APPENDIX F

NATIONAL MARINE FISHERIES SERVICE
BIOLOGICAL OPINION

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Northwest Region
7600 Sand Point Way N.E., Bldg. 1
Seattle, WA 98115

Refer to NMFS No: 2007/08256

May 20, 2008

William A. Wood, Forest Supervisor
Salmon-Challis National Forest
1206 South Challis Street
Salmon, Idaho 83467

Elin Miller, Regional Administrator
U.S. Environmental Protection Agency
Region 10 Pacific Northwest
1200 Sixth Avenue
Mail Stop: RA-140
Seattle, Washington 98101

Re: Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Idaho Cobalt Project. Big Flat Creek, Bucktail Creek, Little Deer Creek, South Fork Big Deer Creek, Big Deer Creek, Blackbird Creek, Panther Creek, Deep Creek, Moccasin Creek, Williams Creek, Perreau Creek, and Salmon River. 6th Field HUCs: 170602030902; 170602031001; 170602031104; 170602031703; 170602031805; 170602032004; 170602032006; 170602032107; and, 170602032201. Lemhi County, Idaho (Two Projects).

Dear Mr. Wood and Ms. Miller:

The enclosed document contains a biological opinion (Opinion) prepared by the National Marine Fisheries Service (NMFS) pursuant to section 7(a)(2) of the Endangered Species Act (ESA) on the effects of constructing, operating, reclaiming, and closing the proposed Idaho Cobalt Project by Formation Capital Corporation in accordance with a Plan Of Operation (POO) approved by the Salmon-Challis National Forest (SCNF) under the General Mining Law of 1872, the Federal Land Policy and Management Act of 1976, and the Surfaces Resources Act of 1955. The SCNF is the lead Federal agency for this consultation, which also includes the issuance of a National Pollution Discharge Elimination System (NPDES) permit by the U.S. Environmental Protection Agency for the discharge of treated mine wastewater into Big Deer Creek under the Clean Water Act. In this Opinion, NMFS concludes that the actions, as proposed, are not likely to jeopardize the continued existence of Snake River spring/summer Chinook salmon or Snake River Basin steelhead, or result in the destruction or adverse modification of designated critical habitats for Snake River spring/summer Chinook salmon or Snake River Basin steelhead. In this Opinion, NMFS also concludes that the actions, as proposed, are not likely to adversely affect Snake River sockeye salmon.

As required by section 7 of the ESA, NMFS provided an incidental take statement with the Opinion. The incidental take statement describes reasonable and prudent measures NMFS considers necessary or appropriate to minimize incidental take associated with these actions. The take statement sets forth nondiscretionary terms and conditions, including reporting requirements, that the Federal agencies and any person who implements the actions must comply with to carry out the reasonable and prudent measures. Incidental take from actions that meet these terms and conditions will be exempt from the ESA take prohibition.

This document also includes the results of our analysis of the actions likely effects on Essential Fish Habitat (EFH) pursuant to section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA), and includes two conservation recommendations to avoid, minimize, or otherwise offset potential adverse effects on EFH. These Conservation Recommendations are a non-identical set of the ESA Terms and Conditions. Section 305(b)(4)(B) of the MSA requires Federal agencies to provide a detailed written response to NMFS within 30 days after receiving these recommendations.

If the response is inconsistent with the EFH conservation recommendations, the SCNF and EPA must explain why the recommendations will not be followed, including the justification for any disagreements over the effects of the action and the recommendations. In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many conservation recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, in your statutory reply to the EFH portion of this consultation, we ask that you clearly identify the number of conservation recommendations accepted.

If you have questions regarding this consultation, please contact Mr. Bill Lind at (208) 378-5697.

Sincerely,

Michael Tehan

for D. Robert Lohn
Regional Administrator

Enclosure

cc:
W. Scales – FCC
D. Miller – FWS
C. Mebane – USGS
T. Bassista – IDWR
B. Edmo - Shoshone-Bannock Tribes
J. Kraayenbrink - BLM
R. Brochu – COE
T. Curet – IDFG
T. Herron – IDEQ
R. Miles - Nez Perce Tribe

Endangered Species Act – Section 7 Consultation
Biological Opinion

&

Magnuson-Stevens Fishery Conservation and Management Act
Essential Fish Habitat Consultation

Idaho Cobalt Project - Big Flat Creek, Bucktail Creek, Little Deer Creek, South Fork Big Deer Creek, Big Deer Creek, Blackbird Creek, Panther Creek, Deep Creek, Moccasin Creek, Williams Creek, Perreau Creek, Salmon River. 6th Field HUCs: 170602030902; 170602031001; 170602031104; 170602031703; 170602031805; 170602032004; 170602032006; 170602032107; and, 170602032201. Lemhi County, Idaho.

Lead Action Agency: USDA Forest Service, Salmon-Challis National Forest

Consultation Conducted By: National Marine Fisheries Service, Northwest Region

Date Issued: May 20, 2008

Issued by: Michael Tehan for D. Robert Lohn, Regional Administrator

NMFS No.: **2007/08256**

TABLE OF CONTENTS

TABLES

APPENDICES

ACRONYMS

4H:1V	4 horizontal to 1 vertical
ATSDR	Agency for Toxic Substances and Disease Registry
BMPs	Best Management Practices
BMSG	Blackbird Mine Site Group
BOD	Biological Oxygen Demand
$CaCO_3$	Calcium Carbonate
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CHART	Critical Habitat Analytical Review Team
COC	Contaminants of Concern
COD	Chemical Oxygen Demand
DEIS	Draft EIS
DO	Dissolved Oxygen
DPS	Distinct Population Segment
DQA	Data Quality Act
DSM	Dynamic System Model
EFH	Essential Fish Habitat
Eh	redox potential
EIS	Environmental Impact Statement
EOG	Electro-Olfactogram
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
FAV	Final Acute Value
FCC	Formation Capital Corporation
FCRPS	Federal Columbia River Power Systems
FEIS	Final EIS
FR	Forest Road
FWS	U.S. Fish and Wildlife Service
GMAV	Genus Mean Acute Values
GPD	Gallons per Day
GPM	Gallons per Minute
H_2S	Hydrogen Sulfide
HDPE	High-Density Polyethylene
HUC	Hydrologic Unit Codes
ICBTRT	Interior Columbia Basin Technical Recovery Team
ICP or Project	Idaho Cobalt Project
IDEQ	Idaho Department of Environmental Quality

IDFG	Idaho Department of Fish and Game
IDL	Idaho Department of Lands
ILL	Incipient Lethal Level
ILT	Incipient Lethal Temperature
ISHO	Idaho State Habitat Office
LOECs	Lowest-Observed-adverse Effect Concentrations
LWD	Large Woody Debris
MP	Milepost
MPGs	Major Population Groups
MSA	Magnuson-Stevens Fishery Conservation and Management Act
MSDS	Material Safety Data Sheet
NAS	National Academy of Sciences
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service
NOEC	No Observed Effect Concentrations
NOI	Notice of Intent
NPDES	National Pollution Discharge Elimination System
ODEQ	Oregon Department of Environmental Quality
Opinion	Biological Opinion
PCEs	Primary Constituent Elements
PFMC	Pacific Fishery Management Council
POO	Plan Of Operation
PVC	Poly-Vinyl Chloride
QA/QC	Quality Assurance/Quality Control
RHCA	Riparian Habitat Conservation Areas
RPMs	Reasonable and Prudent Measures
RUP	Road Use Permit
SCNF	Salmon-Challis National Forest
SMAV	Species Mean Acute Values
SPCC	Spill Control and Countermeasures
SWMP	Stormwater Management Plan
TDS	Total Dissolved Solids
tpd	Tons per Day
TPY	Tons per Year
TSS	Total Suspended Solids
TU_C	Toxicity Units
TWSF	Tailings and Waste rock Storage Facility
UILT	Upper Incipient Lethal Temperature
USFS	U. S. Forest Service
UUILT	Ultimate Upper Incipient Lethal Temperature

VSP	Viable Salmonid Population
WEPP	Water Erosion Prediction Project
WET	Whole Effluent Toxicity

1. INTRODUCTION

The biological opinion (Opinion) and incidental take statement portions of this consultation were prepared by the National Marine Fisheries Service (NMFS) in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973, as amended (16 USC 1531, *et seq*.), and implementing regulations at 50 CFR 402. With respect to designated critical habitat, the following analysis relied only on the statutory provisions of the ESA, and not on the regulatory definition of "destruction or adverse modification" at 50 CFR 402.02.

The Essential Fish Habitat (EFH) consultation was prepared in accordance with section 305(b)(2) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 USC 1801, *et seq*.) and implementing regulations at 50 CFR 600. The administrative record for this consultation is on file at the Idaho State Habitat Office (ISHO) in Boise, Idaho.

1.1. Background and Consultation History

Formation Capital Corporation (FCC) has developed a Plan of Operations (POO) outlining a mineral development project located in the Panther Creek drainage of the Salmon-Cobalt Ranger District, Salmon-Challis National Forest (SCNF). The project is located in T21N, R18E, sections 8, 9, 15, 16, 17, 20, 21, and 22. The FCC property is composed of several mineral deposits acquired by locating and filing mining claims within the Salmon–Cobalt Ranger District of the SCNF. The property consists of unpatented mining claims on National Forest System Lands. The SCNF is the lead Federal action agency for the Idaho Cobalt Project (ICP or Project) and is responsible for the ESA/MSA consultation under a joint Biological Assessment (BA) drafted with the U.S. Environmental Protection Agency (EPA). EPA will be issuing a National Pollutant Discharge Elimination System (NPDES) permit for the ICP and is also an action agency for the purposes of this consultation. The FCC has been granted formal applicant status by the SCNF and the EPA.

The SCNF held an initial public meeting to provide information on the ICP on July 20, 2001, at the City Center in Salmon, Idaho. The SCNF issued a Notice of Intent (NOI) to prepare an Environmental Impact Statement (EIS) for the proposed mining project in the Federal Register on September 10, 2001, and held public scoping meetings on October 10, 2001, in Challis, Idaho, and October 11, 2001, in Salmon, Idaho.

In spring of 2001 the SCNF and FCC developed Plans of Study and began baseline data collection. FCC continued data collection for some disciplines such as water quality through 2006. In 2002, FCC notified the SCNF that the project planning was being placed in an interim status to review issues, collect supplemental environmental baseline information, prepare a modified POO, and obtain adequate funding to carry the Project forward.

In 2004, FCC reinitiated active work on the Project including additional environmental data collection and analysis. The FCC's technical consultants prepared final Technical Reports and evaluations in support of a Modified Plan of Operations. A Modified Plan of Operations was submitted in February 2005; further modifications were submitted in April and June 2006.

In May 2006, FCC submitted an application to EPA for a NPDES permit to discharge treated water to Big Deer Creek and submitted additional revisions to their POO. With the submittal of the NPDES application, the EPA decided to enter the National Environmental Policy Act (NEPA) process as an official cooperating agency. Idaho Department of Environmental Quality (IDEQ) is also a cooperating agency. A revised NOI was published in the Federal Register on November 1, 2006, indicating EPA's involvement as a cooperating agency and the projected Draft EIS (DEIS) distribution in 1st Quarter 2007.

The SCNF, the EIS contractor, and participating agencies have been involved with review of the FCC POO and supporting documents, analysis of effects, development of alternatives, and preparation of the DEIS. In February 2007, a DEIS was completed and distributed to the public for review and comment.

NMFS routinely participated in weekly conference calls on the ICP with the SCNF, EPA, IDEQ, and the U.S. Fish and Wildlife Service (FWS) since 2004. NMFS has reviewed the DEIS, the draft NPDES permit (NPDES Permit Number ID-002832-1), and the final biological assessment (BA) received December 28, 2007. NMFS staff also attended a workshop held on October 25, 2007, in Boise, Idaho, and participated in a field tour of the mining site and surrounding watersheds. Information was also provided to the Salmon-Challis Level I Team in 2007 but no action was requested. NMFS provided the SCNF, EPA, and FCC with draft terms and conditions on April 10, 2008.

The ICP would likely affect tribal trust resources. Because the action is likely to affect tribal trust resources, NMFS contacted the Shoshone-Bannock and Nez Perce Tribes pursuant to the Secretarial Order (June 5, 1997). NMFS provided draft terms and conditions to the Nez Perce and Shoshone-Bannock Tribes on May 01, 2008. No comments have been received from the Nez Perce Tribe. The Shoshone-Bannock Tribe has requested a meeting with NMFS to discuss the Project, but is aware that this Opinion may be finalized before the meeting can be conducted.

1.2. Proposed Action

The activities that are the subject matter of this consultation are FCC's construction, operation, closure, and reclamation of the ICP. This Opinion evaluates two Federal actions associated with these activities, the SCNF's approval of FCC's POO, and the EPA's issuance of the NPDES permit.

The ICP would consist of developing two deposits over the life of the Project, the Ram (with 2.23 million tons of known ore) and the Sunshine (2.57 million tons known ore). Development of each deposit would use underground mining methods. The average rate of mining production would be 280,000 tons per year (tpy), or 800 tons per day (tpd), based on mine operation of 350 days per year. During start-up, the rate would be approximately 400 tpd in the first year, increasing to full production by year three. However, it's possible that production could reach as much as 1,200 tpd. Mining would begin initially on the Ram deposit, with mining of the Sunshine deposit occurring in subsequent years.

Exploration for additional ore reserves is anticipated to continue through the life of the ICP operations. If additional ore tonnage is identified and defined, the production life of the ICP may be extended beyond the currently proposed mine and mill life schedule. Any such extension is beyond the scope of this consultation and would require further ESA review.

1.2.1. Action Overview

There would be three main phases in the life of the ICP: the construction phase (approximately 2 years), the operating phase (10 to 12 years), and the reclamation/closure phase (2 years for surface reclamation, and up to 30 or more years of post mine water treatment and monitoring). There would also be concurrent reclamation in the construction and operating phases as existing disturbed areas or new disturbances are reclaimed post-use.

The construction phase would include improving existing roads, and construction of new roads and facilities. Mine and mill facilities to be constructed would include the Ram mine portal, the tram, the mill/plant, the first phase of the tailings and waste rock storage facility (TWSF), water management pond, water treatment and discharge facilities, new and existing improved roads, borrow areas, and a soil stockpile area. Ancillary facilities would include power lines, fuel storage tanks, water ditches, and the septic and drain field.

The Ram and Sunshine mine portals would be located on the slopes above Bucktail Creek. Declines would be developed from portals located above the groundwater level and would be designed to ensure that water does not drain from the portals. The ore processing mill (flotation) and ancillary facilities would be located on the Big Flat, a relatively flat area located between the drainages of Big Deer Creek and Little Deer Creek. Ancillary facilities would include water treatment, offices, warehouse, change rooms, shipping and receiving docks, emergency sleep quarters, and other structures.

Power for the Project would be obtained from an existing power line that delivers power to the adjacent Blackbird Mine Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) site. Emergency power would be supplied with diesel generating equipment.

The operating phase would include mining, ore processing, and disposal of waste products including tailings, waste rock, water treatment waste, and excess water. Following development of the Ram mine, the Sunshine Mine would subsequently be developed to supplement production from the Ram mine. The mill would initially process approximately 400 tpd; increasing to 800 tpd as the underground Ram mine expands. The TWSF would be expanded to final capacity.

At full production, the mill would produce approximately 32 dry tons of concentrate and 768 dry tons of tailings per day. The proposed action provides for disposal of waste rock and tailings in a 36 acre lined TWSF. The underliner would consist of an impermeable soil (or engineered clay) layer and a synthetic liner. A drainage collection system would be constructed over the liner to collect water that infiltrates the tailings and waste rock. This water would be conveyed to the nearby water management pond.

Tailings would be dewatered prior to placing them in the TWSF. Water treatment waste would be placed in the TWSF and any post closure waste would be hauled offsite. Approximately half of the tailings produced at the mill would be turned into paste tailings and utilized in backfilling the Ram underground workings. Excess treated water would be discharged via a pipeline to an instream diffuser at Big Deer Creek in accordance with a NPDES discharge permit. A comprehensive operational surface water and groundwater quality monitoring and reporting program would be implemented.

The reclamation and closure phase would focus on reclaiming lands disturbed by FCC's mining activities and providing for long-term management of the reclaimed facilities and mine water following cessation of mining and dewatering. There would be limited concurrent reclamation during the operations phase, primarily on completed portions of the TWSF. The reclamation phase would include final shaping, covering, and vegetation of the TWSF, sealing mine portals and demolishing the mill and tram system. Ultimately as they are no longer needed the water treatment system, power line, substation, and roads would be reclaimed.

At the end of the operational mining period, cessation of mine dewatering would be contingent upon monitoring results and projections indicating no unacceptable effects to water quality objectives or Blackbird cleanup goals. If appropriate, the mine workings would be allowed to fill with groundwater, and water from the mines would be released to the watershed. If needed to prevent unacceptable water quality impacts, a bedrock groundwater capture well system would be used to capture mine water that flows out of the mine workings during the closure period. If the bedrock groundwater capture system is insufficient, an alluvial groundwater/surface water capture system in lower Bucktail Creek drainage would be used. Captured water would be treated and discharged into Big Deer Creek through a NPDES permitted facility.

A comprehensive post-operational (closure) surface water and groundwater quality monitoring and reporting program would be implemented. Bonds or other financial security sufficient to allow the U.S. Forest Service (USFS) to reclaim the project would be posted. Agreements with Noranda to utilize existing roads and power lines on private property would be obtained. A more detailed description of each component/phase of the proposed action is included below.

1.2.2. Transportation

The FCC's proposed transportation route for employees and supplies would be via Williams Creek Road (Forest Road [FR] 201) to the Williams Creek Summit, to the Deep Creek Road (FR 101), then to the Morgan - Panther Creek Road (FR 055), to Blackbird Creek Road (FR 115). The FCC estimates that 10 vans and four pickup trucks would be required daily to transport employees to and from the site. The FCC's proposed employee transportation route is also proposed for the transportation of concentrate, equipment, reagents, and other freight.

Table 1 outlines FCC's anticipated list of chemicals, reagents, and operating supplies that would be transported to and from the mine during the operating phase. During the closure and reclamation phase, the majority of the chemicals/reagents (e.g., blasting compounds, ore processing reagents, ore concentrate) with the exception of fuel, lubricants, and possibly water treatment chemicals would no longer be used or transported. If post-closure water treatment is needed, water treatment waste consisting of lime sludge (20 tons) and zeolites (3 tons) would be transported off-site to an approved repository.

All ICP personnel would carpool to the site in FCC vans or pickups. All ICP personnel and contractors would be instructed regarding the ICP Transportation Procedures and Plans and the requirements of the USFS Road Use Permit (RUP) to enhance the safety of access to the site and to reduce the impacts of travel to the site.

Table 1. Chemicals, Reagents, and Operating Supplies – Idaho Cobalt Project

Reagent / Chemical or Product	Annual Use	Container Type	Container Size	Trucks/Year (per day)[1]
AERO® 343 Xanthate	280 tons	Flo Bin	1 ton	14
AERO® 350 Xanthate	308 tons	Flo Bin	1 ton	16
AEROFROTH® 65 Frother	42 tons	Plastic Barrel	55 gallons	3
Sodium Sulfide	100 tons	Sacks	50 pounds	1 – 5
Superfloc	24 tons	Sacks	50 pounds	Ship w/frother
Lime	75 tons	Super Sack	1000 pounds	4
Diesel	750,000 gallons	Fuel Truck	4,500 gallons	150
Gasoline	5,000 gallons	Fuel Truck	4,500 gallons	Ship w/diesel
Cement	2,500 tons	Bulk	20 tons	115
Oils, Lubricants, Grease, Antifreeze	10,000 gallons	Barrel	55 gallons	Ship w/diesel
Propane	40,000 gallons	Fuel Truck	9,400 gallons	5
Anti-scalant	4,000 gallons	250 gallon tote	4,000 gallons	1
Ammonium Nitrate	450 tons	Bulk Container	10 tons	45
Bulk Concentrate	11,200 tons	Sealed Container	16 tons	700 (2)
Water Treatment Chemicals and Reagents				
Polymer Flocculant	20 gallons	Sealed Pail	5 gallons	0[3]
Hydrated Lime	20 tons	Super Sack	1,000 pounds	2
Hydrochloric Acid	400 gallons	Plastic Drum	55 gallons	2
Coagulant	250 gallons	Sealed Tote	50 gallons	0[3]
Methanol	3,000 gallons	Bulk Truck	3,000 gallons	1[1]
Zeolites	3 tons	Super Sack	1,000 pounds	1
Employees				
		Vans		3,500 (10)
		Pickups		1,400 (4)

[1] Average values and actual truck count would vary.
[2] Does not add to truck count, as the product would be backhauled from the refinery. More fully described in FCC's Storm Water Management Plan for the Idaho Cobalt Project (Telesto, 2006a).
[3] Does not add to truck count, as the small quantity would be transported with other materials.

During construction, large mobile equipment, large loads of construction materials, and large equipment being installed at the ICP facility sites would be transported to the site by tractor-trailer. Supplies would be transported by single-frame trucks whenever feasible. Pilot cars would lead tractor-trailer loads and fuel trucks to the site to reduce accident risk. The operators of tractor-trailer loads and fuel trucks would also be instructed regarding the requirements of the Transportation Procedures and Plans.

The type and quantity of supplies to be used by the ICP during operation are listed in Table 1. Single-frame trucks would be used to transport operating supplies. For materials transported via tractor-trailer, whenever possible, the tractor-trailer would be off-loaded in Salmon, Idaho, and supplies reloaded onto a single-frame truck for transport to the ICP.

Approximately 11,200 tons of concentrate are anticipated to be shipped from the ICP mill facility annually. Steel roll-off containers with locking lids would be used to transport the concentrate. Empty containers would be filled at the mill, sealed, and loaded on single-frame trucks for transport to Salmon, Idaho. In Salmon, concentrate containers would be transferred to tractor-trailer equipment for transportation to a processing facility.

The FCC has prepared a spill control and countermeasures (SPCC) plan that addresses management of hazardous materials during shipping and storage. Their plan includes notification of the ICP facility prior to transport of fuels or chemicals, travel only during daylight hours, use of pilot vehicles, and continuous radio contact with pilot vehicle and facility during transport.

1.2.2.1. Road Upgrades

Proposed road improvements are required to improve public safety and reduce potential sediment yield to streams. Proposed mitigation would improve safety, reduce sediment delivery to streams, and reduce spill risks associated with transporting hazardous materials. Appendix A, Figure 1 and Table 1 show the locations and describe the types of road upgrades proposed. Mitigation measures proposed would occur in a phased approach, which would result in resurfacing the entire project access route. The phased work would include:

Phase I – Repair or replace areas with safety and environmental concerns.

1. Reconstruct sections of Morgan Creek – Panther Creek Road (No. 60055), to raise the road grade through the sections lying within the floodplain, shape and drain the subgrade, and place 6 inches of gravel between Deep Creek Road and Blackbird Creek Road.

2. Construct a new section of Williams Creek Road (No. 60021) between milepost (MP) 7.1 and 8.1 to bypass the switchbacks and create a steady grade climbing to the upper bench. Two culverts will be removed from the switchbacks where the existing road crosses North Fork Williams Creek. The stream will be restored to a natural condition

following culvert removal. Recontour and reclaim the replaced section of road between MP 7.1 and 8.1. Shape the subgrade and place 6 inches of gravel from the end of the pavement at MP 4.0 to the end of the new construction at MP 8.1.

3. Construct five turnouts on Blackbird Creek Road (No. 60115) between MP 36.6 and 38.7 to allow safe passing of vehicles.

4. Reconstruct segments between MP 35.7 and 37.4 to raise the grade above the floodplain and improve channel width. Shape the subgrade and place 6 inches of gravel from the Blackbird gate to the mine site.

Phase II – Replace gravel in worn areas.

1. Shape and drain the subgrade of Blackbird Creek Road and place 6 inches of gravel from Morgan Creek – Panther Creek Road to the Blackbird gate.

2. Shape and drain the subgrade of Deep Creek Road, (No. 60101), from the junction with Williams Creek Road to MP 20.85. Place 6 inches of gravel on this road segment.

3. Shape and drain Williams Creek Road from Williams Creek Summit to MP 8.1, with the exception of the segment between MP 12.45 to 12.85. Reinforce the subgrade between MP 11.65 and 12.45, and place 6 inches of gravel.

Phase III – Place surface rock on worn areas not surfaced under Phase I and Phase II.

1. Place 4 inches of gravel on Williams Creek Road between MP 12.45 and 12.85, and from Williams Creek Summit to the Deep Creek Road junction.

2. Place 4 inches of gravel on Deep Creek Road from MP 20.85 to the Morgan Creek to Panther Creek junction.

3. Place 4 inches of gravel on any sections of Phase I and Phase II that show excessive wear. Additionally, the Project Access Route would be treated with dust abatement for safety and to protect the investment in the gravel by keeping the fines in the gravel structure.

An easement through Homestead Entry Survey 71 (Cobalt Townsite) will need to be finalized.

1.2.2.2. On-Site Transportation

A combination of newly constructed roads and existing improved roads are proposed to provide access to the ICP site. All roads would be constructed and improved in accordance with USFS guidelines for road construction. The specifications for road design are provided in the ICP Conceptual Road Design (TTE, 2006).

The proposed facility layout and site roads are shown in Appendix A, Figure 2. The proposed project would require construction of 4.1 miles of new road. Approximately 11.3 miles of existing roads would be improved as necessary to handle ore haulage, larger trucks and increased traffic. Stormwater ditches and sediment control measures on all roads would be constructed in accordance with Best Management Practices (BMPs) for Mining in Idaho guidelines (Idaho Department of Lands [IDL] 1992) to control stormwater runoff.

Primary Roads - All main access roads and roads over which ore or waste rock would be hauled are considered primary roads. Turnouts would be constructed in appropriate locations along the primary roads to allow safe passage of vehicles. Existing and proposed primary roads include:

- A road from the Big Flat to the Ram portal. This road requires improvements, as well as construction of approximately 0.5 miles of new road in two segments. Road distance from the Ram portal to the mill site is approximately 2.8 miles.

- An existing road is proposed for access to the Sunshine portal. The haul distance from the Sunshine portal to the mill site is approximately 1.0 mile.

- About 0.7 mile of new road is proposed for access from the mill site to the TWSF.

Secondary Roads - Roads proposed for daily, year-round use in the operations but are not ore haul routes or main access routes are considered secondary roads. Approximately 1.5 miles of new secondary road construction is proposed for access to the TWSF, soil stockpile, water management ponds, and rerouted access to the upper Bucktail drainage.

Tertiary Roads - Site roads proposed for seasonal use or intermittent use such as roads required to access surface water and groundwater monitoring locations are considered tertiary roads. Tertiary roads are proposed to provide access to the tram tower corridor, the soil stockpile, and surface water and groundwater monitoring locations. This class of roads, as proposed, includes use of about 8.2 miles of existing roads and the construction of approximately 1.4 miles of new roads.

The FCC proposes to negotiate an access agreement with the Blackbird Mine to use the existing mine road on private land (Noranda) through the Blackbird Mine CERCLA site to the Meadow Creek/Bucktail Creek divide. The FCC, for the duration of use, would maintain BMPs, channels, culverts, and other sediment/stormwater control facilities that exist on sections of Noranda property that would be used by FCC. The FCC would upgrade the road from the Noranda water treatment plant gate to the top of the ridge (6.0 miles), including vertical realignment of a 0.3 mile section known as Buddy's Grade.

The primary and secondary roads on the Project site would be maintained year-round, including snow removal during the winter months. Snow removal would generally be accomplished on these roads with the use of a rotary snow blower/plow. The snow would be thrown above or below the road corridor into areas where snowmelt would not create sedimentation or where the snowmelt would be addressed by BMPs. If snow removal by rotary snow blower/plow were not possible, a grader or loader and truck team would remove the snow.

1.2.3. Workforce

The anticipated personnel requirements for the mine, mine maintenance, engineering and geology, and surface facilities support departments during the first year of production would be 69 persons, increasing to 109 persons at full production. The anticipated personnel requirements for the milling operation would be approximately 31 persons. Total Project employment would be approximately 157. The work force numbers may be temporarily higher during construction and start up. During closure and reclamation, the work force would be reduced significantly.

The FCC plans to operate the mine 24 hours per day, five days per week, for approximately 250 days per year. Mill production would operate 24 hours per day, seven days per week, for approximately 350 days per year.

1.2.4. Mine Mill and Facilities

The mill would be located at the Big Flat, an area of relatively flat topography east of the Ram deposit. The mill would include crushing and grinding equipment, flotation cells, concentrate thickeners, tailings thickeners, concentrate filters, tailings filters, and ancillary equipment. Additional facilities located at or near the mill site would include the mine office, change house, maintenance shop, emergency sleeping quarters, and warehouse. Facilities would result in a total surface disturbance area of 10.9 acres. Additionally, there would be a portable explosives magazine located in a suitable site with good access, but removed a safe distance from the mine buildings. There would also be a startup/backup generator.

1.2.4.1. Mill Processing

Trucks or a tramcar would transport ore from the Ram to the mill where it would be dumped into a hopper. Ore from the Sunshine would be hauled by off-highway trucks and dumped into a hopper. A conveyor would lift the ore from the hopper onto the coarse ore stockpile. The stockpile area would have a maximum capacity of approximately 22,000 tons. The ore stockpile area would be a concrete pad designed to collect all surface water runoff, decant, and pipe the water to the process pond.

A flotation mill would be used to process ore from the mine. The milling process would reduce the run-of-mine ore to minus 0.5-inch size in the primary and secondary crushing area. This material would then be reduced to minus 200-mesh size, in a liquid pulp, in the ball mills. The pulp from the ball mills would be conditioned and processed in the flotation circuits, where the ore minerals would be floated away from the waste (gangue). The concentrated ore minerals would be dewatered in a thickener and a drum filter prior to shipment. The waste material, or tailings, would also be dewatered prior to disposal at the TWSF or as backfill in the mine.

At a production rate of 800 tons of ore per day, the mill would produce approximately 32 to 40 dry tons of concentrate per day and approximately 768 dry tons of tailings per day. The concentrate would be shipped to an offsite processing facility. Mill production may vary from 800 to 1,200 tpd, and the operating schedule may vary from 250 to 350 days per year.

The concentrate would be dried to approximately 10% moisture content using a conventional thickener followed by a vacuum filter. The concentrate would be temporarily stored, prior to shipping to the offsite processing facility, in a shed adjoining the mill building. Concentrate would be shipped in modified roll-off containers. Each container would hold approximately 16- to 20-tons of concentrate. The containers would be of steel construction with steel locking lids. Concentrate would be loaded into the containers, and the lids would be closed and locked. Excess concentrate would be removed from the exterior of the containers and the containers would be hoisted onto a truck and clamped into position prior to beginning the journey to the off-site processing facility.

1.2.4.2. Materials and Supplies

Mill reagents used to recover the minerals include sodium xanthate, potassium xanthate, SUPERFLOC™, and lime. An estimate of the annual quantity of the mill materials and supplies is shown in Table 1. The xanthates, frother, and superfloc are biodegradable polymers. The reagent lime would largely be sorbed by the sulfides.

1.2.4.3. Borrow Areas

During construction and reclamation there would be a need for road surfacing materials, drain rock, and riprap. Three borrow areas, including one on the Williams Creek road near the Williams Creek summit (Leesburg East), one along Blackbird Creek, and another in the Bucktail Creek drainage have been identified (Appendix A, Figure 2). All of these borrow areas are on USFS land and have been previously used as borrow sources.

The materials in the borrow area in the Bucktail drainage would be used for surfacing materials for the underground mine roads and other construction and reclamation activities. Approximately 40,000 cubic yards of these materials would be required, and this amount is available at this borrow area. The materials in the Blackbird Creek borrow area have been tested and found to be appropriate for use as road surfacing materials. It is estimated that an additional 20,000 cubic yards of materials would be required for site roads, and is also available at this borrow area. Road improvement work on the Williams Creek Road would utilize borrow from the existing Leesburg East pit located on Forest land in T21N, R20E, section 27. Use of the Leesburg East Pit would require an estimated three acres additional disturbance at this site, which would be reclaimed following its use. All three borrow areas are located outside of riparian habitat conservation areas (RHCAs).

1.2.5. Mine Workings

The mining methods for the Ram and Sunshine deposits would use cut and fill mining. The mining sequence would include drilling between 30 to 40 holes in the ore face, loading these holes with explosive, blasting, scaling loose rock from the back (ceiling) and ribs (walls), mucking the broken ore with a scoop tram, loading a haul truck with the scoop tram, and installing rock bolts for rib and back support. The openings (stopes) created from ore excavation would then be filled with paste tailings and waste rock while the access tunnels would be left open.

The FCC estimates that the mines at full development would produce an average of approximately 51 gallons per minute (gpm) of water. Mine water flows would be collected in sumps in the mine to allow suspended solids to settle. Water storage tanks would be located at the portals for storage of water pumped from the mines. The water would then be pumped to the mill for treatment and reuse, and then to the water management ponds for storage and handling. Water would also be used in the underground mine for dust suppression. Excess water would be treated and discharged to Big Deer Creek.

Ram Portal - The Ram portal is located in an area of steep slopes. A retaining wall for the portal platform would be constructed using an engineered soil reinforcing technique. The platform would contain an office, tool/maintenance shed, surge water tank/pump station, and hoppers for loading ore and waste rock onto trucks or into the tramcar. Other equipment located on the platform would include a transformer, emergency generator, and diesel storage tank.

Mine access to the Ram would be via road from the Big Flat to a portal at 7,060 feet in elevation (Appendix A, Figure 2), and then via an underground decline driven at approximately 12% to the ore zones. The decline would be used to transport people and materials in and out of the mine and as a haul route for the waste rock and ore to the portal, as well as for a portion of the tailings to be brought back into the mine. Two ventilation raises, or airshafts, approximately 8 feet in diameter, would provide ventilation and emergency escape ways. Additional drifts and crosscuts would be developed to provide access to the ore blocks. The total length of mine workings for the Ram deposit is estimated to be about 38,000 feet. There would be approximately 9,800 feet of open stope at any given time. The Ram underground workings would also include a chamber to accommodate tailings dewatering equipment.

An overhead tram would potentially be constructed from the Ram portal to the mill located on the Big Flat. The tram would be built if and when economic conditions justify its construction. The conceptual design for the tram includes an approximate 100 cubic-foot tramcar traveling on track cables and moved by a haul rope. Three or more towers, approximately 45 feet high, would support the track cables. The tramcar would be loaded with either waste rock or ore from a hopper at the Ram portal. Ore would be discharged onto an ore stockpile at the mill crusher. Waste rock would be discharged at the mill and trucked to the TWSF for final disposal.

Sunshine Portal - At the Sunshine, an existing portal would be upgraded to access the internal decline. The existing portal platform would be reshaped and backsloped to control runoff. An office, tool/maintenance shed, diesel storage tank, and water tank would be located on the platform. The decline would be used to transport people and materials in and out of the mine and as a haulage route for ore, waste rock, and tailings. One airshaft, approximately 8 feet in diameter, would provide ventilation and an emergency escape way. Sunshine ore would be hauled directly to the mill in 20-ton trucks and placed in the ore stockpile. Waste rock slash that remains in the mine would be amended with alkaline materials (e.g., limestone, cement) to reduce the potential for leaching of metals after closure.

1.2.6. Mine Dewatering

Mine dewatering would be accomplished by a series of skid-mounted, self-contained pump units. Discharge from the mine dewatering system would be delivered to a holding tank on the portal pad. This tank would be sized to contain the entire backflow from draining the pipeline from the mill on the Big Flat. The tank would be housed in the heated portal pump station to prevent icing in the winter.

The pump station would also house pumps for lifting water to the mill from dewatering the mine and tailings. Pumping from the Ram portal to the mill would be accomplished via a steel pipe with secondary containment. To reach the 8,050 elevation high point at the mill site, this pipeline would be approximately 2,300 feet long. An air intake with a check valve at the high point would allow the line to be self-draining in the event of a pump shutdown. The pipeline would follow the tram right-of-way (Appendix A, Figure 2) and would be winterized to prevent freezing. During an emergency shutdown or production curtailment, the mine pumps would continue to operate.

Secondary containment would include pipe-in-pipe for all areas where the piping is not within the mine or other secondary containment such as the mill building. The system would include leak detection at all low points and at pipe-to-pipe connections. The pumps would have the capacity to handle short periods of high-yield mine inflows caused by structure dewatering. However, it is possible a flow event could occur that exceeds the capacity of the mine pumps. Should that occur, water would build up in the mine sumps until the inflow once again falls below the pumping capacity at which time water levels would begin to return to their long-term operating level.

The Sunshine Mine is not expected to produce enough water for dust suppression or drilling; makeup water would be supplied from the mill circuit. If excess water were produced, it would be pumped back to the mill.

1.2.7. Tailings and Waste Rock Management

The mining operation would produce waste rock and tailings that would have to be contained. This would be accomplished through three mechanisms: (1) A TWSF would be developed to store excess waste rock and tailings; (2) tailings would be converted into a paste and used for backfill in the Ram Mine as mining progresses; and (3) waste rock slash remaining in the mine for use in ramp construction would be amended with alkaline materials to reduce the risk of acid generation and leaching of metals.

1.2.7.1. Waste Rock Production and Characterization

Waste rock would be delivered to the TWSF by truck or tram from the Ram and by truck from the Sunshine. Waste rock was characterized in the baseline geochemical testing program by the analysis of 239 waste rock samples. The majority of the waste rock (approximately 80%) is not expected to present an acid-generation problem as the quartzite has a low pyritic sulfide content. However, approximately 20% of the waste rock is predicted to generate slightly acidic solutions containing variable concentrations of soluble arsenic, cobalt, copper, and zinc. Waste rock would be placed in a separate area of the TWSF and would be layered with tailings in the waste rock area. Commingling of tailings and waste rock as they are placed in the TWSF would result in the hydrologic and geochemical characteristics of the TWSF behaving as though it consisted almost entirely of tailings and reduce risk of metals leaching from waste rock Segregation of waste rock into a separate area would allow drainage from the waste rock to be handled and monitored separately from tailings should the need arise.

1.2.7.2. Tailings Production and Characterization

The mill would produce between 768 and 1,152 dry tons of tailings per day at full production. Tailings would be dewatered in the mill through a thickener and vacuum filter. The dewatered tailings would either be trucked to the TWSF or delivered to the paste plant, also located at the mill, where they would be mixed with water and cement and pumped into the Ram mine for use as paste backfill.

Samples of the tailings solids and the solution stripped from the tailings, after being passed through a filter press, were collected as part of the metallurgical testing activities. The tailings generated by metallurgical testing were then characterized in the baseline geochemical program by a variety of static and kinetic tests to determine the potential for tailings to generate acidic and metal-bearing solutions. During milling, sulfide minerals are removed from the ore. As a result, tailings are relatively low in sulfide minerals (including pyrite) and are considered not to be potentially acid-generating. Results of static acid generation testing (acid-base-analysis or ABA) indicate that the tailings materials are neutral in pH and retain a relatively low level of sulfide-sulfur (approximately 0.05%). However, kinetic tests have indicated there is a potential for long-term release of low levels of metals from the tailings.

Tailings Management - Tailings slurry would be delivered to a dewatering station in the mill, where a thickener and vacuum filter would separate the solids from the liquids. Details on the filter cake are provided in the Conceptual Design of the Tailing/Waste Rock Facility and Water Management Ponds (Telesto, 2006). The final filter cake would be approximately 80% solids after moisture conditioning. If not used as backfill, the filter cake would be trucked from the dewatering facility to the TWSF and end dumped. The tailings would then be leveled and shaped in 2-foot maximum lifts by a small tracked dozer. Compaction of the tailings to 90% standard Proctor density would be achieved by the truck and dozer traffic on the pile.

During winter operations, the working areas would be kept small. Snow would be removed from the working area and placed in the snow removal area. Operational procedures would specify requirements to prevent incorporating snow and frozen tailings into the pile prior to compaction.

1.2.7.3. Tailings Backfill

Tailings would be placed in the TWSF or the Ram mine as backfill. It is estimated that the amount of backfill material required for the mine would consume approximately 40% to 45% of the tailings stream. As the mining method is dependent on backfill as a working platform and for ground support, maintenance of the backfill schedule would be critical to mine production. Backfilling serves the purpose of providing structural support in the mine while reducing the area required for surface tailings storage. Backfill is considered a construction material that is used to create a floor to mine on top of, a rib to mine next to, or a back to mine under. It provides important support to the surrounding rock mass, reducing the ground support requirements in active mining areas. Backfill reduces dilution of the ore by non-economic wall rock.

Tailings, for use as backfill, would have cement added to increase backfill strength and be delivered to the Ram dewatering facility as a paste. Cement would also add alkalinity that would reduce metals mobility in the backfill. The paste functions primarily as a void filler, and its strength need only be sufficient to support mine vehicles working in the stope. It would be a highly viscous mixture of mill tailings, water and cement. Nominal design parameters show a paste consisting of 65% to 70% solids and 30% to 35% water. Solids would include between 96% to 98% tailings and from 2% to 4% Portland cement.

The backfill schedule indicates that backfill would be required within the first six months of mining of the Ram deposit. By the end of the first year of mining, approximately 400 tons of backfill would be required each day. After each cut in a stope is made (approximately 15 feet on the first cut and 10 feet thereafter) and the ore has been removed, backfill material would be placed into the void. Backfill would only be used in ore zones and would not be used in access ramps. By the end of mining the ore zone stopes would be approximately 90% filled with backfill. Approximately 30% of the mine consisting of ramps, access decline and ventilation raises would be left open.

A portion of the backfill would be waste rock (slash) from ramp construction. Some of the slash would be potentially acid generating. Slash left underground will be amended with alkaline material (i.e., limestone, lime, cement) to reduce the leaching of metals from the slash. Slash would be trucked from the ramp being slashed to the mined out area being backfilled, placed on top of previously placed tailings backfill, and subsequently covered by additional tailings backfill. This backfilling sequence would provide a suitable working platform as well as partially isolate the slash. The FCC would submit a plan for USFS approval that provides details on the methods to be used to incorporate the amendment into the slash.

1.2.7.4. Tailings and Waste Rock Storage Facility

The FCC would develop a surface TWSF to store and contain tailings and waste rock material not otherwise disposed of underground, as well as, residuals, or waste products from the water treatment plant. The TWSF would be constructed east of and downslope from the mill on the Big Flat. This location was chosen because of its relatively flat topography, soil characteristics, and distance from active drainages and streams. Specific design elements of the TWSF include:

- Phased design and construction a 36 acre, 1.7 million cubic yard (MCY) initial facility, expanding to 55 acres and 2.5 MCY of tailings and waste rock upon identification of additional ore reserves of similar geochemical characteristics to the materials evaluated in the Final EIS (FEIS);

- Placing tailings and waste rock to allow separate drainage collection systems;

- Layering waste rock and tailings (commingling) in the waste rock area to reduce metals leaching from waste rock;

- Placing a composite under liner system with drainage collection;

- Staged construction and reclamation;

- Collection of runoff from waste rock and tailings with conveyance to the process pond;

- Inclusion of water treatment wastes in covered trenches within the TWSF;

- Should historically contaminated soils be encountered while facilities or road upgrades are being constructed, these soils may be disposed within the TWSF;

- Snow removal storage area with conveyance to the storage pond;

- Diversion of runoff around operating areas of the facility; and

- Use of a soil cap 4 feet thick with 2 feet of clean tailings underneath.

The TWSF would have 4 horizontal to 1 vertical (4H:1V) side slopes, constructed in three 50-foot raises with two 100-foot wide benches. A toe berm would be constructed at the base of the tailings facility to provide containment for seepage and runoff water from the tailings stack and to enhance geotechnical stability. The facility would occupy an area of about 36 acres. The TWSF would be constructed in the following sequence:

- The area would be cleared and grubbed to ensure the surface is free of vegetation, large rocks or boulders, and other debris;

- The topsoil and subsoil would be removed and hauled to the stockpile area;

- A drainage system would be constructed within the subgrade to intercept and remove groundwater from the TWSF foundation soils. The system would consist of a series of French drains constructed upstream and within the footprint of the TWSF and would discharge to engineered wetlands located east of the TWSF;

- The foundation area subgrade would be graded and compacted in-place to create a suitable foundation for the liner;

- The toe berm would be constructed using materials excavated from the water management ponds and other borrow materials as necessary;

- The clay member of the composite liner would be placed. Final design would determine whether a 1-foot clay liner or a geosynthetic clay liner would be used;

- A synthetic liner (such as 60-mil poly-vinyl chloride [PVC]) would be placed over the subgrade in a scheduled construction sequence. Subsequent liner expansions would be installed as needed;

- A drainage collection system would be constructed over the synthetic liner to collect water infiltrating through the tailings and waste rock and to convey flow to the process pond. This system would be constructed within a protective sand layer, which would also act to protect the liner from damage during tailings and waste rock placement;

- Drainage channels would be constructed along the outside slopes of the TWSF, along the toe berm, and on the intermediate benches of the TWSF (as construction progresses) to collect surface drainage from the facility and convey it to the process pond. All drainage channels would be designed to handle runoff from a 25-year, 24-hour storm event. Perimeter drainage channels would be constructed to intercept storm water run-on to the TWSF, direct water around the TWSF, and convey water back to natural flow paths via BMPs and sheet flow;

- Snow would be removed from the active disposal areas prior to placing waste rock or tailings in the TWSF. The snow would be stockpiled in a designated area within the facility;

- Reclamation of the TWSF would occur incrementally after each phase of tailings and waste rock placement is completed. The facility would be regraded to a continuous slope of 4H:1V or less to reduce potential for erosion;

- A cover system consisting of a 60-mil high-density polyethylene (HDPE) geomembrane, a geonet drainage layer over the HDPE, and a 4-foot thick surface soil layer would be constructed over the TWSF to limit infiltration into the tailings and waste rock;

- The cover soil would be revegetated to help reduce infiltration and erosion; and,

- After reclamation, tailings drainage would continue to be conveyed to the process pond as long as closure water treatment is needed. After cessation of water treatment, tailings drainage would be conveyed to an infiltration field located east of the pond.

