are disadvantages in using percentages (a type of ratio data) to make estimates of population means, including that ratios are relatively inaccurate and may not approximate a normal distribution (Sokal and Rohlf 1981). In addition, the midpoint of percentage data is not an ideal measure of central tendency, especially because data between the extremes are excluded. However, actual mean value most likely falls somewhere between (but not on) the extremes, and under the circumstances the midpoint is a reasonable rough estimate of the central tendency. Any other estimates of central tendency made with such few data points and based on ratios would also have disadvantages and some degree of inaccuracy similar to the midpoint.

The midpoint somewhat accounts for the variability in sensitivity of individual test organisms which respond along a normal distribution. In quantal dose responses (e.g., lethality), which follow a normal distribution, a few animals respond to the lowest dose and the highest dose, while larger numbers of animals respond to intermediate doses (Klaassen and Eaton 1991, pp. 20-21). This distribution provides the basis for probit analysis which is used widely to estimate a percent effect at various doses (concentrations) of a chemical. Thus, the midpoint of the range of results of our level of effect represents a gross estimate of the central tendency of response of the listed species.

Estimating Chronic Level of Effect

The most direct way to assess effects on listed species at the chronic criterion level is to compare the NOEC or LOEC for the species to the chronic criterion for the chemical. Then, we can predict the level and magnitude of effect and quantify take directly to the species when exposed

to the chemical at the chronic level. In many cases, a NOEC or LOEC value is not available for the species. In these cases, we estimated an assessment effects concentration using data from surrogate species to model and predict the NOEC or LOEC, and then estimated the level of effect to the listed species based on how a closely related surrogate species would respond when exposed at the chronic criterion. We used two methods to estimate the assessment effects concentrations for listed species in our Biological Opinion; the species sensitivity distribution method or the chronic level of effects model.

Species Sensitivity Distribution

The species sensitivity distribution was used very little to generate a 5th percentile for use estimating chronic level of effect, and ACRs were only used for vernal pool fairy shrimp when data were unavailable to compute a chronic level of effect.

Chronic Level of Effects Model

For the past several years, the USFWS has been involved in a national consultation with the USEPA on the proposed approval of cyanide water quality criteria. The USFWS submitted a draft biological opinion to the USEPA in January 2010 (USFWS 2010e). This draft biological opinion includes a method for quantifying a level of effect from the toxic chemical criteria to threatened and endangered species. We used this method in our Biological Opinion to conduct the jeopardy analysis and incidental take quantifications for our listed species. The methods for adapting the level of effect model described in the USFWS (2010e) draft biological opinion for the proposed approval of cyanide aquatic life criteria to our evaluation of the water quality chronic criteria are described specifically in the USGS final report associated with our interagency agreement with the USEPA (USGS 2010) and briefly below.

The chronic level of effect model is used whenever direct chronic data on a listed species are unavailable to establish an assessment effects concentration (i.e., an estimated NOEC or similar threshold value where effects are not unacceptably adverse) for the listed species. The value is determined by first calculating the ratio of the actual or ICE-estimated acute LC50 value for the listed species to the LC50 value for the surrogate species, and then multiplying this ratio by the assessment effects concentration for the surrogate species (USFWS 2010e). Using the method, we can estimate the level and magnitude of effect on a listed species when exposed at the chronic criterion. Then, by setting the listed species assessment effects concentration to the chronic criterion for a chemical, we can rearrange the equation and calculate the surrogate species assessment effects concentration as the ratio of the surrogate species LC50 to the listed species LC50 multiplied by the chemical's chronic criterion. The resulting value represents the effects concentration for the surrogate species that is equivalent to the effects concentration for the listed species at the chemical's chronic criterion. For example, if the surrogate species was five times less sensitive than the listed species (based on LC50 values), then the surrogate species assessment effects concentration would be five times higher than the chemical's criterion, and the surrogate species would have to be exposed to a concentration five times higher than the chronic criterion to experience an effect of equal magnitude to the effect on the listed species exposed to the chronic criterion.