Waste rock would be delivered to the TWSF by truck or tram from the Ram and by truck from the Sunshine. Waste rock would be placed in approximate 5-foot lifts on the prepared surface by end dumping from mine trucks and would be spread and leveled with a dozer.

1.2.7.5. Tailings Disposal Quality Assurance/Quality Control

To meet material placement specifications for the TWSF, a quality assurance/quality control (QA/QC) plan would be utilized to determine steps for tailings and waste rock placement. Tailings material received from the plant/mill facility would be dewatered to +2% of the optimum moisture content prior to being placed in the TWSF. Once the tailings material has been dewatered to +2% of the optimum moisture content, the material would be end dumped and compacted in two-foot lifts to 90% of the standard Proctor maximum density. During cold weather, dried tailings would be spread and compacted before they freeze. Any non-compacted material that does freeze would be stockpiled. Dried tailings material that cannot be placed because of snowfall events would also be stockpiled. The tailings stockpile would be located on the lined portion of the TWSF. Material that has been stockpiled through the winter that does not meet the stacking requirement of +2% of optimum moisture content would be mixed with newly processed material to ensure that the mixed material meets the stacking requirement. Field verification of the moisture content and density would be conducted once per week and documented.

1.2.8. Water Utilization and Treatment

The ICP's primary demand for water is ore processing. The milling process requires approximately 960 gallons of water per ton of ore processed, which equates to about 768,000 gallons per day (gpd) for the nominal ore production of 800 tpd. Except for water lost to the concentrate and the tailings, the effluent from the milling operation would report to the process pond. This water would mix with mine water and other waters reporting to the process pond and be recycled back to the mill.

The primary source of water for the operation would come from the developed Ram deposit. Mine flow would be a function of the length and depth of mine workings, and flow would increase with the development of the mine. At full development, the POO estimates the Ram would produce approximately 43 gpm and the Sunshine would produce an average of 8 gpm. Additional water for the operation would come from the collection of runoff from the TWSF and stormwater from the ore stockpile area. During startup, process water would be provided by pumping groundwater from water supply wells as needed.

The water supply for the mill is expected to vary throughout the life of the mining operation. Although sufficient water for processing is predicted to be available from the Ram workings and precipitation on the TWSF and ore stockpile, there is the possibility that additional water would be required to support operations. Two water supply wells would be constructed in the Big Flat area to meet additional demand.

Water for human consumption would be supplied as bottled water or from on-site wells. Human consumption water would be less than 100 gpd. Water for showers, toilets, and other human uses would either come from on-site wells or would be site water that has been processed through an on-site treatment plant to produce water of adequate quality for this use. A septic tank and drain field would be permitted and installed north of the mill site.

1.2.8.1. Water Management

Drainage is designed so that mine water, process water, and runoff from the TWSF and ore stockpile go through the treatment plant at a single point of discharge to Big Deer Creek. Water arising from ore processing, groundwater pumped from the mine and drainage from the ore stockpile and the TWSF is stored in the water management pond. Water from the pond is recycled through the mill or is directed to the water treatment plant. The volume of mill processing water/TWSF runoff discharge treated and discharged to Big Deer Creek is limited in the NPDES permit to the net precipitation on the TWSF and water management pond. The single discharge (Outfall 001) is located in Big Deer Creek approximately 100 feet downstream of water quality monitoring station WQ-24.

To estimate impacts to water resources a water and chemical mass balance for the ICP has been developed using a dynamic system model (DSM). The DSM considers the relationships between the Project components and the surrounding water environment, and predicts the impact on water resources throughout the life of the mine and during the post-closure period. The DSM includes specific water balance calculations for each year of the Project life. The ICP water management plan is based on operating a water treatment plant and releasing water in accordance with an NPDES permit in conjunction with temporary storage in a small water equalization pond adjacent to the water treatment plant and a larger water management pond to temporarily store process solutions. The water treatment plant would have the ability to treat up to 150 gpm of water for discharge through the NPDES outfall. Except during periods of very high inflow, the water treatment plant would treat incoming water on an as-received basis, with very little water stored in the water management pond. During periods of high inflow, water would accumulate in the water management pond for treatment during lower inflow periods.

Process Water Characteristics - The primary contaminants of concern (COCs) for the ICP are nitrate, sulfate, arsenic, copper, cobalt, nickel, and zinc. Primary COCs are constituents expected to occur at higher concentrations in the water management pond water compared to natural waters. They may also have significant environmental effects if discharged into surface water or groundwater.

Water Management Pond - The water management pond collects drainage from the TWSF and stores mining and milling process solutions if needed. The pond would be surrounded with an

8-foot high chain link fence for wildlife protection, double lined with HDPE liners, and have a leak detection and recovery system between the primary liner and secondary liner. Protection of the pond liners from potential ice damage (e.g., rub sheet of liner in pond corners) would be provided.

The pond would be sized to contain process waters and would also have the capacity to contain the runoff from a 500-year return period event plus 2 feet of freeboard. The design capacity for the pond is 10 million gallons. After adding the freeboard, the ultimate pond capacity is 12 million gallons. The 10 million gallon active capacity is evenly divided between two cells within the pond. The pond includes a spillway designed to pass a 100-year event to minimize the risk of failure in the event of overtopping.

Prior to commencing construction of the water management pond, FCC would provide a final engineering design to the USFS. The final design would include stability analysis based on actual material and site parameters. It would also include specifications for materials, construction, and QA/QC. The QA/QC plan would specify that construction monitoring would proceed under the supervision of a qualified professional engineer.

In the event projections show that there would be insufficient water for winter operations, adequate water would be retained in the pond to prevent a water shortage. A minimum amount of water would be kept in the pond at all times to hold the liner in place. Water from the mine and effluent from the milling process would flow by gravity to the pond as needed during periods of mill shutdown, or other operational reasons when storage is required. The pipeline from the pond to the mill would be double contained and complete with leak detection at all low points and at pipe-to-pipe connections.

Equalization Pond - The equalization pond would receive water and solids from mine and mill process streams for temporary storage prior to treatment. This pond would serve to equalize inflows from the source waters to provide a more consistent flow quantity to the water treatment equipment. This pond is sized at 90,000 gallons (8 hours storage at 150 gpm plus 25% contingency).

1.2.8.2. Water Treatment

Water pumped from the mine workings at the ICP and runoff/drainage from the TWSF and ore stockpile is predicted to contain elevated concentrations of nitrate, sulfate, and metals (aluminum, cobalt, copper, iron, manganese, and zinc). A water treatment plant designed to process and discharge up to 150 gpm would be installed in the mill to treat excess mine water and runoff/drainage from the TWSF and ore stockpile.

Water Treatment System Design - The water treatment process for the proposed action would utilize ion exchange as a means to remove any residual metals in excess of the NPDES effluent limits. This would eliminate the need for disposal of a stabilized reverse osmosis waste (a potentially large waste stream), but would result in higher levels of sulfate in the discharge water. The treatment process would include biological treatment for removal of nitrate, if needed to meet the nitrate effluent limit. If needed to meet the ammonia effluent limit, ammonia would be removed by an ion exchange process using zeolites, a naturally occurring alumino-silicate mineral. The treatment process would result in treated water (effluent) capable of meeting effluent limits imposed by an NPDES permit for discharge to Big Deer Creek.

A Draft NPDES permit has been issued and is considered as part of this document. Treated water from the water treatment plant would be routed through a pipeline to a surface discharge located on Big Deer Creek below monitoring site WQ-24 (Appendix A, Figure 2). The pipeline would be routed using existing roads where possible. Where no roads exist, the pipeline would follow an alignment that minimizes pipeline length and physical disturbance to soils, vegetation and cultural resources. The pipeline would be buried, and be made of materials suited for this application such as steel, PVC, or HDPE. Where the pipeline crosses existing waterways, a culvert designed to pass the 100-year, 24-hour event would be placed in the stream channel, the pipe placed on top of the culvert, and fill placed over the pipe to prevent the water from freezing. The pipeline would affect jurisdictional wetlands at the crossings of two Bucktail Creek tributaries and at Big Deer Creek.

Rather than an open pipe discharge to Big Deer Creek, an instream diffuser would be used for the discharge. During the initial mine construction phase, the FCC would be required to conduct a field investigation and engineering evaluation to determine the final design of the diffuser. The diffuser would consist of a perforated pipe or pipe with engineered orifices designed to cause rapid mixing of the mine discharge with Big Deer Creek. The pipe would be place on the surface of the streambed or would be buried within the streambed.

1.2.8.3. Stormwater Management Plan

The FCC has proposed a Stormwater Management Plan (SWMP) with the goals of:
(1) Preventing stormwater run-on to proposed facilities; (2) minimizing erosion; and
(3) reducing sediment transport to downstream receiving waters. The FCC would be required to obtain a Stormwater Permit or Permits from EPA prior to beginning construction. The FCC intends to apply for coverage under the Construction General Discharge Permit and Multi-Sector General Discharge Permit. EPA will make decisions on stormwater permits after FCC submits NOIs for these permits. Facilities to be covered under the Permit for the proposed project are as follows: (1) Topsoil and borrow material stockpiles; (2) haul and access roads; (3) parking lots; (4) office buildings; and, (5) ancillary disturbance areas not associated with milling process.

A construction stormwater permit would be required to address construction activities proposed for the site. Snow removal and storage is another component of the SWMP and includes a plan for snow removal for each major facility.

Design Criteria – Separation of clean stormwater runoff would be accomplished through the use of diversion channels to prevent upgradient water from coming into contact with proposed facilities or mined material stockpiles. The diversion channels would be V-shaped channels with

1 foot of freeboard and 1:1 side slopes. These upgradient diversion channels would route clean runoff around proposed facilities and disturbed areas and would distribute flows back to the watershed via sheet flow.

The FCC has proposed that the design storm for the proposed clean water channels, which would exist beyond the life-of-mine, would be either the 25-year, 24-hour storm or the 100-year, 24-hour storm depending on the association of channels with mine roads, process facilities or process materials. The BMP facilities designed to cover mine roads and operations are proposed to handle the 2-year, 24-hour storm.

The sections of road adjacent to the TWSF would be outsloped to minimize concentration of flows. The stormwater runoff generated from these sections of road would be dispersed by the use of slash (stacked timbers and brush). The remainder of roads would be insloped and runoff would report to stormwater diversion channels. Channels would have erosion protection in the form of check dams and riprap at outfalls. Outfalls would be protected with brush barriers, biofiltration swales, or rock structures, to dissipate runoff energy and prevent headcutting.

Stormwater channels installed around proposed facilities would intercept runoff before it interacts with a facility. The intercepted runoff would report to BMP structures, which would also be used to entrap sediment carried by flow in the channels. Design elements of typical BMPs are included in Appendix D to the *Storm Water Management Plan for the Idaho Cobalt Project* (Telesto, 2006).

Specific design elements of the SWMP are:

- Diversion of upslope clean runoff around the proposed TWSF, process ponds, and proposed borrow area;
- Maintenance of existing sheet and overland flow characteristics over undisturbed areas;
- Conveyance of collected runoff to frequently spaced, erosion-protected outfalls;
- Use of available forest slash (partially burned timber and brush), rock sediment basins, silt fencing, and biofiltration swales in BMPs; and,
- Revegetation of mining-disturbed areas, concurrently with operations as practical, to increase erosion protection and reduce sediment loading.

1.2.8.4. Spill Control

The FCC's spill control plan to address management of hazardous materials during shipping and storage would be revised to address any changes in the POO, and would be reviewed and approved by the agencies prior to initiation of construction activities. The plan would include notification of the ICP facilities prior to transport of fuels or chemicals, use of closed trucks, travel only during daylight hours, use of pilot vehicles, and continuous radio contact with pilot vehicle and facility during transport. An approved SPCC plan would be required within 6 months of starting operations.

1.2.8.5. Water Rights

The FCC has applied for water rights on the groundwater from the mines and groundwater from two wells, for mining and milling purposes. Water from the wells would be used initially for drilling and other start-up water needs until the mine pumping and precipitation capture from the TWSF is adequate to supply operating water needs. The wells would also supply water for human use.

1.2.9. Water Resource Monitoring

The SCNF and the ICP would review the POO including the Reclamation Plan and monitoring plans on an annual basis. Water monitoring and stormwater monitoring following all surface disturbing activities, as well as spring runoff, large storm events, and the fall reclamation effort would occur in accordance with approved plans. The results of this monitoring would be documented by FCC and submitted to the USFS. The FCC has prepared a water monitoring plan for the project, the *Operational Water Monitoring Plan for the Idaho Cobalt Project* (Telesto 2007).

The proposed water monitoring plan was prepared to address operational and closure assessment of water resources. The monitoring plan includes a performance-based approach to compliance assessment. For example, groundwater quality data from select operational monitoring wells to be located downgradient of the mines would be evaluated for compliance. If performance criteria exceed pre-established targets (e.g., if the calculated groundwater load were to result in exceedance of a surface water compliance target), a response action would be required by the mine to reduce the groundwater load to acceptable levels.

Components of the water monitoring plan support source identification and/or source allocation to differentiate effects of ICP from those of Blackbird Mine Site Group (BMSG), the monitoring plan includes:

- Groundwater monitoring wells located to the north of the Ram underground workings to monitor for possible groundwater flow along a fault that is present in the area;

- Groundwater monitoring wells located to the south of the Ram underground workings to monitor for possible connections to Blackbird workings;

- Groundwater monitoring wells in lower Bucktail Creek alluvium to evaluate potential effectiveness of capture system;

- Groundwater monitoring wells and piezometers in the area of the Ram bedrock groundwater capture system to determine potential capture effectiveness and monitor dewatering effects;

- Additional groundwater monitoring wells west of Ram Mine to provide information on Ram mine chemical mass loads;

- Groundwater monitoring wells located to the east of the Sunshine underground workings to monitor conditions between Sunshine and Blackbird workings;

- Groundwater monitoring wells located to the west/northwest of the Sunshine underground workings to monitor conditions between Sunshine and West Fork Bucktail Creek;

- Groundwater monitoring wells located in the area of Sunshine groundwater capture system to evaluate potential effectiveness of capture system;

- Monitoring of stormwater outfalls to determine effects of stormwater on streams and to judge effectiveness and adequacy of stormwater controls;

- Installation, testing and monitoring of monitoring wells/recovery wells down gradient of the Ram workings in the first years of mining to determine the effectiveness of FCC's proposed recovery system and to monitor groundwater release from the mine;

- Monitoring of mine water quality (drainage from tailings backfill, groundwater inflows, mine water sumps); and,

- Coordination of ICP and BMSG monitoring activities to ensure consistency and comparability of data.

To collect information needed for final design of the reclamation and closure plan, the modified monitoring plan would also include:

- Monitoring of mine inflow quantity and quality. This information would be used in assessing mine recharge rates after shutdown and in design and operation of the post-closure mine dewatering system; and

- Operational monitoring of mine water quality (drainage from tailings backfill and mine water sumps).

These data would be used in assessing and predicting the effectiveness of tailings and slash waste rock amendment in reducing the leaching of metals to groundwater and would characterize underground mine water sources. The monitoring plan also includes:

- Coordination of ICP and BMSG monitoring activities in the timing, frequency, sampling methods, analytical methods, and QA/QC requirements of monitoring to ensure consistency and comparability of data;

- Consistency with agency guidance and BMSG cleanup requirements in the evaluation of monitoring data to determine compliance with surface water and groundwater water quality standards, performance standards and limits. For surface water, this would include compliance testing compatible with the Blackbird Unilateral Order on Consent statement of work requirements for Big Deer Creek/Panther Creek watershed (compliance based on 96-hour testing). For groundwater, this would include compliance testing based on IDEQ guidance (e.g. test for difference in mean concentrations with standards/limits);

- Consistency with IDEQ requirements on development of a statistically significant baseline data set for groundwater;

- Adaptive management to enhance monitoring as needed to support compliance monitoring;

- Reporting of all monitoring data to the USFS, IDEQ and EPA; and,

- Additional detail regarding the justification for and requirements of a revised water monitoring plan can be found in the Water Resources Technical Report (Hydrometrics, 2006) and Chapter 4 of the EIS.

In order to complete the proposed monitoring, FCC would also install a cable car crossing of Panther Creek to facilitate baseline and operational monitoring of lower Big Flat Creek. The cable crossing would be installed south of the confluence of Big Flat Creek and Panther Creek and would be accessible from FR 055. However, the exact location of this structure would be determined with on-site USFS review. Design and construction of the cable car crossing would be similar to that of the BMSG crossing located upstream on Panther Creek and would consist of two, thirteen-foot high "A" frames anchored with concrete deadmen (2 to 3 yards of concrete), with a 90- to 120-foot cable span (Appendix A, Figure 3). The foot of each A-frame would be bolted into a pre-formed Ecology block embedded into the ground. The support structures would be installed at approximately the same elevation above Panther Creek as the existing road and well above the channel profile. A platform would be built on each "A" frame to allow passenger access. Equipment (e.g., backhoe or excavator) would be required to cross the Creek in order to install the two or three yards of concrete for the deadman. The crossing would be installed in early spring prior to high flow.

1.2.10. Water Management at Closure

At the completion of mining, the decision to cease pumping from the mine would be made based on results of water quality monitoring and predictions of impacts to groundwater and surface water. Post closure groundwater capture wells would be installed and tested during the initial construction phase to confirm that the system would capture a sufficient amount of groundwater to protect downstream water quality. If testing indicated that bedrock wells could not capture enough of the groundwater metals load, an additional groundwater/surface water capture system consisting of an interception trench or series of wells across Bucktail Creek alluvium would be installed downgradient of the Ram mine. The Bucktail capture system would collect alluvial groundwater, and surface water if necessary. The Bucktail capture system would allow collection of additional groundwater and COCs from the Ram and Sunshine mines, and would allow capture and treatment of additional metal load to ensure that the ICP does not contribute to a net increase in metal loading to the South Fork Big Deer, Big Deer, and Panther Creeks. Additional permitting related to disturbance in the streambed (404 and stream alteration permits) would likely be required if the backup lower Bucktail system were required. Additionally, because there would be a larger area of groundwater affected by mine contaminants (between the source and the capture system) the State of Idaho, who regulates groundwater quality, would have to sanction this capture system concept.

The Bucktail capture system would be located in lower Bucktail drainage between monitoring sites WQ-19 and WQ-21, upstream of the unnamed tributary where WQ-11 is located, and upstream of the proposed BT-5 pipeline system (see description of BT-5 diversion under Remedial Actions in Chapter 3, page 3-81 DEIS). The backup capture system would consist of a series of pumpback wells and/or capture trenches within the Bucktail Creek alluvium and provisions for surface water capture at the same location. The surface water diversion structure would use the lower sediment dam with some modifications to the existing outlet structure or a new outlet structure. If the lower sediment dam is removed, an alternative diversion structure along lower Bucktail Creek would be required. The FCC would be required to provide a final design of the groundwater/surface water capture system prior to mining.

The captured water from the bedrock or lower Bucktail capture systems would be pumped to the water treatment plant. The treated water would be piped back to the discharge location in Big Deer Creek near monitoring station WQ-24.

1.2.10.1. Water Management Pond

Prior to reclamation of the water management pond, any remaining water would be treated and discharged. Any sediment or residual material in the pond would be analyzed for pH and metals. If the testing shows leachable metals exceeding the regulatory limits, the sludge would be either stabilized in place and retested, or removed from the site and disposed of in a permitted disposal facility consistent with Federal and state regulations. Following testing, the liner would be folded into the pond, dikes would be pushed into the pond, and the area would be regraded to approximate the pre-construction topography. Following the regrading operation, the area would be covered with growth medium and revegetated. Seedbed preparation and seed application would be performed in one operation with a tractor pulling a chisel tooth harrow and seed drill. The water management pond would be reclaimed after pumping in the Ram and Sunshine Mine pumpback well fields ceases.

1.2.11. Wetland Construction

Jurisdictional wetlands would be impacted by the cable crossing over Panther Creek and the water treatment discharge pipeline which crosses over unnamed drainages along the road to the NPDES discharge location in Big Deer Creek. The discharge pipeline would also affect wetlands near the discharge point in Big Deer Creek.

Cable Crossing of Panther Creek - Construction of footings for the cable crossing would occur at approximately the same elevation above Panther Creek as the existing road and well above the channel profile. Thus the wetland impacts would consist of the construction access and the footings of the cable towers.

Treated Water Discharge Pipeline - The discharge pipeline would follow the access road from the mine area to near the confluence of Little Deer and Big Deer Creeks; along this route the road crosses two drainages that have jurisdictional wetlands. The lower drainage to be crossed is incised about 4- to 6-feet deep and contains a flowing stream (Appendix A, Figure 2). A culvert crossing would be installed for the incised gully.

The pipeline would be constructed across the surface of the newly constructed road bed and would be constructed to prevent freezing. If there is sufficient soil thickness over the culvert pipe, the pipeline would be excavated into the road bed similar to the construction along the bulk of the pipeline route. In the event there is not sufficient soil thickness above the culvert to bury the pipe, the pipe would be brought out of the ground and would lie on the roadway for a short distance. The pipe would once again be buried when it is past the culvert. Wetland topsoil would be salvaged prior to culvert installation and would be in the wetlands mitigation project. Construction stormwater BMP controls such as silt fences, sediment traps, and any other appropriate measures would be used as needed during road construction.

The upper crossing on Ram Gulch is a flat wetlands area and the pipeline would be buried or otherwise constructed to prevent freezing in this area. Methods to eliminate trenching in the wetland could be employed but would require protection from freezing. The discharge pipeline would also cross wetlands for some distance before reaching the terminus point in Big Deer Creek. The discharge would be via an instream diffuser. During the initial mine construction phase, the FCC would be required to conduct a field investigation and engineering evaluation to determine the final design of the diffuser.

In Big Deer Creek, a trench will be excavated to a depth of 1- to 4-feet, extending across no more than 50% of the channel width (personal communication, Ray Henderson, SCNF, April 8, 2008). A backhoe/excavator will be used to conduct the trenching. A 4- to 6-inch pipe will be laid in the trench bottom. Effluent diffuser orifices consisting of smaller diameter pipe will extend from the pipeline to the streambed surface and will be located at regular intervals along the diffuser pipeline. A small volume of streambank (<1 cubic yard) below the ordinary high water mark will need to be excavated to place the pipe. The trench will be backfilled with native material excavated from the site and revegetated. No material will be stored below the ordinary high water mark.

1.2.12. NPDES Draft Effluent Limits

On May 25, 2006, the FCC submitted an application to EPA Region 10 to discharge wastewater to Big Deer Creek under the NPDES permit program. EPA reviewed the application and after several supplementary submittals deemed the application complete on July 14, 2006. The draft NPDES permit contains discharge limitations for 16 potential pollutants, including eight metallic elements, arsenic (a metalloid), ammonia, sulfate, sulfide, total suspended solids (TSS), pH, dissolved oxygen (DO), and temperature (Table 2). The rationale for calculating the effluent limits are identified in the Draft NPDES permit. Outfall monitoring is also specified for iron, aluminum, hardness, chloride, conductivity, total dissolved solids (TDS), Whole Effluent Toxicity (WET), and Expanded Effluent Testing. The list of parameters for ambient stream monitoring is the same with the exception of the addition of silver as a monitoring parameter.

The outfall is proposed to discharge to Big Deer Creek approximately 1,500 feet below the confluence with South Fork Big Deer Creek by way of a diffuser. The outfall is located approximately 3 miles upstream from the confluence with Panther Creek. Access for anadromous species in Big Deer Creek is blocked by a series of impassable falls and cascades approximately 0.7 stream miles upstream from Panther Creek.

Table 2. Outfall Effluent Limits and Monitoring Requirements (draft permit, EPA 2007).[1]

Parameter	Units	Effluent Limits and Monitoring Requirements			
		Maximum Daily Limit	Average Monthly Limit	Monitoring Frequency	Sample Type
Arsenic [1]	μg/L	100	50	Weekly	Grab
Cadmium [1]	μg/L	0.52	0.26	Weekly	Grab
Cobalt [1]	μg/L	141	70.4	Weekly	Grab
Copper [1]	μg/L	4.80	2.40	Weekly	Grab
Lead [1]	μg/L	0.90	0.45	Weekly	Grab
Mercury [1]	μg/L	0.02	0.01	Weekly	Grab
Nickel	μg/L	26.52	13.22	Weekly	Grab
Thallium	μg/L	0.95	0.47	Weekly	Grab
Zinc	μg/L	37.02	18.45	Weekly	Grab
Ammonia (total as N) **(Superceded)**	mg/L	5.62	2.80	2/Month	Grab
Ammonia (total as N) (REVISED 10/1/07)	mg/L	**4.1**	**1.6**	2/Month	Grab
Nitrate + Nitrite	mg/L	100	--	2/Month	Grab
Sulfate	mg/L	1,000	--	2/Month	Grab
Sulfide	μg/L	2	--	2/Month	Grab
TSS	mg/L	30	20	Weekly	Grab
pH	s.u.	Between 6.5 and 9.0 at all times		Weekly	Grab
Dissolved Oxygen	mg/L	Must exceed 6.0 at all times		2/Month	Grab
Temperature	C°	19	--	2/Month	Grab
Iron	μg/L	--	--	Monthly	Grab
Aluminum	μg/L	--	--	Monthly	Grab
Hardness	mg/L	--	--	Monthly	Grab
Chloride	mg/L	--	--	Monthly	Grab
Conductivity	mS/m	--	--	Monthly	Grab
TDS	mg/L	--	--	Monthly	Grab
Whole Effluent Toxicity (WET)	TUC	--	--	1x/6 months	Grab
Expanded Effluent Testing [2]	--	--	--	3x/5 years	Grab

Notes to Table:
1. Expanded effluent testing includes the 126 chemicals listed in 40 CFR § 131.36. This testing shall occur in years 2, 3 and 4 of the permit cycle, and should occur coincident with the September WET testing and other routine monitoring.
2. Metals limits expressed as total recoverable except for mercury which is expressed as total.

1.2.13. Reclamation

The proposed reclamation plan involves approximately 115 acres of surface disturbance. This includes existing roads that would not be reclaimed and would become a part of the post-mining road system in the area. Additionally, some newly constructed Project roads would be incorporated into the post-mining road system. During construction, ICP proposes to reclaim

[1] Effluent limits for Ammonia were lowered by EPA (Personal Communication, Lisa Olson, EPA Permit Writer, Seattle Washington, October 2007) in comparison to the limits in the draft permit (dated approximately February 2007).

about 4.54 miles of substandard and non-essential existing roads in the vicinity of the ICP. The Agencies propose reclaiming an additional 2.95 miles of non-essential existing roads during construction. Once mining has ceased or when no longer required for post-closure water management or other closure activities, all above ground facilities would be demolished, removed from the site, and their former location reclaimed.

The proposed reclamation plan is designed to meet the following goals:

- Conduct reclamation and revegetation concurrently with the mining program, as much as possible. Concurrent reclamation would be performed on areas no longer required for the mining operation;

- Keep all clearing and disturbance to the minimum consistent with Project needs;

- Place waste rock, tailings, roads, structures, diversions, and water management ponds so that they minimize subsequent shaping and recontouring and do not pose a hazard to human health and the environment;

- Reestablish stable and diverse surface topography and hydraulic features that are compatible with the surrounding landscape;

- Establish soil conditions that promote regeneration of stable, diverse, and self-sustaining native plant communities through removal, storage, and redistribution of suitable soil materials;

- Revegetation of all areas disturbed by the operation to stable and diverse native vegetation communities that provide wildlife habitat and minimize erosion;

- Work with the USFS to identify opportunities to improve the post-mining land use of the site through reclamation of existing, unnecessary roads;

- Provide methods, procedures, and practices for seasonal activities, temporary shutdowns, and final reclamation;

- Maintain water quality such that water quality standards are met at the BMSG compliance points in Big Deer Creek (WQ-24), Panther Creek, (WQ-25) and South Fork Big Deer Creek (WQ-22); and,

- Meet NPDES permit stipulations regarding no net increase in copper load in Big Deer Creek.

1.2.13.1. Facility Reclamation

Once the ore reserve is exhausted and mining ceases, surface and underground facilities not needed for reclamation or closure activities would be removed.

Mines - Reclamation of the flat areas adjacent to the Ram and Sunshine portals would include removal of buildings, cables, piping, and concrete pads, regrading, ripping to alleviate compaction, applying available growth medium (and amendments if determined necessary), and revegetating. Adit portals would be sealed to prevent human or animal access. This would include backfilling the entries of adits with clean waste rock and grading the area to fill the portal depression. Grading would include bringing as much of the portal bench fill as possible onto the cut area and reestablishing a continuous hillside slope, to the extent practicable. Abrupt surface features would be smoothed to create uniform grades and to produce a near naturally appearing surface.

Mill - The combined mill, administration building, warehouse, shop, and ancillary facilities (such as piping and tanks) would be demolished and disposed of. Buildings and equipment would be dismantled and removed from the property. Equipment and facilities with salvage value would be sold. All remaining scrap and demolition debris would be disposed of off-site at an approved landfill. Foundations and walls would be demolished to 1 foot below grade and covered with fill to eliminate any safety hazards for wildlife or humans. Sumps or other voids would be backfilled with sufficient soil so that depressions would not occur after settling. Slab foundations would be broken up for adequate drainage, placed in the deeper portions of the regraded fill, and buried under no less than 2 feet of cover.

Regrading of this area would include moving much of the fill into the original cut area to establish a natural looking topography. The edges of the area would be shaped to blend with the surrounding contours. The area would be ripped to relieve compaction prior to topsoil placement. Topsoil would be placed over the area to a minimum depth of 12 inches.

The water treatment plant would be attached to the mill building such that the mill building and equipment can be dismantled and removed without disturbing the treatment operation. Water treatment operations would continue as long as pump-back groundwater wells are operated. At the cessation of pump-back well operation, the water treatment plant would be decommissioned by dismantling and removing all equipment, building, and support structures. Decommissioning and reclamation activities described for the mill process area would be applied to the water treatment plant and surrounding area, including foundation demolition, backfilling, regrading, and reseeding. The equalization pond would be reclaimed in accordance with procedures described for the water management pond.

Tram - Reclamation of the overhead tram would include removal of the structures, pipelines, cables, and concrete pads that comprise the facility. Following removal of the equipment, the disturbed areas within the tramline corridor would be graded and revegetated.

TWSF - The TWSF would be constructed in three phases with construction beginning on the eastern side (the lowest end) of the facility. As stacking on Phase I of the facility nears its ultimate capacity, Phase II would be constructed. Once material placement on Phase I is complete, that phase would be reclaimed. Likewise, as soon as material placement on Phase II is complete, it would be reclaimed. Incremental reclamation of the TWSF would reduce the precipitation catchment area, reducing the amount of excess water captured each year. Reclamation of the TWSF would include grading, cover installation, topsoil placement, and revegetation.

Grading of the TWSF would require minor surface shaping to smooth corners to give a more natural appearance to the pile. The 100-foot setback benches would remain in place and would be backsloped into the pile and sloped to drain laterally. Slopes of the reclaimed pile would be at 4H:1V or less. Setback benches and the pile top would be graded at 3% to drain. If post closure water treatment were required, treatment waste would be shipped off-site or an alternative disposal plan would be developed. The TWSF cover installation would include placing a 60-mil HDPE cover, installing a geonet drainage layer over the HDPE cover, and placing 4 feet of soil cover material on top of the drainage layer. Soil would be placed loose and scarified along the contour to provide micro traps for moisture and seeds. Seeding would be by mechanical means where practical and safe, and by hand where necessary. Seeding would include grasses, forbs, and shrubs. Trees would not be planted to avoid tree root penetration into the cover.

During operation, drainage from the TWSF underliner would report to the water management ponds. Following active mining, for the duration of water pumpback and treatment at the Ram and Sunshine mines, TWSF drainage would continue to report to the water management ponds. After the ponds have been reclaimed, drainage from above the TWSF underliner would report to an infiltration field located east of the water management ponds. The DSM estimates long-term drainage at 0.4 gpm. The drainfield would be designed to accept up to 1 gpm.

Pipelines - Surface pipelines would be removed. To minimize re-disturbing revegetated areas, buried pipelines would be capped and abandoned in place.

1.2.13.2. Soil Salvage

During the construction of the Project, available soil would be stockpiled and stabilized in a discrete location adjacent to the area disturbed by mining-related activities. Soil removed during road and portal construction would be stockpiled downslope of those features. Soil removed during mill and TWSF construction would be stockpiled near the TWSF area (Appendix A, Figure 2). Total topsoil salvage is estimated to be 284,000 cubic yards. Approximately 7 acres would be required for the topsoil stockpile area. Precipitation run-on would be diverted around the stockpile area by perimeter ditches. As topsoil materials are placed in this area, the topsoil would be seeded with a mixture of non-native species (smooth brome, mountain brome, orchard grass, and timothy) at an application rate of 4 pounds per acre each to temporarily stabilize the stockpile.

Soil types and depths vary across the site. In the Ram and the Sunshine areas, the salvageable soil depth is estimated to range from 0- to 8-inches. On the Big Flat, soil depths are estimated to provide between 12- and 14-inches of quality material suitable for reclamation purposes.

Following recontouring of the site, salvaged soil would be taken from the stockpiles and placed over the recontoured surfaces. Prior to placement on disturbed sites, the selected growth medium would be tested for comparison with pre-selected reference sites. Testing would include pH, electrical conductivity, lime, organic matter content, texture, saturation percent, nitrate–nitrogen, phosphorus, potassium, zinc, iron, manganese, and copper. The FCC would review testing results and propose a soil amendment program (if needed) to the USFS. Decomposition of organic matter while growth medium is in stockpile has been documented. Nitrogen fertilizer and organic materials would be added as necessary to ensure adequate plant development.

Seedbed conditioning would consist of ripping or disking the recontoured surface with notched or straight edged discs set together in rows or "gangs" combined with harrows. This step would break up seedbed clods, and turn under and cut brush, limbs, and weeds. Additionally, it would break up surface compaction and anchor any straw or hay mulch that has been applied.

Following grading (or contouring), growth medium placement, and seedbed conditioning, the areas would be revegetated with species appropriate for the specific site and climate. Species used in revegetation would stabilize the area and allow the natural incursion of indigenous species. Lodgepole pine (*Pinus contorta*) is expected to regenerate naturally over the Big Flat and other areas as a result of the 2000 Clear Creek fire. The FCC would collect serotinous lodgepole pine cones from nearby sources and place those cones over the replaced soils for subsequent opening and seed germination.

Grass seed application would be performed with a seed drill, hydroseeding, or hand broadcasting depending upon the terrain. Broadcast seeding would be done in the fall prior to freeze up to produce the optimum germination. Areas seeded to grass would receive a light application of weed-free straw mulch at the time of seeding followed by the application of fertilizer in the spring prior to summer rains.

1.2.13.3. Post-Closure Reclamation Monitoring

All reclaimed sites would be monitored twice a year for a period of 3 years to evaluate the success of the reclamation work. Any areas not meeting the vegetation success criteria would be analyzed to determine the problems and the areas would be revegetated with a modified plan.

In addition, there would be monitoring of ground and surface water for a minimum of 5 years following cessation of pump back Results of this monitoring would be used to evaluate the success of the measures taken to protect the water resources. Any changes in water quality would be evaluated to determine whether the changes are related to the reclaimed mining facilities. Monitoring must demonstrate that water quality meets established standards and is stable over time and a range of hydrologic events prior to bond release.

1.2.14. Financial Assurance

As part of the approval of a plan of operations for the ICP, the USFS would require FCC to post a financial assurance package. This package would ensure that adequate funding is provided to allow the USFS to complete reclamation, post-closure operation, maintenance activities, and necessary monitoring for as long as required to return the site to a stable and acceptable condition. The amount of financial assurance would be determined by the USFS and would be adequate to allow the USFS to complete all necessary reclamation of the ICP at any stage if FCC were to abandon the project. The financial assurance would also cover potential costs to capture and treat water following closure to meet effluent or instream limits. The financial assurance may be in the form of a bond or other financial instrument and would be payable to the USFS in the event that FCC does not perform reclamation actions as required by the ICP POO. The amount of the financial assurance would be calculated in accordance with USFS guidelines.

1.2.15. Design Modifications and Mitigation Measures

The proposed action contains a number of design and operational modifications, mitigation measures and monitoring plans intended to reduce impacts to the environment. An Interagency Oversight Task Force will be formed to provide oversight for the ICP. The Task Force would consist of all permitting agencies including IDEQ, IDWR, NMFS, FWS, USFS, and EPA. A subset of the proposed mitigation measures and monitoring requirements most pertinent to avoiding or minimizing potential adverse effects to ESA-listed species or critical habitat includes:

1. The operator shall obtain agreements for use of existing infrastructure (6850 adit and treatment facility), developed by BMSG as part of the CERCLA remediation action; to address the potential commingling of groundwater from the ICP and the BMSG capture system in Bucktail Creek that would be captured and treated prior to discharge into surface waters.

2. The ICP shall provide an annual report summarizing mining, reclamation, and monitoring activities and projecting proposed activities for the coming year. The ICP shall conduct an annual review with the USFS to determine if activities are in accordance with the approved Plan and if any changes to the Plan or financial assurance are needed.

3. The ICP shall provide notice to the USFS and make appropriate modifications to the POO if there are significant changes to project permits (such as NPDES, 404, or Air Quality).

4. Waste rock (slash) left underground in the Ram and Sunshine mines shall be amended to provide alkalinity to reduce potential for metals leaching.

    a. The ICP shall provide a Waste Rock Amendment Plan as described in the FEIS and Water Resources Technical Report.

    b. The ICP shall provide funding for the SCNF to obtain a third party contractor to assist SCNF in review of the Waste Rock Amendment Plan.

5. Waste rock and tailings materials will be tested throughout the life of the mine to evaluate potential for acid generation and metals leaching.

   a. The ICP shall provide a Geochemical Monitoring Plan as described in the FEIS and Water Resources Technical Report.

6. The ICP shall provide funding for the SCNF to obtain a third party contractor to assist SCNF in review of the Geochemical Monitoring Plan. The ICP shall provide engineering final design for the ponds that includes:

   a. Spillways to reduce risk of failure if overtopping occurs.

   b. Protection of pond liners from potential ice damage.

7. The ICP shall provide a Stormwater Pollution Prevention Plan that includes:

   a. Those permanent water control structures that will exist beyond the life of mine will be designed to handle flow from the 100-year storm event;

   b. Standard state and Federal BMPs will be utilized for Project sediment control;

   c. Soil disturbing construction activities will be conducted during a set construction season to minimize impacts to soils and sediment production; and,

   d. Sediment control monitoring.

8. Modifications to the water resources monitoring plan to provide adequate data to evaluate potential impacts to surface and groundwater.

9. The ICP shall provide a Post-mining Groundwater Capture Plan that includes:

   a. Decision criteria and action (trigger) limits for post-mining water capture. Criteria shall include conditions that would lead to decisions regarding cessation of mine dewatering;

   b. A plan for installation and evaluation of bedrock groundwater, alluvial groundwater and surface water capture systems for the Ram and Sunshine Mines; and,

   c. The groundwater capture system shall be fully installed prior to construction of the mine adits.

10. Enhanced emergency management capabilities will be instituted for medical (including designating a helipad site), spill control and fire situations.

11. Native species and more detailed reclamation procedures will be used for reclamation to ensure achievement of self-sustaining vegetation following reclamation.

12. Additional access road improvements will be made to reduce sediment release and traffic accident risk.

13. Construction workers will be required to utilize van sharing or busing to minimize traffic during the construction period.

14. Any surface vegetation clearing or timber removal will be conducted following USFS guidelines and practices and following USFS approval;

15. The ICP shall institute a weed control plan that conforms to USFS and County guidelines;

16. The ICP will be required to submit a plan to monitor existing wetlands and constructed wetlands to determine impacts to wetlands functions, and shall modify the constructed wetlands as necessary to assure that they are providing suitable wetland habitat to compensate for Project impacts to natural wetlands.

17. The ICP shall submit a final engineering design for the TWSF that includes:

    a. A closure cap that includes a minimum of 4 feet of soil cover material to protect the liner from potential damage from trees growing on the reclaimed surface;

    b. A plan for placement of tailings into the TWSF during winter designed to maintain the design density and moisture content of the dry stack tailings;

    c. Co-disposal of tailings and waste rock in the TWSF to reduce the oxidation rate of the higher permeability waste rock component and reduce long-term risk to the environment from metals release;

    d. Design for construction of the 36 acre TWSF facility; and,

    e. Post-mining monitoring of water quality in, and/or discharging from the TWSF for a period of not less than 5 years.

18. The ICP shall develop a waste rock disposal plan for any material disturbance on the Sunshine portal pad or any other location where preexisting wastes may be disturbed for USFS approval in accordance with all relevant Federal, state, and local laws.

19. The ICP shall submit a final engineering design for the water treatment system and provide SCNF with funding for a third party contractor to assist SCNF in review of the plan. At a minimum, the water treatment plan shall:

    a. Provide treatment capable of meeting effluent limits in the NPDES permit; and,

    b. Minimize the need for disposal of water treatment waste residues.

20. The ICP shall designate a reclamation coordinator who shall be responsible for the following:

    a. Being the primary contact with the SCNF on permit compliance, monitoring and mitigation;

    b. Describing how environmental protection standards contained in plans and permits would be implemented;

    c. Preparing reclamation plans for all proposed surface disturbance. These plans shall be submitted to the USFS and will include interim and final reclamation for the facilities along with an estimate of the costs to complete the work;

    d. The coordinator shall certify that all reclamation work was completed as planned for each facility. The USFS Administrator shall accompany the company coordinator in reviewing all proposed activities; and,

    e. Prior to the 8th year of operation the company shall summarize the results of all testing for closure purposes and submit its plans for final reclamation to the USFS and Interagency Task Force for review and approval.

21. The ICP shall reduce impacts of dust along primary access roads by watering, surfacing, or treating the surface of the road with an approved chemical amendment as directed by the SCNF.

22. The ICP shall comply with USFS Region 4 Reclamation Guidelines except where authorized by the SCNF:

    a. The ICP shall recover soil material from all areas of project disturbance in sufficient quantities to place a minimum of a 1 foot layer on features identified for reclamation in the Reclamation Plan;

    b. Earth fill construction will be confined to the normal operating season unless specifically authorized by the USFS;

    c. All exposed soil materials shall be stabilized and reclaimed in the same season as the disturbance, unless otherwise authorized by the SCNF;

    d. All slopes shall be kept to a minimum;

    e. Surface disturbances shall be recontoured; and,

    f. Reclaimed slopes shall be shaped to prevent the concentration of water except at points specifically designed to handle flows without erosion;

23. The ICP shall obtain a Forest RUP that specifies the conditions under which they can use the Forest roads. The ICP's POO and/or RUP shall include the following:

   a. Access road design shall meet USFS specifications (U.S. Department of Transportation Federal Highway Administration, 2003) for road width, grade, alignment, surfacing, drainage, quality control and signing. Exceptions to these standards may be used only with SCNF approval. Forest Plan requirements for road construction and natural resource protection will be followed. The ICP shall submit designs for road construction and improvements to the USFS for review and approval prior to initiating construction.

   b. Develop a plan for busing of all mine employees. The ICP shall monitor the use of the provided busing to establish the rate of use, and shall furnish an annual summary of use to the SCNF. If an 80% usage rate for all mine employees including management is not achieved on an annual basis, revisions to the plan may be required by the USFS.

   c. The ICP shall require contractors to comply with requirements for van pooling or busing of employees including the 80% participation goal. In addition, the ICP shall ensure that small deliveries or partial loads of materials are delivered to a staging area in Salmon, for consolidation prior to proceeding to the mine site to the extent practicable. Occasional site visitors such as salespeople shall be authorized access to the site as necessary.

   d. The ICP shall develop a written policy for compliance with all SCNF traffic rules and require that all contractors comply with state and Forest rules for oversize and overweight loads.

   e. The USFS must approve all location or design changes for access and haul roads on National Forest System lands.

   f. The ICP shall implement or provide payment to the USFS for deferred road maintenance (such as surface, culvert or bridge replacement), and recurrent (grading, cleaning culverts) maintenance based on road use, as specified in the Road Use Agreement.

24. The ICP transportation and/or spill control plan shall include training requirements for all drivers including a requirement that all new drivers transporting fuel, chemicals, or concentrate make their first trip to the site accompanied by a company representative.

   a. All fuel, chemical supply and concentrate trucks, all tractor-trailer units, and any single unit vehicles more than 24 feet in length shall be accompanied by a pilot car.

   b. No secondary trailers (pups) will be allowed for ICP or their suppliers.

c. Fuel tankers shall contain no more than 4,500 gallons per load.

d. During the construction period, the ICP shall coordinate all use of approved and alternative access routes with the USFS under an approved RUP.

25. Road improvements of the entire 40 mile project access route (Section 1.2.2 of this Opinion, Appendix A, Figure 1 and Table 1).

26. Borrow Areas – Work on the Williams Creek Road would utilize borrow from the existing Leesburg East pit. The pit shall be reclaimed following its use.

27. Road reclamation – Approximately 7.58 miles of site roads shall be reclaimed during the construction phase. All new roads except those roads identified by the agencies as needed for administrative purposes shall be reclaimed at mine closure.

The conservation measures described here and in the consultation initiation package as parts of the proposed action are intended to reduce or avoid adverse effects on listed species and their habitats. NMFS regards these conservation measures as integral components of the proposed action and expects that all proposed project activities will be completed consistent with those measures. We have completed our effects analysis accordingly. Any deviation from these conservation measures will be beyond the scope of this consultation and will not be exempted from the prohibition against take as described in the attached incidental take statement. Further consultation will be required to determine what effect the modified action may have on listed species or designated critical habitats.

1.3. Action Area

'Action area' means all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR 402.02). For purposes of this consultation, the action area includes the ICP mining site, and all waterbodies potentially affected by the mine site, the wastewater discharge, borrow areas, and/or the transportation corridor. The ICP mining site is located in the headwaters of Big Flat Creek, Bucktail Creek, South Fork Big Deer Creek, Little Deer Creek, and Blackbird Creek. Several unnamed tributaries, springs, and wetlands connected to each of these subwatersheds may also be influenced by mine site construction and operations. Each of these streams has been included in the action area for this project.

The proposed discharge pipe will empty into Big Deer Creek, a tributary of Panther Creek. Since the action will discharge treated mine wastewater into Big Deer Creek, the action area includes Big Deer Creek from the discharge pipe downstream to the confluence with Panther Creek. Considering estimated dilution factors outlined in the BA (pages 7-38 to 7-39), the action area also includes the Panther Creek mainstem from Big Deer Creek downstream to its confluence with the Salmon River. As described below in the effects analysis, dilution by the larger Salmon River is expected to reduce concentrations of potential pollutants to insignificant levels, levels unlikely to result in lethal or sublethal effects to ESA-listed species.

The action area also includes the three borrow areas used for the project, including one on the Williams Creek road near the Williams Creek summit (Leesburg East), one along Blackbird Creek, and the other in the Bucktail Creek drainage (Appendix A, Figure 2). All of these borrow areas are on USFS land and have been previously used as borrow sources.

The transportation route plans to use existing USFS roads along Blackbird, Panther, Deep, Moccasin, and Williams Creeks. The route also crosses Perreau Creek near its mouth, and the Salmon River at Shoup Bridge. Because of the potential for spills of mined ore, fuels, acids, and other potentially toxic substances along the route, the action area for the transportation route includes the following streams, and the immediate stream reaches downstream from where spills could occur: (1) Blackbird Creek from the mine site downstream to Panther Creek; (2) Panther Creek, from Blackbird Creek downstream to Big Deer Creek; (3) Deep Creek, from the uppermost road crossing (Forest Road [FR] 101) downstream to its confluence with Panther Creek (approximately 3.5 miles); (4) Moccasin Creek, from the headwaters downstream to the mouth; (5) Williams Creek, from the headwaters downstream to the mouth; (6) Perreau Creek, from the FR 021 bridge downstream to the Salmon River; and, (7) the Salmon River, for 2 miles downstream of the Shoup Bridge. Because of dilution, mixing, and application of the SPCC, effects from spills are not expected to extend beyond these stream reaches.

Snake River sockeye salmon, spring/summer Chinook salmon, Snake River Basin steelhead, and their designated critical habitats are found within the action area for the ICP (Table 3).

Table 3. Federal Register notices for final rules that list threatened and endangered species, designate critical habitats, or apply protective regulations to listed species considered in this consultation.

Species	Listing Status	Critical Habitat	Protective Regulations
Chinook salmon (*Oncorhynchus tshawytscha*)			
Snake River spring/summer run	T 6/28/05; 70 FR 37160	10/25/99; 64 FR 57399	6/28/05; 70 FR 37160
Sockeye salmon (*O. nerka*)			
Snake River	E 6/28/05; 70 FR 37160	12/28/93; 58 FR 68543	ESA Section 9 applies
Steelhead (*O. mykiss*)			
Snake River Basin	T 1/05/06; 71 FR 834	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160

Note: Listing status: 'T' means listed as threatened under the ESA; 'E' means listed as endangered.

2. ENDANGERED SPECIES ACT

The ESA establishes a national program to conserve threatened and endangered species of fish, wildlife, plants, and the habitat on which they depend. Section 7(a)(2) of the ESA requires Federal agencies to consult with the FWS, NMFS, or both, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitats. Section 7(b)(4) requires the provision of an incidental take statement that specifies the impact of any incidental taking and includes reasonable and prudent measures (RPMs) to minimize such impacts.

2.1. Biological Opinion

This Opinion presents NMFS' review of the status of each listed species of Pacific salmon and steelhead[2] considered in this consultation, the condition of designated critical habitat, the environmental baseline for the action area, all the effects of the action as proposed, and cumulative effects (50 CFR 402.14(g)). For the jeopardy analysis, NMFS analyzes those combined factors to conclude whether the proposed action is likely to appreciably reduce the likelihood of both the survival and recovery of the affected listed species.

The critical habitat analysis determines whether the proposed action will destroy or adversely modify designated critical habitat for listed species by examining any change in the conservation value of the essential features of that critical habitat. This analysis relies on statutory provisions of the ESA, including those in section 3 that define "critical habitat" and "conservation," in section 4 that describe the designation process, and in section 7 that sets forth the substantive protections and procedural aspects of consultation. The regulatory definition of "destruction or adverse modification" at 50 CFR 402.02 is not used in this Opinion.

2.1.1. Status of the Species and Critical Habitat

This section defines the biological requirements of each ESA-listed species affected by the proposed action, and the status of each designated critical habitat relative to those requirements. Listed species facing a high risk of extinction and critical habitats with degraded conservation value are more vulnerable to the aggregation of effects considered under the environmental baseline, the effects of the proposed action, and cumulative effects.

2.1.1.1. Status of the Species.

NMFS reviews the condition of the listed species affected by the proposed action using criteria that describe a 'viable salmonid population' (VSP) (McElhany *et al*. 2000). Attributes associated with a VSP include abundance; productivity, spatial structure, and genetic diversity that maintain its capacity to adapt to various environmental conditions and allow it to sustain itself in the natural environment. These attributes are influenced by survival, behavior, and experiences throughout the entire life cycle, characteristics that are influenced, in turn, by habitat and other environmental conditions.

Snake River Sockeye Salmon - The Snake River sockeye salmon, listed as endangered on November 20, 1991 (56 FR 58619), includes populations of sockeye salmon from the Snake River Basin, Idaho (extant populations occur only in the Salmon River drainage). Under the 2005 NMFS hatchery stock policy (June 28, 2005, 70 FR 37204), the progeny of fish from a

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[2] An 'evolutionarily significant unit' (ESU) of Pacific salmon (Waples 1991) and a 'distinct population segment' (DPS) of steelhead (final steelhead FR notice) are considered to be 'species,' as defined in Section 3 of the ESA.

listed population that are propagated artificially are considered part of the listed species and are protected under ESA. Thus, although not specifically designated in the 1991 listing, Snake River sockeye salmon produced in the captive broodstock program are included in the listed ESU.

Within the action area sockeye salmon only occur in the mainstem Salmon River, migrating through the action area during upstream and downstream migrations. Considering the short timeframe sockeye are present in the action area, combined with the very low probability of an accident and hazardous material spill occurring in the vicinity of the Shoup Bridge, the SCNF determined that the proposed action would have "no effect" on Snake River sockeye salmon.

Typically NMFS does not review or concur with "no effect" determinations, a determination made at the discretion of the action agency. However, NMFS has reviewed the determination in this case since sockeye were included in the BA and occur in the action area. A "no effect" determination is only appropriate when the proposed action will not affect a listed species or designated critical habitat. A "not likely to adversely affect" determination is appropriate when potential effects would be insignificant, discountable, or beneficial. Discountable effects are further defined as effects that are extremely unlikely to occur. Although very unlikely, the potential still exists for a spill to occur when sockeye are present in the action area. If sockeye are present near the Shoup Bridge and a spill occurs, adverse effects will occur. This potential for effect, although unlikely makes a "not likely to adversely affect" determination more appropriate. Upon sharing this information with the USFS, the USFS agreed with this and have amended their determination for sockeye salmon to "not likely to adversely affect." Because potential effects to sockeye salmon can be considered discountable for this action due to the protective measures and BMPs included in the proposed action, as supplemented by the terms and conditions imposed in this Opinion, NMFS concurs with this determination and will not further address effects to sockeye salmon in this Opinion.

Snake River Spring/Summer Chinook Salmon - The Snake River spring/summer Chinook salmon ESU, listed as threatened on April 22, 1992, (67 FR 14653), includes all natural-origin populations in the Tucannon, Grande Ronde, Imnaha, and Salmon Rivers. Fish returning to several of the hatchery programs are also listed, including those returning to the Tucannon River, Imnaha, and Grande Ronde hatcheries in Oregon and Washington, and to the Sawtooth, Pahsimeroi, and McCall hatcheries on the Salmon River. Critical habitat was designated for Snake River spring/summer Chinook salmon on December 28, 1993 (58 FR 68543) and was revised on October 25, 1999 (64 FR 57399).

The Snake River drainage is thought to have produced more than 1.5 million adult spring/summer Chinook salmon in some years during the late 1800s (Matthews and Waples 1991). By the 1950s the abundance of spring/summer Chinook had declined to an annual average of 125,000 adults and by the mid-1960s, further declined to an average of about 60,000 adults. Adult returns counted at Lower Granite Dam reached all-time lows in the mid-1990s, and numbers have begun to increase since 1997. Over a 10-year period from 1992 to 2001, which includes the year of listing (1992), returns of wild/natural fish ranged from 183 in 1994 to 12,475 in 2001, and averaged 3,314 salmon adults. The estimated smolt production capacity of 10 million smolts for rivers in Idaho, coupled with historic smolt to adult return rates of 2% to 6 %, indicate Idaho could produce wild/natural runs of 200,000 to 600,000 adults (Fish

Passage Center 2002; Fish Passage Center 2003). The relatively low numbers of the last decade are reflected throughout the entire distribution of Chinook salmon subpopulations scattered throughout the Grande Ronde, Imnaha, Tucannon, and Salmon River Basins. Redd counts and estimates of parr and smolt densities generally indicate that fish production is well below the potential, and continuing to decline.

Despite fluctuations in the number of adult returns, the general trend in adult returns since 1977 has been a gradual population decline with episodic oscillations (McClure et al. 2003). Chinook salmon numbers were higher since 2000 than during the 24 previous years of record (Fish Passage Center 2004). Although there were record returns in 2000 and 2001, and relatively high returns from 2002 to 2004, ESU numbers are in general very low in comparison to historic levels (Beven et al. 1994). Average returns of adult Snake River spring/summer Chinook salmon (averaging 3,314 over a recent 10-year period) are also low in comparison to interim target species recovery levels of 44,766 for the Snake River Basin. The low returns amplify the importance that a high level of protection be afforded to each adult Chinook salmon, particularly because a very small percentage of salmon survive to the life stage of a returning, spawning adult, and because these fish are in the final stage of realizing their reproductive potential (approximately 2,000 to 4,000 progeny per adult female) (Behnke 2002).

Habitat impairment is common in the range of this ESU. Spawning and rearing habitats are impaired by factors such as tilling, water withdrawals, timber harvest, grazing, mining, and alteration of floodplains and riparian vegetation (NMFS 2004a). Mainstem Columbia River and Snake River hydroelectric developments have altered flow regimes and estuarine habitat, and disrupted migration corridors (Raymond, 1979; NMFS 2000). Competition between natural indigenous stocks of spring/summer Chinook salmon and spring/summer Chinook of hatchery origin has likely increased due to an increasing proportion of naturally reproducing fish of hatchery origin (Behnke 2002).

Compared to the greatly reduced numbers of returning adults for the last several decades, exceptionally large numbers of adult Chinook salmon returned to the Snake River drainage in 2000 and in 2001. These large returns are thought to be a result of favorable ocean conditions (Logerwell *et al.* 2003; Meeings and Lackey 2005), and above average flows in the Columbia River Basin when the smolts migrated downstream. These large returns are only a fraction (5% to 10%) of the estimated returns of the late 1800s (Behnke 2002). Recent increases in the population are not expected to continue, and the long-term trend for this species indicates a decline. Detailed information on the range-wide status of Snake River spring/summer Chinook salmon under the environmental baseline is described in Chinook salmon status reviews (Myers *et al.* 1998; BRT 2003; NMFS 2004a). Habitat improvements would not necessarily correspond to increased salmon productivity because a myriad of other factors can still depress populations, but diminished quality would probably correspond to reduced productivity (Regetz 2003). Additional information on the biology, status, and habitat elements for Snake River spring/summer Chinook salmon is described in the status review updates (BRT 1998, 2003; NMFS 2004a; Good *et al.* 2005).

The status of Snake River spring/summer Chinook salmon in the Upper Salmon River Basin was evaluated in a comprehensive manner by the USFS, in the description of the environmental baselines for the 1994 watershed-scale BAs for the anadromous salmonid fourth field hydrologic unit codes (HUCs) (USDA Forest Service [USFS]1993). More recently, NMFS and its Federal partners updated the status of the Snake River spring/summer Chinook salmon ESU by identified Interior Columbia Basin Technical Recovery Team (ICBTRT) populations and for each fifth field HUC for the remand of the Federal Columbia River Power Systems (FCRPS) Opinion (Cooney 2004; NMFS 2004a; ICBTRT 2005). The Idaho Department of Fish and Game (IDFG) conducts annual aerial and ground-based redd counts throughout the Upper Salmon River Basin and provides some estimates of parr-to-smolt survival from various pit-tag studies (StreamNet 2007).

Within the Snake River spring/summer Chinook salmon ESU, independent populations have been grouped into larger aggregates that share similar genetic, geographic (or hydrographic), and/or habitat characteristics (ICBTRT 2005; McClure et al. 2003). This ESU was broken down into five Major Population Groups (MPGs) with 31 extant independent populations. These MPGs consist of the Lower Snake, Grande Ronde/Imnaha, South Fork Salmon, Middle Fork Salmon, and the Upper Salmon. The project is located in the Upper Salmon MPG, which is further delineated into nine independent populations (Table 4).

Chinook salmon from Big Deer, Panther, Deep, and Moccasin Creeks are part of the historic Panther Creek population (ICBTRT 2005; NMFS 2006b). Williams Creek, Perreau Creek, and the mainstem Salmon River are included in the Lower Salmon Chinook salmon population (ICBTRT 2005; NMFS 2006b).

The ICBTRT determined that the Panther Creek Chinook salmon population was extirpated during the 1960s due to legacy mining and the heavy metal wastes deposited in Lower Panther Creek from the Blackbird Mine operations (ICBTRT 2003; ICBTRT 2005). Near the mouth of Panther Creek, a chemical barrier formed that prevented anadromous fish passage and eventually caused the demise of the Panther Creek Chinook salmon population. Recovery has been slow, but Chinook salmon redds were observed downstream of the Blackbird Mine again in Panther Creek in 1990 and 1991 (IDEQ 2001). In June and July 2001, the IDFG planted 1,064 Chinook salmon adults at four sites in Panther Creek for harvest. In the fall of 2001, over 80 redds were observed in Panther Creek and were suspected to be attributed to the planted fish. Chinook salmon have since been observed spawning in Panther Creek as far upstream as Moyer Creek in 2005, 2006, and 2007 (USDA Forest Service 2007).

The Lower Salmon River Chinook population contains both spring and summer run Chinook. This population does not currently meet viability criteria because neither abundance/productivity risk nor spatial structure/diversity risk meets the criteria for a viable population. The 20-year delimited recruit per spawner point estimate is above replacement (1.25), but less than the 1.45 required at the minimum threshold abundance. The 10-year geometric mean abundance (123) is only 6% of the minimum threshold abundance. Substantial improvements in abundance/productivity status (reduction of risk level) will need to occur before the population can be considered viable. Also, the population currently does not meet the criteria for a “maintained” population (NMFS 2006b).

The current status of the MPG was determined by applying the ICTRT's six MPG-level viability criteria (ICTRT 2005). Viability assessments for all populations in the MPG were completed before considering the MPG-level criteria. Assessment of abundance/productivity risk level has not been completed for the North Fork Salmon River and Panther Creek populations.

Table 4. Characteristics of independent populations in the Upper Salmon River Spring/summer Chinook MPG.

Population	Extant/ Extinct	Size	Threshold Abundance	Minimum Productivity
North Fork Salmon River	Extant	Basic	500	1.90
Lemhi River	Extant	Very Large	2,000	1.2
Salmon River Lower Mainstem	Extant	Very Large	2,000	1.2
Pahsimeroi River	Extant	Large	1,000	1.45
East Fork Salmon River	Extant	Large	1,000	1.45
Yankee Fork Salmon River	Extant	Basic	500	1.90
Valley Creek	Extant	Basic	500	1.90
Salmon River Upper Mainstem	Extant	Large	1,000	1.45
Panther Creek	Extinct	Intermediate	1,000	1.2

Note – Minimum abundance and productivity values represent levels needed to achieve a 95% probability of persistence over 100 years. Shaded populations or portions of those populations occur in non-wilderness portions of the SCNF and within the action area.

The Upper Salmon River MPG currently does not meet MPG-level viability criteria. For the MPG to be considered viable, a minimum of five of the nine independent populations in the MPG must be considered viable. Currently, none of the nine populations in the MPG meet population level viability criteria. For a detailed discussion and updates regarding the status and viability of Snake River spring/summer Chinook salmon and its independent populations, please refer to the following website: http://www.idahosalmonrecovery.net/.

Within the action area, Chinook salmon have only been documented in the mainstem Salmon River and Panther Creek (USDA Forest Service 2007; Kuzis 2004).

Snake River Steelhead - The Snake River Basin steelhead DPS, listed as threatened on August 18, 1997 (62 FR 43937), includes all natural-origin populations of steelhead in the Snake River basin of southeast Washington, northeast Oregon, and Idaho. One of the hatchery stocks (originating from Dworshak Reservoir) in the basin is listed under the B-Run Program (Pollard, personal communications 2004), and six hatchery stocks are included in the DPS under the 2005 NMFS hatchery stock policy (June 28, 2005, 70 FR 37204). Although Snake River Basin steelhead were originally listed as a threatened ESU, they were recently reclassified as a threatened DPS (January 5, 2006; 71 FR 834).

The Snake River basin is believed to have produced up to half of the steelhead in the Columbia River basin historically, but natural runs have been declining in abundance over the past several decades (BRT 2003). Counts of wild and hatchery-origin steelhead returning to the Snake River basin declined sharply in the early 1970s, increased modestly from the mid 1970s through the 1980s, and declined again during the 1990s (Fish Passage Center 2004).

With a few exceptions, recent annual estimates of steelhead returns to specific production areas within the Snake River are not available. Annual return estimates are limited to counts of the aggregate returns over Lower Granite Dam. Returns to Lower Granite Dam remained at relatively low levels through the 1990s. The 2001 run size at Lower Granite Dam was substantially higher relative to the 1990s. The 2002 through 2006 return years have declined annually but continue to remain higher than the 1990s return years. Although steelhead numbers have dramatically increased, wild steelhead have comprised only 10% to 23% of the total returns since 1994. Consequently, the large increase in fish numbers does not reflect a change in steelhead status based on historic levels. The long-term trend for this species indicates a decline. The recent 5-year (2002 to 2006) mean abundance (40,941 natural returns) is approximately 79% of the interim recovery target level. This is a dramatic increase over the previous 5-year (1997 to 2001) mean abundance level of 19,717 fish counted at Lower Granite Dam.

Significant factors contributing toward declining steelhead populations include mortality associated with the mainstem dams along the Columbia and Snake Rivers, losses from harvest, loss of access to more than 50% of their historic range, and degradation of habitats used for spawning and rearing (NMFS 2004b). Possible genetic introgression from hatchery stocks is another threat to this DPS since wild fish comprise such a small proportion of the populations (Behnke 2002). Detailed information on the current range wide status of Snake River Basin steelhead, under the environmental baseline, is described in the steelhead status review (Busby et al. 1996), the status review update (BRT 2003), and the DPS listing (January 5, 2006, 71 FR 834).

The ICBTRT identified 29 independent populations in the Snake River Basin steelhead DPS, grouped into six MPGs (NMFS 2006c). Of the 29 populations, 25 are extant, three extirpated, and one (North Fork Clearwater), blocked from its historic habitat. Steelhead in the action area are included in the Salmon River MPG, which includes twelve independent populations (Table 5). Eight of these populations are classified as supporting A-run steelhead and four are classified as supporting B-run steelhead. Population size designations, based on intrinsic potential habitat, range from Basic to Large (NMFS 2006c). The Panther Creek and Pahsimeroi populations occur within the action area.

The Panther Creek steelhead population is found in the Panther Creek drainage, an area encompassing Big Deer Creek, Deep Creek, and Moccasin Creek in the action area. As an intermediate sized population, the ICBTRT recommends a minimum abundance threshold of 1,000 naturally produced spawning adults to produce a sufficient intrinsic productivity to achieve a 5 % or less risk of extinction over 100 years. The ICBTRT has identified one major spawning area and three minor spawning areas within the Panther Creek steelhead population. Panther, Big Deer, Moccasin, and Deer Creeks are located within the major spawning area identified for the Panther Creek population. Although the ICBTRT does not document any of these streams in the current spawning range for the population, each of these streams has been identified as having high intrinsic potential for spawning (ICBTRT 2005).

Table 5. Characteristics of independent populations in the Salmon River steelhead MPG.

Population	Extant/ Extinct	Life History	Size	Threshold Abundance	Minimum Productivity
Little Salmon River	Extant	A-Run	Intermediate	1,000	1.2
South Fork Salmon	Extant	B-Run	Intermediate	1,000	1.2
Secesh River	Extant	B-Run	Basic	500	1.4
Chamberlain Creek	Extant	A-Run	Intermediate	1,000	1.2
Lower Middle Fork	Extant	B-Run	Large	1,500	1.13
Upper Middle Fork	Extant	B-Run	Large	1,500	1.13
Panther Creek	Extant	A-Run	Intermediate	1,000	1.2
North Fork Salmon	Extant	A-Run	Basic	500	1.4
Lemhi	Extant	A-Run	Intermediate	1,000	1.2
Pahsimeroi River	Extant	A-Run	Intermediate	1,000	1.2
East Fork Salmon	Extant	A-Run	Intermediate	1,000	1.2
Upper Salmon Mainstem	Extant	A-Run	Intermediate	1,000	1.2

Note – Minimum abundance and productivity values represent levels needed to achieve a 95% probability of persistence over 100 years. Shaded populations or portions of those populations occur in non-wilderness portions of the SCNF and within the action area.

Aquatic habitat in the Panther Creek drainage has been severely degraded through mining activity, substantially affecting the presence and distribution of steelhead within the population. The current range of the population is significantly reduced from historic conditions. Loss of occupancy in the single major spawning area, which contained approximately 67% of the historic intrinsic potential habitat, has reduced occupancy to only 33% of the historic range. The population does not currently meet the ICBTRT viability criteria for abundance/productivity risk (moderate risk) or spatial structure/diversity (high risk) However, because there is only one major spawning area in the population, the lowest risk level this population will be able to achieve is Moderate (ICBTRT 2005).

The Pahsimeroi River steelhead population includes the Pahsimeroi River, and the mainstem Salmon River and it's tributaries from the Pahsimeroi River downstream to its confluence with the Lemhi River. Williams Creek and Perreau Creek are included in this population. The ICBTRT has identified three major spawning areas and two minor spawning areas within the Pahsimeroi River steelhead population. Williams Creek is classified as having a high intrinsic potential for steelhead spawning, and is located within one of the two identified minor spawning areas for the population. Neither the mainstem Salmon River reach in the action area nor Perreau Creek were included as part of a spawning area or rated as having any intrinsic potential for spawning (ICBTRT 2005).

Like Panther Creek, habitat is degraded in Perreau Creek due to historic and more recent mining during the 1980s (IDEQ 2001). The lower ends of Perreau and Williams Creeks are seasonally disconnected from the Salmon River due to water withdrawals and by agricultural irrigation diversions. Although spawning has not been reported recently for this population in the action area, the population is rated at a moderate risk because of spawning activity in the three major spawning areas in the Pahsimeroi River watershed. The Pahsimeroi River steelhead population does not currently meet viability criteria because abundance/productivity risk has been rated as moderate risk. Improvement in abundance/productivity status will need to occur before the population can be considered viable (ICBTRT 2005).

Considering population viability assessments completed for all 12 populations in the MPG, the Salmon River steelhead MPG currently does not meet MPG-level viability criteria. For the MPG to be considered viable, a minimum of six of the 12 extant independent populations in the MPG must be considered viable. The current recovery planning objective for the MPG is for the Chamberlain Creek, Secesh River, South Fork Salmon, Upper Middle Fork Salmon River, and Upper Salmon River Mainstem populations (plus one additional Large or Intermediate population) to be rated as viable. One must be rated as highly viable. Currently, none of the 12 extant populations in the MPG meet population level viability criteria (NMFS 2006c). For a detailed discussion and updates regarding the status and viability of Snake River Basin steelhead and its independent populations, please refer to: http://www.idahosalmonrecovery.net/.

Within the action area, steelhead have been documented in the mainstem Salmon River, Deep Creek, Little Deer Creek, lower Big Deer Creek, Blackbird Creek, and Panther Creek (USDA Forest Service 2007; Kuzis 2004).

2.1.1.2. Status of Critical Habitat.

NMFS reviews the status of designated critical habitat affected by the proposed action by examining the condition and trends of primary constituent elements (PCEs) throughout the designated area. The PCEs consist of the physical and biological features identified as essential to the conservation of the listed species in the documents that designate critical habitat (Table 6).

At the time that each habitat area was designated as critical habitat, that area contained one or more PCEs within the acceptable range of values required to support the biological processes for which the species use that habitat. Of the six types of sites identified by NMFS and discussed above, the action area for this consultation provides freshwater spawning, rearing and migration habitat for Snake River spring/summer Chinook salmon and Snake River Basin steelhead.

The action area also provides adult and juvenile migratory habitat for Snake River sockeye salmon. Sockeye salmon spawning and rearing habitat in the action area is currently limited to Petit, Alturas, and Redfish Lakes and their inlet and outlet streams. Therefore, sockeye salmon spawning and rearing habitat will not be influenced by the proposed action. Sockeye salmon migratory critical habitat would only be affected if a spill occurs alongside the mainstem Salmon River. Because NMFS considers the risk of adverse effects from spills occurring along U.S. Highway 93 to be discountable, adverse effects to the migration PCE for sockeye salmon are not expected, and NMFS concurs with the USFS determination of “not likely to adversely affect” sockeye salmon designated critical habitat. Consequently, effects to sockeye salmon critical habitat will not be further discussed in this Opinion.

The essential features associated with the freshwater spawning, rearing and migratory sites potentially affected by this action include water quality, water quantity, substrate/spawning gravels, forage/food, riparian vegetation, and access/safe passage.

Table 6. Types of sites and essential physical and biological features designated as PCEs, and the species life stage each PCE supports.

Site	Essential Physical and Biological Features	ESA-listed Species Life Stage
Snake River Steelhead[a]		
Freshwater spawning	Water quality, water quantity, and substrate	Spawning, incubation, and larval development
Freshwater rearing	Water quantity & floodplain connectivity to form and maintain physical habitat conditions	Juvenile growth and mobility
	Water quality and forage[b]	Juvenile development
	Natural cover[c]	Juvenile mobility and survival
Freshwater migration	Free of artificial obstructions, water quality and quantity, and natural cover[c]	Juvenile and adult mobility and survival
Snake River Spring/summer Chinook Salmon		
Spawning & Juvenile Rearing	Spawning gravel, water quality and quantity, cover/shelter, food, **riparian vegetation**, and **space**	Juvenile and adult.
Migration	Substrate, water quality and quantity, water temperature, water velocity, cover/shelter, food[d], riparian vegetation, space, **safe passage**	Juvenile and adult.
Snake River Sockeye Salmon		
Spawning & Juvenile Rearing	Spawning gravel, water quality and quantity, **water temperature**, food, **riparian vegetation**, and **access**	Juvenile and adult.
Migration	Substrate, water quality and quantity, water temperature, water velocity, cover/shelter, food[d], riparian vegetation, space, **safe passage**	Juvenile and adult.

a Additional PCEs pertaining to estuarine, nearshore, and offshore marine areas have also been described for Snake River steelhead. These PCEs will not be affected by the proposed action and have therefore not been described in this Opinion.
b Forage includes aquatic invertebrate and fish species that support growth and maturation.
c Natural cover includes shade, large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
d Food applies to juvenile migration only.

Snake River Spring/Summer Chinook Salmon Critical Habitat - Critical habitat was designated for Snake River spring/summer Chinook salmon on December 28, 1993 (58 FR 68543), and was revised on October 25, 1999 (64 FR 57399). Critical habitat is designated in the Upper Salmon River basin to include all river reaches presently or historically accessible to

Snake River spring/summer Chinook salmon. Critical habitat includes the stream bottom, the water, and the adjacent riparian zone, which is defined as the area within 300 feet of the line of high water of a stream channel or from the shoreline of a standing body of water.

Habitat impairment is common in the range of this ESU, including the SCNF. Spawning and rearing habitats have been impaired by factors such as tilling, water withdrawals, timber harvest, grazing, mining, and alteration of floodplains and riparian vegetation. According to the ICTRT, the Panther Creek population was extirpated because of legacy and modern mining-related pollutants creating a chemical barrier to fish passage (Chapman and Julius 2005). Mainstem Columbia and Snake River hydroelectric developments have altered flow regimes and estuarine habitat, and disrupted migration corridors.

During all life stages spring/summer Chinook salmon require cool water that is relatively free of contaminants. Water quality impairments in the designated critical habitat of this ESU include inputs from fertilizers, insecticides, fungicides, herbicides, surfactants, heavy metals, acids, petroleum products, animal and human sewage, dust suppressants (e.g., magnesium chloride), radionuclides, sediment in the form of turbidity, and other anthropogenic pollutants. Pollutants enter the surface waters and riverine sediments from the headwaters of the Salmon River as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in salmon tissue. This species also requires rearing and migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

Within the action area, Chinook salmon designated critical habitat occurs in the mainstem Salmon River, mainstem Panther Creek, Blackbird Creek, lower Big Deer Creek, and Deep Creek.

Snake River Basin Steelhead Critical Habitat - Critical habitat for Snake River Basin steelhead was designated on September 2, 2005, with an effective date of December 31, 2005 (70 FR 52630). Critical habitat on the SCNF includes significant reaches in the Middle Fork Salmon and Upper Salmon River basins; Table 21 in Federal Register details the streams within the Snake River Basin steelhead geographical range but excluded from critical habitat designation. Designated critical habitat for the Snake River Basin steelhead only includes the stream channel, with a lateral extent as defined by the ordinary high-water line.

The Snake River Basin Critical Habitat Analytical Review Team (CHART) concluded that all occupied areas contain spawning, rearing, or migration PCEs for this species. The CHART concluded that many of the watersheds within the SCNF have high conservation values. The complex life cycle of steelhead gives rise to complex habitat needs, particularly during the freshwater phase (Spence *et al.* 1996). Spawning gravels must be of a certain size and free of sediment to allow successful incubation of the eggs. Eggs also require cool, clean, and well-oxygenated waters for proper development. Juvenile steelhead need abundant food sources, including insects, crustaceans, and other small fish. They need places to hide from predators (mostly birds and bigger fish), such as under logs, root wads and boulders in the stream, and beneath overhanging vegetation. They also need places to seek refuge from periodic high flows (side channels and off channel areas) and from warm summer water temperatures (coldwater springs, cool tributaries, and deep pools). Returning adults generally do not feed in fresh water but instead rely on limited energy stores to migrate, mature, and spawn. Like juvenile steelhead, the adults also require cool water and places to rest and hide from predators.

Like Chinook salmon, steelhead require cool water that is relatively free of contaminants during all life stages. Water quality impairments previously identified for Chinook salmon also occur across the range of Snake River Basin steelhead. Steelhead require rearing and migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

The CHART identified several management activities that have affected the PCEs in the designated critical habitat on the SCNF, including grazing, irrigation impoundments and withdrawals, mineral mining, and road building and maintenance. Mining and roads have affected the PCEs in many parts of the basin including the action area (NMFS 2005).

Within the action area, steelhead designated critical habitat occurs in the mainstem Salmon River, mainstem Panther Creek, lower Williams Creek, lower Big Deer, lower Little Deer, and Deep Creek, from the mouth upstream to its confluence with Little Deep Creek.

A more thorough discussion of the present condition of PCEs within designated critical habitat areas and the human activities that affect PCE trends are further described in the environmental baseline section of this Opinion.

2.1.2. Environmental Baseline

The ‘environmental baseline’ includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). An environmental baseline that does not meet the biological requirements of a listed species may increase the likelihood that adverse effects of the proposed action will result in jeopardy to a listed species or in destruction or adverse modification of a designated critical habitat.

NMFS describes the environmental baseline in terms of the biological requirements for habitat features and processes necessary to support all life stages of each listed species within the action area. The biological requirements of salmon and steelhead in the action area vary depending on the life history stage present and the natural range of variation present within that system (Groot and Margolis 1991; NRC 1996; Spence et al. 1996). During spawning migrations, adult salmon generally require clean water with cool temperatures and access to thermal refugia, DO near 100% saturation, low turbidity, adequate flows and depths to allow passage over barriers to reach spawning sites, and sufficient holding and resting sites. Fish select spawning areas based on species-specific requirements of flow, water quality, substrate size, and groundwater upwelling.

Embryo survival and fry emergence depend on substrate conditions (e.g., gravel size, porosity, permeability, and oxygen concentrations), substrate stability during high flows, and cold water temperatures (i.e., 55°F or less for most species). Habitat requirements for juvenile rearing include seasonally suitable microhabitats for holding, feeding, and resting. Migration of juveniles to rearing areas, whether the ocean, lakes, or other stream reaches, requires unobstructed access to these habitats. Physical, chemical, and thermal conditions may all impede migrations of adult or juvenile fish.

Each ESA-listed species considered in this Opinion resides in or migrates through the action area. Thus, for this action area, the biological requirements for salmon and steelhead are the habitat characteristics that support successful adult and juvenile migration, adult holding, spawning, incubation, rearing, and growth and development to smoltification. The habitat features likely to be affected by the proposed actions are water quality, water quantity, substrate, passage, and riparian vegetation.

The ICP site is mostly located on flat-topped mountains and moderate to steep V-shaped canyons at elevations ranging from 6,100 to 8,100 feet. The site is drained by Little Deer, Bucktail and Big Flat Creeks. Bucktail Creek flows into South Fork Big Deer Creek, Big Deer Creek, and Panther Creek. Big Flat and Little Deer Creeks flow directly into Panther Creek.

Most water sources in and around the ICP project area are affected by some degree by past mining activities and ongoing cleanup activities associated with the Blackbird Mine, an inactive mine located adjacent to the ICP (Appendix A, Figures 1 and 2). The Blackbird Mine site covers approximately 830 acres of private patented mining claims and 10,000 acres of unpatented mining claims within the SCNF. Mining activity resulted in about 14 miles of underground workings, a 12-acre open pit, 4.8 million tons of waste rock deposited in numerous piles, and 2 million tons of tailings disposed at a tailings impoundment (Hydrometrics 2006).

The Blackbird Mine site spans the Bucktail Creek and Meadow/Blackbird Creek drainages. These drainages flow into Panther Creek. Acid rock drainage from the waste rock piles, the underground workings, the tailings impoundment, and tailings deposited along streams have resulted in the release of elevated levels of hazardous substances to the environment (groundwater, surface water, and soils), including but not limited to copper, cobalt, and arsenic. These releases have contributed to elevated levels of dissolved copper and cobalt in Panther Creek and some of its tributaries. Contaminated soil, sediments, waste rock, and tailings were also released from the Blackbird Mine site during high water flows from thunderstorms and snowmelt and deposited in soil along the banks of downstream creeks (referred to as overbank deposits/soil) including Panther Creek and its tributaries (Hydrometrics 2006).

Several actions have been conducted and are ongoing at the Blackbird Mine site to address the release of contaminants and the effects to natural resources that resulted from the releases. Response actions at the site have included emergency actions to address imminent releases from the West Fork Tailings Impoundments; non time-critical removal actions conducted in the Bucktail Creek, Meadow/Blackbird Creek, and Panther Creek; and investigations and studies to complete the Remedial Investigations/Feasibility Study. A Record of Decision (ROD) for the site was issued in February 2003 by the EPA, and the BMSG is currently implementing the remedy outlined in the ROD under a Unilateral Administrative Order issued by the EPA. Although there have been significant improvements in surface water and sediment quality as the result of ongoing cleanup actions, streams in the area of the Blackbird Mine and the ICP continue to exceed water quality standards as the result of historic mining at the Blackbird Mine Site (Hydrometrics 2006). The ICP POO does not authorize operations that will adversely affect the Blackbird Mine Superfund Site remedial or restoration activities, or damage to any Blackbird Mine Superfund Site remedial or restoration infrastructure by the plan operator or its agents, employees, or contractors. Actions that result in adverse impacts to the Blackbird Mine Superfund Site remedial or restoration activities shall constitute non-compliance with the POO.

In evaluating current condition and potential future trends, it's also important to note that in July, 2000, a lightning-caused wildfire began in the Clear Creek subwatershed that became one of the largest wildfires in Idaho's recent history. The Clear Creek Fire covered approximately 206,379 acres in the heart of the Panther Creek watershed. The fire was considered stand replacing within the proposed ICP project area including the upper Big Flat, Big Deer Creek, and Blackbird Mine areas (IDEQ 2001). Thus the current conditions are changing as the landscape recovers from the fire. In addition, following the fire, there was a series of high intensity rain/thunderstorm events that initiated a series of debris flows and slides affecting Panther Creek.

Aquatic habitat conditions have been extensively sampled in the ICP project area. The FCC funded Aquatic Baseline studies in 2001, 2002 and 2004, as well as summarizing existing aquatic information (Kuzis 2004). In addition, the BMSG-funded aquatic surveys completed in 2002, 2003, and 2005 (Stantec 2004; Ecometrix 2006). Baseline conditions within the action area were evaluated for the Project at the watershed scale. The SCNF based its evaluation of the environmental baseline on the "matrix of pathways and indicators" (NMFS 1996). This method assesses the current condition of instream, riparian, and watershed factors that collectively provide properly functioning aquatic habitat essential for the survival and recovery of the ESA-listed species. The environmental baseline will be described in more detail for each watershed in the action area below.

2.1.2.1. Panther Creek

Panther Creek is a fifth order stream draining about 529 square miles of the Salmon River Mountains in east-central Idaho. Stream flow patterns are typical snowmelt runoff driven, with peaks in May or June and lows in fall and winter. Average annual flow at the mouth of Panther Creek is about 265 cubic feet per second (cfs) with mean monthly flows ranging from 83 to 136 cfs (IDEQ 2001). There is a main access road paralleling most of the length of Panther Creek.

Historically, Chinook salmon spawned in Panther Creek. However, the Chinook runs began to decline about 1940 and dropped following development of the Blackbird Mine. Extensive fish kills occurred in Panther Creek during March, April, and July of 1954 (USFS 1993). An annual average of 51 Chinook redds were counted in Panther Creek between 1954 and 1962 (IDFG 1965). Redd counts were discontinued after 1967 and no redds were observed during periodic field checks from 1968 to 1977 (Platts et. al. 1979). In the late 1980's and early 1990's Chinook salmon redds and juveniles were occasionally observed in the lower reaches of Panther Creek (USFS 1993), but water quality conditions from Blackbird Creek prevented migration to the upper portions of Panther Creek. Panther Creek from Blackbird Creek to Napias Creek is on the IDEQ 303(d) list of impaired surface waters for metal, copper and unknown toxicity (IDEQ 2003).

In June and July 2001 IDFG planted 1,064 adult Chinook salmon for harvest in four sites along Panther Creek. In the fall of 2001 IDFG and the Shoshone Bannock Tribe conducted spawning surveys and over 80 redds were observed (Kuzis 2004). It is likely the Chinook observed spawning were fish that had been planted in Panther Creek. Surveys completed in May 2003 found juvenile Chinook indicating there has been successful reproduction (Stantec 2004). In 2005, the IDFG conducted a helicopter redd survey in September. The survey was an index flight and not a total count of redds in the target area. Eighteen salmon redds were identified on Panther Creek in the survey, with seven located between Moyer Creek and Blackbird Creek, eight between Blackbird Creek and Big Deer Creek, and three between Big Deer and Clear Creek. The IDFG biologists noted that visibility conditions were not ideal, and that the actual number of redds present could be slightly greater than the 18 identified in the survey (Ecometrix 2006). Some of the spawning Chinook salmon observed in 2005 may have been returns from those produced by stocked adults in 2001.

The BA states that it's currently unknown if the rainbow trout in Panther Creek are actually steelhead. Historically, Panther Creek had Chinook salmon and steelhead habitat (USFS 2005). Therefore, rainbow trout in Panther Creek are assumed to be steelhead for purposes of this consultation.

Habitat is generally "functioning at unacceptable risk" in the Panther Creek watershed. Only streambank stability, floodplain connectivity, RHCAs, and the increase in drainage network matrix indicators are "functioning appropriately" in the watershed. Sediment, refugia, and peak/baseflows are "functioning at risk." All other matrix indicators were rated "functioning at unacceptable risk" for the Panther Creek watershed. Of note, Panther Creek is listed on the 303(d) list from Blackbird Creek to the confluence of Napias Creek for metals and copper contamination. Also, although there are no physical migration barriers in Panther Creek, water quality problems below Blackbird and Big Deer Creeks essentially blocked migration up and down Panther Creek in the past. Fisheries survey data and water quality monitoring results suggest that this situation is reportedly improving (USFS 2005).

2.1.2.2. Big Deer Creek

Big Deer Creek is a tributary to Panther Creek, draining an area of roughly 44 square miles. A natural cascade is located about 0.7 miles upstream from its mouth blocking upstream fish passage. The Big Deer Creek watershed is a third order stream draining Blackbird Mountain to the south and Gant Ridge to the north. South Fork Big Deer Creek and Bucktail Creek are also found within the watershed. The streams headwaters originate in the Frank Church River of No Return Wilderness. The average annual discharge in Big Deer Creek is approximately 36 cfs, with peak flows averaging 144 cfs in June and low flows averaging 11 cfs in January (IDEQ 2001). Portions of lower Big Deer Creek between Panther Creek and South Fork Big Deer were burned in the Clear Creek fire.

This watershed has been impacted by historic mining activities. Waste rock and tailings from the Blackbird Mine site drain into Bucktail Creek which discharges chemically polluted water into South Fork Big Deer Creek. Copper and iron concentrations in Big Deer Creek below the South Fork have exceeded the lethal limits for most forms of aquatic life (USFS 1993). However, ongoing clean-up efforts and remediation activities including collection and storage of contaminated water from Bucktail Creek for treatment at the Blackbird Creek drainage collection pond have significantly improved water quality conditions. The 303(d) list identifies Big Deer Creek from the confluence of South Fork Big Deer Creek to Panther Creek as water quality limited for sediment, pH, and metals. Bucktail Creek is on the 303(d) list from its source to its confluence with South Fork Big Deer Creek for metals contamination (IDEQ 2003).

There are no roads along Big Deer Creek. However, there is a tertiary access road that parallels a short section of South Fork Big Deer Creek just upstream from the confluence of Bucktail Creek, and several old mining roads cross through the headwaters of Bucktail Creek. After a slide into Bucktail Creek, three dams were constructed across Bucktail Creek. Water from the upper impoundment is pumped back through the mountain to the water treatment plant located in the headwaters of Blackbird Creek. The lower two dams were constructed to trap sediment along Bucktail Creek during the construction of the upper pump back impoundment. Sediment from these dams is cleaned out every few years (USFS 2005).