After calculating the surrogate species assessment effects concentration, the magnitude of effect can be estimated using the chronic exposure-response curve for the surrogate species (see USFWS 2010e). Regression was used to model or fit the relationship between a criterion chemical and toxic effects to listed species based on the surrogate species chronic toxicity data for fish and invertebrates, as described in USGS (2010). Chronic toxicity studies for surrogate species were reviewed for data acceptability and the selected response and treatment data were smoothed and transformed only where appropriate (see data input and output rules described below). Procedures for conducting transformation and smoothing (calculating moving averages of response data) followed rules and limitations specified in USFWS (2010e). The regression was run by inputting the surrogate species assessment effects concentrations calculated for each of the listed species and obtaining the predicted response variable. The level of effect and confidence limits were then calculated based on the percent difference of the response variable of the test organisms from the response of control organisms.

In summary, we set the listed species assessment effects concentration to the chemical's chronic criterion concentration, and then calculate the surrogate species assessment effects concentration to obtain the magnitude of effect (on the listed species) from the exposure-response relationship for the surrogate species.

The assumptions for this model are that relative differences in sensitivity to a chemical from acute exposures are good approximations of differences in sensitivity to chronic exposures between surrogate and listed species, and the slope of the exposure-response curves for surrogate and listed species are reasonably similar.

The data input rules for selecting chronic toxicity tests suitable for use in the model include the following:

(1) valid test and quality controls with at least 80% survival of controls,
(2) monotonic treatment/response (i.e., the response [y] only moves in one direction as treatment [x] increases and data or transformed data graphically appear normally distributed), and
(3) treatment input values must be within range of tested values (data points for regression) with at least five points within treatment input range (four points accepted if insufficient information is available for a listed species or family).

Data output rules for interpreting regression results from the model include:

(1) regression must be significant ($P \leq 0.05$) with the adjusted $R^2 > 0.6$,
(2) when multiple tests are available and other test parameters are similar, use test with largest F-ratio (a larger F ratio indicates there is greater predictive ability and greater confidence in a relationship, or the more certainty that there is a difference between the groups),
(3) when multiple tests are available and F values similar for the same species and endpoint, use the most conservative maximum likelihood estimator while considering the confidence interval spread, and

(4) when available, use tests from surrogate species within the same family, or consider life history similarities as a possible surrogate selection tool (Banks et al. 2010).

Typically, we rejected more studies on a chemical than we found acceptable based on our data rules, which adds to the uncertainty of results by reducing sample size available for the model. For example, in evaluating the chronic level of effect of copper on fish, we reviewed 19 separate chronic studies and rejected four of these studies. From the 15 studies accepted, we reviewed 44 separate chronic test endpoint evaluations and rejected 18 evaluations based on our data rules, resulting in 27 acceptable test evaluations used to estimate level of effect. Eight evaluations were used to evaluate the survival endpoint representing early life stages of fathead minnow, brook trout, rainbow trout, steelhead, bluegill, fountain darter, and spotfin chub. Eleven evaluations were used to evaluate the growth endpoint (length, mass, or biomass) representing early life stages of rainbow trout, steelhead, brook trout, bluegill, and fathead minnow. Eight evaluations were used to evaluate the reproduction endpoint representing early life stages of fathead minnow, bluegill, and brook trout. Although the overall number of studies was reduced using our data rules, the quality of studies used in the assessment was better controlled and was necessary to maintain consistency in result interpretation.

Where appropriate data are unavailable for conducting the regression model according to the USFWS (2010e) draft biological opinion for the proposed approval of cyanide aquatic life criteria, an alternative method was used similar to the acute level of effect quantification. As chronic results are not as standardized as acute, there are several endpoints in the chronic database including NOECs (measured and estimated), LOECs, EC10s, and EC20s. We considered these endpoints to represent an effect level between 0% and 20%. When chronic data are unavailable, we use the 5th percentile of a closely related taxonomic surrogate to represent the threshold value for the two categories, 0% to 20% or 21% to 100%. If the criterion falls below the threshold, we assume the upper end of the category at 20% for level of effect. When the criterion is greater than the threshold, falling into the 21% to 100% level of effect category, we assume 100% effect to the population. Although assuming 100% effect is conservative, without data to refine the estimate it is considered the best approach without available science.