Big Deer Creek above the confluence with the South Fork has had little if any disturbance and provides suitable spawning and rearing habitat for resident bull trout, redband/rainbow, and westslope cutthroat trout. Fish population sampling in Big Deer Creek in 2001 (Kuzis 2004), 2003 (Stantec 2004), 2005 (Ecometrix 2006; USDA Forest Service 2007) found only redband/ rainbow trout or cutthroat trout. In general, there were very few fish downstream of the confluence with South Fork Big Deer Creek. The lack of fish below the South Fork reflects the ongoing impacts of chemically contaminated water. The 2005 USFS survey crew sampled above and below the Big Deer cascade and found westslope cutthroat trout above the falls, and rainbow/steelhead trout below the falls (USDA Forest Service 2007). Big Deer Creek is not considered spawning habitat for Chinook salmon due to the steep cascade/falls located 0.7 miles upstream from the mouth of Big Deer Creek and degraded water quality associated with the Blackbird mining activities (USFS 2005). It is not known if steelhead spawn in Big Deer Creek. However, due to the presence of rainbow/ steelhead, NMFS assumes that both Chinook salmon and steelhead can and do rear in this lower 0.7 miles.

Habitat is generally “functioning at unacceptable risk” in the Big Deer Creek watershed. Only physical barriers, streambank stability, floodplain connectivity, large woody debris (LWD), RHCAs, and the increase in drainage network matrix indicators are “functioning appropriately” in the watershed. Sediment and peak/baseflows are “functioning at risk.” All other matrix indicators were rated “functioning at unacceptable risk” for the Big Deer Creek watershed. The IDEQ 303 (d) listed segments include: (1) Big Deer Creek, from the confluence of the South

Fork downstream to Panther Creek for sediment, pH, and metals; (2) South Fork Big Deer Creek from its confluence with Bucktail Creek to Big Deer Creek for metals contamination; and (3) Bucktail Creek from its source to its confluence with the South Fork for metals contamination (IDEQ 2003).

2.1.2.3. Little Deer Creek

Little Deer Creek is a second order stream draining the ICP site which flows to the northeast into Panther Creek. The watershed has a drainage area of 6.2 square miles. There are no 303(d) listed stream segments for Little Deer Creek (IDEQ 2003). The entire upper portion of the Little Deer drainage was severely burned in the Clear Creek fire. As a result, during a storm event in October 2000, the Little Deer Creek channel destabilized and unraveled. Most of the streambank vegetation was burned in the fire and it appeared that woody debris jams in the channel had come apart. There were long stretches of deep (>5 foot) downcuts and areas of overland mud flows where the channel spread out and there was no distinct thalweg. In the 2004 sampling effort there had been significant regrowth of riparian vegetation and clear channels were becoming established (Kuzis 2004).

There are no roads or other development in the Little Deer drainage. The proposed ICP project would add less than 1/2 mile of new roads in the headwaters of Little Deer Creek. Activities proposed in this drainage would be located far from any channels and riparian areas in the watershed. Juvenile Chinook were collected in the lowest reaches of Little Deer Creek in 2002 (USFS 2005).

Habitat is generally “functioning at risk” in the watershed. Although sediment, physical barriers, floodplain connectivity, increase in drainage network, and RHCAs are “functioning appropriately,” all other indicators are either “functioning at risk” (LWD, peak/baseflows, road density/location, disturbance regime) or “not properly functioning” (all others).

2.1.2.4. Big Flat Creek

Big Flat Creek is a 3-mile long, second order tributary to Panther Creek, and the primary drainage from the ICP site. Big Flat Creek has a drainage area of 1.6 square miles, and an average stream gradient of 21%. Approximately a quarter of the total length of Big Flat Creek has a gradient of 30% or higher. In addition, over 80% of Big Flat Creek is a highly confined valley with steep valley walls.

Much of the Big Flat drainage was burned severely in the 2000 Clear Creek fire and almost all riparian vegetation was burned. There has been substantial recovery of understory shrubs as noted in the 2004 Macroinvertebrate Sampling report.

Big Flat Creek has one road crossing, located in the headwaters and within the project area. About 2,000 feet below the road crossing, near upper Big Flat Creek, the channel is covered with loose rock and scree from the canyon walls. Flows are almost entirely subsurface at this location and, in some places, flowing water can be heard under the rocks for significant stretches (Kuzis 2005). Because of the subsurface flows, Big Flat Creek contains no fish habitat, and no matrix of pathways and indicators was created for this drainage. There are no stream segments for Big Flat Creek listed on the IDEQ 303(d) list.

2.1.2.5. Blackbird Creek

Blackbird Creek is a second order tributary to Panther Creek. Mean annual flow in Blackbird Creek is 12 cfs, with mean monthly discharge ranging from 4 cfs to 48 cfs (IDEQ 2001). Blackbird Creek is on the IDEQ 303(d) list from Blackbird Reservoir to its confluence with Panther Creek for sediment, pH, and metals contamination (IDEQ 2001).

The water quality in Blackbird Creek is impaired due to Blackbird Mine which was a major supplier of cobalt during World War II. Operations at the mine ceased in 1982 and the site is now undergoing regulated cleanup. The site is divided by a ridge into two drainage basins: the Big Deer Creek basin to the north, and the Blackbird Creek basin to the south (including Meadow, West Fork Blackbird, and Blackbird Creeks). Disturbance due to historic mining is spread over approximately 830 acres of primarily private patented mining claims but includes areas of unpatented claims on National Forest. The Blackbird Creek watershed has the highest density of roads in the entire Panther Creek drainage.

The area is undergoing a remediation cleanup, with the EPA as the lead agency, that includes removal of mill facilities, expansion of a water treatment facility, capping of waste rock, and removal of tailings from along streambanks and impoundments. Cleanup activities are still occurring and agreements between the agencies and companies are ongoing to meet cleanup goals. The majority of the activities for the cleanup have occurred on patented lands (784 acres).

Only the lower 2 miles of Blackbird Creek have suitable gradients for steelhead and Chinook salmon spawning and rearing. The habitat conditions in Blackbird Creek have historically been poor, due to chemical pollution from leeching, streambank degradation, low numbers of pools, low amounts of LWD, and high stream temperatures (USFS 1993). However, surveys completed in 2003 found juvenile Chinook salmon and bull trout in the lower 100 yards of Blackbird Creek indicating that conditions have improved significantly (Stantec 2004).

Habitat is generally "functioning at unacceptable risk" in the Blacktail Creek watershed. Only the increase in drainage network pathway is considered to be "functioning appropriately" in the watershed. The LWD, refugia, width to depth ratio, floodplain connectivity, and change in peak/base flow indicators are "functioning at risk" in the watershed. All other matrix indicators were rated "functioning at unacceptable risk" for the Blacktail Creek watershed.

2.1.2.6. Williams Creek

Williams Creek is a tributary to the Salmon River, with a mean annual flow of 10 cfs, and a range of mean monthly flows from 40 cfs (June) to 3 cfs. During the irrigation season the lower reaches of the creek may be dewatered although in some years there may be sufficient flow for fish passage (IDEQ 2001). Williams Creek is 16.1 miles long with 4 miles of stream at less than 4% gradient (which are mostly private lands below the Forest boundary), 8.5 miles of stream at gradients between 4% and 10%, and 3.6 miles of gradient greater than 10%.

Forty percent of these stream miles were historically accessible to anadromous fish, including all portions below the Forest boundary and two additional miles on Forest lands. There is no information on the current extent of Chinook use. However, "natural" adult steelhead have been observed in the lower reaches of Williams Creek since the IDFG began planting hatchery outmigrants into the Salmon River at Shoup Bridge.

Habitat is generally "functioning at unacceptable risk" in the Williams Creek watershed, although habitat is generally in worse condition in the lower private reaches than on SCNF lands. Only the sediment, chemical contamination/nutrients, streambank condition, and floodplain connectivity pathways are "functioning appropriately" in the watershed. The temperature, refugia, change in peak/base flow, and increase in drainage network indicators are "functioning at risk." All other matrix indicators were either trending toward or rated as "functioning at unacceptable risk" for the Williams Creek watershed.

2.1.2.7. Moccasin Creek

Moccasin Creek is a first order tributary to Napias Creek. Moccasin Creek is approximately 7 miles long with an average gradient of 15%. The main access road to the ICP project parallels about 2.5 miles of Moccasin Creek where the gradient averages about 8%. The road confines the creek in a narrow valley. Because the valley is so narrow, the roadfill functions as one of the banks of this small creek. During low flow years, portions of the creek flow subsurface, which has restricted the collection of sediment and water temperature data by the USFS (Kuzis 2004).

Bull trout were the only fish species sampled in Moccasin Creek during 2001 electrofishing surveys (Kuzis 2004). A falls in Napias Creek, located about 0.5 miles from the mouth, has been identified as a complete barrier to upstream distribution for anadromous fish, blocking access to Moccasin Creek.

2.1.2.8. Deep Creek

Deep Creek is a major tributary in the Panther Creek drainage. Mean annual flow in Deep Creek is 20 cfs, with a mean monthly maximum flow at 80 cfs and a minimum flow at 6 cfs (IDEQ 2001). Deep Creek is part of the Deep-Moyer watershed that has been designated in whole or part as a priority watershed in PACFISH for anadromous fish recovery during ESA section 7 consultation for amending the SCNF Forest Plan.

Deep Creek headwaters are fairly steep and transition into deeply cut canyons. Approximately 53% of the length of the 24.4 mile drainage has gradients between 4% and 10%, 43% of the drainage length has gradients less than 4%, and gradients in the remaining 3% exceeding 10%.

The ICP access road parallels the lowest 4.2 miles of Deep Creek. The channel morphology through the project location can be best described as a Rosgen type B2a. The channel is steep and moderately to deeply entrenched. Deep Creek is incised in predominantly small boulder/cobble bed material with lesser amounts of small cobble and gravel materials present. This channel type is a high energy and low sediment supply stream type, with corresponding low bedload transport rates. The channel bed and streambanks are normally stable and contribute little to sediment supply.

Deep Creek supports significant fisheries resources and has been identified as a historic producer of both Chinook salmon and steelhead (NPPC 1991). While numbers of Chinook salmon and steelhead spawners have declined dramatically within the Panther Creek drainage from historical numbers and no Chinook salmon have been observed in Deep Creek in recent years, the stream is still considered a potential anadromous fish production tributary of the Panther Creek system. Deep Creek, to a point near the mouth of Little Deep Creek, is thought to support habitat for steelhead/rainbow trout and Chinook salmon. Currently Deep Creek supports rainbow and/or steelhead trout. No Chinook were observed in lower Deep Creek following IDFG outplants in 2001. Adult steelhead have not been observed spawning in the watershed. Introduced brook trout (*Salvelinus fontinalis*) have been identified in the lowermost reaches of Deep Creek in recent years.

Habitat is generally "functioning at risk" in the Deep Creek watershed. Although temperature, sediment, chemical contamination/nutrients, physical barriers, LWD, width to depth ratio, streambank condition, change in peak/base flows, increase in drainage network, and RHCAs are "functioning appropriately," all other indicators are either "functioning at unacceptable risk" (pool frequency/quality) or "functioning at risk" (all others).

2.1.3. Effects of the Action

"Effects of the action" means the direct and indirect effects of an action on the listed species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Effects of the action that reduce the ability of a listed species to meet its biological requirements may increase the likelihood that the proposed action will result in jeopardy to that listed species or in destruction or adverse modification of a designated critical habitat.

The potential for mine-related adverse effects on ESA-listed salmon, steelhead, and their critical habitat, varies depending upon a variety of factors. The potential for effects varies based upon site-specific features such as topography, location of the ore bodies, the mining method, toxicity of minerals present, chemicals used, mine management and engineering, haul road location and design, control and disposal of mine wastes, reclamation methods, monitoring effectiveness, proximity of critical habitat, and life stages of fish that may be affected.

The BA provides a detailed analysis of the effects of the proposed action on Snake River spring/summer Chinook salmon and Snake River Basin steelhead. The analysis in the BA uses

NMFS' Matrix of Pathways and Indicators (NMFS 1996) and the information in the BA to evaluate elements of the proposed action that have the potential to affect the listed fish or essential habitat features of their critical habitat.

The potential adverse effects of the proposed action on ESA-listed salmon and steelhead and their critical habitat can be broadly categorized into the following:

1. Fine sediment delivery to critical habitat during and after mine operation;

2. Accidental spill or discharge of toxic substances during and after mine operation;

3. Water quality and quantity issues during mine operation; and,

4. Effects to riparian vegetation in the project area.

2.1.3.1. Effects on Listed Species

Construction activities associated with installation of the NPDES Outfall along Big Deer Creek and the cable car crossing in Panther Creek are likely to mobilize sediments and temporarily increase downstream turbidity levels. Road construction, reconstruction, use, and maintenance also have the potential to affect water quality by increasing sediment delivery to streams along the transportation corridor. Although facility construction and reclamation will also generate sediment, the relatively flat nature of the construction site, application of erosion control practices, and the facilities' distance from streams occupied by ESA-listed salmon and steelhead (over 2 miles) are expected reduce the likelihood of effects from facility construction and reclamation to negligible levels.

Water quality could also be affected through increases in stream temperature and/or through chemical contamination. Clearing of riparian vegetation could result in stream temperature increases. Chemical contamination may occur any time construction equipment is working within or adjacent to the stream channel, should an accidental spill occur along the transportation corridor, or as a result of effluent discharge.

Suspended Sediment/Sediment Deposition. The most critical aspects of sediment effects are related to timing, duration, intensity, and frequency of exposure (Bash *et al.* 2001). Depending on the level of these parameters, turbidity can cause lethal, sublethal, and behavioral effects in juvenile and adult salmonids (Newcombe and Jensen 1996). For salmonids, turbidity has been linked to a number of behavioral and physiological responses (i.e., gill flaring, coughing, avoidance, and increase in blood sugar levels) which indicate some level of stress (Bisson and Bilby 1982; Berg and Northcote 1985; Servizi and Martens 1987). The magnitude of these stress responses is generally higher when turbidity is increased and particle size decreased (Bisson and Bilby 1982; Servizi and Martens 1987; Gregory and Northcote 1993). Although turbidity may cause stress, it has been shown that moderate levels of turbidity (35 to 150 nephelometric turbidity units) accelerate foraging rates among juvenile Chinook salmon, likely because of reduced vulnerability to predators (camouflaging effect). Turbidity and fine sediments can reduce prey detection, alter trophic levels, reduce substrate oxygen, smother redds, and damage gills, among other deleterious effects (Bjornn 1991; Spence *et al.* 1996).

Quantifying turbidity levels and their effect on fish species and their habitat is complicated by several factors. First, turbidity from an activity will typically decrease as distance from the activity increases. The time needed to attenuate these levels depends on the quantity of material in suspension (e.g., mass or volume), particle size, the amount and velocity of ambient water (dilution factor), and the physical/chemical properties of the sediments. Second, the impact of turbidity on fish is not only related to the turbidity levels but also to the particle size of the suspended sediments.

Cable Car and Outfall Construction - Turbidity is likely to exceed ambient levels and potentially affect ESA-listed fish species present during construction of the cable car crossing on Panther Creek and during construction of the NPDES outfall on Big Deer Creek. For the cable car crossing, a backhoe or excavator would be required to cross Panther Creek in order to install concrete for the deadman. The crossing would be installed in early spring prior to high flow. Equipment crossings would deliver two separate, temporary pulses (minutes) of sediment as the equipment crosses the creek. Machinery will not operate from the channel and no construction will occur in the active channel.

At the cable car crossing, turbidity is expected to be short-lived and highly localized due to the short work window, construction occurring outside the active channel, and low flow conditions during the construction period. The Project includes measures to reduce or avoid sediment delivery to streams, including using typical state and Federal erosion control measures. Both species are present in Panther Creek at and downstream from the construction area, but the temporary nature and small size of the sediment plumes should not result in prolonged exposure to listed salmonids. It is highly unlikely that sediment plumes will extend across the wetted stream width from an equipment stream crossing, which will allow fish to avoid exposure to undesirable turbidity levels by swimming to adjacent, less turbid habitat (i.e., behavioral response, similar to what might occur under natural high flow events). Based on sediment generating projects completed by the Nez Perce National Forest and others, sediment suspended as a result of this phase of the Project is not expected to extend any further than 300 feet downstream from this disturbance.

Construction details were not well described in the BA for installation of the effluent pipe. In Big Deer Creek, a backhoe/excavator will excavate a trench to a depth of 1- to 4-feet, extending across approximately 50% of the channel width. Although not specified in the proposed action, trenching will likely occur from the streambank, but trenching itself will occur within the active channel of Big Deer Creek. In addition, approximately 1 cubic yard of streambank will need to be removed to place the pipeline. Based on the small disturbance area and the short duration for this phase of the project, sediment-related effects are expected to be short-term, localized, and of reduced severity. Although any sediment generated is expected to settle within about 300 feet, and well before habitat occupied by anadromous fish (approximately 2.3 miles downstream). This sediment would later be mobilized at high flows and could potentially affect ESA-listed fish or degrade instream habitat conditions. Because Big Deer Creek is currently on the 303(d) list for sediment, additional BMPs are warranted to ensure that sediment generated as a result of the diffuser installation is properly minimized.

Road Construction/Reconstruction – Construction of a road network can greatly accelerate erosion rates in a watershed (Haupt 1959; Swanson and Dyrness 1975; Swanston and Swanson 1976; Beschta 1978; Gardner 1979; Cederholm and Reid 1987). Sediment generated through road construction and reconstruction can reach streams through surface erosion and mass movements of destabilized soil, the effects of which can be both dramatic and long-lasting (Meehan 1991). Unpaved road surfaces continually erode fine sediments, adding significant amounts of sediment to streams (Reid and Dunne 1984; Swanston 1991). Roads and related ditch networks are often connected to streams, providing a direct conduit for sediment. On steep slopes, road construction or improper maintenance can greatly increase landslide rates relative to undisturbed forest (Swanson and Dryness 1975; Swanston and Swanson 1976; Furniss et al. 1991), delivering large pulses of sediment to streams.

Increases in sediment supply beyond the transport capability of the stream can cause channel instability, aggradation (sometimes to the extent that perennial streams become intermittent) (Cederholm and Reid 1987), widening, loss of pools, and a reduction in gravel quality (Sullivan et al. 1987; Furniss 1991; Swanston 1991). For salmon, these changes can mean reduced spawning and rearing success when spawning areas are covered, eggs and fry suffocate or are trapped in redds, food abundance is reduced, and over-wintering habitat is reduced (Cederholm and Reid 1987; Hicks et al. 1991).

Roads built in riparian areas often eliminate part of the riparian vegetation (Furniss 1991), reducing large wood recruitment and shade. Riparian roads also constrain the natural migration of the stream channel where channel migration zones are present. Roads can intercept, divert, and concentrate surface and subsurface water flows, thereby increasing the watershed's drainage network (Hauge et al. 1979; Furniss et al. 1991; Wemple et al. 1996). This can change peak and base stream flows and increase landslide rates. Stream crossings can restrict channel geometry and prevent or interfere with migration of adult and juvenile anadromous fish (Furniss et al. 1991). Culverts also can be a source of sedimentation, especially if they fail or become plugged with debris (Furniss et al. 1991; Murphy 1995).

The ICP will require construction of 4.1 miles of new road and reconstruction/improvement of 11.3 miles of existing road (Appendix A, Figure 1). The Project will also decommission 7.5 miles of current site roads in order to achieve a net reduction of roads in the Big Flat and Bucktail Creek drainages. New road construction will occur in the vicinity of the portals, the mill, and the TWSF. New road segments will be located in the headwaters of the Big Flat and Bucktail Creek drainages, over 2 miles upstream from stream reaches occupied by listed anadromous fish. All roads will be constructed and improved in accordance with USFS guidelines for road construction. Stormwater ditches and sediment control measures will be constructed in accordance with IDL's BMPs for Mining in Idaho,[3] which include but are not limited to rolling dips, road sloping, use of straw bales, sediment traps, silt fences, straw mulch, broadcast seeding, etc. The flat nature of this area, construction to USFS guidelines, IDL BMPs,

[3] http://www.idl.idaho.gov/bureau/Minerals/bmp_manual1992/bmp_index.htm

and the location of the new roads over 2 miles away from occupied habitat make the likelihood of effects to anadromous fish from these new road segments unlikely. In addition, the obliteration and reforestation of the 7.5 miles of existing road should reduce the potential for long-term effects, while the obliteration of the 4.1 miles of new road upon completion of the project should help ensure that any adverse effects would not be long-term.

Roads along the transportation route will be reconstructed to improve safety and to reduce sediment delivery to the streams. The majority of roads along the transportation route parallel or cross fish-bearing streams, or drain directly into streams occupied by fish. Based on known distribution, effects from road-generated sediment on ESA-listed salmon and steelhead are most likely to occur in lower Deep Creek, Blackbird Creek, Panther Creek (between Deep Creek and Blackbird Creek), and lower Williams Creek (steelhead only). Although sediment inputs are expected to increase substantially during periods of road reconstruction, efforts to resurface the road, improve drainage, and elevate the road above the floodplain should improve effects from the haul road over its existing condition.

Two culverts will be removed from the Williams Creek Road during the road reconstruction phase. The existing culverts on North Fork Williams Creek will be removed and the channel restored as part of the reconstruction effort. No effects to steelhead are expected to occur from the culvert removal as the crossings are located high in the watershed, in a high gradient stream, several miles upstream from any documented distribution of steelhead in Williams Creek mainstem. However, specific details about the final design of the Williams Creek road and culvert removal have yet to be finalized and were therefore not included in the BA. Because this will potentially involve inwater work, NMFS will need to review the timing and engineering design for culvert replacements prior to project implementation to ensure that sufficient BMPs are established to ensure this phase of the project will not result in effects not considered in this Opinion.

The USFS used the Water Erosion Prediction Project (WEPP) model (Flanagan and Livingston 1995) to evaluate sediment related effects for the existing condition and the operational period from project hillslopes and site roads in the Bucktail and Big Flat Creek drainages, and from the access road between Williams Creek and Blackbird Creek. As described in the BA (USDA Forest Service 2007), sediment modeling of the planned road upgrades showed that sediment production should actually decrease along the transportation route. The model predicted that improvements to the transportation route from Williams Creek to the gate at Blackbird Creek would decrease sediment leaving the road by approximately 50% compared to existing conditions. From the gate up to the project site, sediment is predicted to decrease 82% over existing conditions. The decommissioning of the existing roads at the project site would also result in a net decrease in road density in the Big Flat and Bucktail Creek drainages. Considering the results of the model, the USFS concluded that these actions would actually result in long-term improvements to instream substrate conditions.

Road construction and reconstruction activities will increase the potential for turbidity and surface runoff. However, NMFS believes that the upgrades proposed and the conservation measures described should be adequate to keep turbidity and erosion to levels that should not cause lethal take during construction or reconstruction efforts. Turbidity levels may be high enough to cause indirect, non-lethal take of juvenile steelhead and/or Chinook salmon from short-term harassment, feeding disruptions, and/or avoidance of the area. Based on the modeling and effects described above, it is reasonably likely that the proposed action will have very small, short-term negative effects caused by increased sediment yield, but should result in long-term positive benefits to the survival and recovery of ESA-listed species by reducing overall road density and reducing chronic sediment delivery.

Road Use and Maintenance – Road maintenance can have both short-term and long-term effects on Snake River steelhead and spring/summer Chinook salmon. Surface erosion from forest roads affect the fine sediment budget in streams and may impose a chronic condition of sediment inputs that directly affect the stream substrate and the health of aquatic life (Luce et al. 2001). Planned activities such as placement of cross drains, ditching, grading and graveling may result in disturbances that typically create short-term increases in sediment delivery that taper off after disturbed areas become compacted or after several runoff events occur. Maintenance can also correct problems with the road surface and drainage and thereby reduce levels of sediment delivery from an existing road (baseline condition). Beneficial effects of maintenance typically persist for one or more seasons, depending on a variety of factors such as amount of traffic, precipitation, and physical properties of the road surface. Benefits of proper maintenance include minimization of erosion or sediment delivery from ditches and road surfaces; however, improper maintenance can exacerbate erosion or sediment delivery to streams. Placement of dips (road drains) if installed properly, will reduce ruts and gullies along the roadbed and direct water flow and sediment away from streams. In contrast, removal of the material deposited at the base of the road-cut during maintenance operations interferes with the natural slope-forming process, removes a favorable site for vegetation growth, and initiates slope erosion processes (Megahan and Kidd 1972). Road maintenance also includes removal of roadside vegetation and can impair stream functions by decreasing shade and reducing recruitment of LWD along streamside roads.

Traffic will increase exponentially once construction begins and will continue through closure and reclamation. Daily traffic will include up to 10 vans and four pickup trucks to transport employees to and from the site. In addition, transportation of concentrate, equipment, reagents, and other freight will result in an additional estimated 1,084 round trips per year (Table 1). Without proper road maintenance, a traffic increase of this magnitude would certainly increase surface erosion beyond baseline levels.

Road maintenance for the ICP will be conducted in accordance with SCNF road maintenance procedures. Guidelines for road maintenance on the SCNF were independently consulted upon, receiving concurrence with a "not likely to adversely affect" determination in 2003 (I/NWR/2003/00526), and following amendment in 2007 (I/NWR/2007/01352). The SCNF Road Maintenance Programmatic was designed to limit the effect of road maintenance activities on ESA-listed species and correct "the legacy of landscape conditions that have accumulated over time." The programmatic outlines protocols for: (1) Surface blading and replacement; (2) blasting; (3) ditch cleaning; (4) flood damage repair; (5) vegetation clearing; (6) sign, bridge, drainage culvert and cattle guard maintenance; (7) culvert replacement; (8) dust abatement; and (9) snow plowing.

The road maintenance programmatic includes measures designed to minimize sediment production and delivery, to maintain or improve road drainage, to avoid the introduction of dust abatement chemicals to streams, and to keep fuel storage areas outside of RHCAs. The programmatic further directs that authorized road maintenance by non-federal entities on public lands will meet the federal management agencies' standards and guidelines. NMFS expects that application of the general design criteria from the roads programmatic currently in place on the SCNF will effectively minimize the risk of adverse effects occurring from road use, maintenance, and deterioration on aquatic habitats in the action area.

Toxic Spills/Discharge. Water quality could be further affected through chemical contamination. Chemical contamination could occur: (1) when construction equipment is working within or adjacent to the stream channel; (2) should a spill occur in the transportation corridor; (3) as a result of effluent discharge; or (4) through groundwater contamination.

Cable Car and Outfall Construction - Heavy machinery operations adjacent to streams or in wetlands raises concern for the potential of an accidental spill of fuel, lubricants, hydraulic fluid or similar contaminant into the riparian zone, or directly into the water where they could adversely affect habitat, injure or kill aquatic invertebrates, or directly impact ESA-listed species. Petroleum-based contaminants such as fuel, oil, and some hydraulic fluids, contain polycyclic aromatic hydrocarbons, which can cause chronic sublethal effects to aquatic organisms (Neff 1985). Ethylene glycol (the primary ingredient in antifreeze) has been shown to result in sublethal effects to rainbow trout at concentrations of 20,400 mg/L (Staples 2001). Brake fluid is also a mixture of glycols and glycol ethers, and has about the same toxicity as antifreeze.

As previously described, work will occur in the active channel for both the cable car and discharge pipe installation, increasing the potential for this form of chemical contamination to occur. However, it's unclear in the proposed action if equipment used on this project will be clean and free of fuel and lubricant leaks, or whether the equipment will be inspected prior to beginning and during this work. It's also unclear what provisions will be in place regarding the fueling and maintenance of equipment in RHCAs. NMFS believes that additional fuel spill and equipment leak contingencies and preventions will be necessary to effectively minimize the risk of negative impacts to ESA-listed fish and fish habitat from toxic contamination during this phase of the operation. However, because of the short duration of each project and the need for only one piece of equipment at each site, it's highly unlikely that antifreeze, brake, or transmission fluid, will be present on-site or spilled in volumes or concentrations large enough to harm salmonids in or downstream from project sites.

Accidental Spills - Toxic spills have the potential to enter streams in the action area either through a spill occurring during transport, or through an on-site spill and subsequent discharge into nearby waterbodies. An accidental spill of fuels or toxic chemicals being transported to or on-site at the ICP could adversely affect ESA-listed salmon and steelhead if the spill reached any of the action area streams. The potential for and magnitude of those effects would be dependent upon a number of variables, such as: (1) proximity to streams; (2) whether the spill reached a stream; (3) accident severity; (4) amount of material spilled; (5) volume and attributes of receiving waters at the time of the spill; (6) type of chemical; (7) form of chemical (dry or liquid); (8) transportation container; (9) weather; (10) spill response time; (11) effectiveness of spill containment; and (12) salmonid life stage(s) present and exposed. In addition to considering each of these variables in the BA, the potential for spill for each chemical was analyzed in relation to the number of trips per year each chemical would be transported to the site.

As bulk liquids, the USFS determined that diesel, gasoline, and antiscalant, presented the highest risk of release should an accident occur (Table 7). As containerized liquids in containers <100 gallons, Aerofroth 65 Frother, polymer flocculent, methanol, hydrochloric acid, oils, lubricants, grease, and antifreeze presented a moderate risk of release should an accident occur.

The transportation route plans to use existing USFS roads along Blackbird, Panther, Deep, Moccasin, and Williams Creeks. The route is approximately 40 miles long, and crosses Perreau Creek near its mouth and the Salmon River at Shoup Bridge. Although not all currently occupied, all streams along the transportation route, except Moccasin Creek, historically provided habitat for Snake River spring/summer Chinook salmon and steelhead. Except for Moccasin Creek, steelhead could be affected by spills into any of these stream systems. Chinook would most likely be affected by spills alongside the Salmon River, Panther Creek, or Blackbird Creek, but could also be affected by large spills or spills near the mouths of the smaller tributaries.

The likelihood of a spill occurring along U.S. Highway 93 and entering the Salmon River has been estimated to be relatively small (USDA 2007). Similarly, spills into Williams or Perreau Creeks are not expected to result in hazardous conditions in the mainstem Salmon River. Although the potential for direct effects to ESA-listed salmon and steelhead from a transportation spill exists at the Shoup Bridge, the bridge has good approach visibility and is sufficiently wide that the probability of a spill is small. The distance traveled from Salmon along the highway is relatively short. Although the distance from Challis is longer, the highway road is in good condition and is relatively safe to drive along. NMFS considers the risk of adverse effects from spills occurring along U.S. Highway 93 to be negligible.

The Beartrack Mine, which utilized the same transportation route as proposed for the ICP, had no accidents associated with the transportation of hazardous materials to the mine site. There were two incidents with hazardous materials transport. The first incident involved a fuel truck that pulled off the road at Mile Post 12 onto a false shoulder of snow to avoid an oncoming vehicle that did not yield to the pilot car. The fuel truck did not tip over, but approximately 4 gallons of fuel were spilled from the truck’s fuel tank. This material was cleaned up with absorbent pads and removed to an approved disposal site. The second incident involved a Lime truck at Mile Post 8. While negotiating a switchback the right rear wheels of the truck went off the road shoulder above the culvert. There was no vehicle turnover and no material spilled. As a result of this incident additional barriers were installed on the inside of this curve above the culvert and the curve was widened.

Table 7. Risk of a material release in case of an accident during transport to the ICP project.

Material	Amount Transported per Trip	# Trips per Year	Form	Container Type/Size	Risk of Release in Event of Spill[1]
AERO 343 Xanthate	40,000 lbs	14	Dry	Flo Bin/ 1 ton	Low
AERO 350 Xanthate	40,000 lbs	16	Dry	Flo Bin/ 1 ton	Low
AEROFROTH 65 Frother	28,000 lbs	3	Liquid	Plastic Barrel/ 55 gal	Moderate
Sodium Sulfide	20-100 tons	5	Dry	Sacks/ 50 lbs	Low
Superfloc	8	3	Dry	Sacks 50 lbs	Low
Diesel	4,470 gal	167	Liquid	Fuel Truck/ 4,500 gal.	High
Gasoline	30 gal	150	Liquid	Fuel Truck/4,500 gal	High
Lime	37,500 lbs	4	Dry	1000 Super Sack	Low
Cement	4,4000	115	Dry	Dry Bulk 22 tons	Low
Oils, Lubricants, Grease, Antifreeze	60 gal	150	Liquid	Barrel/ 55 gal	Moderate
Propane	9,400 gal	5	Gas	Fuel Truck/ 9,400 gal	Low
Antiscalant	4,000 gal	1	Liquid	250 gal. tote/ 4,000 gal	High
Ammonium Nitrate	10 tons	40	Dry	Bulk Container/ 10 tons	Low
Bulk Concentrate	16 tons	700	Dry	Sealed Container/ 16 tons	Low
Water Treatment Chemicals & Reagents					
Polymer Flocculent	20 gal	1	Liquid	Sealed Pail/ 5 gal	Moderate
Hydrated Lime	2 tons	1	Dry	Super Sack/ 1000 lbs	Low
Methanol	500 gal	2	Liquid	Plastic Drum/ 55 gal	Moderate
Hydrochloric Acid	200 gal	2	Liquid	Plastic Drum/ 55 gal	Moderate
Zeolites	3 tons	1	Dry	Super Sack	Low

[1]Containerized Solid= Low Risk, Containerized Liquids=Mod Risk, Bulk Liquids = High Risk

Considering the number of times a chemical would be transported to the mill site each year, the USFS used annual truck miles and accident frequency to further refine the likelihood that each chemical would be involved in an accident and potentially spilled during the lifetime the mine. Estimated accident frequencies along the access routes were based on a general figure of 9.8 accidents per million miles traveled on Forest Service roads (USFS 1997). This figure is based on historic accident records for general traffic conditions without special accident control or mitigative measures (e.g., use of pilot cars). An estimated 1,083 trips per year of hazardous materials is equivalent to 43,320 miles per year for transport of hazardous substances. Of those miles, 18,628 miles (43%) are in close proximity to a stream.

The analysis revealed that only seven of the 18 different materials transported to the mine would be expected to have an accident occurring near a stream in less than 1,000 years. Only three of these materials were expected to have a risk of occurring once in less than 100 years. Spills of cement, bulk concentrate, and diesel, gasoline, oil, lubricant, grease, and antifreeze[4] were considered the most likely to occur. Table 8 identifies the seven materials determined most likely to be spilled during the life of the project.

[4] Diesel, gasoline, and other oils will be shipped on the same trucks. Gasoline, oils, etc. will be in small quantities.

The USFS conducted a literature review for toxicity data for each material being transported to the site. Review included EPA's Ecotox Database (http://cfpub.epa.gov/ecotox/), the Pesticide Action Network Database (http://www.pesticideinfo.org/Index.html), the Material Safety Data Sheets (MSDS) from the chemical manufacturer, and other refereed literature. Considering the spill response plan in place, exposure was assumed to be short-term (i.e., lasting a few hours at most, for sub-acute to acute durations), making their use of the 96-hour LC_{50} a conservative approach to gauging toxicity.

Table 8. Transported materials with a risk of accidents near stream of less than once in one thousand years.

Material	Amount Transported per Trip	Number of Trips per Year	Annual Truck Miles	Annual Accident Frequency	Years Between Accident	Accident Frequency Near Stream	Years Between Accidents Near Streams
Bulk Concentrate	16 tons	700	26,705	0.2617	4	0.1125	9
Diesel Gasoline Oils, Lubricant, Grease, Antifreeze[1]	4,470 gal 30 gal 60 gal	150	5,723	0.0561	18	0.0241	41
Cement	44,000	115	4,387	0.0430	23	0.0185	54
Ammonium Nitrate	10 tons	40	1,526	0.0150	67	0.0064	156
AERO 350 Xanthate	40,000 lbs	14	534	0.0052	191	0.0023	444
AERO 343 Xanthate	40,000 lbs	16	610	0.0060	167	0.0026	389

[1] Diesel, gasoline, oil, lubricant, grease, and antifreeze will be shipped on the same trucks. Gasoline, oils, etc. will be in small quantities.

Spill effects analyses in the BA were conducted specifically for, gasoline, diesel, and antiscalant, which were classified as high-risk materials. A spill analysis was also conducted for cement, ammonium nitrate, and bulk concentrate since they are associated with higher accident probabilities; and for sodium sulfide, and SUPERFLOC^{TM}, because of their higher toxicities. The effects of the remaining transported materials, including AERO 343 and 350 xanthate, AEROFROTH 65 Frother, lime, propane, polymer flocculant, hydrated lime, methanol, zeolites, and hydrochloric acid were not analyzed because of their low accident probabilities, and because none were categorized as being highly toxic to fish or as having a high spill risk (Table 9). NMFS believes that the approach used by the USFS in the BA is conservative one, and has therefore followed a similar approach for this Opinion.

Table 9. Materials selected for toxic effects analysis.

Material	Years between Accidents near Streams	Spill Risk	Toxicity
Gasoline	40	High	moderately toxic
Diesel	40	High	slightly toxic
Antiscalant	1,244	High	not acutely toxic
Bulk Concentrate	9	Low	insoluble – not toxic
Cement	54	Low	slightly toxic (pH)
Ammonium Nitrate	156	Low	moderately toxic
Sodium Sulfide	1,244	Low	highly or moderately toxic depending on source
SUPERFLOC™	2,073	Low	highly toxic

A detailed summary of the potential effects of each transported material is presented in the BA (USDA Forest Service 2007) for listed salmon and steelhead. Toxicity levels are specified for each material, as are the quantities needed to reach toxic levels in receiving waters of Williams Creek, the Salmon River below Williams Creek, Deep Creek, Blackbird Creek, Panther Creek below Blackbird Creek, and in the Salmon River below Panther Creek. Several of the transported materials would require relatively small amounts to be spilled into the receiving waters to reach toxic levels. SUPERFLOC™ would require the least amount of material to be spilled to achieve toxic conditions (Table 10).

The actual probabilities for spills of any transported material may be lower than predicted in the spill risk analysis due to mitigation (safety) measures proposed for transport of hazardous materials and fuels. The proposed action includes notification of the ICP facility prior to transport of fuels or chemicals, travel only during daylight hours, use of single frame trucks, use of pilot vehicles, and continuous radio contact between the pilot vehicle and the facility during transport. Pilot cars would lead tractor-trailer loads and fuel trucks to the site to reduce accident risk. All vehicles transporting hazardous materials will be accompanied from U.S. Highway 93 to the mine site by a pilot vehicle equipped with a two-way radio to warn both the transport vehicles and oncoming traffic. The pilot vehicle will also carry a spill response kit designed to contain spills. Spill boxes will also be located at various points along the route where the road runs adjacent to streams. Compliance with precautionary measures to reduce the probability of an accidental spill of hazardous material during transport or storage at the project site listed in the Spill Prevention and Response Plan (FCC 2002) should both minimize the potential for and the severity of adverse affects on Snake River spring/summer Chinook salmon, steelhead, and their habitat. This considered, adverse impacts to streams from spills of hazardous materials were not predicted by the USFS in the BA (USDA Forest Service 2007). The following section will discuss in more detail those materials either most likely to be spilled or those that would be most toxic to fish should a spill occur.

Table 10. Toxic thresholds for key materials at risk of release in the event of a one minute spill at low flow.

			Blackbird Creek	Williams Creek	Deep Creek	Panther Creek below Blackbird Creek	Salmon River below Williams Creek	Salmon River below Panther Creek
Low Flow cfs			***2.5***	***3***	***4.1***	***20***	***700***	***1,300***
Material	**Toxicity (mg\l)**							
LIQUIDS								
Gasoline	2.7	gallons	0.004	0.005	0.006	0.031	1.102	2.046
Diesel	18	gallons	0.02	0.03	0.04	0.20	6.98	12.97
Diesel	25	gallons	0.03	0.04	0.06	0.28	9.70	18.01
Antiscalant	2,660	gallons	2.60	3.11	4.26	20.76	726.76	1,349.70
SOLIDS								
Cement	92	lbs	0.86	1.03	1.41	6.89	241.22	447.98
Ammonium Nitrate	8.01	lbs	0.08	0.09	0.12	0.60	21.00	39.00
Sodium Sulfide, Hydrated	0.55	lbs	0.0052	0.0062	0.0084	0.041	1.44	2.68
SUPERFLOC™	0.22	lbs	0.0021	0.0025	0.0034	0.0165	0.58	1.07

Note: Bulk concentrate not considered toxic so excluded from this table.

Diesel – Of the substances being transported, diesel has been identified as posing the highest risk of a spill affecting ESA-listed salmonids and critical habitat (USDA Forest Service 2007). This is because diesel is delivered in large quantities (4,500 gallons), will be hauled to the mine site about 150 times per year, containers are aluminum and easily ruptured, and the substance is a liquid that can rapidly flow down gradient into nearby streams. In addition, diesel has a high accident probability for locations near streams, with a spill estimated to occur once every 40 years (USDA Forest Service 2007). Because of its relatively high probability for an accident due to the frequent delivery and its high risk of spill, the USFS determined that diesel was the material with the highest potential for adverse impacts to the aquatic environment.

The 96-hour LC_{50} reported in the BA for diesel and rainbow trout was a range of 18 to 25 mg/L (Conoco 2000). NMFS agrees that this is an appropriate range for evaluating acute effects on ESA-listed salmonids. The wide range of toxicity for diesel reflects variations in the petroleum compounds which occur for each source of crude, where it was refined, and the time of year the petroleum was produced. Petroleum products which are refined for use during the summer have different toxicity than products refined during the winter because the additives used to maintain the desired viscosity are applied in different proportions.

Table 10 shows that a relatively small spill of 0.28 gallons, if not contained, would cause toxicity in Williams, Blackbird, Deep, or Panther Creeks. In the Salmon River diesel below Williams Creek, a spill of as little as 10 gallons would be toxic. Diesel spills have the potential to affect listed salmon, steelhead, and their habitat. Diesel spills are not uncommon in Idaho, including but not limited to: (1) Little Salmon River (1993, 900 gallons diesel + 900 gallons gasoline), 133 dead rainbow trout and brook trout; (2) Lochsa River (2003, 6,300 gallons diesel), no documented fish kill or effect to critical habitat; and the (3) Middle Fork Clearwater River (2002; 10,000 gallons diesel), no documented fish kill or effect to critical habitat.

Two additional spills in the Salmon River basin warrant further discussion. On August 19, 1983, the IDFG reported a diesel fuel spill of 2,800 gallons into the Little Salmon River. The IDFG calculated that approximately 30,000 fish total (all species) died in the kill and also reported aquatic insect mortality. On September 6, 1989, a diesel spill occurred on Johnson Creek that resulted in 400 gallons of diesel reaching Johnson Creek. The precise effects of this spill and the emergency response to it were not well documented. However, depressed populations of aquatic insects were reported for 3.5 miles downstream and diesel odor was evident in stream substrates a year after the spill. The IDFG notes do indicate seeing Chinook salmon and steelhead showing obvious signs of stress.

A large diesel spill of 4,500 gallons in Blackbird Creek, Williams Creek, Deep Creek, Panther Creek, or the Salmon River would kill Chinook salmon and steelhead juveniles, adults, alevins, and eggs downstream of the accident, depending on the time of year. Any diesel spilled into action area streams would tend to travel downstream in a slug and dissipate slowly. Diesel from a spill could mix with spawning gravels and sand and be retained in the stream substrate for a year or more, and thereby negatively affect salmon and steelhead eggs, alevins, and juveniles for several years. Large amounts of petroleum products can suffocate aquatic organisms by coating their gills. Diesel fuel, like most petroleum products, contains toxic organic compounds that adversely affect water appearance and odor.

The effects of a diesel spill on Chinook salmon and/or steelhead would occur from disrupting the uptake of oxygen at the gill interface. At toxic concentrations mortality would occur as the result of suffocation. Direct mortality of ESA-listed salmon and steelhead, as well as other aquatic organisms, can also occur from physical coating, entanglement, or ingestion of fine oil droplets. Some of the compounds used in diesel are not readily biodegradable and settle into the sediments of the affected stream. This can result in sublethal exposure, which may or may not be fatal, as hydrocarbons are incorporated into the food chain (USDA Forest Service 2007).

Based upon information presented in Table 10, the effects of a 4,500 gallon diesel spill into occupied streams would be extreme and would likely result in mortality of all life stages of ESA-listed salmon or steelhead present at that time. A diesel spill of this magnitude would likely continue to affect salmon and steelhead eggs, alevins, and juveniles for several years should it be allowed to settle into stream substrates. The magnitude and extent of effect would vary tremendously depending upon the amount of material spilled. Although a higher likelihood of spill and higher risk of delivery to streams than other materials hauled to the site, accidents near streams are not expected to occur but once every 41 years (Table 8), well beyond the proposed life of the ICP. In addition, proposed improvements to the transportation corridor (turnout construction, removal of switchbacks, etc.), transport during daylight hours only, use of pilot cars, and application of the proposed spill and response measures (FCC 2002) should effectively limit the severity and the likelihood that a spill would occur. Consequently, NMFS does not anticipate a large spill of diesel fuel to occur as a result of the project.

Gasoline – CITGO (2007) reported that various grades of gasoline exhibited a range of lethal toxicity (LC_{100}) from 40 mg/L to 100 mg/L in ambient stream water for rainbow trout. CHEVRON identifies a 96-hour LC_{50} for rainbow trout and unleaded gasoline at 2.7 mg/L (Chevron 2005); this is the concentration used in the BA effects analysis. A gasoline spill would require a loading of only 0.004, 0.005, 0.006, 0.031, 1.102, and 2.046 gallons/minute for toxicity to be observed in Blackbird Creek, Williams Creek, Deep Creek, Panther Creek below Blackbird Creek, Salmon River below Williams Creek, and Salmon River below Panther Creek, respectively (Table 10).