Frequency of Effects

In analyzing the effects of the action on the listed species being impacted, we have considered the best available laboratory information to estimate how these species or surrogates would respond from exposure to toxic chemicals in natural settings. Laboratory toxicity assessments provide good information on the direct assessment of magnitude of exposure, but water quality criteria also include a temporal element of exposure based on modeling of natural systems, and this temporal element differs from most laboratory assessments (as described below). Chemical concentrations in a natural water body fluctuate due to stream flow, effluent quality, and other factors; therefore, aquatic organisms experience variable rather than constant concentrations over time and thus the EPA includes with the criterion a time period over which exposure is averaged. In contrast, exposure to organisms in laboratory studies is more consistent and includes a standardized exposure duration and frequency.

The CMC and CCC are described temporally as measures of 1 hour and 4 days, respectively, not to be exceeded more than once every 3 years (i.e., the average concentration measured in ambient water over 1 hour or 4 days should not exceed the criterion more than once every 3 years). In acute toxicity laboratory assessments, the standard time for measuring response is typically 48 or 96 hours, while chronic toxicity test responses are measured from 7 days to multiple years for fish or an entire lifecycle for invertebrates. Therefore, there are temporal differences in the criteria as compared to the protocols for acute and chronic toxicity testing in the laboratory. EPA derives the CMC and CCC from acute and chronic laboratory tests, respectively. Therefore, we believe it is appropriate to use the results of those acute and chronic tests with their duration and frequency components along with the duration and frequency components of the criteria in describing how we view the frequency of any adverse effects from this action. Specifically, any adverse acute effects are considered to occur one time for 1 hour every 3 years (consistent with the duration and frequency components of the CMC) and are predicted based on acute toxicity test results taken from studies that typically last 48 or 96 hours. Similarly, any adverse chronic effects are considered to occur one time for a period of 4 days every 3 years (consistent with the duration and frequency components of the CCC) and are predicted based on chronic toxicity test results taken from studies that extend 7 days, multiple years, or entire lifecycles depending on the test organism.

Magnitude of Exceedence of Criteria

As mentioned previously, the CMC and CCC should not be exceeded more than once every 3 years. In evaluating adverse effects to listed species, it is important to have some sense of the magnitude of the exceedence. In their Water Quality Handbook, USEPA provides some background regarding the exceedence of criteria (USEPA 2012). "[USEPA] recommends an average frequency for excursions of both acute and chronic criteria not to exceed once in 3 years. In all cases, the recommended frequency applies to actual ambient concentrations, and excludes the influence of measurement imprecision. [USEPA] established its recommended frequency as part of its guidelines for deriving criteria . . . . [USEPA] selected the 3-year average frequency of criteria exceedence with the intent of providing for ecological recovery from a variety of severe stresses. This return interval is roughly equivalent to a 7Q10 design flow condition [note: 7Q10 is the lowest 7-day average flow in a 10-year period]. Because of the nature of the ecological recovery studies available, the severity of criteria excursions could not be rigorously related to the resulting ecological impacts. Nevertheless, [USEPA] derives its criteria intending that a single marginal criteria excursion (i.e., a slight excursion over a 1-hour period for acute or over a 4-day period for chronic) would require little or no time for recovery. If the frequency of marginal criteria excursions is not high, it can be shown that the frequency of severe stresses, requiring measurable recovery periods, would be extremely small. [USEPA] thus expects the 3-year return interval to provide a very high degree of protection."

Because USEPA develops the criteria with the intention that any allowable exceedence will be "slight" or "marginal," in analyzing potential effects to listed species we believe it is appropriate to only compare toxicity data to the CMC or CCC rather than also attempting to analyze the potential effects of an unknown concentration that substantially exceeds the CMC or CCC.

2.1.11 Additional Method Uncertainties

Laboratory studies vary in many of the physicochemical characteristics (i.e., pH, hardness, organic and inorganic ligands, temperature, etc.) of their test media. For example, Sorenson (1991, pp. 125, 132) states that changing test temperature produces striking effects on zinc toxicity and zinc toxicity varies with age of test organism. Hansen et al. (2002c, p. 72) found that water pH influences the toxicity of cadmium and zinc to bull trout, a listed endangered species. Eisler (1988a, p. 53) states that toxic effects to arsenicals to aquatic species are significantly modified by numerous abiotic factors including temperature, pH, organic content, phosphate concentrations, suspended solids, and the presence of other substances. Physicochemical characteristic addressed in the water quality criteria are as follows: metals criteria are based on hardness, ammonia criteria are based on temperature and pH, and pentachlorophenol is based on pH.