Transported with diesel, gasoline poses both a high risk of accident and a high risk of delivery to streams should an accident occur. Approximately 30 gallons of gasoline will be hauled to the mine site about 150 times per year in a separate compartment of the diesel fuel trailers. As with diesel, containers will be aluminum and easily ruptured, and, as a liquid, it will be able to rapidly flow down gradient into nearby streams. Gasoline has the same accident probability as diesel, with a spill near streams estimated to occur once every 40 years (USDA Forest Service 2007).

The bulk of the available literature on gasoline relates to the environmental impact of monoaromatic (benzene, toluene, ethylbenzene, xylenes: BTEX) and diaromatic (naphthalene, methylnaphthalenes) constituents. In general, non-oxygenated gasoline exhibits some short-term toxicity to freshwater and marine organisms, especially under closed vessel or flow-through exposure conditions in the laboratory. The components which are the most prominent in the water-soluble fraction and cause aquatic toxicity, are also highly volatile and can be readily biodegraded by microorganisms. This material is expected to be readily biodegradable following a spill (USDA Forest Service 2007; CHEVRON 2005).

Based upon information presented in Table 10, the effects of a 30-gallon gasoline spill into occupied streams would be extreme and would likely result in mortality of all life stages of ESA-listed salmon or steelhead present immediately downstream from the spill. The magnitude and extent of effect would vary dependent upon the amount of water in the receiving waterbody and the amount of material spilled. Although there is a higher likelihood of spill and higher risk of delivery to streams than with other materials hauled to the site, accidents near streams are not expected to occur but once every 41 years (Table 8), well beyond the proposed life of the ICP. In addition, proposed improvements to the transportation corridor, transport during daylight hours only, use of pilot cars, and application of measures proposed to prevent spills from occurring and to respond to a spill to limit the severity (FCC 2002), NMFS does not anticipate a gasoline spill to occur as a result of the project.

Antiscalant – The USFS reported the 96-hour LC_{50} value for rainbow trout of 2,660 mg/L for CYQUEST DMA (anionic polyacrylamide) (USDA Forest Service 2007). To achieve this concentration, an antiscalant spill would require a loading of approximately 2.60, 3.11, 4.26, 20.76, 726.76, and 1,349.70 gallons/minute for toxicity to be observed in Blackbird Creek, Williams Creek, Deep Creek, Panther Creek below Blackbird Creek, Salmon River below Williams Creek, and Salmon River below Panther Creek, respectively (Table 10).

Antiscalant will be transported to the mill in loads of 16 250-gallon totes (4,000 gallons total), but will only be transported to the site once per year. Due to the very low transport rate, the estimated accident interval for this material near a receiving waterbody is estimated to occur only once every 6,220 years. However, because antiscalant is transported as a liquid, it remains a high risk to be delivered to streams in the event of an accident. Based upon information presented in Table 10, the dumping of one tote of antiscalant would be sufficient to result in mortality of ESA-listed salmonids in the smaller streams (i.e., Blackbird, Williams, or Panther Creeks). Multiple totes would need to be ruptured before toxic thresholds would be reached in the Salmon River. In either case, the magnitude and extent of effects would vary dependent upon the amount of water in the receiving waterbody and the amount of material spilled. Considering the spill analysis, combined with only one trip per year, and the fact that the proposed spill prevention and control measures (FCC 2002) should further reduce the likelihood and severity of a spill, NMFS expects that antiscalant will not likely be spilled during the life of the project. Therefore, the risk of an antiscalant spill into action area streams is negligible.

Cement – The USFS reported the 96-hour LC_{50} for rainbow trout at 92 mg/L for cement (USDA Forest Service 2007). To achieve this concentration, a cement spill would require a loading of approximately 0.86, 1.03, 1.41, 6.89, 241.22, and 447.98 lbs. for toxicity to be observed in Blackbird Creek, Williams Creek, Deep Creek, Panther Creek below Blackbird Creek, Salmon River below Williams Creek, and Salmon River below Panther Creek, respectively (Table 10).

Cement will be transported to the mill in loads of 4,000 lbs. (22 tons) per trip, with roughly 115 trips per year. Due to the higher transport rate, the estimated accident interval for this material near a receiving waterbody is higher than most materials at once every 54 years. Because cement is transported in bulk and dry, it's a low risk to be delivered to streams in the event of an accident. However, based upon information presented in Table 10, the effects of dumping of one truckload of concrete into occupied streams would be extreme and would likely result in mortality of all life stages of ESA-listed salmon or steelhead present. The magnitude and extent of effect would vary dependent upon the amount of water in the receiving waterbody and the amount of material spilled. However, considering the spill analysis, combined with the fact that the proposed spill prevention and control measures (FCC 2002) should further reduce the likelihood and severity of a spill, NMFS expects that cement will not likely be spilled during the life of the project, making the risk of a cement spill into action area streams negligible.

Ammonium Nitrate – The USFS reported the 96-hour LC_{50} for rainbow trout at 8.01 mg/L for ammonium nitrate. To achieve this concentration, a spill of ammonium nitrate would require a loading of only 0.08, 0.09, 0.12, 0.60, 21.00, and 39.00 lbs. for toxicity to be observed in Blackbird Creek, Williams Creek, Deep Creek, Panther Creek below Blackbird Creek, Salmon River below Williams Creek, and Salmon River below Panther Creek, respectively (Table 10).

Ammonium nitrate will be transported to the mill in a bulk container in loads of 10 tons each, transported to the site roughly 40 times per year. Due to a moderate transport rate, the estimated accident interval for this material near a receiving waterbody is estimated at once every 156 years. Because this material is transported in bulk and dry, it's a low risk for delivery to streams in the event of an accident. However, based upon information presented in Table 10, the effects of dumping of one truckload of ammonium nitrate into occupied streams would be extreme and would likely result in mortality of all life stages of ESA-listed salmon or steelhead present. The magnitude and extent of effect would vary dependent upon the amount of water in the receiving waterbody and the amount of material spilled. Considering the spill analysis and the proposed spill prevention and control measures (FCC 2002), NMFS does not expect a spill of ammonium nitrate to occur during the life of the project, making the risk of effects from ammonium nitrate negligible.

Sodium Sulfide, Hydrated – The USFS reported 96-hour LC_{50} values for rainbow trout ranging from 0.55 mg/L to 1.64 mg/L (USDA Forest Service 2007). At the more toxic concentration, a spill of this material would require less than a pound of material to reach toxic levels in the smaller tributaries. A loading of only 0.0052, 0.0062, 0.0084, 0.041, 1.44, and 2.68 lbs. for toxicity to be observed in Blackbird Creek, Williams Creek, Deep Creek, Panther Creek below Blackbird Creek, Salmon River below Williams Creek, and Salmon River below Panther Creek, respectively (Table 10).

Sodium sulfide will be transported in loads of 20- to 100-tons, contained in individual 50-pound sacks. This material will be delivered to the site approximately five times per year, resulting in an estimated accident interval near a receiving waterbody only once every 1,244 years. Because sodium sulfide is shipped dry and in 50-pound sacks, the risk of delivery to a stream should an accident occur is considered low risk. Based upon information presented in Table 10, the dumping of only one sack of sodium sulfide into an occupied stream would likely result in mortality of all life stages of ESA-listed salmon or steelhead present. As with other materials, the magnitude and extent of effect would vary dependent upon the amount of water in the receiving waterbody and the amount of material spilled. However, considering the spill analysis, combined with the fact that the proposed spill prevention and control measures (FCC 2002) should further reduce the likelihood and severity of a spill, NMFS expects that this material will not likely be spilled during the life of the project, making the risk of a spill into action area streams negligible.

SUPERFLOC™ 330 Flocculant – CYTEC (2001) reported the 96-hour LC_{50} value for rainbow trout at 0.022 mg/L for SUPERFLOC™ (water flea [*Daphnia magna*] 0.17 mg/L). However, according to CYTEC customer service, SUPERFLOC™ 330 flocculant was discontinued about 10 years ago, and has been replaced by SUPERFLOC® 577 Flocculant. The MSDS for SUPERFLOC® 577 does not reference an acute toxicity for rainbow trout, but does identify a 96-hour LC_{50} range of >10 to 100 mg/L for zebrafish (*Danio rerio*), and >10 to 100 mg/L for water flea. Lacking information specific to rainbow trout for SUPERFLOC® 577, the USFS analysis used the more specific and more conservative value of 0.022 mg/L in their analysis.

A SUPERFLOC® spill would require a loading of only 0.0021, 0.0025, 0.0034, 0.0165, 0.58, and 1.07 lbs. for toxicity to be observed in Blackbird Creek, Williams Creek, Deep Creek, Panther Creek below Blackbird Creek, Salmon River below Williams Creek, and Salmon River below Panther Creek, respectively (Table 10).

SUPERFLOC® will be transported in loads of eight, 50-pound sacks per trip, with only three trips per year. Therefore, the corresponding estimated accident interval involving this material near a receiving waterbody was only once every 2,073 years. In addition, SUPERFLOC® has been assigned a low spill risk level because it's transported dry, in 50-pound sacks. Based upon information presented in Table 10, the effects of dumping of one sack of SUPERFLOC® into occupied streams would be extreme and would likely result in mortality of all life stages of ESA-listed salmon or steelhead present. The magnitude and extent of effect would vary dependent upon the amount of water in the receiving waterbody and the amount of material spilled. Considering the spill analysis, combined with the fact that the proposed spill prevention and control measures (FCC 2002) should further reduce the likelihood and severity of a spill, NMFS expects that SUPERFLOC® will not likely be spilled during the life of the project, making the risk of a spill into action area streams negligible.

During the analysis of SUPERFLOC® it became clear that specific reagents and formulas are subject to change over time. It also became clear that the toxicity of certain reagents and formulas varies dependent upon the manufacturer. Therefore, this analysis only applies to the specific brand or formulation specified in the BA and this Opinion. A switch to a different product by FCC will require an additional evaluation to ensure that the new product is not any more toxic than products analyzed in this Opinion.

Outfall Discharge – ICP's NPDES permit will include discharge limitations for 16 potential pollutants, including eight metallic elements, arsenic, ammonia, sulfate, sulfide, TSS, pH, DO, and temperature. The outfall will discharge effluent into Big Deer Creek approximately 3 miles upstream from its confluence with Panther Creek. Although remediation of the Blackbird Mine Site has reduced the legacy pollutant concentrations in Big Deer Creek, background concentrations still need to be considered when evaluating the Project's effect on water quality.

The ICP effluent has the potential to adversely affect ESA-listed Chinook salmon or steelhead downstream from the outfall. However, neither species occurs in the immediate vicinity of the outfall because access is blocked by the barrier 2.3 miles downstream on Big Deer Creek. Since they are not present at the outfall, neither species will be exposed to concentrations as toxic as those coming directly out of the pipe. Because Big Deer Creek is already on EPA's list of 303(d) waterbodies for metal contaminants, it's important to consider background levels of metals in Big Deer Creek before analyzing potential effects to fish. Table 11, summarizes this information, along with the estimated pollutant discharge concentrations and limits both at the outfall, and downstream from the falls on Big Deer Creek where ESA-listed salmonids are present.

Except for sulfates, EPA effluent limits are set at end-of-pipe and without a mixing zone. The ICP water treatment plant has been designed to meet the more restrictive limits with no dilution planned for the discharge. For many potential pollutants, the estimated concentration is less than the effluent limits specified in the NPDES permit, providing a potential cushion of safety over the estimated discharge concentration. Nevertheless, this analysis assumes effluent concentrations at the limits specified in the NPDES permit. Effluent limits were established to meet State of Idaho water quality criteria. For toxic chemicals of concern, these limits are computed on the basis of acute and chronic effects to coldwater aquatic biota. Where EPA has not developed criteria specific to aquatic life, limits were designed to meet the needs of other beneficial uses. For example, the criterion for thallium was set to limits for human health and fish consumption, for nitrates to protect agricultural water supply, and for sulfate as a secondary drinking water supply.

Table 11. Outfall Estimates of Pollutant Discharge, Effluent Limits, Acute/Chronic Water Quality Criteria, and Big Deer Creek Background/Expected Concentrations.

Parameter	Effluent Estimates		Effluent Limits & Monitoring Requirements		EPA WQ Criteria		Background Conc. Big Deer Creek[1,3]	Estimated Conc. Big Deer Creek @ Cascades[2,3]
	Max. Daily Value	Avg. Daily Value	Max. Daily Limit	Avg. Monthly Limit	Acute	Chonic		
Arsenic (µg/L)	<50	<50	100	50	340	150	0.0008	2.8
Cadmium (µg/L)	<0.1	<0.1	0.52	0.26	0.52	0.37	0.00006	0.01
Cobalt (µg/L)	<50	<50	141	70.4	--	86	0.021	3.9
Copper (µg/L)	<2.0	<2.0	4.80	2.40	4.6	3.5	6.9	6.6
Lead (µg/L)	<0.3	<0.3	0.90	0.45	13.88	0.54	0.00062	0.03
Mercury (µg/L)	0.0018	0.001	0.02	0.01	--	0.012	0.00005	0.001
Nickel (µg/L)	<5.0	<5.0	26.52	13.22	145	16.1	0.001	0.74
Selenium (µg/L)	4	2	--	--	20	5	--	--
Thallium (µg/L)	<0.2	<0.2	0.95	0.47	--	10	0.00001	0.03
Zinc (µg/L)	<10	<10	37.02	18.45	36.2	36.5	0.006	1.0
Ammonia (as N)(mg/L)	1	1	4.1	1.6	5.6	2.34	0.022	0.1
Nitrate + Nitrite (mg/L) [4]	<100	<100	100	--	--	--	--	--
Sulfate (mg/L)[5]	<840	<840	250	--	--	--	7.3	53.6
Sulfide (µg/L)	--	--	2	--	--	--	--	--
TSS (mg/L)	30	15	30	20		--	--	--
pH (s.u.)	9	7.5	6.5 - 9.0 at all times		--	--	--	--
Dissolved O_2 (mg/L)	--	--	> 6.0 at all times		--	--	--	--
Temperature (Summer/winter) (ºF)	55/35	55/40	66	--	Max. Daily 66.2		--	--
Iron (µg/L)	300	30	--	--	--	--	--	--
Aluminum (µg/L)	200	20	--	--	--		--	--
Magnesium (mg/L)	100	10	--	--	--	--	--	--
Manganese (µg/L)	50	.005	--	--	--	--	0.0127	2.8

[1] Background concentrations were collected at Water Quality Site 24, on Big Deer Creek, below confluence with South Fork Big Deer Creek (collected May 2001 to October 2005).

[2] Big Deer Creek at the cascades is the upstream distribution limit of anadromous fish, 2.3 miles downstream from the outfall.

[3] Concentrations assume the maximum outfall flow of 0.33 cfs, the 7Q10 flow statistic of 5.6 cfs for Big Deer Creek, and the NPDES average monthly limit.

[4] Limit is based on EPA water quality criteria value for agricultural water supply, no limit has been defined for cold water aquatic life.

[5] A mixing zone has been requested for sulfate. End-of-pipe concentration is estimated to be less than 840 mg/L. End of mixing zone effluent limit is 250 mg/L.

Toxicity of cadmium, copper, lead, nickel, and zinc is hardness dependent. Therefore, Idaho's aquatic life water quality criteria for metals are expressed as a function of hardness measured in mg/L of calcium carbonate ($CaCO_3$). As water hardness of the receiving water increases, the toxicity of the metals decreases and the numerical value of the criteria increases. Idaho Water Quality Standards stipulate minimum and maximum hardness values of 25 mg/L and 400 mg/L, respectively [IDAPA 58.01.02.210.03(c)(i)]. For the purposes of calculating a conservative value for metals criteria, the Technical Support Document for Water Quality-based Toxics Control (EPA 1991) recommends using the 5th percentile of hardness values measured in the receiving water. In the case of the ICP, the 5th percentile of 16 hardness measurements collected from Water Quality Site 24 on Big Deer Creek is 20.5 mg/L (i.e., soft water). Accordingly a hardness value of 25 mg/L was used in calculating numeric criteria for hardness dependent metals for the ICP. In estimating background concentrations and pollutant concentrations in Big Deer Creek below the falls, the USFS used a worst-case scenario (USDA Forest Service 2007). Concentrations for the last two columns in Table 11 assume the maximum outfall flow of 0.33 cfs, and the 7Q10 flow statistic of 5.6 cfs for Big Deer Creek. The following section will discuss in more detail those effluent parameters most likely to result in adverse effects to ESA-listed fish species.

<u>Arsenic</u> – Mining of ore containing arsenic can pollute surface waters as the metalloid leaches from tailings ponds and waste piles. It can later be mobilized in acid drainage from abandoned mines (Nelson et al. 1991). The discharge of mining wastes into freshwater systems has resulted in high concentrations of arsenic in sediments and benthic invertebrates (Pedlar et al. 2002). Although elemental arsenic is insoluble in water, many arsenates are highly soluble. Therefore, arsenic compounds occur naturally in western streams and lakes (Nelson et al. 1991). Arsenic is highly persistent in water, with a half-life > 200 days (Vincoli 1997). Typical concentrations for background freshwater streams and rivers are less than 1 µg/L (USDA Forest Service 2007). According to water quality monitoring data from 2001 until 2005, background concentrations of arsenic in Big Deer Creek are currently estimated at 0.0008 µg/L (Table 11).

Arsenic toxicity can be altered by a number of factors including pH, redox potential (Eh), organic matter, phosphate content, suspended solids, presence of other toxicants, speciation of the chemical itself, and the duration of exposure to arsenic. Although the BA states that arsenic does not readily bioconcentrate in aquatic species, a literature review conducted as part of this analysis suggests that this may not be accurate. Rainbow trout exposed to 60 to 240 mg/L of arsenic resulted in increases in whole-body arsenic concentrations ranging from 8.1 to 13.5 µg/g (McGeachy and Dixon 1991). After feeding rainbow trout a diet of four dietary arsenicals over an eight-week period, Cockell and Hilton (1998) found that carcass arsenic concentration showed a dose-response relationship to dietary arsenic concentration and exposure rate. Following aqueous and dietary exposure of lake whitefish (*Coregonus clupeaformis*) to arsenic, Pedlar and Klaverkamp (2002) noted significant accumulation of arsenic in the stomach, pyloric caeca, intestines, liver, kidney, and scales. They also found increased concentrations of arsenic in the gallbladder, gonads, spleen, gill, and bone.

Similarly, metals have also been found to accumulate in aquatic invertebrates (Woodward et al. 1994; Beltman et al. 1999). Metal accumulation occurs via uptake across the gills, through the gut, and through adsorption to the exoskeleton (Beltman et al. 1999). Woodward et al. (1994) noted a study where metal concentrations in aquatic invertebrates in Clark Fork River were two to fourteen times greater than for the same taxa in other less-contaminated rivers. Although susceptible to bioaccumulation, aquatic invertebrates appear to be more tolerant of arsenic than fish, with concentrations of 3.0 to 14.0 mg/L and 10 to 20 mg/L reported as non-toxic to mayfly and dragonfly nymphs respectively (Nelson et al 1991). Rainbow trout fed a diet of metal-contaminated aquatic invertebrates from the Clark Fork resulted in 6- to 10-fold greater tissue concentrations of arsenic than control groups (Woodward et al. 1994). Considering this information, it's possible that indirect effects could potentially occur to ESA-listed salmon and steelhead feeding on arsenic-contaminated macroinvertebrates.

Acute toxicities have been reported at a 96-hour LC_{50} of 10.8 mg/L for arsenic and rainbow trout (Hale 1977). Nelson et al. (1991) described significant chronic effects to coho salmon (*O. kisutch*) from a six-month exposure to 300 μg/L of arsenic trioxide, where the normal increase in plasma thyroxine was delayed causing a transitory reduction in gill sodium-potassium ATPase activity. Although treated coho showed no direct effects in growth or survival, treated fish were less successful in seaward migration than control fish. However, Cockell and Hilton (1988) noted altered feeding behavior, and reduced growth and feed consumption by rainbow trout exposed to four dietary arsenicals. Rankin and Dixon (1994) observed significant reductions in growth at 9.64 mg/L of waterborne arsenite. Reduced appetite and direct metabolic impact were believed to be the cause for the reduced growth. It's also important to note that trout exposed to this concentration also suffered 10% mortality, mainly due to erosion of the mandibular and olfactory regions of the head. All fish exposed to this concentration also showed inflammation of the gallbladder wall.

The EPA has identified acute water quality criteria for arsenic of 340 μg/L and chronic water quality criteria of 150 μg/L. The NPDES permit limits effluent to 100 μg/L maximum daily load and 50 μg/L average monthly load. The FCC estimates effluent concentrations of arsenic of < 50 μg/L at the end-of-pipe. Both the effluent limits and estimated concentration at the outfall are below EPA's acute and chronic water quality criteria. At the end-of-pipe, maximum concentrations allowed by the NPDES permit limits are well below the 96-hour LC_{50} value of 10.8 mg/L reported by Hale (1977) for rainbow trout, and well below the 9.64 mg/L concentration of waterborne arsenite identified by Rankin and Dixon that led to reduced growth and mortality. Furthermore, considering dilution, mixing, and background concentrations of 0.0008 μg arsenic/L in Big Deer Creek, arsenic concentrations instream are estimated to be even lower at approximately 2.8 μg/L downstream of the falls where anadromous fish reside (Table 11). Therefore, NMFS does not expect the discharge of arsenic to result in direct effects to Chinook salmon or steelhead in Big Deer Creek. However, the potential for indirect effects is less certain. Although arsenic concentrations are not expected to rise to a range toxic to aquatic invertebrates, increased arsenic entering Big Deer Creek via the effluent could lead to bioaccumulation of arsenic in macroinvertebrates. Contaminated aquatic invertebrates later preyed upon by salmon and steelhead could then lead to potential accumulation of arsenic in salmonid tissues and organs. Although there is some degree of uncertainty regarding indirect effects of arsenic on salmon and steelhead in Big Deer Creek, stream surveys suggest that Chinook are absent and steelhead are only present in low numbers in lower Big Deer Creek (Kuzis 2004). Because indirect effects, if any, would be localized only to individual fish in lower Big Deer Creek, it's highly unlikely that enough fish would be affected that bioaccumulation will rise to a level sufficient to appreciably reduce species abundance/productivity at the larger population, MPG, or ESU scales.

Cadmium – Cadmium occurs naturally in the aquatic environment as a sulfide salt, often in association with zinc and lead ores (Nelson et al. 1991). Natural concentrations are often in the range of 0.002 to 0.08 µg/L, with background concentrations of dissolved cadmium in Idaho waters generally in a range of less than 0.02 to 0.1 µg/L (Mebane 2006). Cadmium has no known biological use and is considered one of the most toxic metals. While released through natural processes, anthropogenic cadmium emissions have greatly increased levels of cadmium in the environment. In aquatic systems, cadmium quickly partitions to sediment, but is readily remobilized through a variety of chemical and biological processes (USDA Forest Service 2007). The half-life of cadmium in water is relatively short because, once in water, cadmium is considered highly soluble and highly mobile (Mebane 2006). According to water quality monitoring data from 2001 until 2005, background concentrations of cadmium in Big Deer Creek are currently estimated at 0.00006 µg/L (Table 11).

Cadmium is a known teratogen, carcinogen, and probable mutagen to which freshwater organisms are considered the most sensitive (Vincoli 1997; Eisler 1985a). Acute toxic effects may include the death of animals, birds, or fish, and death or low growth rate in plants. In freshwater organisms, effects of cadmium toxicity include decreased growth, inhibited reproduction, and population alterations (Vincoli 1997). Cadmium is extremely toxic to aquatic animals. In comparative acute toxicity testing of all 63 atomically stable heavy metals in the periodic table, cadmium was the most toxic metal (Mebane 2006). The acute toxicity of cadmium is due to its role as a calcium antagonist. It's pathological effects tend to be less severe at higher water calcium levels (i.e., harder water) (Scott et al. 2003; Vincoli 1997). In other words, the harder the water, the lower the toxicity. Toxicity of cadmium can also be influenced by pH; becoming more toxic in low-alkalinity water. Because the carbonate and hydroxide forms of cadmium are insoluble, cadmium is precipitated out at high pH values, reducing its overall toxicity to fish (Nelson et al. 1991). Uptake of cadmium from waterborne exposure occurs primarily through the gills, versus dietary uptake which occurs in the gastrointestinal tract (Scott et al. 2003; Szebedinszky et al. 2001). It's been conservatively estimated that adverse effects to fish or wildlife are either pronounced or probable when cadmium concentrations exceed 3 µg/L in freshwater or 100 µg/g (parts per billion) in diet (Eisler 1985a).

Cadmium salts (chlorides and sulphates) in stream sediments can be mobilized and made bioavailable if pH is lowered (more acidic) or because of high oxidation (redox potential). If rooted aquatic macrophytes or adjacent wetland plants are present in freshwater systems, cadmium bound in sediments may be transferred to the plant tissue and, when the plants die and decompose, become bioavailable in the aquatic ecosystem to other biota (Eisler 1985a).

Although the BA states that cadmium does not bioconcentrate in aquatic species, a literature review conducted as part of this analysis suggests that this is not the case. The concentration of cadmium found in fish tissues is expected to be much higher than the average concentration of cadmium in the water from which the fish was taken (Vincoli 1997). Benoit et al. (1976) found residues of cadmium in brook trout kidney, liver, gill, gonad, spleen, muscle, and red blood cells following exposure. Hamilton et al. (1987) described a close dose-response relationship between exposure concentrations and free-cadmium concentrations in brook trout liver and kidneys. They also found that whole-body residues of cadmium increased significantly in brook trout at all exposures from 3.6 to 60.6 µg/L, demonstrating a dose-dependent response to cadmium concentrations. Residues were highly correlated with both cadmium exposure concentrations and mortality after 30 days of exposure. Cadmium has been documented to accumulate in tissue from both waterborne and dietary exposure, and has also been shown to enter circulation and accumulate significantly in the liver and kidney (Scott et al. 2003; Szebedinszky et al. 2001; McGeer et al. 2000).

Eisler (1985a) reported the bioconcentration factor for cadmium in rainbow trout to be 260 times (x) for gill tissue, 17x for liver tissue, 26x for kidney, and zero for spleen and heart tissues over ambient concentrations in the aquatic environment. Biomagnification from one trophic level to another in aquatic systems has also been demonstrated for cadmium, going from phytoplankton to zooplankton. However, fish fed cadmium-contaminated cladocerans for four days showed no change in body burdens. For the whole organism, the bioconcentration factor for rainbow trout was 33x, was 2,550x for *Chlorella vulgaris* (a species of green algae), 2,200x for chironomids, and 1,630x for mayflies in the genus *Ephemerella* (Eisler 1985a).

As summarized by Thurston et al. (1979), Davies found a no effect concentration for rainbow trout in hard water (hardness 326 mg/L) between 13.5 µg/L (no effect) and 21 µg/L (20% mortality), versus a no effect concentration in soft water (hardness 31 mg/L) between 0.7 µg/L (no effect) and 1.5 µg/L (10% mortality). Nelson et al. (1991) reported acute and chronic lowest observed effect levels for cadmium of 3.9 µg/L and 1.1 µg/L respectively. After exposing three generations of brook trout to 0.06 to 6.4 µg/L of cadmium, Benoit et al. (1976) noted significant mortality of first- and second-generation spawning males at 3.4 µg/L. This concentration also significantly retarded the growth of second and third generations.

Lethal concentrations of cadmium ranging from 0.8 to 9.9 µg/L have been identified for several species of aquatic insects, crustaceans, and fishes. According to Mebane (2006), chronic and acute toxicity of cadmium in the aquatic environment is dependent on water hardness. For example, using a hardness correction, Mebane (2006) calculated LC_{50}s of 467 µg/L for a species of caddisfly (*Arctopsyche* species), 150.86 µg/L for the blue-winged olive mayfly (*Baetis tricaudatus*), and 15.29 µg/L for a species of Cladocera (*Ceriodaphnia dubia*). For acute exposure to cadmium, fish in the genus *Oncorhynchus* were found to be the most sensitive of 58 genera evaluated. Mebane reported Chinook salmon hardness-corrected LC_{50}s ranging from 2.27 to 49.90 µg/L for acute cadmium exposures (dependent on life stage). For rainbow trout/steelhead, hardness-corrected LC_{50}s ranged from 0.55 to 9.13 µg/L. For chronic exposures to cadmium, hardness-corrected LC_{50}s were identified as 1.72 µg/L for Chinook salmon, and 1.36 to 2.22 µg/L for rainbow trout/steelhead (Mebane 2006). The author also identified lowest-observed-adverse effect concentrations (LOECs) for chronic exposure to cadmium at 1.3 µg/L for Chinook salmon, and a range of 1.3 to 7.02 µg/L (not corrected for water hardness) for rainbow trout.

Eisler (1985) reported that sublethal aquatic effects of cadmium are associated with concentrations of 0.7 to 570 µg/L, noting decreased growth, inhibited reproduction, and population alterations. Studies by Scott et al. (2003) and Riddell et al. (2005) have also revealed that exposure to low levels of cadmium can result in sublethal effects to salmonids. Scott et al. (2003) found that exposure to sublethal levels of cadmium can result in cadmium accumulation in the olfactory rosette, nerve, and bulb, impairing olfactory function of juvenile rainbow trout. This accumulation in the olfactory system from waterborne exposure was greater than any other organs of accumulation other than the gill. Exposure inhibited normal behavioral and physiological responses of rainbow trout to alarm substance (a chemical signal released from specialized epidermal cells in fish skin when attack by a predator causes sufficient skin damage), disrupting their predator-avoidance strategy. Normal behavioral responses to alarm substance were eliminated by waterborne exposure to 2 µg/L cadmium for 7 days (moderate hardness of 120 mg/L hardness as $CaCO_3$). Scott cited an addition study by Tjälve and Gottofrey that used similar waterborne cadmium concentrations (1 µg/L and 10 µg/L for one week in 40 mg/l hardness as $CaCO_3$), where cadmium accumulated in the olfactory system of brown trout (*Salmo trutta*). Inhibition of a predator avoidance strategy has the potential to affect survival of ESA-listed salmonids at both the individual and population scale.

Riddell et al. (2005) noted a change in foraging preference, behavior, and fish condition factor following exposure to sublethal levels of cadmium (0.5 µg/L and 5.0 µg/L for 30 days; 156 mg/l hardness as $CaCO_3$). Brook trout exposed to elevated levels of cadmium switched from a preference for the more mobile and nutritious *B. tricaudatus* to the non-mobile, less nutritious *Chironomus tetans* (bloodworm head). Independent of prey choice, the capture efficiency of cadmium-stressed trout decreased by 20% to 55% with increasing cadmium concentrations. Fish condition factor, determined by the length and width of the fish pre- and post-treatment, was also negatively affected, declining by 12% to 18% in exposed fish as opposed to increasing by 34% in control fish. In this study, alteration of diet and reduced capture efficiency associated with exposure to sublethal levels of cadmium negatively affected fish growth, a condition that would decrease the likelihood that these fish would survive in the wild. Sloman et al. (2003) also noted a behavioral response in rainbow trout exposed to cadmium. Rainbow trout exposed to sublethal concentrations (150 µg/L [15% of 96-hour LC_{50} of 1,000 µg/L]) of cadmium for 24 hours became subordinate when paired with non-exposed fish, severely confounding the ability of exposed fish to become dominant.

The EPA has identified acute water quality criteria for cadmium at 0.52 µg/L, and chronic water quality criteria at 0.37 µg/L. The NPDES permit limits effluent to 0.52 µg/L maximum daily load, and 0.26 µg/L average monthly load. The FCC estimates effluent concentrations of cadmium at < 0.1 µg/L at the end-of-pipe. Both the effluent limits and estimated concentration at the outfall are at or below EPA's acute and chronic water quality criteria. At the end-of-pipe, maximum concentrations allowed by the NPDES permit limits are below the no effect concentration of 0.7 µg/L reported by Thurston et al. (1979) for rainbow trout in soft water, and well below the 3.64 µg/L concentration identified by Benoit et al. (1976) that led to significant mortality. Furthermore, considering dilution, mixing, and background concentrations of 0.00006 µg cadmium/L in Big Deer Creek, cadmium concentrations instream are estimated to be even lower at approximately 0.01 µg/L downstream of the falls where anadromous fish reside (Table 11). Although it is less certain that the EPA water quality criteria for cadmium are set low enough to avoid bioaccumulation in potential prey species or to avoid sublethal effects to ESA-listed salmonids, the mixing and subsequent dilution to concentrations of 0.01 μg/L by the time cadmium-laden water reaches waters inhabited by salmon and steelhead makes it less likely that exposure to cadmium would result in lethal or sublethal effects.

Cobalt – Although naturally occurring in uncontaminated waters at no more than a few micrograms per liter, cobalt can occur in elevated concentrations in water affected by runoff from mines containing cobalt-bearing ores (Marr et al. 1998; Nagpal 2004). In the United States, concentrations of cobalt in freshwater streams range from <1 μg/L in undisturbed natural streams, to levels between 11 and 50 μg/L in streams passing through mining districts and regions with heavy agricultural land use (Nagpal 2004). Background concentrations of cobalt in Big Deer Creek are currently estimated at 0.021 μg/L (Table 11). Cobalt and its salts are highly persistent in the aquatic environment, with a half-life >200 days (Vincoli 1997). The concentration of cobalt has been reported to significantly correlate to pH (inverse) and suspended solids (positive) in water (Nagpal 2004).

Cobalt is a known carcinogen, and has been documented as mutagenic in humans. Although little data exists on the toxicity of cobalt, acute exposure can result in death to animals, birds, or fish. Cobalt and its salts reportedly have high chronic toxicity to aquatic life, resulting in effects such as shortened life spans, reproductive problems, lower fertility, and potential changes in appearance or behavior (Vincoli 1997). Nagpal (2004) suggested that aquatic invertebrates are more sensitive to cobalt exposure than either plants or fish. Cobalt may bioconcentrate, with cobalt concentrations in fish expected to be somewhat higher than the average concentration in the water from which the fish was taken (Vincoli 1997).

Nagpal (2004) found that rainbow trout were the most sensitive fish species to cobalt exposure. Studies reviewed by Nagpal identified chronic LC_{50}s of 470 to 490 μg/L for 28-day embryo-larval toxicity tests, and a 144-hour LC_{50} of 520 μg/L (Marr et al. 1998) for fry. Marr et al. (1998) also identified 14-day no observed effect concentrations (NOEC) and LOEC for rainbow trout fry growth and survival of 132 and 255 μg/L respectively. Marr et al. identified a 96-hour LC_{50} of 1,406 μg/L for rainbow trout, but also identified a time-independent incipient lethal level (ILL) of 346 μg/L. Because cobalt concentrations high enough to eventually lead to 100% mortality caused no mortality until after 72-hours of exposure, with most fish dying between 72- and 192-hours, the study suggested that the 96-hour LC_{50} would not likely be sufficient to adequately protect rainbow trout from cobalt exposure.

ESA-listed salmon and steelhead could be indirectly affected through effects to prey items. Effects of metals have been shown to affect invertebrate communities by decreasing plecopteran and trichopteran densities, decreasing total taxa richness, decreasing biomass, and increasing the relative abundance of chironomids (Beltman et al. 1999). Nagpal (2004) found that *D. magna* and *C. dubia* exhibited chronic effects when exposed to low cobalt concentrations, noting an LOEC of roughly 8 μg /L was sufficient to cause reproductive effects in these invertebrates. To protect aquatic life from cobalt exposure, Nagpal (2004) recommended acute concentrations <110 μg/L and chronic concentrations <4 μg/L (30-day average).

Sublethal effects have also been noted for both Chinook salmon and steelhead in the form of avoidance behavior. Hansen et al. (1999b) found that Chinook salmon were more sensitive to elevated cobalt concentrations than steelhead, avoiding concentrations of 24 µg/L versus 180 µg/L. This study suggests that every day behavior and habitat use would be affected at concentrations much lower than those reported for acute or chronic mortality.

Mebane (2007) recently completed a white paper at the request of NMFS to evaluate the protectiveness of a site-specific cobalt concentration of 86 µg/L for Chinook salmon and steelhead in Panther Creek, Idaho (Appendix B). In this effort, Mebane conducted an extensive literature review for cobalt and its potential effects on aquatic biota, including a detailed review of a 2004 study of toxicity of cobalt to fish and invertebrate species for Blackbird Mine cleanup targets (Pacific EcoRisk 2005). Acute and chronic cobalt toxicity levels were tested for rainbow trout, mottled sculpin (*Cottus bairdi*), two mayflies (*Serratella tibialis* and *Centroptilum conturbatum*), one caddisfly (*Brachycentrus americanus),* and one midge (*C. tentans*). In rainbow trout, acute testing revealed 96-hour LC_{50}s ranging from 800 to 1,360 µg/L, NOECs ranging from 750 to 950 µg/L, and LOECs ranging from 290 to 470 µg/L. Chronic toxicity tests revealed a NOEC of 101 µg/L, and a LOEC of 242 µg/L. *C. conturbatum* was determined to be the most sensitive of all species tested to cobalt concentrations with LC_{50}s ranging between 2,000 and 9,400 µg/L. *C. tentans* was the most sensitive invertebrate to chronic cobalt exposure, with an EC_{20} (concentration adversely affecting 20% of test population) of 2,370 µg/L. Considering this data and literature reviewed, Mebane (2007) concluded that appreciable adverse effects to steelhead and Chinook salmon populations or their habitat associated with sustained concentrations of cobalt up to 86 µg/L would be unlikely.

The EPA has not identified an acute water quality criterion for cobalt, but has identified a chronic water quality criterion of 86 µg/L. The NPDES permit limits effluent to 141 µg/L maximum daily load, and 70.4 µg/L average monthly load. The FCC estimates effluent concentrations of cobalt at < 50 µg/L at the end-of-pipe. Both the effluent's chronic limit and estimated concentration at the outfall are at or below EPA's chronic water quality criteria. At the end-of-pipe, maximum concentrations of cobalt allowed by the NPDES permit limits are in a range are well below LC_{50}s reported by Nagpal (2004), and also well below the NOEC, LOEC, and ILL toxicity levels reported by Marr et al. (1998). Although EPA's maximum daily limit of 141 µg/L is higher than the acute recommendation of 110 µg/L by Napgal (2004), dilution in Big Deer Creek is expected to lower cobalt concentrations below the falls to approximately 7.9 µg/L (acute) and 3.9 µg/L (chronic) (Table 11). This concentration is also below Nagpal's recommended 4.0 µg/L for chronic cobalt exposure. Therefore, both the EPA water quality criteria for cobalt and the estimated concentrations from the ICP effluent appear to be set low enough to avoid adverse effects to potential prey species and to avoid sublethal effects to ESA-listed salmonids.

<u>Copper</u> – Copper is naturally occurring, plentiful, and essential to the normal growth and metabolism of all living organisms. At low concentrations it's an essential element for both plants and animals, but at slightly higher concentrations can be toxic to aquatic life (Eisler 1998a; Vincoli 1997). Although copper is relatively insoluble in water, it becomes more soluble as pH drops and can thereby be introduced into streams via acid mine drainage (Nelson et al. 1991). Once in aquatic systems, it can dissolve or bind to organic and inorganic materials either

in suspension or in sediment. Ambient monitoring studies by the USGS for dissolved copper at 811 sites across the United States have revealed background copper concentrations ranging from 1 to 51µg/L, with a median of 1.2 µg/L (Hecht et al. 2007). Background concentrations of copper in Big Deer Creek are currently estimated at 0.0456 µg/L (Table 11). Copper is highly persistent in the aquatic environment, with a half-life >200 days (Vincoli 1997). Scientific literature indicates that copper toxicity decreases as water hardness and alkalinity increases (Nelson et al. 1991). It can also be influenced by sodium (Welsh et al. 2008).

Dissolved copper is highly toxic to a broad range of aquatic species, including algae, macrophytes, aquatic invertebrates, and fishes (Hecht et al. 2007). Copper is a sodium antagonist (Sloman et al. 2003), and acute exposure can result in death to animals, birds, or fish. Chronic exposure can affect life spans, reproduction, fertility, growth, osmoregulation, sensory function, metabolism, carcinogenicity, mutagenicity, teragenicity, appearance and behavior (Vincoli 1997; Hecht et al. 2007; Eisler 1998a). Mortality to fish from copper occurs when insoluble copper-protein compounds form on gill surfaces, causing the sloughing of gill epithelia and eventual suffocation (Nelson et al. 1991). Copper may also bioconcentrate in organisms, with concentrations in fish expected to be considerably higher than the average concentration in the water from which the fish was taken (Vincoli 1997). Loss of invertebrate taxa richness has been reported to occur at copper concentrations as low as 5 µg/L. Direct exposure to dissolved copper can impair and destroy olfactory sensory neurons which are important for finding food, avoiding predators, migration, recognizing kin, reproducing, and avoiding pollution (Hecht et al. 2007).

Hecht et al. (2007) conducted a comprehensive literature review regarding copper and its effects on fish. Table 12 is an excerpt from that publication, summarizing both acute and chronic effect concentrations reported in the literature. As summarized in Table 12, acute mortality of various life stages of salmon and steelhead exposed to dissolved copper occur a low concentrations, with 96-hour LC_{50}s ranging from 9 to 57 µg/L. A growing body of literature reviewed demonstrates that low levels of dissolved copper result in sublethal effects to salmon and steelhead as well. Dissolved copper is a neurotoxicant that can directly damage the sensory capabilities of salmonids at low concentrations, concentrations at or slightly above ambient levels. Sensory system effects are generally among the more sensitive fish responses and underlie important behaviors involved in growth, reproduction, and ultimately survival (i.e., predator avoidance, migration, etc.). Salmonids begin actively avoiding dissolved copper at concentrations as low as 0.75 µg/L, experience reduced growth from concentrations as low as 1.9 µg/L, experience delayed outmigration at 5 µg/L, and experience problems homing and spawning at concentrations in the 10 to 25 µg/L range (Table 12).

As reported by Hecht et al. (2007), several recent studies highlight some important aspects of dissolved copper olfactory toxicity (Baldwin et al. 2003; Sandahl et al. 2004; Sandahl et al. 2007). Impairment of olfaction (i.e., smell) can be measured by an electrophysiological technique called the electro-olfactogram (EOG). The EOG measures olfactory response of a population of receptor neurons in fish. Reductions in the EOG amplitude of copper-exposed fish compared to unexposed fish reflect functional losses in sensory capacity. Dissolved copper's toxic effect to olfactory sensory neurons is observable as a reduction in or elimination of the EOG amplitude to a recognizable odor. Baldwin et al. (2003) found that the neurotoxic effects of copper in coho salmon manifest over a timescale of minutes. At 10 minutes, EOG amplitude reductions were observed in juvenile coho exposed to 2, 5, 10, and 20 µg/L dissolved copper above experimental background (3 µg/L). After 30 minutes at 2 µg/L dissolved copper above experimental background, the EOG amplitude from juvenile coho to odors was reduced by approximately 25% compared to controls; after 30 minutes in 20 µg/L dissolved copper amplitude was reduced by 80%. Sandahl et al. (2004) reported similar effects following 7 days of exposure (both in EOG reductions and copper concentrations). This result indicated that the juvenile olfactory system does not appear to be able to adapt or otherwise compensate for continuous copper exposure for durations up to 7 days.