2.1.11.1 Mixtures

Water quality criteria are derived from data conducted by testing chemicals in isolation. In the aquatic environment, chemicals occur in a mixture or in combination with conventional pollutant stressors such as temperature, pH, or dissolved oxygen. Effluent discharges are not composed of a single chemical either; rather, they include several chemicals as demonstrated in NPDES permits for stormwater and effluents from WWTPs and mining. Organisms tested in treatments with multiple chemicals or other stressors such as temperature and pH produce responses that can be greater or less than expected compared to treatment from a single chemical. Thus, the response from interactions of multiple stressors can change the toxicity of a single chemical. Interactive effects can be additive, synergistic, antagonistic, or produce other effects (Sorenson 1991, p. 333-334; Klaassen and Eaton 1991, p. 17). Interactive effects can be additive (the combined effect of two stressors is equal to the sum effect of each stressor given alone), synergistic (the combined effect is much greater than the sum of the effect from each stressor given alone), or exhibit potentiation (one stressor does not have a toxic effect unless it is in the presence of another stressor).

Tests conducted with metal mixtures show that responses often differ from isolated chemicals. Eisler (1998a, p. 11) notes that copper interacts with numerous elements including iron, zinc, molybdenum, manganese, nickel, selenium, silver, cadmium, mercury, and lead. These interactions can be beneficial or harmful to exposed organisms, but the patterns of copper accumulation, metabolism, and toxicity frequently differ from those produced by copper alone. Cooper et al. (2009, p. 1523), conducted acute and chronic bioassays with copper, lead, and zinc mixtures on *Cerodaphnia dubia* and *Daphnia carinata*. The authors found that no significant effects on acute survival occurred when chemicals were tested individually at or below water quality criteria concentrations, but some constituents had an effect when presented in a mixture (Cooper et al. 2009, p. 1526). Another study of mixtures conducted by Spehar and Fiandt (1986, p. 918) tested arsenic, cadmium, chromium, copper, mercury and lead (chemicals often found together in sewage and industrial wastes). Results showed that metal mixtures at criterion maximum concentrations (acute) were found to reduce the survival of rainbow trout and *Ceriodaphnia dubia* and at chronic criterion concentrations showed adverse effects on production of daphnid young and growth of fathead minnows. In specific tests with bull trout

conducted by Hansen et al. (2002c, p. 72), test organisms were found to be more sensitive to a mixture of cadmium and zinc compared to cadmium only.

We recognize that chemicals in the environment rarely occur in isolation, and these mixtures could produce greater or lesser effects in endangered species compared to their responses from one chemical alone. These unknown interactive effects add uncertainty to the protectiveness of current water quality criteria. However, evaluating potential effects of these chemical mixtures is beyond our technical capabilities at this time given the scope of this consultation and the added complexity and uncertainty it would add to the analysis.

2.1.11.2 Metals

Although most of the metals criteria in this consultation take into account water hardness, it has long been recognized that several other water quality characteristics can also affect their toxicity. Schubauer-Berigan et al. (1993) found that the acute toxicity of metals (cadmium, copper, nickel, lead and zinc) in hard water can be greatly affected by test pH even within the range of pH 6 to 9. In tests with varying regimes of hardness, pH, alkalinity, and natural organic matter, Hoang et al. (2004, p. 91) concluded that nickel toxicity to fathead minnows was influenced by hardness and also by natural organic matter, alkalinity and pH. This is probably most well explained in relation to copper toxicity (Erickson et al. 1996 and USEPA 2003).

Criteria have been proposed which take into account these factors. By incorporating these other influential parameters into the equation, the biotic ligand model integrates the concept of bioavailability into ambient water quality criteria (Niyogi and Wood 2004, p. 6177). In simplest terms, the biotic ligand model incorporates the competition of the free metal ion with other naturally occurring cations (e.g., $Ca2^{+}$, Na^{+}, $Mg2^{+}$, H^{+}) together with complexation by abiotic ligands (e.g., dissolved organic matter, chloride, carbonates, sulfide) for binding with the biotic ligand which is the site of toxic action on the organism (Niyogi and Wood 2004, p. 6177). While biotic ligand models have been developed for several different metals, copper appears to be the most advanced in development (Niyogi and Wood 2004, p. 6180). In their 2007 Revision for the Aquatic Life Ambient Water Quality for Copper, USEPA recommends the biotic ligand model as the basis for aquatic life criteria. Input parameters to the model include: pH, dissolved organic carbon, percent humic acid, temperature, major cations (calcium, magnesium, sodium, and potassium), major anions (chloride and sulfate), and dissolved inorganic carbon (USEPA 2007, p. 12). In addition to copper, biotic ligand models are also available for silver, nickel, and zinc. As of 2004 no models for cadmium or lead had been published (Niyogi and Wood 2004, p. 6186-6187), but they were in various stages of development according to Keithly et al. (2004, p. 693).

The most recent information from the biotic ligand model development illuminates the fact that mechanisms occurring in aquatic systems are complex and varied. The metal criteria proposed for approval do not take these relationships into account. It is beyond the scope of this consultation to evaluate any of the biotic ligand models but they appear to be more comprehensive in their consideration of influential factors related to possible adverse affects of these metals on aquatic species.

Internal Regulation

For all essential elements, each species has an optimal range of concentrations required for normal metabolic functioning and Muyssen and Janssen (2001, p. 2053) note that acclimation to metals in the natural environment is well documented. Acclimation- induced sensitivity shifts may considerably affect the relevance of the derived water quality criteria guidelines. By monitoring zinc tolerance for *Daphnia magna*, Muyssen and Janssen (2001, p. 2053) found changes in zinc tolerance as a function of the zinc acclimation concentration in the culture medium and acclimation influenced their sensitivity by a factor of two. In addition, long-term zinc deficiency resulted in decreased fitness of the daphnids (Muyssen and Janssen 2001, p. 2059). These authors conclude that without pre-acclimating test organisms to conditions similar to their natural habitats for a sufficient time period, laboratory tests may overestimate the toxicity of naturally occurring substances (e.g., metals) and essential elements (e.g., zinc) (Muyssen and Janssen 2001, p. 2058). Furthermore, they believe water quality criteria based on laboratory testing that does not account for these factors may cause a deficiency of elements essential to aquatic life (Muyssen and Janssen 2001, p. 2058).

According to Sorenson (1991), for essential elements such as zinc and selenium there are two schools of thought debated by toxicologists, homeostatic regulation and ecomix concept. The concept of homeostatic regulation is defined by maintenance of essential element levels through physiological requirements for survival and a balanced supply of metal even with varying concentrations in the external environment. Alternatively, the ecomix concept argues that nutrient accumulation occurs in the same ratio as the elements occur in the environment. Sorenson (1991, pp. 121-122) indentifies studies supporting both theories specific to zinc. In their BE, USEPA (2008) considers homeostatic regulation as the internal control mechanism for all essential elements in this consultation.

These regulatory pathways of acclimation and homeostasis in aquatic organisms may result in an overestimation of adverse affects to endangered species which are based on laboratory studies, most often conducted with no acclimation time to the test chemical.

2.1.11.3 Bioaccumulation/Bioconcentration

Bioconcentration and bioaccumulation factors (BCF/BAF) are indices used to evaluate potential risk of chemicals accumulating in tissue from multiple uptake pathways. The BCF accounts for accumulation of a chemical in tissue based on uptake from water exposure only, while the BAF accounts for accumulation from both water and dietary exposures. Generally the BAF is derived from measurements in natural environments and BCF under laboratory conditions (McGeer et al. 2003, p. 1018).

McGeer et al. (2003, p. 1018) note that using the BCF/BAF model for describing the relationship between bioaccumulation and potential effects for naturally occurring inorganic substances such as metals, complex internal metal dynamics of uptake, internal sequestration, storage, active elimination, and nutrient essentiality or the potential for adverse effects go unrecognized. Furthermore, as a result of a host of factors that influence metal uptake and accumulation, BCF values for metals are likely not to be independent of exposure. McGeer et al. (2003, p. 1019, 1032) found that the accumulation of zinc, cadmium, copper, lead, nickel, and silver in aquatic

biota were generally remarkably consistent, where total body/tissue concentration varied little over a wide range of exposure concentrations, exposure conditions, and species. The authors concluded that it is virtually impossible to derive a meaningful BCF value (representative of the BCF for each metal) even when BCFs are limited to the exposure range where chronic toxicity might be expected (based on water-quality guidelines) (McGeer et al. 2003, p. 1033). The inverse correlation between BCF and exposure for these metals produces the possibility of inadequate and faulty predictions for hazard as exemplified when hazard identification based on BCFs indicates that hazard is reduced as concentration increases, a conclusion inconsistent with the toxicological literature. Therefore, McGeer et al. (2003, p. 1033) conclude that although bioaccumulation is a characteristic of the metals examined, the BCF parameter does not properly characterize metal accumulation nor its relation to the potential for toxic impacts. Understanding and predicting bioaccumulation of metals is one of the key requirements in understanding their fate and toxicity in aquatic environments and for environmental protection measures.