Table 12. Selected examples of adverse effects with copper to salmonids or their preya (from Hecht et al. 2007)

Species (life stage)	Effect	Effect Conc. (µg/L)[b]	Effect Statistic	Hardness (mg/L)[c]	Exposure Duration	Sources
Sensory and behavioral effects						
Coho salmon (juvenile)	Reduced olfaction & compromised alarm response	0.18 –2.1	EC10 to EC50	120	3 hours	Sandahl et al. 2007
Chinook salmon (juvenile)	Avoidance in laboratory exposures	0.75	LOEC	25	20 minutes	Hansen et al. 1999a
Rainbow trout (juvenile)	Avoidance in laboratory exposures	1.6	LOEC	25	20 minutes	Hansen et al. 1999a
Chinook salmon (juvenile)	Loss of avoidance ability	2	LOEC	25	21 days	Hansen et al. 1999a
Atlantic salmon (juvenile)	Avoidance in laboratory exposures	2.4	LOEC	20	20 minutes	Sprague et al. 1965
Atlantic salmon (adult)	Spawning migrations in wild interrupted	20	LOEC	20	Indefinite	Sprague et al. 1965
Chinook salmon (adult)	Spawning migrations in wild apparently interrupted	10 – 25	LOEC	40	Indefinite	Mebane 2000
Coho salmon	Delays & reduced downstream migration dCu-exposed juveniles	5	LOEC	95	6 days	Lorz and McPherson 1976; 1977
Rainbow trout	Loss of homing ability	22	LOEC	63	40 weeks	Saucier et al. 1991
Ecosystem effects						
N/A[d]	Ecosystem function: Reduced photosynthesis	2.5	LOEC	49	≈ 1 year	Leland and Carter 1985
N/A[d]	Ecosystem structure: loss of invertebrate taxa richness in a mountain stream	5	LOEC	49	≈ 1 year	Leland et al. 1989
Other sublethal effects						
Chinook salmon	Reduced growth (as weight)	1.9	EC10	25	120 days	Chapman 1982
Rainbow trout		2.8	EC10	25	120 days	Marr et al. 1996
Coho salmon		21 – 22	NOEC	24–32	60 days	Mudge et al. 1993
Steelhead		45 to >51	NOEC	24–32	60 days	Mudge et al. 1993
Direct Lethality[e]						
Chinook salmon (fry)	Death	19	LC50	24	96 hours	Chapman 1978
Coho salmon (fry)	Death	28 – 38	LC50	20–25	96 hours	Lorz and McPherson 1976
Steelhead/ rainbow (fry)	Death	9 – 17	LC50	24–25	96 hours	Chapman 1978; Marr et al. 1999

Species (life stage)	Effect	Effect Conc. (µg/L)[b]	Effect Statistic	Hardness (mg/L)[c]	Exposure Duration	Sources
Coho salmon (adult)	Death	46	LC50	20	96 hours	Chapman and Stevens 1978
Steelhead (adult)	Death	57	LC50	42	96 hours	Chapman and Stevens 1978
Coho salmon (juvenile)	Death	21 – 22	NOEC	24–32	60 days	Mudge et al. 1993
Steelhead (juvenile)	Death	24 – 28	NOEC	24–32	60 days	Mudge et al. 1993
Steelhead (egg-to-fry)	Death	11.9	EC10	25	120 days	Chapman 1982

a Abbreviations: LOEC = Lowest observed adverse effect concentration (and most LOEC values given are not thresholds, but were simply the lowest concentration tested); NOEC = No observed adverse effect concentration; LC50 = the concentration that kills 50% of the test population; ECp = effective concentration adversely affecting (p) percent of the test population or percent of measured response, e.g., 10% for an EC10, etc.; and Indefinite = field exposures without defined starting and ending times. NA = not applicable.
b Effects and exposure durations stem from laboratory and field experiments, therefore in some experiments multiple routes of exposure may be present (i.e., aqueous and dietary) and water chemistry conditions will likely differ (see reference for details).
c Hardness is reported, as it can influence the toxicity of copper.
d This study examined ecosystems consisting of a number of species or unidentified species.
e Acute sensitivity of salmonids to copper probably varies by life stage, and the swim-up fry stage is probably more sensitive than older juvenile life stages such as parr and smolts or adults.

Using EOG measurements in combination with a predator avoidance assay, Sandahl et al. (2007) presented the first evidence that impaired olfaction resulted in a direct suppression of predator avoidance behavior (alarm response) by juvenile coho salmon at environmentally relevant dissolved copper exposures (≥2.0 µg/L; 3 hour exposure). Another fish sensory system, the lateral line, is also a target for the neurotoxic effects of dissolved copper. It's composed of mechanosensory neurons (hair cells) that respond to surface water vibrations, flow, and other types of mechanical cues in the aquatic environment. The lateral line system thereby mediates shoaling, pursuit of prey, predator avoidance, and rheotaxis (orientation to flow). In a recent study, dissolved copper killed 20% of hair cells in zebrafish following 3 hours of exposure to ≥20 µg/L (Linbo et al. 2006, as cited in Hecht et al. 2007).

Review of water quality data in Big Deer Creek show seasonal increases in copper during peak runoff that have been decreasing over time. However, these peaks appear to reflect increases in total copper with the dissolved fraction remaining relatively constant. Since total copper is not bioavailable to fish, these seasonal peaks are not anticipated to affect ESA-listed salmonids.

In addition, prior to the commencement of discharge, the permittee must provide a copper loading demonstration plan demonstrating to EPA and IDEQ that there will be no net increase in copper mass loading to the Big Deer Creek watershed as a consequence of mining activity. This demonstration must be approved by IDEQ. As described in the State's certification of the permit, the permittee must demonstrate to IDEQ, prior to the commencement of discharge, that there will be no net increase in copper mass loading to the Big Deer Creek watershed as a consequence of mining activity. Because Big Deer Creek is listed as a "high priority" waterbody under the total maximum daily load program, this requirement is necessary in order to comply with State water quality standards at IDAPA 58.01.02.054.04. Prior to discharge, the permittee must prepare a written plan that: (1) describes the measures that will be implemented (if any) to ensure that, notwithstanding the addition of copper from the discharge, the total mass load of copper remains constant or decreases in the Big Deer Creek watershed; and, (2) includes a schedule for the implementation of these measures. The written plan must be submitted to EPA and the IDEQ regional office. The plan must be approved by IDEQ prior to discharge, and implemented in accordance with the approved plan.

The EPA has identified acute water quality criteria for copper at 4.6 µg/L, and chronic water quality criteria at 3.5 µg/L. The NPDES permit limits effluent to 4.8 µg/L maximum daily load, and 2.4 µg/L average monthly load. The FCC estimates effluent concentrations of copper of < 2.0 µg/L at the end-of-pipe. Both the effluent limit and estimated concentration at the outfall are below EPA's acute and chronic water quality criteria. At the end-of-pipe, maximum concentrations allowed by the NPDES permit limits are below the 96-hour LC_{50} values of 9 to 57 µg/L previously discussed, but above the 0.75 µg/L concentrations resulting in avoidance, the 1.9 µg/L concentrations affecting growth, and within the 5 to 25 µg/L range where copper concentrations have been shown to affect migration and spawning behaviors. These concentrations are also in excess of the 5 µg/L level where copper has been shown to begin to result in loss of invertebrate taxa. Considering dilution, mixing, and background concentrations of 6.9 µg copper/L in Big Deer Creek, copper concentrations instream are estimated to be approximately 6.6 µg/L downstream of the falls where anadromous fish reside (Table 11). However, because FCC will be required meet the no net increase standard for copper and submit a copper loading plan to achieve this standard for approval prior to first discharge, NMFS does not expect concentrations of copper contributed from the effluent to result in direct or indirect effects to Chinook salmon or steelhead in Big Deer Creek.

Lead – Lead is a relatively rare mineral (Eisler 1998b), occurring naturally as the mineral galena (Nelson et al. 1991). Lead is neither essential nor beneficial to living organisms, negatively affecting survival, growth, reproduction, development, behavior, learning, and metabolism (Eisler 1998b). Lead has high acute toxicity to aquatic life and birds (Eisler 1998b; Vincoli 1997; Nagpal 1987). It and its compounds are soluble in water, with bioavailability increasing in waters with low pH, low organic content, and low metal salt content (USDA Forest Service 2007). Background concentrations of lead in Big Deer Creek are currently estimated at 0.00062 µg/L (Table 11). Lead is highly persistent in water, with a half-life >200 days (Vincoli 1997). Waste associated with lead mining activities has directly affected fish through toxic exposure and indirectly affected fish through poisoning of prey items (Eisler 1998b).

Lead is most often precipitated to sediments in aqueous environments. Adsorption of lead by aquatic animals is affected by the age, gender, and diet of the organism, as well as the particle size, chemical species and presence of other compounds in the water. Species sensitive to lead are more affected by dissolved lead than total lead (USDA Forest Service 2007). In aquatic organisms, concentrations of lead are greatest in algae and benthic organisms and lowest in predators; showing no real evidence of biomagnification in the food chain (Eisler 1998b). Nelson et al. (1991) reported that toxicity of lead to rainbow trout is inversely related to oxygen concentration. Toxic effects are more pronounced at higher water temperatures, reduced pH, in comparatively softer waters, in younger life stages, and after longer exposures (Eisler 1998b). Mortality of fishes from lead occurs from exposure to waters containing lead salts which causes resulting from a suffocating film of coagulated mucus to form over the body and gills (Nelson et al. 1991). Like cadmium, lead is a calcium antagonist and neurotoxin and has been documented to have significant effects on behavior (Sloman et al. 2003).

Lead is more toxic to fish in soft water than hard water, although 96-hour LC_{50}s vary greatly (Nagpal 1987). For rainbow trout, Nagpal (1987) cited studies by Davies et al. (1976) where 96-hour LC_{50} acute toxicities ranged from a high of 471,000 µg/L in hard water (290 mg/L $CaCO_3$) to a low of 1,179 µg/L in soft water (32 mg/L $CaCO_3$). Hale (1977) reported a 96-hour LC_{50} of 8,000 µg/L for rainbow trout. Nagpal also summarized results for studies conducted by Holcombe et al. (1976) with brook trout in soft water (44 mg/L $CaCO_3$), where 96-hour LC_{50} acute toxicities ranged from 4,100 µg/L total lead to 3,362 µg/L dissolved lead. However, when exposed to tetramethyllead, Eisler (1998b) reported acute toxicities to rainbow trout at only 3.5 µg/L.

Fish chronically exposed to waterborne lead exhibit various signs of lead poisoning, including: (1) Spinal curvature, (2) anemia; (3) darkening of the dorsal tail region; (4) degeneration of the caudal fin; (5) destruction of spinal neurons; (6) ALAD (delta-aminolevulinic acid dehydratase) inhibition in blood, bone, gill, liver, and renal tissues; (7) reduced ability to swim against a current; (8) destruction of the respiratory epithelium; (9) basophilic stippling of erythrocytes; (10) elevated lead concentrations in blood, bone, gill, liver, and kidney; (11) muscular atrophy; (12) paralysis; (13) renal pathology; (14) growth inhibition; (15) retardation of sexual maturity; (16) altered blood chemistry; (17) testicular and ovarian histopathology; and (18) death (Eisler 1998b).

Davies et al. (1976) reported chronic effects in the form of black tails and spinal deformities in early life-stage (embryo to 19 month) rainbow trout in soft water (28 mg/L $CaCO_3$). Pre-hatch rainbow were affected at concentrations between 4.1 and 7.6 µg/L, while effects were noted in fry at concentrations as low as 7.2 to 14.6 µg/L. Only 5% of the fish were reportedly affected at these lower concentrations. However, 32% of the fish were reported to have spinal deformities where rainbow trout were exposed to lead concentrations of 27 µg/L. While studying long-term effects on rainbow trout fry and fingerlings, Davies et al. (1976) noted that rainbow developed spinal deformities in soft water at much lower lead concentrations than in hard water. In soft water (28 mg/L $CaCO_3$), spinal deformities were observed in 44% of the fish exposed to 31 µg/L and 97% of the fish exposed to 62 µg/L. In hard water (353 mg/L $CaCO_3$), no spinal deformities were observed in fish exposed to 190 µg/L, while 10% of the fish exhibited spinal deformities when exposed to 380 µg/L.

Two additional studies cited by Nagpal (1987) suggested similar results for spinal deformities. Holcombe et al. (1976) noted spinal deformities in brook trout exposed to concentrations of 58 to 119 µg/L (total lead), and at 39 to 84 µg/L (dissolved lead) (hardness 44 mg/L $CaCO_3$). Two month exposure of juvenile rainbow trout to lead resulted in spinal deformities at concentrations ranging between 48 and 83 µg/L lead (hardness 34 mg/L) (Sauter et al. 1976, as cited in Nagpal 1987).

Lead has been shown to bioconcentrate in aquatic species, accumulating more in invertebrates than vertebrates. Inorganic lead is poorly accumulated in fish, with larger organic lead compounds such as tetraalkyllead more toxic than smaller compounds like trialkyllead (USDA Forest Service 2007). In vertebrates, concentrations of lead tend to localize in hard tissues such as bone or teeth (USDA Forest Service 2007; Eisler 1998b), but can also accumulate in liver and kidney tissues (Nelson et al. 1991). Although lead has been shown to concentrate in aquatic species, there is little evidence for biomagnification (USDA Forest Service 2007; Eisler 1998b; Nagpal 1987).

Nagpal (1987) used data by Davies et al. (1976) that identified maximum acceptable lead concentrations for rainbow trout ranging from 4.1 to 7.6 µg/L in setting British Columbia chronic water quality standards for lead exposure. Based on this study, Nagpal recommended a maximum concentration of 4.0 µg/L for moderately soft water (i.e., 20 to 40 mg/L $CaCO_3$).

The acute toxicity of lead to aquatic invertebrates varies tremendously, reported in concentrations ranging between 100 and 100,000 µg/L. Nagpal (1987) reported 48-hour LC_{50}s ranging from 450 µg/L to 1,910 µg/L for Daphnia, with toxicity increasing as water softened. Daphnids were found to be 11 times more sensitive to lead in soft water than in hard, where chronic toxicity limits ranged from 9 to 16 µg/L in soft water (52 mg/L $CaCO_3$), to 85 to 193 µg/L in hard water (151 mg/L $CaCO_3$). However, prey species important to rearing salmonids do not appear to be as sensitive to lead, with 7-day LC_{50}s of 16,000 and 32,000 µg/L in soft water (44 mg/L $CaCO_3$) for the mayfly (*Ephemerella subvaria*) and the caddisfly (*Hydropsyche bettine*), respectively. For other species of stonefly, caddisfly, and mayfly studied, 14-day LC_{50}s in soft water (50 mg/L $CaCO_3$) ranged from 3,500 µg/L to 64,000 µg/L (Nagpal 1987).

The EPA has identified acute water quality criteria for lead at 13.88 µg/L, and chronic water quality criteria at 0.54 µg/L. The NPDES permit limits effluent to 0.90 µg/L maximum daily load, and 0.45 µg/L average monthly load. The FCC estimates effluent concentrations of lead at < 0.3 µg/L at the end-of-pipe. Both the effluent limit and estimated concentration at the outfall are below EPA's acute and chronic water quality criteria. At the end-of-pipe, maximum concentrations allowed by the NPDES permit limits are well below the lowest 96-hour LC_{50} acute values, and below the 3.5 and 4.1 µg/L previously discussed for the onset of chronic toxicities to rainbow trout and aquatic invertebrates. Furthermore, considering dilution, mixing, and background concentrations of 0.00062 µg lead/L in Big Deer Creek, lead concentrations instream are estimated to be even lower at approximately 0.03 µg/L downstream of the falls where anadromous fish reside (Table 11). Therefore, NMFS does not expect concentrations of lead contributed from the effluent to result in direct or indirect effects to Chinook salmon or steelhead in Big Deer Creek.

<u>Mercury</u> – Mercury and its compounds have no known normal metabolic function and its presence in living organisms is undesirable and usually hazardous. Mercury contamination in the environment has been associated with; (1) mining and smelting operations; (2) fungicides used in agriculture; (3) the manufacturing of chlorine and sodium hydroxide; (4) use as a slime control agent in the pulp and paper milling processes; (5) the production of plastics and electrical operations; (6) the byproduct of producing electricity from coal; and (7) careless waste disposal practices (including incineration and landfilling of certain mercury-containing products) (Eisler

1987). Mercury is insoluble but highly persistent in water, with a half-life > 200 days. Once released into water, microorganisms convert mercuric salts into methyl-mercury, which is then taken up by algae and enters the food chain (Vincoli 1997).

Mercury can enter the aquatic environment from aerial deposition, surface runoff and spills, and via contaminated groundwater. Elevated levels of mercury in living organisms in mercury-contaminated areas may persist for as long as 100 years after the source of pollution has been discontinued (Eisler 1987). While the residence time of atmospheric mercury is relatively short at about 11 days, it's relatively much longer in oceanic waters, soils, and sediments (at least 1,000 years). In general, the number of mercury-contaminated fish and wildlife habitats has progressively increased, almost all as a direct result of anthropogenic activities. Eisler (1987) reported that mercury levels in river sediments have increased fourfold since pre-cultural times, and twofold to fivefold in sediment cores from lakes and estuaries. Background concentrations of mercury in Big Deer Creek are currently estimated at 0.00005 µg/L (Table 11).

Mercury is a heavy metal, which as an element is a liquid near room temperature and pressure. Mercury is a known mutagen, teratogen, and carcinogen, causing embryocidal, cytochemical, and histopathological effects. Forms of mercury with relatively low toxicity, such as insoluble mercuric sulfide, can be transformed into forms of very high toxicity such as methylmercury or mercuric chloride by some biological processes (Eisler 1987). Microorganisms can covert inorganic and organic mercury compounds into highly toxic methylmercury or dimethylmercury in the water column or sediments and under aerobic or anaerobic conditions (Eisler 1987). In contaminated waters, almost all mercury in fish is methylmercury (Nagpal 1989). Nutrient content of the water, pH, redox conditions, suspended sediment load, sedimentation rates, and microbial activity all affect the methylation of mercury in aquatic ecosystems (Eisler 1987).

Inorganic and methylmercury have high acute and chronic toxicities to aquatic life. Chronic effects can include shortened life span, reproductive problems, lower fertility, and changes in appearance or behavior in exposed animals. These effects can be seen long after first exposure(s) to mercury (Vincoli 1997). In testing of rainbow trout and inorganic mercury, acute 96-hour LC_{50}s ranged from 155 to 200 µg/L for juvenile fish, to 28-day LC_{50}s of 4.7 µg/L and <0.1 µg/L for embryo-larva in static and flow through tests respectively. For organic mercury, acute 96-hour LC_{50}s ranged from 5 to 42 µg/L for juvenile rainbow trout, to 24 µg/L for rainbow trout larva, to 65 µg/L for juvenile brook trout (Eisler 1987). Hale (1977) reported a 96-hour LC_{50} of 33.0 µg mercury/L for rainbow trout. In yearling coho salmon, Lorz et al. (1978) reported a 96-hour LC_{50}s of 240 µg/L for inorganic mercury, and 38.9 µg/L for methylmercury. Additional studies have reported acute toxicities for rainbow trout exposed to mercuric chloride ranging from 10 µg/L to 1,000 µg/L (Illiopoulou-Georgudaki and Kotsanis 2001).

Mercury has a tendency to both bioconcentrate (Vincoli 1997) and biomagnify through food chains (Eisler 1987). Fish and shellfish have a strong tendency to concentrate mercury in their bodies, often in the form of methylmercury, a highly toxic organic compound of mercury. Because of biomagnification, species of fish high in the food chain (e.g., piscivorous fishes like northern pike, bull trout, northern pike-minnow, and lake trout), contain higher concentrations of mercury than herbivorous or insectivorous fishes. Consequently, salmon and most trout species are characterized by relatively lower levels of mercury in their tissues. At relatively low concentrations, mercury has been shown to affect reproduction, growth, behavior, metabolism, blood chemistry, osmoregulation, and oxygen exchange of marine and freshwater biota. Organomercury compounds, especially methylmercury, are significantly more likely to result in sublethal effects and/or bioaccumulation than inorganic mercury (Eisler 1987).

Bioaccumulation of mercury is markedly enhanced by higher water temperatures, softer water, reduced pH, organism age, reduced organic content of the water, and in the presence of zinc, cadmium, or selenium in solution. Summarizing a study by Ribeyre and Boudou (1984) 30-day exposure to 0.1 µg/L methylmercury, Eisler (1987) reported increased bioconcentration factors ranging from 28,300 in the brain to 238,000 in the spleen. Although whole body levels up to 100 mg/kg are reportedly not lethal to rainbow trout, levels of 20 to 30 mg/kg have been associated with reduced appetite, loss of equilibrium, and hyperplasia of gill epithelium. However, brook trout were more sensitive, showing a toxic response and death by whole body residues of only 5 to 7 mg/kg mercury (Eisler 1987).

Significant adverse sublethal effects of mercury have been observed among various aquatic species at water concentrations as low as 0.03 µg/L. In trout, reductions in growth of rainbow trout occurred when exposed to mercury for 64-days at concentrations of 0.04 µg/L methylmercury, and 0.11 µg/L of phenylmercury. Brook trout growth was affected when exposed to 0.79 µg/L organomercury for 21 days, and enzyme disruption occurred in embryos immersed for 17 days in water containing 0.88 µg/L methylmercury (Eisler 1987).

Nagpal (1989) summarized both acute and chronic toxicity data for macroinvertebrates. Acute toxicity for invertebrates was found to be dependent upon species, developmental stage, and overall environmental conditions. For inorganic mercury, *Daphnia* sp. were found to be the most sensitive invertebrate, with LC_{50}s of 1.4 to 4.4 µg/L for *D. magna* and 2.2 µg/L for *D. pulex*. In chronic toxicity tests of inorganic mercury and *D. magna,* adverse effects occurred at concentrations ranging from 0.72 to 1.82 µg/L. In chronic tests for methylmercury, adverse effects were observed in *D. magna* at concentrations less than 0.04 µg/L.

The EPA has not identified an acute water quality criterion for mercury, but has identified a chronic water quality criterion at 0.012µg/L. The NPDES permit limits effluent to 0.02 µg/L maximum daily load, and 0.01 µg/L average monthly load. The FCC estimates effluent concentrations of mercury at 0.0018 µg/L daily maximum, and 0.001 µg/L average daily value at the end-of-pipe. Both the chronic limit and estimated concentration at the outfall are at or below EPA's chronic water quality criteria. At the end-of-pipe, concentrations allowed by the NPDES permit limits are well below the lowest 96-hour LC_{50} values of 0.1 µg/L previously discussed for the onset of acute toxicities to rainbow trout, and below the 0.03 µg/L value reported for onset of chronic toxicities for various aquatic species. Furthermore, considering dilution, mixing, and background concentrations of 0.00005 µg mercury/L in Big Deer Creek, mercury concentrations instream are estimated to be even lower at approximately 0.001 µg/L downstream of the falls where anadromous fish reside (Table 11). Consequently, NMFS does not expect concentrations of mercury contributed from the effluent to result in direct or indirect effects to Chinook salmon or steelhead in Big Deer Creek.

Nickel – Elemental nickel is one of the most common metals found in surface waters, occurring naturally due to the weathering of rocks. Nickel is a transition metal, and is generally considered to be an essential macronutrient (Pane et al. 2003). Anthropogenic sources of nickel include mining, combustion of coal, petroleum and tobacco, manufacture of cement and asbestos, food processing, textile and fur fabrication, laundries, car washes, electroplating, and smelting (Forest Service 2007; Vincoli 1997). Background levels of nickel in unaltered freshwater waterbodies are generally in the range of 1 to 10 µg/L, but can range as high as 1,000 µg/L in highly contaminated waterbodies (Pane et al. 2003; Eisler 1998b). Background concentrations of nickel in Big Deer Creek are currently estimated at 0.001 µg/L (Table 11). Unlike other divalent metals, nickel has not been well studied in terms of toxicity, mode of action, or bioavailablity (Keithly et al. 2004; Brix et al. 2004).

Nickel is a confirmed carcinogen in animals causing lung and nasal tumors. It has also been shown to cause teratogenic and mutagenic effects in test animals. Nickel is highly persistent in water, with a half-life >200 days. Both it and its compounds have high acute toxicity to aquatic life, with toxicity increasing as water softens (Vincoli 1997). Recent studies suggest that acute toxicity to nickel is primarily due to issues with respiration in rainbow trout, occurring at levels approximating the 96-hour LC_{50} (Pane et al. 2003; Brix et al. 2004).

Hale (1977) reported a 96-hour LC_{50} for nickel of 35,500 µg/L for rainbow trout. More recently, Pane et al. (2003) identified a 96-hour LC_{50} for rainbow trout and nickel at 15,300 µg/L in moderately hard water (approximately 140 mg/L $CaCO_3$). This value is similar to 96-hour LC_{50}s reported for coho salmon alevins (16,700 µg/L) and juveniles (18,000 µg/L), and rainbow trout alevins (25,100 µg/L), juveniles (7,800 – 10,900 µg/L), and adults (31,700 µg/L) from earlier studies (Eisler 1998b). Referencing recent studies for rainbow trout, Brix et al. (2004) reported a geometric mean median LC_{50} of 13,000 µg/L. In their own study, they arrived at a 96-hour LC_{50} concentration of 20,800 µg/L (hardness of 91 mg/L $CaCO_3$) for nickel and rainbow trout. The authors felt that this result was comparable to that reported by Nebeker et al. (1985), who had reported a 96-hour LC_{50} of 10,000 µg/L (33 mg/L $CaCO_3$) for nickel and rainbow trout. Normalizing these two studies to a hardness of 50 mg/L, results in 96-hour LC_{50}s of 12,800 and 14,200 µg/L respectively.

Nickel also bioconcentrates, with nickel concentrations in fish expected to be somewhat higher than the average concentration in the water from which the fish was taken (Vincoli 1997). However, there is little evidence that biomagnification of nickel occurs along food chains (Eisler 1998b). The USFS (2007) reported that chronic effects have been noted for fish in soft water at concentrations as low as 2,000 µg/L. Giattina et al. (1982) identified an avoidance threshold of roughly 23.9 µg/L for nickel and rainbow trout. In chronic exposures, Brix et al. (2004) summarized studies reporting a NOEC of 35 µg/L and a LOEC of <35 µg/L for nickel and rainbow trout. However, in their own study, Brix et al. (2004) found no significant effects to rainbow trout embryo survival, swim-up, hatching, or fingerling survival and growth when subjected to concentrations up to 466 µg/L. The authors noted that although nickel accumulated on the gill in an exponential manner, it appeared to plateau in trout plasma at concentrations around 118 µg/L.

Pane et al. (2004) exposed rainbow trout to nickel concentrations ranging from 243 to 2,034 µg/L for 42 to 99 days to study chronic effects. No mortalities occurred in chronic exposures up to 394 µg/L, versus 33% mortality in chronic exposures of 2,034 µg/L. Results from this study indicated that the greatest accumulations of nickel in the gill, kidney, and plasma, noting that the plasma was the primary sink for nickel in rainbow trout. Noting that most previous studies exposed fish to nickel concentrations much higher than would be expected to occur in natural stream systems (3,200 to 14,000 µg/L), Pane et al. (2004) exposed rainbow trout to much lower concentrations to observe potential chronic effects. Rainbow trout were exposed to concentrations that fall within the range often found in watersheds heavily impacted by mining and industrial activity (243 to 394 µg/L). Concentrations in this range were reportedly near 1% of 96-hour LC_{50}s reported in previous acute exposure studies. These levels impaired the exercise physiology of rainbow trout, and acting as a limiting stressor by decreasing maximal rates of oxygen consumption during strenuous exercise. Based on these results, the authors determined that respiratory effects of chronic exposure to low levels of nickel were quite subtle and only noticed after strenuous aerobic exercise. In summary, they suggested that chronic impairment of a critical organ like the gill could depress the overall fitness of the fish, potentially leading to impaired predator avoidance, prey capture, and/or migration success.

ESA-listed salmon and steelhead could be indirectly affected through effects to prey items. Long-term exposure of aquatic invertebrates to nickel has been shown to have adverse effects at concentrations as low as 500 µg/L (USDA Forest Service 2007). Keithly et al. (2004) summarized studies reporting NOEC's ranging from 10 to 220 µg/L for *D. magna*, in water hardness ranging from 51 to 205 mg/L. For the amphipod *Hyalella azteca*, the Keithly et al. (2004) reported a 96-hour LC_{50} of 3,045 µg/L, a NOEC of 29 µg/l, and a LOEC of 58 µg/L. However, the caddisfly *C. dubia* was found to be more sensitive to nickel, with 48-hour LC_{50}s ranging between 81 to 400 µg/L, NOEC's ranging from <3.8 to 5.8 µg/L, and LOEC's ranging from <3.8 to 9.6 µg/L (50 to 253 mg/L $CaCO_3$). Toxicity decreased with increasing water hardness. Nickel bioconcentration increased in *H. azteca* with increasing test concentrations.

The EPA has identified acute water quality criteria for nickel at 145 µg/L, and chronic water quality criteria at 16.1 µg/L. The NPDES permit limits effluent to 26.52 µg/L maximum daily load, and 13.22 µg/L average monthly load. The FCC estimates effluent concentrations of nickel at < 5.0 µg/L at the end-of-pipe. Both the effluent limit and estimated concentration at the outfall are below EPA's acute and chronic water quality criteria. At the end-of-pipe, maximum concentrations allowed by the NPDES permit limits are well below the lowest 96-hour LC_{50} value of 7,800 µg/L previously discussed for the onset of acute toxicities to juvenile rainbow trout. They are also below concentrations reported for onset of chronic toxicity to rainbow trout (i.e., avoidance at 23.9 µg/L, NOEC at <35 µg/L). Furthermore, considering dilution, mixing, and background concentrations of 0.001 µg nickel/L in Big Deer Creek, nickel concentrations instream are estimated to be even lower at approximately 0.74µg/L downstream of the falls where anadromous fish reside (Table 11). Although below concentrations acutely and chronically toxic to *C. dubia* by the time the mixed effluent reaches waters occupied by ESA-listed salmonids, effluent up to 26.52 µg/L at the end-of-pipe could result in chronic effects to various prey items for listed salmonids. However, because the effluent will quickly mix and become diluted upon entrance to Big Deer Creek, NMFS does not expect prolonged exposure of aquatic invertebrates to nickel concentrations this high in Big Deer Creek. Consequently, NMFS does not expect concentrations of nickel contributed from the effluent to result in direct or indirect effects to Chinook salmon or steelhead in Big Deer Creek.

Selenium – Selenium is a widely available and naturally occurring metalloid, generally available in the environment due to the weathering of rocks. It's particularly abundant with sulfide minerals of various metals, such as iron, lead, and copper (Eisler 1985b). In uncontaminated waters, selenium is typically found in concentrations of 0.1 to 0.4 µg/L (Palace et al. 2004; Kennedy et al. 2000). Selenium is an essential element, that can bioconcentrate and become toxic at concentrations slightly above the homeostatic requirement (Miller et al. 2007; Eisler 1985b). Selenium is highly persistent in water, with a half-life greater than 200 days (Vincoli 1997).

Selenium and its compounds have high acute toxicity to aquatic life (Vincoli 1997), with the major symptom of toxicity in fish occurring in the form of larval teratogenic deformities (Miller et al. 2007; Palace et al. 2004). Teratogenisis is restricted to the egg-larval stage of development as the larvae utilize selenium-contaminated yolks (Palace et al. 2004). Toxicity of selenium varies among fish species, varying by the form of selenium and the life stage of the fish. It can exist in four oxidation states, with sodium selenite (NA_2SEO_3) generally found to be the most toxic (Miller et al. 2007; Eisler 1985b). Selenite is generally more toxic to early life stages and the magnitude of effects increases with temperature (Eisler 1985b). Other documented effects of selenium on salmonids include decreases in egg incubation period, hatch rate, post swim-up juvenile survival, and juvenile growth (Van Kirk and Hill 2007). Miller et al. (2007) cited studies documenting 96-hour LC_{50}s for rainbow trout ranging from 4,200 to 9,000 µg/L for sodium selenite, and 3,200 to 4,700 µg/L for selenate. Other studies have reported similar 96-hour LC_{50}s for selenium and rainbow trout, ranging from 4,200 to 12,500 µg/L (Eisler 1985b).

Selenium bioconcentrates in fish, with the concentration of selenium found in fish tissues expected to be somewhat higher than the average concentration of selenium in the water from which the fish was taken (Vincoli 1997). When available in both diet and water, teleosts have been shown to accumulate selenium in the liver, kidney, gills (Eisler 1985b), eggs, and muscle (Kennedy et al 2000). However, selenium is generally thought to accumulate in fish tissues more through the diet than through the waterborne exposure, and is later transferred to offspring through the egg (Kennedy et al. 2000; Van Kirk and Hill 2007). Miller et al. (2007) suggested that the most significant effect of exposure to excess selenium occurs in the form of egg accumulation and the subsequent larval deformities. Kennedy et al. (2000) further supported this concept, noting that bioaccumulation of selenium in fish can cause reproductive failure, egg and embryo mortality, and embryonic deformities and malformations. Skin lesions, cataracts, swollen gill filament lamellae, myocarditis, and liver and kidney necrosis have also been documented in fish from chronic exposure to selenium (Miller et al. 2007). Studies suggest that there's an extremely narrow margin between normal and toxic concentrations of selenium in tissues (Kennedy et al. 2000).

For chronic exposure, anemia and reduced hatch success of rainbow trout has been documented at selenium concentrations ranging from 47 to 83 µg/L, while growth inhibition has been documented after 21 days of exposure to 250 µg/L selenium (Eisler 1985b). Because of the bioaccumulative nature of selenium, transferring selenium from the female parent to offspring through the egg (Kennedy et al. 2000; Van Kirk and Hill 2007), it has been suggested that the concentration of waterborne selenium is not a good predictor of selenium toxicity to fish. Replacement of EPA's selenium freshwater criterion by whole-body tissue concentration has been recommended and is currently being evaluated by EPA (Van Kirk and Hill 2007). Currently, EPA's draft freshwater chronic criterion is expressed as a concentration in whole-body fish tissue of 7.91 µg/g, dry weight. If fish tissue samples exceed 5.85 µg/g during summer or fall, fish should be monitored during the winter to determine if selenium exceeds 7.91 µg/g (EPA 2004). However, because the concentrations in the draft criterion were based on juvenile bluegill, a debate currently exists on whether these concentrations can be directly related to salmonids. Mortality has been reported for rainbow trout, Chinook salmon, striped bass (*Morone saxatilis*), and bluegill (*Lepomis macrochirus*) when selenium tissue concentrations increase to 4 to 10 µg/g (Kennedy et al. 2000). The authors, citing other research efforts, noted additional effects of selenium concentration on growth at 3 to 6 µg/g, smoltification at 9.5 µg/g, teratogenic effects at 15 µg/g, and effects on reproduction at 8 to 16 µg/g.

It has been found that selenium salts can be converted to methylated forms by microorganisms. These forms are readily accumulated by aquatic invertebrates. At concentrations of 47 to 53 µg/L, selenium has been shown to retard growth of freshwater green algae, and shift the species composition of freshwater algal communities (Eisler 1985b).

Currently, EPA has identified acute water quality criteria for selenium at 20 µg/L, and chronic water quality criteria at 5 µg/L. The NPDES permit does not propose effluent limits for selenium, but FCC estimates effluent concentrations of 4.0 µg/L maximum daily, and 2.0 µg/L average daily at the end-of-pipe. The estimated concentration at the outfall is below both EPA's acute and chronic water quality criteria, which are also well below the lowest 96-hour LC_{50} value of 3,200 µg/L previously discussed for the onset of acute toxicities to juvenile rainbow trout. These concentrations are also below those reported for onset of chronic toxicity to rainbow trout (i.e., anemia and reduces hatch success at 47 µg/L). However, because of its tendency to bioaccumulate, it's unclear whether the water quality criteria established for selenium are low enough to sufficiently keep selenium from accumulating in steelhead and/or Chinook salmon tissue. Tissue samples from non ESA-listed resident salmonids downstream from the discharge and upstream from the falls would provide a better indicator of the potential effects of selenium on ESA-listed anadromous fish. Although there is some degree of uncertainty regarding indirect effects of selenium on salmon and steelhead in Big Deer Creek, stream surveys suggest that Chinook are absent and steelhead are only present in low numbers in lower Big Deer Creek (Kuzis 2004). Because indirect effects, if any, would be localized only to individual fish in lower Big Deer Creek, it's highly unlikely that bioaccumulation will rise to a level sufficient to appreciably reduce species abundance/productivity at the larger population, MPG, or ESU scales.

<u>Thallium</u> – Thallium is naturally found in trace amounts in the earth's crust. Thallium and its compounds may enter the environment during mining and smelting operations, through industrial discharges, or spills (Vincoli 1997). As a byproduct of burning coal and smelting other metals,

such as cobalt, thallium is a trace contaminant of the raw materials and may be released into the environment. Once released into the environment, thallium remains in air, water and soil as a contaminant for a long time and is not broken down (Agency for Toxic Substances and Disease Registry [ATSDR] 1995). Thallium is insoluble in water, but readily forms soluble compounds when exposed to air or water (Vincoli 1997). Water concentrations of thallium in rivers in the United States and Canada that receive mining operations effluents range from 0.7 to 88.3 µg/L (ATSDR 1992; Pickard et al. 2001). Thallium concentrations in tailings ponds in New Brunswick, Canada ranged from 27 to 1,620 µg/L (Zitko et al. 1975). Background concentrations of thallium in Big Deer Creek are currently estimated at 0.00001 µg/L (Table 11).

Thallium is a non-volatile, heavy metal that was used in the manufacture of rat, ground squirrel, roach, and ant poisons (ATSDR 1992). Thallium and its compounds have high acute toxicity to aquatic life (Vincoli 1997). In general, aquatic invertebrates are more sensitive to thallium than fish, with a 96-hour LC_{50} for daphnids at 2,200 µg/L, while freshwater fish demonstrate a 96-hour LC_{50} of 120 mg/L (WHO 1996). Zitko et al. (1975) found information was limited regarding the toxicity of thallium to aquatic fauna, but summarized earlier studies reporting mortality of fish and aquatic invertebrates after 72 hours of exposure to thallium concentrations of 10,000 to 60,000 µg/L, and 2,000 to 4,000 µg/L respectively. Although Pickard et al. (2001) estimated a 96-hour LC_{50} of 3,200 µg/L for thallium and rainbow trout, they found 100% mortality of rainbow trout in 24-hour exposure to concentrations as low as 2,500 µg/L. Rainbow trout mortality was also noted in their study at concentrations as low as 1,560 µg/L, the lowest concentration used in their study.

For the fathead minnow (*Pimephales promelas*), LeBlanc and Dean (1984) reported a 96-hour LC_{50} of 860 µg/L for thallium. In that same study, the authors also reported that no fathead minnow embryos survived thallium concentrations >720 µg/L. Survival increased to 22% at 350 µg/L and 75% at 200 µg/L. Embryos were not affected at thallium concentrations <200 µg/L. Post-hatch, no fathead minnow larvae survived concentrations over 350 µg/L, and larvae survival was significantly reduced at concentrations as low as 40 µg/L (LeBlanc and Dean 1984).

According to the ATSDR, thallium is a known carcinogen that bioaccumulates in fish and shellfish (ATSDR 1995). The bioconcentration factors for thallium in aquatic organisms are presented in Table 13, with the highest factor of 1,430 reported for juvenile Atlantic salmon (*Salmo salar*) (Zitko et al. 1975) and 50,000 for the freshwater diatom, *Stephanodiscus hatzschii*. On a weight basis, thallium may be as toxic to fish and other aquatic organisms as copper (Zitko et al. 1975).

According to Twiss et al. (2004), bioconcentration factors for thallium vary with the species of freshwater phytoplankton, the type of thallium compounds in the water, and the concentration of potassium in the water affected. This study found that: (1) *Chlorella* species concentrated less thallium than freshwater diatoms; (2) inorganic thallium had higher bioconcentration factors than organic thallium compounds; and (3) bioconcentration factors for both species of freshwater phytoplankton declined as potassium concentrations in the water increased.

Table 13. Bioconcentration factors for thallium in aquatic organisms.

Aquatic Taxon	Bioconcentration Factor	Reference
Marine clams	18.2	Zitko and Carson 1975
Marine mussels	11.7	Zitko and Carson 1975
Atlantic salmon (juveniles)	27 - 1,430	Zitko and Carson 1975
Bluegill	34	Barrows et al. 1978
Higher Plants (Riparian & Aquatic)	0.05-594	Cataldo and Wildung 1983; Wallwork-Barber et al. 1985
Freshwater Phytoplankton	780 - 50,000*	Twiss et al. 2004

* Depends on species of phytoplankton and concentration of potassium in the water

Thallium is readily absorbed by plants and can enter the food chain, including aquatic food chains (Cataldo and Wildung 1983; Wallwork-Barber et al. 1985). Heit and Klusek (1985) demonstrated that concentrations of thallium in the omnivorous white sucker (*Catostomus commersoni*), carnivorous yellow perch (*Perca flavescens*), and brook trout were similar in their axial muscle tissue (<0.07 to 3.0 mg/kg dry weight), independent of the water pH. In New Brunswick, Canada, Atlantic salmon in waters contaminated by mining wastewater contained from 1.2 to 89 mg thallium/kg fish tissue, with the highest concentration found in the gills (Zitko et al. 1975).

Sublethal effects of thallium include deleterious effects to the prey supply of juvenile anadromous salmonids. Using a rotifer (*Brachionus calyciflorus*), Zeleznock (2004) calculated LD_{50}s for thallium exposures at different concentrations of potassium. Increased levels of potassium saturate the uptake sites in the rotifer cell membranes and protect the aquatic invertebrate from toxic thallium intake. At 8 µg/L, thallium can reduce the growth of aquatic plants including algal cells, the base of the aquatic food web (WHO 1996).

The EPA has not identified an acute water quality criterion for thallium, but has identified a chronic water quality criterion of 10 µg/L. The NPDES permit limits effluent to 0.95 µg/L maximum daily load, and 0.47 µg/L average monthly load. The FCC estimates effluent concentrations of thallium at < 0.2 µg/L at the end-of-pipe. Both the effluent limits and estimated concentration at the outfall are well below EPA's chronic water quality criteria. At the end-of-pipe, maximum concentrations allowed by the NPDES permit levels are well below effect levels reported for acute or chronic adverse effects to aquatic biota summarized in this analysis. Furthermore, considering dilution, mixing, and background concentrations of 0.00001 µg thallium/L in Big Deer Creek, thallium concentrations instream are estimated to be approximately 0.03 µg/L downstream of the falls where anadromous fish reside (Table 11). Therefore, NMFS has concluded that levels of thallium from the effluent should not rise to concentrations that will result in direct, indirect, or sublethal effects to ESA-listed salmonids.

<u>Zinc</u> - Zinc is a naturally occurring, common metallic element that exists as a variety of salts. Solubility ranges from low to high, depending on the salt formed (Vincoli 1997), the pH, and the alkalinity (Eisler 1993). Most zinc salts are both highly soluble and persistent in water, with half-lives >200 days (Vincoli 1997). Zinc is a trace element, essential to all living biota (Holcombe et al. 1979; Eisler 1993). Background zinc concentrations rarely exceed 40 µg/L in natural waters, but can be found as high as 99 mg/L in heavily contaminated waters. Most zinc

introduced into aquatic systems is eventually partitioned into the sediment, where its release is enhanced by conditions of high DO, low salinity, and low pH (Eisler 1993). Background concentration of zinc are estimated to be 0.006 µg/L in Big Deer Creek (Table 11).

Zinc and its salts have high acute toxicity to aquatic life (Vincoli 1997). Zinc speciates into the toxic aquo ion $[Zn(H_2O)_6]^{2+}$, other dissolved chemical species, and various inorganic/organic complexes. Zinc is most toxic to aquatic species in softer water (Vincoli 1997), and conditions of low pH, low alkalinity, low DO, and higher stream temperatures (Eisler 1993). Zinc uptake can occur via the gill or through the digestive tract (Köck and Bucher 1997; Galvez et al. 1998). The gill epithelium can be physically damaged by zinc, and is the primary target of zinc toxicity in fish (Skidmore 1970; Eisler 1993).

In soft water (20 mg/L $CaCO_3$), Sprague et al. (1965) identified an ILL for rainbow trout of 600 µg Zn/L. However, salmonids may be able to at least partially acclimate to increased concentrations of zinc (Chapman 1978; Stubblefield 1999; Alsop et al. 1999). At levels acutely toxic to non-acclimated fish, Chapman (1978) noted markedly decreased mortality in sockeye salmon (*O. nerka*) acclimated to 242 µg/L zinc. Stubblefield et al. (1999) reported similar results, documenting an ILL of 695 µg/L for non-acclimated adult rainbow trout at 131 hours of exposure, versus 2,025 µg/L at 168 hours for fish acclimated at half the ILL or 324 µg/L. After 30 days of exposure to concentrations ranging from 50 to 450 µg/L of zinc (hardness 20 to 120 mg/L $CaCO_3$), all rainbow trout acclimatized with no apparent effect on growth, whole-body sodium or calcium concentrations, zinc tissue levels, metabolic rate, or fixed velocity swimming performance (Alsop et al. 1999). However, critical swimming velocity (U_{Crit}) was significantly reduced in fish exposed to zinc.

In a review of existing literature, Eisler (1993) described an even wider range of acute toxicities to zinc for freshwater fish, describing 96-hour LC_{50}s ranging from 60 µg/L to 40,900 µg/L. Hale (1977) reported a 96-hour LC_{50} for zinc at 550 µg/L for rainbow trout, while Lorz and McPherson (1977) reported a 96-hour LC_{50} for zinc at 4,600 µg/L for coho salmon. Chapman (1978) described a 96-hour LC_{50} for zinc at 140 µg/L for rainbow trout, 460 µg/L for Chinook salmon, and 750 µg/L for sockeye salmon. In brook trout, 96-hour LC_{50}s for zinc have been reported at 960 µg/L (Chapman 1978) and 2,000 µg/L (Holcombe et al. 1979). Alsop et al. (1999) found that zinc was 5.4 times more toxic to rainbow trout in soft water than hard water, as evidenced by 96-hour LC_{50}s of 869 µg/L in hard water (120 µg/L $CaCO_3$) and 162 µg/L in soft water (20 µg/L $CaCO_3$).

Several recent studies have found that elevated levels of calcium significantly decrease zinc uptake in both the gill and whole body tissues of rainbow trout (Alsop et al. 1999; Barron and Albeke 2000; De Schamphelaere and Janssen 2004). De Schamphelaere and Janssen (2004) found that toxicity of zinc to rainbow trout was influenced by both pH and concentrations of Ca^{2+}, Mg^{2+}, Na^+, and H^+. Depending upon pH and various concentrations of Ca^{2+}, Mg^{2+}, Na^+, and H^+, the authors reported 96-hour LC_{50}'s ranging from 130 and 2,280 µg Zn/L. In these tests, survival was more sensitive than growth, suggesting that toxicity of zinc is primarily acute in nature.