In USEPA's BE (2008), it appears that BCFs and BAFs were used interchangeably. However, this may not be appropriate. DeForest et al. (2007, p. 236) investigated trends in metal bioaccumulated data by evaluating data for BAFs and trophic transfer factors. BAFs are frequently 100 to 1000 times larger than BCFs for the same metal and species (DeForest et al. 2007, p. 236). These authors state that identification procedures based on persistence, bioaccumulation, and toxicity used in aquatic hazard identification for chemicals is problematic for metals for 3 reasons: (1) they are naturally persistent in the environment, (2) both essential and non-essential metals are naturally bioaccumulated and internally regulated using different strategies (e.g., active excretion, storage), and (3) the toxicity is highly influence by geochemical factors that influence metal bioavailability. DeForest et al. (2007, p. 237) hypothesize that organisms regulate non-essential metals to varying degrees because the mechanisms for regulation are not metal specific. The authors note that research on the aquatic toxicology of metals increasingly focuses on mechanisms for linking aqueous metal concentrations to critical tissue concentration and that mechanistic chronic toxicity and bioavailability models for metals are undergoing development (DeForest et al. 2007, p. 237).

In their compilation of BAFs for various metals, DeForest et al. (2007, p. 240) found that an inverse relationship typically exists for a variety of metals and metalloids. This is similar to the findings of McGeer et al. (2003) with laboratory-derived BCFs. A high BAF observed for a metal may be driven by exposure concentration or other metal and species-specific factors that influence metal bioaccumulation. DeForest et al. (2007, p. 242) state that BAF and BCF comparisons between the same species, or at least the same taxa, are most relevant due to differences in how aquatic biota regulate and store metals. Consequently, using one BAF interchangeable with BCFs for all aquatic species does not provide an accurate measure of risk to each species. An evaluation done by DeForest et al. (2007, p. 242), shows BAFs tend to be considerably larger than BCFs. However, the authors point out that higher BAF values do not necessarily point to greater hazard. Their results clearly demonstrate that identifying the most relevant BAF for a given exposure scenario is critical and not taking concentration dependent into account can lead to overestimation of hazard/risk (DeForest et al. 2007, p. 243). Authors believe that use of a fixed BAF is not a viable approach for assessing bioaccumulation hazard of metals. Analysis conducted by DeForest et al. (2007, p. 244) identified selenium as possibly

posing hazard to wildlife via secondary poisoning at water concentrations equal to chronic ambient water quality criteria being evaluated in this consultation, while cadmium, copper, lead, and zinc were predicted to pose no hazard to wildlife at water concentration equal to ambient water quality criteria.

Clearwater et al. (2002, p. 269) states that the uptake and toxicity of dietborne metals in fish is not as well characterized as waterborne mechanisms. There are also differences between the toxicity of natural diets compared to laboratory-prepared feeds which can influence a risk assessment. The authors recommend that more studies are needed to examine the effects of water quality on dietborne metals, and these studies should monitor and report parameters such as flow rates, temperature, pH, alkalinity, hardness, salinity, and anion and cation concentrations along with waterborne metals to improve interpretation (Clearwater et al. 2002, pp. 305, 308).

We acknowledge that there is substantial variation and uncertainty when using BCF and BAF values to predict accumulation in tissue and to estimate toxic effects in an organism based on the predicted whole body residue values. As noted above and in the BE, this uncertainty is especially notable for metals and essential nutrients. As with chemical mixtures, using BCF and BAF values could produce estimates of toxic effects that are greater or lesser that those experienced by endangered species in the wild, and this adds uncertainty to the protectiveness of current water quality criteria. However, the BCF and BAF values or currently the more accepted and best available technique to estimate how water borne chemicals may effect listed species.

3 References Cited

References cited within Appendix 1 are included in the References Cited (Section 10) of the Biological Opinion.