After exposing three generations of brook trout to elevated levels of zinc, Holcombe et al. (1979) couldn't find any significant harmful effects from zinc concentrations ranging from 2.6 to 534 µg/L. Neither growth nor survival was significantly reduced in first-generation brook trout after 28 weeks of exposure to zinc concentrations of up to 1,360 µg/L. However, the study did conclude that the embryo-larval and early juvenile stages of brook trout were the most sensitive to zinc. As cited in Holcombe et al. (1979), Sinley et al. (1974) similarly found that zinc did not affect the growth of rainbow trout in either hard or soft water at concentrations of 2,200 or 547 µg/L respectively. De Schamphelaere and Janssen (2004) reported a 30-fold variation in zinc toxicity and rainbow trout (NOEC 32.7 – 974 µg/L), where increased concentrations of Ca^{2+}, Mg^{2+}, Na^{+}, and H^{+} decreased chronic toxicity by factors of 12, 3, >2, and 2, respectively. In these tests, calcium, magnesium, and sodium all reduced chronic zinc toxicity. Similarly, Barron and Albeke (2000) also determined that zinc uptake was significantly reduced at the gills in rainbow trout following acclimation or exposure to calcium.

Zinc can bioconcentrate in fish, with the concentration of zinc found in fish tissues expected to be considerably higher than the average concentration of zinc in the water from which the fish was taken (Vincoli 1997). Holcombe et al. (1979) found that the greatest amount of zinc accumulated in the gills, liver, kidney, and opercular bone tissues in brook trout. Bioavailability of zinc from sediments is enhanced in waters with high DO, low salinity, low pH, and high levels of inorganic oxides and humic substances (Eisler 1993).

In a review of existing literature, Eisler (1993) described a wide range of acute toxicities to zinc in freshwater aquatic invertebrates, with 96-hour LC_{50}s ranging from a low of 32 µg/L to a high of 40,930 µg/L. In chronic exposure, a gradual decrease in growth rate has been noted in mayfly larvae (*Epeorus latifolium*) after four weeks of exposure to zinc in concentrations of 30 µg/L (Eisler 1993). Zinc is readily bioaccumulated in aquatic invertebrates, an important food source for rearing juvenile salmonids (Bowen et al. 2006).

The EPA has identified acute water quality criteria for zinc at 36.2 µg/L, and chronic water quality criteria at 36.5 µg/L. The NPDES permit limits effluent to 37.02 µg/L maximum daily load, and 18.45 µg/L average monthly load. The FCC estimates effluent concentrations of zinc at < 10 µg/L at the end-of-pipe. The estimated concentration at the outfall is below EPA's acute and chronic water quality criteria, but the maximum daily load is not. This considered, at the end-of-pipe, maximum concentrations allowed by the NPDES permit limits are well below levels identified above as likely to be acutely or chronically toxic to ESA-listed salmonids or their food source. Furthermore, considering dilution, mixing, and background concentrations of 0.006 µg zinc/L in Big Deer Creek, zinc concentrations instream are estimated to be even lower at approximately 1.0 µg/L downstream of the falls where anadromous fish reside (Table 11). Therefore, NMFS does not expect concentrations of zinc contributed from the effluent to result in direct or indirect effects to Chinook salmon or steelhead in Big Deer Creek.

<u>Mixed Metals</u> – When combined in solution, certain combinations of metals have been shown to have different toxic effects on aquatic biota than when considered individually. In combination, certain combinations of metals can act in an antagonistic, additive, or synergistic manner. Additive toxicity occurs where the overall toxicity of a mixture is exactly equal to that predicted from the individual toxicities of the metals in solution (Finlayson and Verrue 1982). As

described by Nelson et al. (1991), antagonistic metal reactions, which reduce the toxicity of metals primarily through precipitation, result in solutions that are less toxic than simple metal solutions. For example, calcium markedly counteracts the toxic effects of copper, lead, magnesium, and zinc through an antagonistic reaction. Conversely, combinations of metals can also be synergistic, where their joint effect is actually greater than the sum of the separate effects. For example, copper, cobalt, zinc, and cadmium, all of which occur in the effluent, are known to act synergistically. Of the metals in the effluent, the metals of primary concern with potential synergistic effects are cobalt and copper, particularly because copper loading is an issue in the watershed from past mining activities.

In studies with Chinook salmon in soft water (20-22 mg/L $CaCO_3$), mixtures of copper and zinc were shown to be antagonistic, while mixtures of copper/cadmium and zinc/cadmium both acted in an additive manner. Mixtures of the three metals displayed antagonistic toxicity (Finlayson and Verrute 1982). Marr et al. (1998) exposed rainbow trout fry to cobalt, copper, and mixtures of the two metals. In this study, they found that cobalt acted as an antagonist when first mixed with copper, but later acted in an additive or slightly synergistic manner, making it difficult to predict short-term mortality from this mixture.

As described in the BA (USDA Forest Service 2007), behavioral avoidance of metals has been demonstrated both in the laboratory and field at very low concentrations. The avoidance response is a species-specific form of adaptive fish behavior occurring at sublethal concentration levels (Svecevičius 1999). Avoidance behavior has been documented in salmonids to a variety of metals and mixture of metals. However, the response is highly variable depending on water chemistry, species studied, and mixture of metals encountered. Hansen et al. (1999b) reported that behavioral avoidance to copper and cobalt mixtures in soft water differed greatly between rainbow trout and Chinook salmon. Rainbow trout avoided 1.6 µg/L copper and 180 µg/L cobalt individually, but the response was noted for a mixture of the two metals at significantly lower concentrations, a mix of 2.6 µg/L copper and 2.4 µg/L cobalt. Chinook salmon were more sensitive than rainbow trout, avoiding mixtures of 1.0 µg/L copper and 0.9 µg/L cobalt (Hansen et al. 1999b).

Studies in the Clark Fork and Coeur d'Alene Rivers, both of which have been impacted by metals from mining activities, have demonstrated the avoidance response in salmonids. In the Clark Fork River, metal contamination is considered the primary cause of reduced fish populations. Studies of avoidance behavior to metal mixtures found that rainbow trout were more sensitive than brown trout, which in part may explain why rainbow trout populations appear to be more severely affected than brown trout populations (Woodward et al. 1995a; Hansen et al. 1999c). Cutthroat trout (*O. clarki*) avoided a metals mixture of 6 µg/L copper, 0.3 µg/L cadmium, 0.6 µg/L lead, and 28 µg/L zinc (Woodward et al. 1997). Rainbow trout avoided all metal concentrations tested from 10% to 1,000 % of a fixed ratio of ambient metal concentrations (12 µg/L copper, 1.1 µg/L cadmium, 3.2 µg/L lead, and 50 µg/L zinc) (Hansen et al. 1999c). In the Coeur d'Alene River study (Goldstein et al. 1999), adult Chinook salmon avoided the South Fork (mining impacted) versus the North Fork (the control) due to the higher ambient concentration of a mixture of heavy metals; cadmium, lead, and zinc. The findings indicate that natural fish populations will avoid tributaries with high metals contamination.

Fish can also be affected indirectly by the combined effects of metals. As previously stated, metals accumulate in aquatic invertebrates, occurring via uptake across the gills, through the gut, and through adsorption to the exoskeleton (Woodward et al. 1994; Beltman et al. 1999). Consequently, indirect effects are likely to occur to ESA-listed salmon and steelhead feeding on metals-contaminated macroinvertebrates. Woodward et al. (1994) found that both survival and growth were significantly reduced in rainbow trout fed metal-contaminated invertebrates from the Clark Fork River. By day 42, rainbow trout fed contaminated invertebrates were 15% smaller than the control group; by day 91, survival was 50% that of control fish. A follow up study with young-of-the-year rainbow trout and brown trout also revealed reduced growth and elevated whole body metal concentrations following 88 days exposure to simulated Clark Fork River water and a diet of aquatic invertebrates collected from the river (Woodward et al., 1995b). Rainbow trout fed the contaminated diet exhibited constipation and reduced feeding activity.

The potential combined effect of metals is addressed in the EPA draft permit through the requirement for WET testing. WET testing is used to assess aggregate toxic effects to aquatic organisms from all pollutants contained in a facility's wastewater (effluent). The EPA uses WET testing to implement the Clean Water Act's prohibition of the discharge of toxic pollutants in toxic amounts, measuring wastewater's effects on specific test organisms' ability to survive, grow and reproduce. The draft permit requires chronic toxicity testing using standard test organisms, the water flea (*C. dubia*) and the fathead minnow. Toxicity testing is required to be completed twice a year during low flow conditions in September and February. The chronic tests for daphnia reproduction and fathead minnow growth provide a sensitive test for the combined effects of metals and other potentially toxic constituents in the effluent. Although not anticipated to occur, provided NPDES permit effluent limits are met, effects observed in the test species would be an indication of potential chronic effects to ESA-listed salmonids found in Big Deer Creek. The outcome of the test is interpreted in chronic toxicity units (TU_C). TU_C are equal to the reciprocal of the effluent concentration that causes no observable effect in chronic toxicity tests. The draft permit recommends that the limit for this test is 1 TU_C (EPA 2007b).

Ammonia (as N) – Ammonia occurs naturally as a byproduct of metabolism, although high concentrations often occur as a result of anthropogenic inputs (e.g., sewage treatment plants, agricultural runoff, etc.) (Burkhalter and Kaya 1977). Ammonia is highly soluble, but not persistent in water with a half-life of less than two days (Vincoli 1997). Ammonia can be converted to nitrite (NO_2) and nitrate (NO_3) by bacteria, which are forms usable by plants.

Total ammonia includes the sum of both un-ionized ammonia (NH_3) and ionized ammonia (NH_4^+). The toxicity of ammonia solutions is primarily due to NH_3 (Vincoli 1997). Ammonia toxicity is influenced by pH, water temperature, DO concentration, salinity, and the carbon dioxide-carbonic acid equilibrium (MacDonald et al. 1991; EPA 1999). Un-ionized ammonia, which is directly toxic to aquatic organisms, is a problem at higher pH values. At a given temperature, the higher the pH, the greater the amount of un-ionized ammonia will be present for a given amount of total ammonia (NMFS 1999). It’s important to note that NPDES permit limits established by EPA for the ICP are set based on total ammonia.

Concentrations of ammonia acutely toxic to fish may cause loss of equilibrium, hyper-excitability, increased breathing, cardiac output, oxygen uptake, and, in extreme cases, convulsions, coma, and death. At lower concentrations ammonia has many effects on fishes, including a reduction in hatching success, reduction in growth rate and morphological development, and pathologic changes in tissue of gills, livers, and kidneys (EPA 1999, as cited in USDA Forest Service 2007).

When updating the ambient water quality criteria for ammonia, EPA (1999) calculated genus mean acute values (GMAV) and species mean acute values (SMAV) for various genus and species. Expressed in total ammonia nitrogen, EPA reported a GMAV of 21.95 mg/L for the genus *Oncorhynchus*, and SMAV's of 11.23 mg/L for rainbow trout and 17.34 mg/L for Chinook salmon (pH = 8). From the set of GMAVs, EPA calculated a final acute value (FAV) of 14.32 mg N/L for total ammonia. However, because the rainbow trout SMAV is lower at 11.23 mg N/L, the FAV was lowered to this level.

Noting that EPA standard toxicity protocols require testing be conducted on resting, unfed, and undistressed fish, Wicks et al. (2002) performed a series of acute toxicity tests on swimming and resting rainbow trout. Because ammonia tends to become internally elevated during periods of activity, the authors hypothesized that acute LC_{50}s might vary between resting and swimming fish. Swimming rainbow trout were exposed to ammonia levels ranging from 0 to 58 mg N/L, and resting rainbow trout were exposed to levels ranging from 0 to 378 mg N/L. Mortality of both the resting and swimming fish increased linearly with increasing concentrations of ammonia. However, mortality increased much more quickly with increasing ammonia concentrations for swimming fish. The 96-hour LC_{50}s varied significantly between the two groups, dropping from 207 mg N/L in resting trout to 32.4 mg N/L in swimming rainbow.

In Gila trout (O. *gilae*), Fuller et al. (2003) found that all fish survived exposures to ammonia up to concentrations of 4.5 mg/L total ammonia (0.36 mg/L un-ionized ammonia), and recorded a 96-hour median lethal concentration of 5.86 mg/L total ammonia (0.47 mg/L un-ionized ammonia) (pH 8.5, water temperature 59ºF). Examining previous studies, the authors concluded that Gila trout appeared to have similar resistance to ammonia as rainbow trout.

Exposure to sublethal concentrations of ammonia may affect the swimming performance of salmonids (Shingles et al. 2001; Wicks et al. 2002). Muscle fatigue has been linked to ammonia accumulation and could lead to reduced swimming performance in trout (Shingles et al. 2001). Wicks et al. (2002) exposed coho salmon to sublethal levels of un-ionized ammonia (NH_3) to see if the ability to swim would be affected by low levels of ammonia. U_{crit} was recorded for coho salmon exposed to NH_3 levels of 0.02, 0.04, and 0.08 mg/L. Although not noting a significant difference between the U_{crit} of control fish versus those exposed to 0.02 mg/L, swimming performance was significantly reduced at 0.04 and 0.08 mg NH_3/L. The authors also found a correlation between the plasma and U_{crit} of individual fish, noting that plasma ammonia increases as the U_{crit} decreases. Studies by Shingles et al. (2001) documented similar results, noting increased ammonia plasma levels and a significant reduction in U_{crit} for trout exposed to increased concentrations of ammonia. At the pH range required by the NPDES permit (pH 6.5 – 9.0), and a wide range of water temperatures between 50 and 68°F, the 1.6 mg/L average monthly limit required in the NPDES permit correlates to NH_3 levels ranging from approximately 0.0013 mg/L to 0.035 mg/L (EPA 1986). Therefore, because NH_3 levels are expected to remain below the 0.04 mg/L levels previously described as the lowest levels causing sublethal effects in coho salmon, NMFS expects that levels established for total ammonia in the NPDES permit should be adequate to avoid sublethal effects from effluent discharge.

Aquatic invertebrates are generally more resistant to acute ammonia toxicity than fish, with acute toxicity decreasing substantially with decreasing temperatures (EPA 1999). Ammonium is readily taken up by aquatic biota, so an increase in ammonium concentrations tends to diminish rapidly in the downstream direction (MacDonald et al. 1991). However, ammonia is not expected to bioaccumulate in fish (Vincoli 1997).

The EPA has identified acute water quality criteria for total ammonia at 5.6 mg/L, and chronic water quality criteria at 2.34 mg/L. The NPDES permit limits effluent total ammonia to 4.1 mg/L maximum daily load, and 1.6 mg/L average monthly load. The FCC estimates effluent concentrations of ammonia at 1 mg/L at the end-of-pipe. Both the effluent limits and estimated concentration at the outfall are below EPA's chronic water quality criteria. At the end-of-pipe, maximum concentrations allowed by the NPDES permit levels are below those reported for acute and chronic adverse effects to anadromous fish summarized in this analysis. Furthermore, considering dilution, mixing, and background concentrations of 0.022 mg ammonia/L in Big Deer Creek, ammonia concentrations instream are estimated to be approximately 0.1 mg/L downstream of the falls where anadromous fish reside (Table 11). Therefore, NMFS does not expect concentrations of ammonia contributed from the effluent to result in direct or indirect effects to Chinook salmon or steelhead in Big Deer Creek.

<u>Nitrate plus Nitrite</u> - Nitrate will be present in ICP mine drainage water as a residual from explosives used in underground blasting operations. EPA applies the permit limit to the combined form of nitrogen as "nitrate plus nitrite" because nitrite nitrogen is relatively unstable and is easily oxidized to the nitrate form. Nitrite is not expected to be present in ICP wastewaters (USDA Forest Service 2007).

Both oxidized forms of nitrogen, nitrate and nitrite, are known to be toxic to salmonids and other aquatic life as reviewed by Lewis and Morris (1986), Westin (1974), and Kroupova et al. (2005). Elevated ambient nitrite concentrations are also a potential problem for fish since nitrite is actively taken up across the gills in competition with chloride and disrupts multiple physiological functions (Kroupova et al. 2005, as cited in USDA Forest Service 2007). However, nitrite is not expected to be present in surface waters, as it's generally short-lived in natural environments, rarely existing in concentrations toxic to salmonids (Spence et al. 1996). Therefore, this effects analysis will focus on nitrates, the expected form in wastewater.

Idaho Water Quality Standards do not specify a numeric criterion for nutrients since stream and river productivity varies greatly due to natural water chemistry. Instead the Idaho Standards identify a narrative criterion to prevent eutrophication from human sources. The narrative standard states, "*surface waters of the state shall be free from excess nutrients that can cause visible slime growths or other nuisance aquatic growths impairing designated beneficial uses.*" The draft permit refers to the narrative criteria in the Idaho Water Quality Standards, but does not recommend a numeric criterion for interpreting the narrative standard. Implementation of the narrative standard in this fashion assumes that changes in eutrophication can be readily detected in a system that is already impaired by metals and that waste water treatment changes can rapidly be made to remedy the problem. This is not an adequately conservative approach to protect ESA-listed fish. A permit limit based on prevention of impairment needs to be developed for the ICP. The draft permit adopts a numeric criterion for nitrate plus nitrite (100 mg/L), based on protection of agricultural stock water, but this high concentration does not address the indirect effect of nutrient enrichment on aquatic life.

Nitrate toxicity in aquatic animals is caused by inhibition of the oxygen carrying ability of hemoglobin, and by disruption of osmoregulation at high concentrations (McGurk et al 2006). Toxicity to aquatic biota is directly related to increasing nitrate concentrations and exposure times (Camargo et al. 2005). Excessive nitrate concentrations are a concern from both the standpoint of toxicity, which may occur at very high concentrations, and from the standpoint of stimulating excessive algal growth and resulting negative effects on DO and pH regimes (USDA Forest Service 2007). Nitrates at high concentrations negatively impact domestic water supply, livestock drinking water, and are a direct toxicant to aquatic organisms. The maximum concentration of nitrates is 10 mg/L for human drinking water and 100 mg/L for livestock water (EPA 2003b).

Westin (1974) reported 96-hour median tolerance limits of 5,800 mg/L nitrate for Chinook salmon fingerlings, and 6,000 mg/L nitrate for rainbow trout fingerlings. Camargo et al. (2005) reviewed the laboratory studies of nitrate toxicity to fish, invertebrates, and amphibians. The primary source for salmonid toxicity values (rainbow trout, Chinook salmon, cutthroat trout, and coho salmon) cited by Camargo was a study by Kincheloe et al. (1979). The Kincheloe study reported toxicity values in the range of 1.1 to 7.6 mg/L nitrate (no observed and lowest observed effects). However, the validity of the Kincheloe et al. (1979) study was questioned by a recent study, where McGurk et al. (2006) reported toxicity values 2 to 3 orders of magnitude higher than reported by Kincheloe et al. (1979). For example, the most sensitive test reported by McGurk et al. (2006) noted sublethal effects for chronic exposures in the range of 6.25 to 25 mg/L for lake trout (*S. namaycush*) and lake whitefish embryos respectively; most other endpoints were reported at greater than 100 mg/L. McGurk et al. (2006) conclude that a low nitrate concentration that causes toxicity is valid only “for situations of long-term chronic exposure and only during the early life stages of fish. It does not support the use of the 2.9 mg/L nitrate value currently used as a risk benchmark in Canada for situations that involve short-term acute exposures or situations that involve exposures to adult salmonids, which are relatively insensitive to nitrate.”

Nitrates at high concentrations are a concern for toxic effects on aquatic biota, and at relatively low concentrations pose a concern for the deleterious effects associated with nitrogen enrichment. Camargo et al. (2005), in summarizing the literature up to that time, concluded that keeping nitrate levels below the drinking water standard of 10 mg/L would provide safe conditions for most sensitive freshwater species. McGurk et al. (2006) noted 96-hour LC_{50}s of 1,121 mg/L for lake trout swim-up fry, and 1,903 mg/L for lake whitefish swim-up fry. Chronic 130- to 150-day exposure LC_{50}s for embryo to swim-up fry were 190 to 64 mg/L respectively.

However, sublethal effects on developmental timing and fry body size were noted at nitrate concentrations of 6.25 and 25 mg/L. McGurk et al. (2006) further reported that adult salmonids are relatively insensitive to nitrate in comparison to early stages.

Aquatic invertebrates (particularly insects) have been identified as the most sensitive group to acute toxicity from nitrates, although little is known regarding chronic exposure (McGurk et al. 2006). Camargo et al. (2005) estimated short-term safe levels of nitrate of 6.7 – 9.6 mg/L and 4.5 – 6.5 mg/L for two net spinning caddisflies. Otherwise, the reported toxicity values for freshwater invertebrates, as summarized by Camargo, were an order of magnitude higher. In this paper, the authors recommended nitrate levels below 10 mg/L to protect sensitive freshwater biota. Concentrations of nitrate greater than 60 mg/L have been found to kill the tadpoles of many amphibians, although behavior and survivorship for some species has been affected by nitrate concentrations as low as 11 mg/L (Chambers et al. 2001).

An increase in fish production could occur from a small or moderate increase in primary production, potentially becoming beneficial to many forest streams. However, should excess plant growth occur increased plant respiration could deplete oxygen levels and result in adverse effects (MacDonald et al. 1991). Nutrient addition using fertilizers has been used to enhance productivity for aquatic communities expressly to increase salmonid populations; for example, with sockeye salmon in British Columbia (Stockner and Macisaac 1998) or as with the beneficial effect of salmon carcasses in the Pacific Northwest (Cederholm et al. 1999). However, nutrient enhancement efforts have all been accompanied by careful studies to determine the appropriate level of nutrient addition.

Although continuous discharge of nitrate in effluent for extended periods of months or years is generally not toxic, it can result in habitat alteration and changes in abundance and composition of riverine plants and animals depending upon the resulting nutrient concentrations (Chambers et al. 2001). Stimulation of excess algal productivity occurs at a much lower concentration than for potential toxicity. Ferreira et al. (2006) noted that microbial nitrogen demands can apparently be met at relatively low levels in streams, at levels one to two orders of magnitude lower than that found in polluted streams. They further suggested that even minor increases in dissolved nitrogen in streams can cause eutrophication which can lead to significant shifts in microbial dynamics and ecosystem function. Lacking a site-specific criterion, the literature provides some generic values that can be used for developing scenarios for this purpose. Golterman (1975) suggested a generic criterion of 0.3 mg/L for nitrate-nitrogen. Cline (1973), cited by MacDonald et al. 1991, suggested that <0.3 mg/L nitrate would *probably* (emphasis added) prevent eutrophication in forested systems.

The EPA published guidance in 2000 establishing nutrient criteria based on eco-regional specific nutrient conditions, but these recommendations have not been widely adopted by the states, and have not been adopted by the State of Idaho. The concentration for total nitrogen recommended by EPA for the Western Forested Mountains is 0.12 mg/L (EPA 2000), of which roughly 0.08 mg/L would be represented by nitrates (pers. comm. C. Mebane, USGS, March 2008). To get a better understanding of acceptable Idaho-specific levels of Nitrogen or Nitrates, NMFS conducted a review of Idaho watersheds where excess levels of nitrogen were a limiting factor and Total Maximum Daily Load (TMDL) targets had been developed. In the Teton TMDL the target for Moody Creek was 0.3 mg/L (nitrite plus nitrate) to remove excess nutrients. In the American Falls Reservoir TMDL the total nitrogen target is set at 0.85 mg/L. In the Blackfoot River and Jim Ford Creek TMDLs, the nitrogen targets are not to exceed 0.3 mg/L and 0.23 mg/L as total inorganic nitrogen, respectively. NMFS believes that target levels in this range would also be appropriate for application to Big Deer Creek.

The highest nitrate concentrations will occur near the point of discharge, roughly 2.3 miles upstream from the upper distribution limit of ESA-listed anadromous fish. As described in the BA, dilution should be considered when evaluating the potential effects of nitrates since the effect occurs via growth of periphyton on stream substrate and not directly at the pipe outlet. However, because few contributing watersheds occur between the outfall and the falls downstream, nitrate concentrations will not be further diluted at the falls. At this point, the effluent should be well mixed with Big Deer Creek. A margin of safety is provided by the fact that nitrate is readily taken up by aquatic biota, so an increase in nitrate concentrations tends to diminish rapidly in the downstream direction (MacDonald et al. 1991).

Table 14 lists estimated nitrate concentrations in Big Deer Creek directly below the outfall, at Panther Creek below Big Deer Creek, and at Panther Creek at the mouth. Concentrations in this table factor in both the estimated effluent flow volume, and the estimated average and maximum flows.

In minimally disturbed watersheds, background nitrite + nitrate levels in the Salmon River basin range from undetectable levels to 0.225 mg/L (Ott and Maret 2003). Within the action area, DEQ has sampled nitrate + nitrite levels at least once in several locations, with the highest levels found at 0.12 mg/L in Panther Creek and 0.05 mg/L in Bucktail Creek. These levels were used to calculate the estimated total concentration in Table 14.

Table 14 Estimated nitrate concentrations downstream from Outfall at Draft NPDES permit limit of <100 mg/L (modified from BA Table 7-20, USDA Forest Service 2007).

	September Q-80 (cfs)	Dilution	Estimated Nitrate Concentration (mg/L)	Estimated Total Concentration (mg/L)
Outfall Average Flow	0.25	1.0	--	
Big Deer Ck. @ Outfall	6.9	27.6	3.6	3.65
Panther Ck. below Big Deer Ck.	51.1	204.4	0.5	0.62
Panther Ck. @ mouth	70.9	283.6	0.4	0.52
Outfall Maximum Flow	0.33	1.0	--	
Big Deer Ck. @ Outfall	6.9	20.9	4.8	4.85
Panther Ck. below Big Deer Ck.	51.1	154.8	0.6	0.72
Panther Ck. @ mouth	70.9	214.8	0.5	0.62

From a toxicity standpoint, resulting nitrate concentrations should be compared to the range of 6.25 mg/L to 10 mg/L, the lowest level where sublethal effects to fish embryos have been observed (McGurk et al. 2006), and the 10 mg/L safe level recommended by Camargo et al. (2005). For eutrophication, the estimated instream nitrate concentration is compared to 0.3 mg/L, the target recommended by Cline (1973, as cited by MacDonald et al. 1991) to avoid eutrophication in forested streams. At the draft NPDES permit limit of <100 mg/L, nitrate concentration is estimated to be 3.65 to 4.85 mg/L in Big Deer Creek at the outfall, 0.62 to 0.72 mg/L at Panther Creek at the Big Deer Creek confluence, and 0.52 to 0.62 mg/L at the mouth of Panther Creek.

Therefore, predicted nitrate concentrations are not expected to be in the acute or chronically toxic range for anadromous fish in the action area. However, the NPDES permit level of <100 mg/L and concentrations in excess of 0.3 mg/L does not appear to be sufficient to prevent indirect effects related to nutrient enrichment. Although not likely to affect ESA-listed salmonids at the population-scale, anadromous fish could be affected indirectly in Big Deer Creek and Panther Creek as a result of habitat alteration and/or changes in abundance and composition of prey species and/or riverine plants.

Considering the information presented in Table 14 NMFS believes it is necessary to further reduce the maximum daily load to 10 mg/L to lower the risk of nutrient enrichment and prevent effects to sensitive aquatic biota (prey species for ESA-listed salmonids).

Sulfate/Sulfides – Sulfur is a critical element in living matter, it participates in several structural, metabolic and catalytic activities; and is a critical component of photosynthesis. Sulfur compounds are naturally occurring within the ore body at the ICP in the form of metallic sulfide minerals. In contact with aerated water the sulfides are oxidized to sulfate ions. Once liberated, both sulfates and sulfides would be present in wastewater from the mill and in the discharge at the outfall (USDA Forest Service 2007). Sulfate is usually found in low concentrations in most freshwater environments, although it can be found in high concentrations where sulfate-containing ores or anthropogenic activities exist (Davies 2007). Background concentrations of sulfate in Big Deer Creek are currently estimated at 7.3 mg/L (Table 11).

Sulfate toxicity testing generally indicates that invertebrates are more sensitive to sulfate than salmonids. As stated in the BA (USDA Forest Service 2007), BC Research (1998) evaluated toxicity to early life stages of rainbow trout, using 7-day salmonid embryo viability tests, and reported a NOEC of 1,060 mg/L SO_4 and a LOEC of 3,500 mg/L. An EC_{50} (median effective concentration) calculated for embryo viability from this study was 1,477 mg SO_4/L. Other studies have reported 96-hour LC_{50}s for rainbow trout ranging from 5,000 mg/L in soft water (25 mg/L $CaCO_3$), to 9,900 mg SO_4/L in hard water (250 mg/L $CaCO_3$); and 96-hour LC_{50}s for coho salmon ranging from 5,742 mg/L in soft water (25 mg/L $CaCO_3$), to 9,875 mg SO_4/L in hard water (250 mg/L $CaCO_3$) (Singleton 2000).

Davies and Hall (2007) obtained recent LC_{50} values for sulfate toxicity at low hardness (25 mg/L) for sensitive invertebrates which is similar to the hardness values in Big Deer Creek: *H. azteca*: 569 mg/L, *D. magna*: 1, 194 mg/L. Davies (2007) reevaluated previous studies by Frahm (1975) who reported toxic impacts at low concentrations of sulfate to the aquatic moss, *Fontinalis antipyretica*. Davies (2007) concluded that toxicity at the very low concentrations of sulfate reported earlier by Frahm (1975) were in error, and were actually due to the toxicity of the potassium ion. Davies (2007) did not report LC_{50}s, but his results do not show toxicity effects at sulfate concentrations less than 600 mg/L at low hardness values similar to those found in Big Deer Creek.

Soucek (2005) reported acute toxicity values (48- or 96-hour LC_{50}s) for four invertebrates, which ranged from 512 mg/L sulfate for *H. azteca* to 14,134 mg/L for *C. tentans*, and found that hardness had a strong influence on sulfate toxicity for test crustaceans (Soucek 2005; Soucek 2007). There is growing evidence suggesting that some ions, particularly sulfate, are less toxic in harder waters (Davies and Hall 2007; Soucek 2007; Soucek and Kennedy 2005). Davies and Hall (2007) identified 48-hour LC_{50}s for *D. magna* and sodium sulfate increasing from 1,194 mg/L to 3,203 mg/L, with a change in water hardness from 25 to 100 mg/L $CaCO_3$. They also identified 96-hour LC_{50}s in *H. azteca* increasing from 569 mg/L to 5,259 mg/L, with a change in water hardness from 25 to 100 mg/L $CaCO_3$. Soucek (2007) documented *H. azteca* sulfate LC_{50}s increasing from less than 1,900 mg/L in 100 mg/L hardness, to over 4,000 mg/L at 500 mg/L hardness.

The degree of hazard from sulfide to aquatic life is dependent on the temperature, pH, and DO. At lower pH and in an anaerobic environment a greater proportion of sulfur occurs in the form of the toxic undissociated hydrogen sulfide (H_2S). The fact that H_2S is oxidized in well-aerated water by natural biological systems to sulfates or is biologically oxidized to elemental sulfur has caused investigators to minimize the toxic effects of H_2S on fish and other aquatic life (USDA Forest Service 2007).

The EPA has not established toxics criteria for sulfates or sulfides, and Idaho has not adopted any numeric criteria for these pollutants to protect aquatic life or human health in their Water Quality Standards. The EPA set the sulfate maximum daily limit in the draft NPDES permit at 250 mg/L based on the secondary drinking water standard, and not on toxicity to aquatic life. The maximum daily limit for sulfide was set at 2 µg/L (1:1 ratio), based on the chronic criteria for protection of freshwater aquatic life. A review of the literature indicates that the potential toxicity of sulfates to salmonids and invertebrates is 2 to 3 times higher (Davies and Hall 2007, Soucek and Kennedy 2005, Soucek 2007) than the draft permit limit of 250 mg/L. For impact evaluation purposes, the NOEC for an early life stage of a salmonid species of 1,060 mg/L is considered protective of listed species (BC Research 1998) and will be used to evaluate the potential effluent toxicity to fish that may occur in Big Deer Creek.

The FCC has formally requested a modification of the draft NPDES Permit to include a mixing zone for sulfate. Treated water discharged into Big Deer Creek is predicted to have sulfate concentrations ranging from approximately 400 to 840 mg/L at the end-of-pipe. Within the mixing zone, the ICP discharge would be diluted to meet the water quality standard of 250 mg/L established in the draft NPDES permit.

The actual length and width of the mixing zone would depend on the configuration of the effluent diffuser and on authorization of the mixing zone by IDEQ. According to IDEQ mixing zone policy[5], in defining a mixing zone, several guidelines should be followed, including:

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[5] www.deq.**idaho**.gov/water/data_reports/surface_water/monitoring/**mixing_zone**s.cfm

1. The mixing zone should not interfere with existing beneficial uses.

2. Water quality within a mixing zone may exceed chronic water quality criteria so long as chronic water quality criteria are met at the boundary of any approved mixing zone.

3. Acute water quality criteria may be exceeded within a zone of initial dilution inside the mixing zone.

4. The mixing zone may not be acutely toxic to biota significant to the receiving water's aquatic community.

5. The mixing zone should be limited to 25% of the width and volume of the stream to allow a zone of passage for aquatic life.

For the purposes of this analysis, the no observed effect value of 1,060 mg/L sulfate concentration is compared to the chronic value requirement in Item Number 2 above (Acute values would be much higher than chronic values and are therefore addressed by evaluating the chronic criteria). The mixing zone is evaluated assuming that it does not exceed 25% of the width or volume of flow. Based on the nature of flow in Big Deer Creek (turbulent), it's estimated that the mixing zone would extend several hundred feet in length.

Sulfate concentrations within the regulatory mixing zone were estimated based on ICP discharge and ambient Big Deer Creek water quality and flow estimates from the DSM modeling. A summary of the basis for the estimates and resultant instream sulfate concentrations are provided in Table 15. This table evaluates combinations of flow conditions, average flows (Scenario 1) and low flow conditions (Scenario 2) at both full mixing and only 25% of mixing with Big Deer Creek. Low flow conditions combined with a 25% mixing zone provides the worst-case scenario that fish may be exposed to. At the point of discharge, sulfate concentrations immediately adjacent to the diffuser ports would be similar to the discharge, 400 to 840 mg/L. At the edge of the assumed regulatory mixing zone (25% mixing), sulfate concentrations would range from approximately 34 to 166 mg/L during average conditions (Scenario 1) and low flow (7Q10) conditions (Scenario 2) respectively (Column 5 in the Table).

Outside of the regulatory mixing zone, after mixing with the entire volume of Big Deer Creek is achieved, sulfate concentrations in Big Deer Creek are predicted to increase from approximately 7.3 mg/L (baseline condition) to approximately 14 mg/L during average flow conditions and 53.6 mg/L during low flow (7Q10) conditions (Column 4 in the Table). The worst-case scenario of low stream flow and 25% mixing area results in a predicted sulfate concentration of 166.1 mg/L, much lower than the estimated protective value of 1,060 mg/L. Given these assumptions, NMFS concludes that levels of sulfates/sulfides from the effluent should not rise to levels that are likely to result in direct, indirect, or sublethal effects to ESA-listed salmonids. However, because NMFS' analysis relies on the mixing zone as described, any difference in the mixing zone from that specifically analyzed in this Opinion may require reinitiation of consultation.

Table 15. Sulfate concentrations within and outside of the mixing zone (USDA Forest Service 2007).

Scenario 1: Average Conditions Big Deer Creek Q50, Average Outfall Flow, Average Sulfate Discharge Concentration				
	Q50 (cfs) (1)	Sulfate Concentrations Before Mixing (mg/L) (2)	Sulfate Concentrations in Big Deer Creek After Full Mixing (mg/L)	Sulfate Concentrations in Big Deer Creek After Mixing with 25% of Big Deer Creek Flow (mg/L)
Outfall Average Discharge	0.25	581	---	---
Big Deer @ Outfall	20.72	7.3	14.1	33.7
Scenario 2: Low Flow Conditions Big Deer Creek 7Q10, Maximum Outfall Flow, 90th Percentile Sulfate Discharge Concentration				
Outfall Average Discharge	0.33	840	---	---
Big Deer @ Outfall	5.6	7.3	53.6	166.1

Notes:

(1) **Flows:** Discharge flow estimate are from the ICP NPDES Permit application (FCC, 2006). Big Deer Creek average flow estimates from DSM results (see pg A-26, Appendix A, Water Resources Technical Report, Hydrometrics, 2006). Big Deer Creek 7Q10 estimated from duration curve developed from DSM daily flow results for the 100-year simulation period. For comparison to the 5.6 cfs value, USGS estimates 7Q10 to be 4.9 cfs from USGS StreamStats model (see Mebane, 2007). The DSM value was chosen for this analysis for consistency with other DSM predicted values that are based on the watershed model.

(2) **Sulfate:** Big Deer Creek sulfate concentration during base flow condition as used in DSM model (see Table B-3b DEIS). ICP discharge concentrations from DSM results (see pg A-24, Appendix A, Water Resources Technical Report, Hydrometrics, 2006).

Total Suspended Solids – TSS include both organic and inorganic particulate matter in water and refer to the portion of total solids retained on a 2 µm (or smaller) filter. Total solids are the material left from a liquid mixture (e.g., effluent) after evaporation and drying at a defined temperature (EPA 2003a).

Factors affecting TSS concentrations and distributions in receiving waters, include: flow rate, temperature, soil erosion, urban runoff, wastewater and septic system effluent, decaying plants and animals, and bottom-feeding fish. The decrease in water clarity caused by TSS can affect the ability of fish to see and catch food. Suspended sediment can also clog fish gills, reduce growth rates, decrease resistance to disease, and prevent egg and larval development. When suspended solids settle to the bottom of a waterbody, they can smother the eggs of fish and aquatic insects, as well as suffocate newly hatched insect larvae. Settling particulates can coat substrates and fill in spaces between rocks typically used by high value aquatic organisms eaten by salmonids (Wilber and Clarke 2001; Mitchell and Stapp 1992).

Four categories of effects resulting from exposure to TSS are recognized in fish: lethal, paralethal, sublethal, and behavioral (Newcombe and Jensen 1996). These four effect categories are defined as follows: lethal effects are those that result in mortality; paralethal effects are those that reduce the population in time such as reduced growth rate; sublethal effects are reduced feeding rate, or feeding success and physiological stress; and behavioral effects are avoidance, alarm or movement from cover. Although concentration and duration of exposure are the primary drivers of TSS effects on fish, other factors influence the degree of the effects. Particle size affects the ability of fish to clear the gills of TSS (Servizi and Martens 1987). Environmental factors such as temperature and DO affect tolerance to TSS by further stressing the animal (Servizi and Martens 1991). Also, the availability of refugia will influence the ability to avoid exposure (Bisson and Bilby 1982).

Following discharge, the size of particles entrained in the receiving water varies with flow characteristics (e.g. velocity, gradient, turbulence, and temperature). Deposition of suspended sediment is related to particle size and diminished flow. Temperature stratification can prevent TSS from mixing with portions of water columns or extend its downstream attenuation. The very fine particle fraction (<0.06mm) tends to stay in suspension for the length of the fluvial system. These suspended solids can directly cause toxicity to aquatic biota or can settle to the bottom of the receiving waterbody and cause toxicity to the benthic community that serves as a prey base for other aquatic biota.

Large quantities of TSS in a waterbody often correlate to higher concentrations of bacteria, nutrients, pesticides, and metals in the water. These pollutants may attach to inorganic and organic particles on the land and be carried into waterbodies with stormwater or attach to particulates in effluent and be carried downstream. Pollutants bound to solids may settle to the bottom, or remain suspended through fluvial systems, and release into the water column at variable rates (Wilber and Clarke 2001; COE 1999).

Salmonid response to TSS is dependent on environmental factors including duration of exposure and temperature (Servizi and Martens 1992). In completing a thorough review of the literature, Newcombe and Jensen (1996) summarized data from a wide range of studies, noting various levels of effect to varying concentrations of suspended sediment. Although not likely lethal at low concentrations, 24-hour exposure to levels as low as 16.5 to 25 mg/L were reported to reduce feeding activity and behavior of salmon and trout. After 2-hour exposure to suspended sediment concentrations of only 35 mg/L, cutthroat trout ceased feeding and sought cover. Extended exposure resulted in more serious effects, even at lower concentrations. For example, after 720 hours of exposure to only 18 mg/L, rainbow trout abundance was reportedly reduced.

The FCC estimates their maximum daily TSS concentration at 30 mg/L, with an average daily value of 25 mg/L. The NPDES permit limits ICP effluent to a maximum daily value of 30 mg/L, with a 20 mg/L average monthly load. These TSS concentrations appear to exceed levels previously discussed that would result in paralethal and/or sublethal effects to salmon and steelhead. However, these concentrations are estimated at the end-of-pipe and will be diluted 20-fold by mixing in Big Deer Creek before reaching habitat occupied by ESA-listed fish species downstream of the falls. Therefore, NMFS concludes that levels of TSS from the effluent should not rise to levels that are likely to result in significant effects to ESA-listed salmonids.

pH – The pH is a measure of the concentration (activity) of hydrogen, or hydronium, ions in water. Specifically, pH is the negative log of the hydrogen ion concentration. The pH of natural waters reflects the acid-base equilibrium achieved by various dissolved solids and gases, and is an important factor in the chemical and biological interactions found in waterbodies. On the pH scale of 0 to 14, waters of 0 to 7 are acidic, and waters from 7 to 14 are alkaline. Elevated hydrogen ion concentrations at low pH are directly toxic to fish, causing osmoregulatory problems (NMFS 1999).

Changes in pH also affect the solubility or toxicity of metals such as aluminum, manganese, zinc, copper, and cadmium in the water column and sediments, thereby affecting the exposure dose of metals to aquatic organisms (NMFS 1999). Metals exacerbate the physiological response to the increased hydrogen atom, increasing both their mobility and bioavailability to aquatic organisms (Spence et al. 1996). There are two types of metals: type I metals (e.g., cadmium, copper, and zinc), that are less toxic as the pH decreases; and type II metals (e.g., lead) that are more toxic at lower pH levels (USDA Forest Service 2007). Aluminum is the metal of greatest concern at low pH values, while un-ionized ammonia (directly toxic to aquatic organisms), is a problem at higher pH values. At a given temperature, the higher the pH, the greater the amount of un-ionized ammonia that will be present for a given amount of total ammonia (NMFS 1999).

Rainwater without anthropogenic acids has a pH generally between 5.0 and 5.6. The buffering capacity of a waterbody is related to alkalinity, a trait that is determined by soil type and parent geology. Waters with high alkalinity are able to neutralize or buffer a certain amount of acidic inputs. Alkalinities are lowest during periods of high surface runoff (winter and spring) and highest during periods when groundwater discharge dominates stream flow (summer and fall) (NMFS 1999).

There is little species-specific information for pH effects on anadromous fish. Highly acid-sensitive fish species (e.g. fathead minnow and striped bass) may suffer decreased reproductive success. Below pH 6.0, reproductive success of lake trout declines in some waters, and lake and rainbow trout do not survive in the wild in aquatic habitats at pH 5.5 to 5.0. Vulnerable life stages of Chinook salmon are sensitive to pH below 6.5 and possibly at pH greater than 9.0 (NMFS 1999). Winter et al. (2005) found water at pH 10 acutely toxic to juvenile rainbow trout.

Considering indirect effect to the salmonid food base, some insect larvae including those of the mayflies, stoneflies, and caddisflies are sensitive to low pH in the range of 5.5 to 6.0 (NMFS 1999). In the pH range of 6.5 to 6.0, anticipated effects are a small decrease in species richness of phytoplankton, zooplankton, and benthic invertebrate communities resulting from the loss of a few highly acid-sensitive species (NMFS 1999).

The FCC has estimated the maximum daily pH of the effluent to be 9.0, with an average daily pH of 7.5. The NPDES permit requires that water in Big Deer Creek be in the range of 6.5 to 9.0 at all times. Based on the tolerances previously described for fish and their prey base, pH values in this range are not expected to result in adverse effects to ESA-listed fish species as a result of the effluent discharge.

<u>Temperature</u> – Temperature controls many physical and chemical processes in water, affecting concentrations of numerous constituents, such as: dissolved oxygen, pH, hardness and alkalinity, and the toxicity of some constituents such as, ammonia, organics, metals, and nitrogen. Temperature affects a number of biological interactions, which may affect or alter ecological

regimes, including, competition, predation, metabolic function, disease, and prey forage. The National Academy of Sciences (NAS 1972, in McCullough et al. 2001) recommendations for water temperature exposure for protection of aquatic life specify maximum acceptable temperatures for prolonged exposures (greater than one week), winter maximum temperatures, short-term exposure to extreme temperature, and suitable reproduction and development temperatures. Comprehensive literature reviews, issue papers, and guidance have recently been completed and most of the following information is from these sources (McCullough et al. 1999, 2001; EPA 2003a).

Temperature is important in controlling many physiological and behavioral processes in salmon and steelhead (McCullough et al. 2001). Data from many experiments provide evidence that temperatures tolerated by salmonids (and other species of fish as well) are a function of at least three factors: (1) the acclimation temperature; (2) the magnitude of the difference between the acclimation temperature and the elevated temperature; and (3) the duration of exposure to the elevated temperature (Tang et al. 1987, in McCullough 1999). The acclimation temperature is the temperature of the water the fish are living in before being exposed to the elevated temperature. The elevated temperature that a salmonid can tolerate increases with increasing acclimation temperature.

The upper incipient lethal temperature (UILT) is an exposure temperature, given a previous acclimation to a constant acclimation temperature, in which 50% of the fish will die within 7 days (McCullough et al. 2001). The duration for which a salmonid can tolerate an elevated temperature decreases with increasing temperature. As acclimation temperatures rise above 68ºF, the UILT essentially estimates the ultimate upper incipient lethal temperature (UUILT). The UUILT for salmonids is quite consistent, ranging from about 73.4-78.8 ºF among species (McCullough et al. 2001). Sockeye salmon are more sensitive than Chinook with a 0.9-1.8 ºF lower UUILT. Rainbow trout have a higher UUILT (about 1.8 ºF) than Chinook salmon. However, steelhead smolts appear to be much more sensitive than Chinook smolts. Although UUILT values for salmonids reported in the literature are up to 78.8 ºF, fish in the field will not necessarily be acclimated to warm temperatures as they are in laboratory tests of UUILT. Given that wild fish are often more sensitive than domesticated laboratory stocks and are often acclimated to temperatures lower than laboratory fish, mortality in the field could be induced at in the field may be 1.8 - 3.6 ºF lower than the UUILT values derived in the laboratory. Field data suggest maximum distribution of salmonids is limited to waters that have a temperature of about 3.6 – 5.4 ºF less than the UUILT derived from laboratory experiments (McCullough et al. 2001).

The incipient lethal temperature (ILT) for a given fish species is determined by acclimating the fish to one temperature and then subjecting them instantaneously to another temperature. Acute thermal shock leading to death can be induced by rapid shifts in temperature (McCullough 1999). When salmonids are acclimated below 53.6 ºF, substantial (50%) lethality (LT50) can be expected to occur almost instantly (1-60 seconds) at temperatures above 86-93.2 ºF. For spring Chinook salmon acclimated at 59 ºF and subjected to water at 85.5 ºF, survival time can be calculated to be 3.2 minutes (McCullough 1999).

Thermal shock can also cause sublethal effects and indirect mortality even with short duration exposures. Juvenile Chinook salmon and rainbow trout acclimated to 59-60.8 ºF and transferred to temperature baths in the range of 78.8 to 86 ºF suffered significantly greater predation than controls (Coutant 1973). Coho salmon and steelhead trout acclimated to 50 ºF and transferred to 68 ºF water suffered physiological changes including hyperglycemia, hypocholestorolemia, increased blood hemoglobin, and decreased blood sugar regulatory precision (Wedemeyer 1973). Predation of sockeye salmon fry by juvenile coho salmon increased with an 18 ºF increase in temperature (Sylvester 1972). Rainbow trout acclimated at 46.4 ºF and transferred to 75.2 ºF water remained at the surface for 1 to 2 minutes, expended large amounts of energy swimming haphazardly, then became lethargic at the surface. After about 1 hour, the fish could be handled with very little evasive response (Smith and Hubert 2003). In a fluctuating environment, multiple-day exposure to near lethal or lethal temperatures may create cumulative effects (McCullough et al. 2001). Multiple stresses can load or limit physiological systems, reduce growth rates, initiate disease, reduce osmoregulatory ability, and result in death when the physiological tolerance is overwhelmed (Budy et al. 2002; McCormick et al. 1998; Wedemeyer et al. 1990).

The smoltification process is tightly linked to temperature. At the time of smoltification, anadromous salmonids experience reduced ATPase levels at constant or acclimation temperatures greater than 51.8-55.4 ºF that may result in delay or residualization (McCullough et al. 2001). It appears that temperatures between 59 and 68 ºF can block smoltification during migration of smolts to the ocean (McCullough et al. 2001). Temperatures above 64.4 ºF may inhibit feeding in smolts. At elevated temperatures (i.e., temperatures that cause changes in the smoltification process), several outcomes are possible. First, if smoltification cannot be completed then residualization occurs. Juvenile salmon usually wait for another year before out-migrating to the ocean. Remaining in freshwater for a second year subjects these juveniles to increased predation and sublethal effects associated with rearing in an area that reaches critical temperatures. Second, survival of smolts that reach the ocean depends heavily upon the degree of smoltification and general health to volitionally enter saltwater. If elevated temperatures have inhibited smoltification, reduced growth, or increased stress, then ocean survival can be decreased.

Adult salmonids are very sensitive to temperature and appear to have lethal tolerances 3.6-5.4 ºF lower than the juvenile fish typically used in lethality testing (McCullough et al. 2001). The National Academy of Sciences (1972, in McCullough 1999) recommended that a safety factor of -3.6 ºF be added to the UILT. This safety factor was meant to minimize mortality but did not consider disease incidence or other sublethal effects, which were poorly understood at that time (McCullough 1999). The UILT also does not consider other stressors that may be present, such as high turbidity, low dissolved oxygen, or chemical toxicants, that may cause fish to die at temperatures lower than the UILT (McCullough 1999).

Loss of equilibrium occurs at lower temperatures than lethality. In addition, as temperatures increase, incidence of disease increases. Holding adult Chinook at 57.2 ºF for 1.5 months resulted in no mortality, but holding for a similar amount of time at 66.2 ºF resulted in nearly total mortality from disease (McCullough 1999). Columbia River steelhead, acclimated at

66.2 ºF, had a lethal threshold of 69.8 ºF and have also been shown not to migrate until river temperatures fall below 69.8 ºF (McCullough et al. 2001). Of note, is that as temperature increases and causes adult salmon to hold in thermal refugia, the additional expenditure of energy while holding and not migrating may cause pre-spawning mortality through expenditure of limited energy reserves (McCullough et al. 2001). MacDonald (2000) noted that in a tributary to the Frazier River, 25% mortality of Chinook salmon spawning runs occurred when mean summer temperatures were frequently above 68 ºF and reached a high of 73.4 ºF.

Migration behavior of adult Chinook salmon can be changed by water temperature. Chinook and other salmon are assumed to delay migration until water temperature is conducive. Extensive delays can prevent adult salmon from reaching spawning grounds in time to spawn successfully. Generally, spring and fall Chinook salmon migrate upstream when temperatures are between 51.8 and 68 ºF (McCullough et al. 2001). A migration threshold at a temperature of 69.8 to 71.6 ºF is documented by numerous studies across major migratory salmonid species in the Columbia River (McCullough et al. 2001; McCullough 1999). On the Tucannon River in Washington, spring Chinook salmon stopped migrating when water temperature reached 69.8 ºF. Similarly, adult sockeye salmon migrations were also blocked at 69.8 ºF according to one study. Another study reports that some passage occurred even when temperatures exceeded 72 ºF. In the Okanogan River steelhead migration did not begin until temperatures fell below 73 ºF, though optimal migration temperature was about 57.9 ºF (McCullough 1999). Snake River sockeye salmon are reported to delay migration at temperatures greater than 69.8 ºF (Quinn et al. 1997). At temperatures above 68-69.8 ºF, less than 50% of fall Chinook entering the Snake River successfully passed Lower Granite Dam (Karr et al. 1998). Migrations during temperatures a few degrees lower resulted in almost twice as many fish reaching Lower Granite Dam.

Spawning success can change as a result of the temperature experienced by adult salmon before spawning. Pre-hatch mortality and developmental abnormalities were higher and alevin size was smaller in Chinook salmon held at 63.5-66.2 ºF for 2 weeks before spawning compared to Chinook salmon held at 57.2-59.9 ºF for the same period of time (McCullough 1999). As a result of these and similar observations, temperatures of 42.8-57.2 ºF have been recommended for holding of Chinook salmon broodstock (McCullough 1999). Pre-spawning mortality caused by elevated temperature and extended holding times are similar for sockeye salmon survival; 51% of adults survived in a diel temperature regime of 48.9-73.9 ºF while 96% survived in a diel regime of 52-60 ºF (McCullough 1999). Elevated holding temperature has also been shown to result in lower health and reproductive indices (McCullough et al. 2001).

Under stressful conditions mortality rates of gravid, and therefore sensitive, adult migrants are likely to increase at the population or run level. Furthermore, gametes inside body cavities of exposed adults are highly susceptible to thermal damage. Delayed pre-spawning mortality due to disease, stress, and exhaustion will increase in upstream areas (Wilkie et al. 1997). Also spawning vigor (penetration into low order streams, mate selection, redd construction, and other such essential behaviors) may be severely impaired and result in lower rates of survival of progeny. Pre-spawning mortality rates of salmon often exceed 20-40% after passage through the lower Snake River (Wilkie et al. 1997).

The NPDES permit sets a maximum daily limit of 66 ºF for the ICP effluent. FCC estimates that the effluent will have a maximum daily value of 55 ºF in the summer and 35 ºF in the winter. Average daily temperatures of the effluent are estimated to range from 55 ºF in the summer, to 40 ºF in the winter. Effluent temperatures ranging from 35 to 55 ºF should not affect background temperature levels in Big Deer Creek. These temperatures are also not in a range where adverse effects to steelhead or Chinook salmon are likely to occur as a result of effluent discharge.

Dissolved Oxygen – Low levels of DO in water can cause direct and indirect effects to fish, as well as create additional stress by causing an increase in toxicity of metals such as zinc. Sublethal effects of reducing DO below saturation can include metabolic, feeding, growth, behavioral, and productivity effects. Behavioral responses include avoidance of low DO sites or patches and curtailment of migration if DO levels drop too low across the entire stream corridor. Physiological changes to low DO, include elevation in both rate and amplitude of breathing, decreased heart rate, increased stroke volume of the heart, and altered metabolic rate (Ruggerone 2000). In situations where demand of DO exceeds input, fish kills may occur.

Productive streams exhibit diurnal cycles in water-column DO concentrations due to photosynthesis and respiration. Although fish can detect and will attempt to avoid reduced concentrations of DO, average measurements of DO do not reflect the damage that can occur during diurnal minima. Other important factors include the length and frequency of fish exposure to the low DO level. In several species studied, fish growth appeared to be determined by the daily minimum of DO, not the average or maximum. Studies reviewed in NMFS (1999) indicate possible 5-20% reductions in growth of juvenile coho salmon between 6.5-8 mg/L DO.

Reductions in DO can decrease swimming performance in both adult and juvenile fish, affecting the ability to migrate, forage and avoid predators (NMFS 1999; Spence et al. 1996). Any reduction in DO below saturation at high water temperatures increases the risk of adverse effects to salmonids. Subyearling and smolt life stages are very sensitive to low DO. Dahlberg et al. (1968) found that a reduction in DO to 7.5 mg/L resulted in a 5% reduction in swimming speed. Dahlberg noted that swimming speed declined markedly below 7-8 mg/L DO. As cited in Spence et al. (1996), Bjornn and Reiser (1991) concluded that although thresholds for survival were generally low (3.3 mg/L), growth and food conversion efficiency are affected at DO levels of 5 mg/L, and DO levels of 8 to 9 mg/L are needed to ensure unimpaired normal physiological functions of salmonids. The ecological significance of increased stress and reduced swimming ability has only recently been increasingly verified and associated with latent declines in production and survival (Servizi and Martens 1991; Wilkie et al.1997; Wedemeyer et al. 1990; Budy et al. 2002).

Sublethal effects that occur below 8 mg/L may control survival and success of juvenile salmonids in nature through reduced growth and size observed in juvenile salmonids at DO concentrations below saturation. Swimming speed in juvenile salmon declines markedly at DO concentrations below 7-8 mg/L (NMFS 1999). Results of several growth experiments summarized for coho salmon (Warren et al. 1973, as cited in ODEQ 1995) show that growth rate appears closely related to DO concentrations below 6.0-6.5 mg/L. The Oregon Department of Environmental Quality (ODEQ's) issue paper reports that concentrations around 8 to 6.5 mg/L resulted in measurable reductions in swim speed and maximum attainable growth. The paper also cites laboratory studies that note changes in oxygen transfer efficiency occurring and blood not fully saturated with oxygen at levels near 6.5 mg/L.

Warm water levels and water temperatures work in synergy with DO concentrations to cause a range of adverse effects to salmonids. This range includes acute lethal toxicity, inability to complete essential foraging and predator avoidance behaviors, area avoidance, migration delays, increased stress, reductions in growth, and slower swimming speed. Low DO concentrations increase the acute toxicity of various toxicants such as metals (e.g., zinc) and ammonia (Rand and Petrocelli 1985; NMFS 1999). Also, TSS and toxicants may increase sensitivity to low concentrations of DO. For example, any toxicant which damages the gill epithelium can decrease the efficiency of oxygen uptake.

Three components of effluent can affect receiving water DO levels: (1) the actual concentration of DO by volume discharged into the waterbody; (2) the load of chemical oxygen demand (COD) material; and (3) the load of biological oxygen demand (BOD) material. These components cumulatively affect different areas within receiving waters based upon rate of assimilation and degradation, fate and transport. The EPA has identified a DO concentration limit of >6 mg/L at all times to meet the Idaho State water quality criteria for aquatic life. A 20-fold mixing of the effluent will occur in Big Deer Creek at low streamflow and full discharge. When considered in addition to the predicted low BOD and COD levels (1 mg/L maximum daily) in the effluent, the ICP is not expected to exceed the 6 mg/L limit identified for DO in the NPDES permit. Based on DO effect levels described above, DO levels >6 should be adequate to avoid the effects of low DO on ESA-listed salmonids downstream from the discharge pipe.

<u>Iron</u> – Iron is pivotal to metabolism in almost all living organisms (Andersen 1997). Iron is an abundant, naturally occurring metal that can be found in a variety of forms. Oxidation of ferrous minerals in iron-containing ores is a primary source of environmental acids (Nelson et al. 1991). Acid mine drainage often contains elevated levels of dissolved ferrous iron (Fe^{2+}) which converts to insoluble ferric (Fe^{3+}) species, including particulate ferric hydroxide ($Fe(OH)_3$), as pH is elevated and oxygenation takes place (Smith and Sykora 1976). Iron and most of its compounds are insoluble and highly persistent in water, with a half-life >200 days (Vincoli 1997).

Iron can be mobilized in acid mine drainages, precipitating out of solution in the form of ferric oxide. These oxide and hydroxide precipitates are the more toxic forms of iron, and have been shown to suffocate trout eggs and to coat gills suffocating other aquatic biota. The toxicity of iron to fish increases with decreasing pH. For example, brook trout 96-hour LC_{50}s shifted from 1,750 µg iron/L at pH 7.0, to 480 µg iron/L at pH 6.0 (Nelson et al. 1991). Anderson (1997) exposed brown trout fertilized eggs, yolk-sac larvae, and start-fed fry to various concentrations of iron (0 to 35 mg/L), finding only the start-fed fry were highly susceptible to waterborne iron. Iron accumulation affecting the fry was thought to increase with the development of the gills. Smith and Sykora (1976) noted sharp declines in coho salmon alevin survival when exposed to iron concentrations of 6,000 to 12,000 µg/L.

The EPA has not identified water quality criteria for iron, nor have they identified any NPDES permit limits for the ICP. Although there are no toxic criteria established for iron in section 210 of the Idaho Water Quality Standards, a value of 5,000 µg/L is recommended in the EPA Blue Book for irrigation water supply. In their NPDES application, FCC estimates maximum daily effluent concentrations of iron at 300 µg/L, and average daily concentrations of 30 µg/L at the end-of-pipe. At these concentrations, EPA determined that the discharge shows no reasonable potential of violating water quality standards, and an iron effluent limit was not developed for the ICP. Although no limit has been developed, iron will be monitored on a monthly basis with the other pollutants to ensure that limits do not exceed those anticipated by FCC. Based on review of the data cited above, NMFS concludes that levels of iron predicted for the effluent are not likely to rise to concentrations that will result in direct, indirect, or sublethal effects to ESA-listed salmonids.

Aluminum – Elemental aluminum is one of the most abundant metals in the earth's crust, and because of its frequent use it enters the environment from both point and non-point sources (Vincoli 1997). It has been identified as one of the most important toxicants of aquatic systems acidified by anthropogenic introduction of acids (Dussault et al. 2004; Exley 2000) and is highly persistent in water with a half-life >200 days (Vincoli 1997). Solubility is dependent upon pH, increasing as pH increases or decreases from near neutrality. Solubility also increases at lower temperatures or with the addition of complexing ligands. It is lowest between pH 6 and 8 (Winter et al. 2005). As previously stated, changes in pH affects the solubility or toxicity of metals, thereby affecting the exposure dose of metals to aquatic organisms. Aluminum is the metal of greatest concern at low pH values (NMFS 1999).

Aluminum has moderate acute and high chronic toxicity to aquatic biota (Vincoli 1997). Natural water chemistry appears to be more effective at protecting fish from acute toxicity than chronic toxicity (Exely 2000). Inorganic monomeric aluminum species are generally considered the most toxic, including Al^{3+}, $AlOH^{2+}$, $Al(OH)_2^+$, $Al(OH)_3$, an $Al(OH)_4^-$ (Winter et al. 2005). Exely (2000) showed that rainbow trout fry avoided elevated concentrations of waterborne aluminum, although the response varied depending upon both concentration and pH. When exposed to aluminum concentrations ranging from 17 to 540 µg/L, fry avoided concentrations as low as 34 µg/L at pH levels of 5.0. However, avoidance noted at pH levels between 5 and 5.75 did not occur at a pH of 6.0, where fish did not avoid concentrations as high as 540 µg/L. No mortality or signs of acute toxicity were noted for rainbow trout at these concentrations or pH levels.

Waterborne aluminum is the main toxicant affecting fish in acidic waters, acting as a gill toxicant by causing ionoregulatory and respiratory effects (Winters et al. 2005; Dussault et al. 2004; Allin and Wilson 1999; McDonald et al. 1991; Mueller et al. 1991). If pH in acidic waters is increased quickly, aluminum solubility decreases which increases toxicity as aluminum polymerizes and precipitates in fish gills. Mortality in brook trout exposed to 200 µg/L aluminum for 56 days was significantly higher at pH 5.3 than either pH 6.1 or 7.2 (Cleveland et al. 1991). Although Winters et al. (2005) found that aluminum readily accumulated on rainbow trout gills at pH levels ranging between 6 and 8, accumulation was eliminated at all pH levels by the presence of natural organic matter (aluminum bonded to the organic matter versus fish gills).

Following 53 to 69 days of exposure of juvenile rainbow trout to aluminum concentrations ranging from 0 to 80 µg/L, Dussault et al. (2004) noted aluminum accumulation in both the gills and liver increased with increasing waterborne concentrations of aluminum. The study documented issues with trout swimming speed and cardiac output, describing a significant relationship between aluminum accumulation at the gill and both decreased swimming speed and increased heart rate. Prior to this study Allin and Wilson (1999) had shown similar results in fish behavior, noting dramatic changes in swimming behavior of juvenile rainbow trout exposed for 34 days to aluminum concentrations of 30 µg/L (pH 5.2). Trout were more sluggish after only 1 day of exposure; spending considerably less time swimming, more time holding position, and demonstrating dramatically fewer burst-swimming episodes. At the conclusion of their study, the authors theorized that if fish experienced similar changes to routine swimming behavior in the wild as observed in the laboratory, chronic exposure to aluminum would likely affect a fish's ability to forage, avoid predation, migrate, or reproduce.

The EPA has not identified water quality criteria for aluminum, nor have they identified any NPDES permit limits for the ICP. However, in their NPDES application, FCC estimates maximum daily effluent concentrations of aluminum at 200 µg/L, and average daily concentrations of 20 µg/L at the end-of-pipe. At these concentrations, EPA determined that the discharge shows no reasonable potential of violating water quality standards, and an aluminum effluent limit was not developed for the ICP. Although no limit has been developed, aluminum will be monitored on a monthly basis with the other pollutants to ensure that limits do not exceed acute and chronic toxicity values described above. Provided pH levels are maintained between 6.5 and 9.0, as required in EPA's effluent limits, NMFS concludes that levels of aluminum in the effluent are not likely to rise to concentrations that will result in direct, indirect, or sublethal effects to ESA-listed salmonids.

<u>Magnesium</u> – Magnesium is a naturally occurring mineral, found in both terrestrial soils and aquatic sediment, posing very little threat to the environment (Vincoli 1997). As described by El-Mowafi and Maage (1998), magnesium is an essential element for fish, where deficiency has been found to result in retarded growth, anorexia, sluggishness, high mortality, reduced ash content, and reduced concentrations of magnesium and calcium in the whole body and vertebrae. Magnesium is nearly insoluble (0.01%) and highly persistent in water, with a half-life >200 days. Anthropogenic sources of magnesium include discharges and spills from industrial and municipal waste treatment plants (Vincoli 1997).

Magnesium and its salts have slight acute toxicity to aquatic life, and little information exists regarding its short-term toxicity to plants, birds, or terrestrial animals. The EPA does not currently consider magnesium a hazardous substance (Vincoli 1997). Mount et al. (1997) reported 24-, 48-, and 96-hour LC_{50}s of 3,520, 2,840, and 2,120 mg/L $MgCl_2$ for fathead minnows, and 24-, and 48-hour LC_{50}s of 1,560 and 3,330 mg/L $MgCl_2$ for *D. magna*. For rainbow trout eggs exposed to $MgCl_2$, the Pesticide Action Network's PAN Pesticides Database (2008) reported a mean 28-day LC_{50} of 1,355 mg/L, ranging from a low of 119.9 mg/L to a maximum of 1,507 mg/L. After 14 days, Shearer and Åsgård (1992) noted elevated whole body concentrations of magnesium in fish exposed to 150 mg/L and 1,000 mg/L, further noting that fish exposed to the higher concentration were significantly smaller than fish exposed to lesser concentrations. Mortalities began to occur in rainbow trout after 2 days of exposure to the

1,000 µg/L concentration, increasing to 48% total mortality at the end of the experiment (14 days). In the absence of dietary magnesium, the authors concluded that rainbow trout are able to meet their magnesium requirement at waterborne concentrations of 46 mg/L.

EPA has not identified water quality criteria for magnesium, nor have they identified any NPDES permit limits for the ICP. However, in their NPDES application, FCC estimates maximum daily effluent concentrations of magnesium at 100 mg/L, and average daily concentrations of 10 mg/L at the end-of-pipe. At these concentrations, EPA determined that the discharge shows no reasonable potential of violating water quality standards, and a magnesium effluent limit was not developed for the ICP. Although no limit has been developed, magnesium will be monitored on a monthly basis with the other pollutants to ensure that limits do not exceed acute and chronic toxicity values described above. Based on review of the data cited above, NMFS concludes that levels of magnesium in the effluent are not likely to rise to concentrations that will result in direct, indirect, or sublethal effects to ESA-listed salmonids.

<u>Manganese</u> – Manganese is a naturally occurring metallic element weathered from rock, broadly distributed throughout surface waters, soil, aquatic sediments, and groundwater (Reimer 1999; Stubblefield et al. 1997). Manganese is an essential trace element for both plant and animal life (Reimer 1999; Vincoli 1997). It and its compounds vary in their solubility in water from very soluble to nearly insoluble, and are highly persistent in water with a half-life >200 days (Vincoli 1997). Manganese solubility increases at low pH, while the presence of high concentrations of chlorides, nitrates, and sulfates may also increase solubility and increase aqueous mobility and plant uptake. Undisturbed freshwater concentrations of manganese vary widely, seldom exceeding 100 µg/L, but infrequently ranging to 10,000 µg/L. Highest concentrations of manganese are typically observed during spring runoff (Reimer 1999). Manganese is a common constituent of point and nonpoint discharges from mining and smelting operations, and is often present in elevated concentrations in proximity to these activities (Stubblefield 1997).

Manganese and its compounds reportedly have moderate acute toxicity to aquatic life (Vincoli 1997). Previously, several states have employed a water quality standard of 1,000 µg/L for manganese. However, this standard was apparently based on limited toxicological data and did not take water hardness into consideration. The toxicity of manganese is inversely correlated to water hardness, becoming less toxic with increasing water hardness (Reimer 1999; Stubblefield et al. 1997). Reimer (1999) appears to have completed the most thorough review of manganese acute and chronic toxicities to date, an effort completed in support of establishing ambient water quality criteria for the Province of British Columbia. Reimer's results take hardness into consideration. This analysis will focus primarily on results of testing in soft or moderately soft waters, since this information is most applicable to conditions within the action area.

Exposure of juvenile longfin dace (*Agosia chrysogaster*) to manganese resulted in a 96-hour LC_{50} of 13,000 µg/L at 224 mg/L $CaCO_3$ (Lewis 1978). For salmonids, Reimer (1999) identified 96-hour LC_{50}s ranging from a low of 2,400 µg/L (hardness 25 mg $CaCO_3$) in coho salmon to a high of 4,830 µg/L in rainbow trout (hardness 38 mg $CaCO_3$). Results for brown trout were intermediate with a 96-hour LC_{50} of 3,770 µg/L (hardness 34 mg $CaCO_3$). Aquatic invertebrates appeared to be more sensitive to manganese concentrations, exhibiting 96-hour LC_{50}s of 3,600 and 5,800 µg/L for *H. azteca* and *C. tentans* respectively (hardness 25 mg/L $CaCO_3$). Although finding a 48-hour LC_{50} of 800 µg/L for *D. magna* at 25 mg/L $CaCO_3$ hardness, Reimer used results from his chronic toxicity studies (summarized below) to theorize that *D. magna* sensitivity to water hardness may have artificially influenced the test result. Reiser calculated an IC_{25} concentration of 5,300 µg/L that he considered to be a more effective measure of toxicity than either the LOEC or NOEC concentrations.

Chronic toxicity tests on rainbow trout revealed a seven-day early life stage EC_{50} concentration of 14,600 µg/L (hardness 25mg/L). Because the chronic value actually came out higher than the acute values previously reported, Reimer (1999) suggested that a less sensitive life stage may have been used in the chronic study than the acute study. Because of excessive deaths of *D. magna* at hardness levels of 25mg/L, the author was not able to determine chronic toxicity values in soft water. However, he did calculate IC_{25} concentrations of 5,400 and 9,400 µg/L at water hardnessess of 100 and 250 mg/L $CaCO_3$.

According to Vincoli (1997), the concentration of manganese found in fish tissues is expected to be about the same as the average concentration of manganese in the water from which the fish was taken. Although not specifically stating that bioaccumulation of manganese does occur, Reimer (1999) reported higher concentrations of manganese in liver and gill tissue than in muscle tissue.

The EPA has not identified water quality criteria for manganese, nor have they identified any NPDES permit limits for the ICP. However, in their NPDES application, FCC estimates maximum daily effluent concentrations of manganese at 50 µg/L, and average daily concentrations of 0.005 µg/L at the end-of-pipe. At these concentrations, EPA determined that the discharge shows no reasonable potential of violating water quality standards, and a manganese effluent limit was not developed for the ICP. Although no limit has been developed, manganese will be monitored on a monthly basis with the other pollutants to ensure that limits do not exceed acute and chronic toxicity values described above. Both of these concentrations fall well below acute and chronic toxicities reported in this analysis, even before dilution. Therefore, NMFS concludes that levels of manganese from the effluent should not rise to concentrations that will result in direct, indirect, or sublethal effects to ESA-listed salmonids.

Water Quantity/Groundwater: The FCC has applied for water rights to the groundwater from the mines and from two wells. Water would be used initially for drilling and other start-up water needs until the mine pumping and precipitation capture from the TWSF is adequate to supply operating water needs. A groundwater capture system will be set up downgradient from the mines to capture and test groundwater for compliance with surface water quality standards. As proposed, the USFS estimates that the ICP will result in a baseflow reduction in Big Deer Creek of 1%, which will be completely restored post operation (USDA Forest Service 2007).

For salmon and other aquatic organisms, flows determine the amount of available water and space, the types of micro- and macro-habitats, and the seasonal patterns of disturbance to aquatic communities. High flows redistribute sediments in streams, flushing fine sediments from spawning gravels and allow recruitment of gravels to downstream reaches. Extreme high flows are essential for developing and maintaining healthy floodplains, moving and depositing sediment, recharging groundwater aquifers, dispersing vegetation propagules, and recruiting and transporting LWD (Spence et al. 1996). Low flow conditions can affect rearing juveniles by reducing the amount of refugia available for protection from predators, by elevating water temperatures, reducing the availability of food, and increasing competition for space and food sources (Gregory and Bisson 1997).

Streamflow is also important in facilitating downstream movement of salmonid smolts. Dorn (1989) found that streamflow increases triggered downstream movement of coho salmon in a western Washington stream. Similarly, Spence (1995) also found short-term flow increases are an important stimulus for smolt migration in four populations of coho salmon. Chinook salmon can gradually move downstream over several weeks or months. Thus the normal range of streamflows might be required to maintain normal temporal patterns of migration in a particular basin. Streamflow is also important in determining the rate at which smolts move downstream, although factors influencing the speed of migration remain poorly understood (Spence et al. 1996).

Where water is withdrawn from smaller rivers and streams, seasonal or daily flow fluctuations can adversely affect fish, macroinvertebrates in littoral areas, aquatic macrophytes, and periphyton (Ploskey 1983). Fluctuating water levels can delay spawning migrations, impact breeding condition, reduce salmon spawning area (Beiningen 1976), dewater redds and expose developing embryos, strand fry (CRFC 1979), and delay downstream migration of smolts. The literature suggests that diversions can contribute to low flows and are likely to inhibit or delay salmonid smolt migration. This delay could limit fish survival and reduce potential numbers of returning adults (NPPC 1986).

Surface water quality impacts from the ICP were evaluated through multiple methods and tools, including the DSM, baseline surface water hydrology and chemistry characterization, and a watershed hydrology model (Hydrometrics 2006). The mine dewatering process would alter groundwater flow and is expected to reduce the groundwater inputs into Big Flat and Bucktail Creeks, thereby causing reduction of flow into the South Fork Big Deer and Big Deer Creeks. The changes were modeled using DSM Version 4.0 (Hydrometrics 2006).

The predicted changes from the ICP would be expected to occur only for the duration of the mining operation when active dewatering is taking place (10 to 14 years). The largest alterations of stream flow would occur in Bucktail, South Fork Big Deer, and Big Flat Creeks. Bucktail Creek and Big Flat Creeks are small streams that have probably never provided fish habitat. The South Fork Big Deer Creek is a small stream with steep gradient, likely providing limited resident fish habitat only in the lowest reach. Salmon and steelhead do not occur in any of these streams. The predicted downstream change in Big Deer Creek would be a baseflow reduction of approximately 1% (Table 16).

Table 16. Predicted changes in stream baseflow due to ICP operations (USDA Forest Service 2007).

Stream	% Flow Reduction During Operations
Big Flat Creek	3%
Bucktail Creek	44%
South Fork Big Deer Creek	11%
Big Deer Creek	1%
Panther Creek	0

Mine dewatering would create small decreases in flow in the fish-bearing Big Deer Creek. However, NMFS does not expect a small baseflow reduction of 1% to alter the wetted width or instream habitat characteristics of Big Deer Creek. Therefore, a baseflow reduction of only 1% should not appreciably affect available habitat for anadromous fish in Big Deer Creek.

Effects to water quality could occur as a result of groundwater contamination. In an effort to ensure this does not occur, FCC has installed a network of 29 groundwater monitoring wells around the project site. Fourteen new wells and three replacements will be installed to complete the groundwater monitoring effort. The ICP has outlined a water monitoring plan to address operational and closure assessment of water resources (Telesto 2007). The monitoring plan includes a performance-based approach to compliance assessment. For example, groundwater quality data from select operational monitoring wells to be located downgradient of the mines would be evaluated for compliance. If performance criteria exceed pre-established targets (e.g. if the calculated groundwater load were to result in exceedance of a surface water compliance target), a response action would be required by the mine to reduce the groundwater load to acceptable levels. Post closure groundwater capture wells installed and tested during the initial construction phase will confirm that the system would capture a sufficient amount of groundwater to protect downstream water quality. If testing indicates that bedrock wells could not capture enough of the groundwater metals load, an additional groundwater/surface water capture system consisting of an interception trench or series of wells across Bucktail Creek alluvium would be installed downgradient of the Ram mine. The Bucktail capture system would collect alluvial groundwater and surface water, if necessary. The Bucktail capture system would allow collection of additional groundwater and COCs from the Ram and Sunshine mines, and would allow capture and treatment of additional metal load to ensure that the ICP does not contribute to a net increase in metal loading to the South Fork Big Deer, Big Deer, Panther Creek system. With this groundwater monitoring and response system in place, NMFS does not expect adverse effects to water quality from groundwater contamination to occur.

Riparian Vegetation: Disturbance of riparian vegetation has the potential to result in decreased shade and increased solar radiation, which potentially could further increase water temperatures in the action area. Riparian vegetation will be disturbed in the vicinity of the outfall and cable car crossings alongside Big Deer and Panther Creeks. Elevated water temperatures may adversely affect salmonid physiology, growth, development, alter life history patterns, induce disease, and may exacerbate competitive predator-prey interactions (Spence et al. 1996).

However, the small amount of vegetation disturbed as a result of this project is negligible when considered in context of the stream reach scale, particularly with RHCAs currently classified as "Functioning Appropriately" for both Big Deer Creek and Panther Creek.

2.1.3.2. Effects on Critical Habitat

Critical habitat within the action area has an associated combination of physical and biological features essential for supporting freshwater spawning, rearing, and migration of the ESA-listed salmon and steelhead. The critical habitat elements most likely to be affected by the proposed action include water quantity, water quality (sediment and chemical contamination), riparian vegetation, substrate/spawning gravel, forage/food, and safe fish passage. Impacts to these critical habitat elements are likely to impact the action area's ability to provide for successful spawning, rearing, and/or migration. The effects of the proposed action on these features are summarized as a subset of the habitat-related effects that were discussed above.

Water quantity – Withdrawal of water from the Big Deer Creek watershed will potentially affect the water quantity PCE for rearing juvenile steelhead and Chinook salmon. However, as previously stated, baseflow reductions of only 1% are expected to occur Big Deer Creek, an amount not likely to appreciably diminish the conservation value of affected critical habitats downstream.

Water quality – Chemical contamination and turbidity have the potential to affect the water quality PCE for freshwater migration, spawning, and/or rearing habitat. Effects to water quality could occur: (1) when construction equipment is working within or adjacent to the stream channel; (2) should a spill occur in the transportation corridor; (3) as a result of effluent discharge; or (4) through groundwater contamination.

As previously discussed, construction equipment operating in the vicinity of action area streams will result in a temporary minor degradation of water quality through the introduction of fine sediment and associated increased turbidity. Increased turbidity generated as a result of the cable car installation is expected to be short-lived and highly localized due to the short work window, construction occurring outside the active channel, and low flow conditions during the construction period. Sediment control measures proposed are expected to effectively minimize sediment delivery from these activities. However, BMPs are not well described for the installation of the pipeline on Big Deer Creek or the removal of the two culverts on North Fork Williams Creek. The proposed action does not describe dewatering the work areas when removing the culverts or trenching within the active channel. Although sediment suspended as a result of these activities is expected to settle out long before reaching critical habitat downstream, additional BMPs are necessary to ensure that critical habitat is not further degraded by the pipeline installation. Although degradation will occur in the reach immediately downstream of the proposed action, this amount is not likely to appreciably diminish the conservation value of designated critical habitat at the 5th field HUC.

There is also a potential for temporary toxic chemical contamination, primarily from petroleum based fuels and lubricants. As previously stated species effects section, it's unclear in the proposed action if equipment used on this project will be clean and free of fuel and lubricant leaks, or whether the equipment will be inspected prior to beginning and during this work. It's also unclear what provisions will be in place regarding the fueling and maintenance of equipment in RHCAs. Additional fuel spill and equipment leak contingencies and preventions will be necessary to effectively minimize the risk of negative impacts to fish habitat from toxic contamination. Any degradation would occur in the immediate vicinity of the proposed action, and not in amounts likely to appreciably diminish the conservation value of designated critical habitat at the 5th field HUC.

Road construction, reconstruction, use, road maintenance will all generate sediment and increase the potential for surface runoff and turbidity. Turbidity levels may periodically spike in the project area during and immediately following rain events. However, the majority of these roads are already in existence and actively contributing sediment to action area waterbodies. Efforts to resurface the road, improve drainage, obliterate existing roads, and elevate the road above the floodplain should work to reduce effects from the haul road over its existing condition (based on modeling effort previously described). NMFS believes that the road upgrades proposed and the conservation measures described should be adequate to keep turbidity and erosion to levels to a minimum. However, to best minimize sediment delivery to action area streams, completion of all road upgrades in environmentally sensitive areas, in road segments bordering streams, and along road segments draining directly to streams should be completed in the first phase of project implementation. Although degradation will occur in reaches bordering roadwork, the amount is not likely to appreciably diminish the conservation value of designated critical habitat at the 5th field HUC.

Water quality could also be affected by an accidental spill of toxic materials as they're being transported to the site, or through an on-site spill and subsequent discharge into nearby waterbodies. Depending upon the location, an accidental spill of fuels or toxic materials could adversely affect the water quality in salmon and steelhead spawning, rearing, or migratory habitat. Many materials hauled to the site will result in acutely toxic concentrations of chemicals, with the magnitude, duration, and extent of exposure varying with the chemical spilled, whether it's a bulk or liquid, if it's delivered to a stream, the size of the receiving waterbody, etc. The speed of the response and effectiveness of the clean-up activities will also determine the duration of effect to water quality. Considering proper application of the proposed spill avoidance measures and spill control plan, NMFS does not expect an accidental spill to occur as a result of the project.

Water quality in Big Deer Creek and possibly lower Panther Creek could also be affected by chemical contamination from the effluent outfall. Each potential pollutant and their potential effect to fish and fish habitat were described in detail in the previous section. Except for excess levels of nitrate, effluent meeting levels established in the NPDES permit should ensure that water quality is protected in Big Deer and Panther Creeks. Degradation will occur in lower Big Deer Creek, but these amounts are not likely to appreciably diminish the conservation value of designated critical habitat at the 5th field HUC.

Substrate/Spawning Gravel – As previously discussed, construction equipment and roadwork is expected to result in increased sediment delivery to action area streams. Suspended sediments generated by the project are expected to settle out and be deposited within 300 feet of the point of introduction. Sediment related effects are most likely to be realized to steelhead along Williams Creek, or Chinook salmon and/or steelhead from roadwork along Deep Creek, Panther Creek, or Blackbird Creek. The amount of suspended sediment is not expected to be substantial, and the proposed roadwork could actually result in an improvement over existing condition. Sediment control measures are expected to effectively limit the amount of sediment delivered from project-related activities.

However, as previously described, BMPs are not well described for the installation of the pipeline on Big Deer Creek. The proposed action does not describe dewatering the work area when trenching within the active channel. Sediment suspended as a result of this activity is expected to settle out long before reaching critical habitat downstream. However, this stream segment is already on the 303(d) list for sediment, with additional sediment potentially further degrading this PCE. Additional BMPs will be necessary to ensure that no long-term effects to substrate or spawning gravel quality occur.

Riparian Vegetation (spring/summer Chinook salmon) – A small-scale alteration of streamside vegetation will occur along Panther Creek from installation of the cable car crossing. Riparian vegetation will also be altered along Big Deer Creek from the installation of the outfall, but will not be considered further in this analysis due to the very small amount of vegetation disturbed and that it occurs upstream from the impassable waterfall, approximately 2.3 miles upstream from the nearest Chinook salmon rearing habitat. Although not specified in the BA, review of the plans, the small size of the structures, and the description of this particular action suggest that the disturbance of riparian vegetation for installation of the cable car will not likely exceed 25- to 30-feet in length on either side of Panther Creek. The localized removal of this small amount of riparian vegetation should not alter riparian functions and processes in this segment of Panther Creek. Consequently, the effects of riparian vegetation alteration should be negligible when considered in the context of the stream reach and/or watershed.

Forage/Food – Chemical contamination resulting from an accidental spill or through effluent discharge has the potential to affect the forage/food PCE for juvenile salmon or steelhead. Mortality of aquatic invertebrates from a spill of toxic materials would be dependent upon the type and amount of material spilled. Since toxicity is expected to attenuate in a downstream direction, mortality from a spill is not likely to extend more than a mile or two downstream. As discussed in the previous section, concentrations of pollutants from the effluent are not expected to rise to a level acutely toxic to aquatic invertebrates in Big Deer Creek. However, as discussed above, it's less certain that pollutants will not reach levels chronically toxic to aquatic invertebrates. It's also not certain to what extent various pollutants might affect community assemblage or bioaccumulate in the prey base, indirectly affecting fish survival and growth downstream.

Safe Passage – Although no physical barriers will be constructed as a result of the project, migratory fish passage could be affected through chemical contamination resulting from an accidental spill or effluent discharge. Because spills are not expected to occur, and effluent discharged into Big Deer Creek is expected to be within the limits established in the NPDES permit, fish passage is not expected to be affected by the project.

2.1.3.3. Cumulative Effects

'Cumulative effects' are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Cumulative effects that reduce the ability of a listed species to meet its biological requirements may increase the likelihood that the proposed action will result in jeopardy to that listed species or in destruction or adverse modification of a designated critical habitat.

Between April 1, 2000, and July 1, 2006, the population of Lemhi County increased by 1.6%[6]. Thus, NMFS assumes that future private and state actions will continue within the action area, increasing as population density rises. As the human population in the action area continues to grow, demand for agricultural, commercial, or residential development is also likely to grow. However, because the SCNF administers approximately 93% of the area in the Panther Creek watershed, the effects of new development caused by that demand are not likely to reduce the conservation value of the habitat within the action area. Any actions taking place on the SCNF would correspondingly be subject to section 7 consultation and would not be considered cumulative effects.

The SCNF identified no specific private or state actions that are reasonably certain to occur in the future that would affect anadromous fish or their habitat within the action area. Similarly, NMFS is not aware of any specific future non-Federal activities within the action area that would cause greater effects to a listed species or a designated critical habitat than presently occurs. Therefore, for purposes of this analysis, NMFS assumes that private and state actions will continue to have impacts on habitat at the same level as that reflected in the environmental baseline.

2.1.4. Conclusion

After reviewing the status of Snake River spring/summer Chinook salmon, Snake River Basin steelhead, their designated critical habitats, the environmental baseline for the action area, the effects of the proposed action, and cumulative effects, NMFS concludes that the action, as proposed, is not likely to jeopardize the continued existence of the affected species and is not likely to destroy or adversely modify designated critical habitats for those species. These conclusions are based on the following considerations.

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[6] U.S. Census Bureau, State and County Quickfacts, Lemhi County. Available at http://quickfacts.census.gov/qfd/states/16/16059.html

In spite of some uncertainties, the likelihood of jeopardizing listed salmon or steelhead through harm from sublethal effects or outright mortality is highly improbable due to the following circumstances:

1. Transportation spills of toxic substances have a low likelihood of occurring along the transportation corridor. Conservation measures designed to avoid the risk of accident and spill, combined with the application of the proposed SPCC plan should effectively avoid or minimize the risk of spill.

2. NPDES permit levels identified for the effluent of this project generally protect aquatic biota from acute toxicities. The WET testing required as a condition of the NPDES permit should ensure that chronic toxicity levels do not occur to aquatic biota.[7]

3. With the majority of work occurring outside the active channel, and only two crossings of Panther Creek proposed, it's not expected that any fish will be killed by crushing during project implementation,

4. Turbidity and sediment deposition are expected to be temporary and effectively minimized. Turbidity generated is expected to be in the form of temporary pulses, settling out within 300 feet of the disturbance, and not expected to extend across entire channels. In addition, road obliteration, surfacing, and drainage improvements are expected to reduce sediment delivery over baseline conditions.

5. Some fish could potentially be exposed to stress from temporary water quality degradation and fish displacement due to noise and construction activity. Any stress experienced by those fish is likely to be brief. Additionally, most juveniles will be large enough to move to less turbid adjacent habitats and avoid possible negative effects.

6. In spite of the specific uncertainties mentioned above, the proposed action contains measures sufficient to avoid appreciably diminish VSP parameters, especially abundance/productivity. This is true at the population level, and should therefore not diminish the likelihood of the survival and recovery of the species at the MPG or ESU-scales.

The destruction or adverse modification of critical habitat is unlikely due to the following circumstances:

1. Construction related BMPs should effectively minimize the disturbance of instream habitat and riparian vegetation. Turbidity and sediment deposition are expected to be temporary and effectively minimized. Reseeding areas disturbed by ground disturbing activities with native species should ensure that long-term effects are avoided.

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[7] The conclusions for toxics included in this biologic opinion are based on site-specific conditions including, among other things, the lack of mixing zones (except for sulfate), dilution factors, and the distance between the outfall and the nearest ESA listed fish. Because this analysis is specific to the ICP, it cannot be relied upon to determine the effects of toxics in other areas or circumstances.

2. Any significant chemical contamination is likely to be quickly discovered and addressed through proposed water quality monitoring efforts.

3. Chemical contamination is not expected to occur. Concentrations are not likely to reach acutely toxic levels, nor are they likely to reach chronically toxic levels that could alter the biological community in a manner that would appreciably alter PCEs of proposed or designated critical habitat at spatial and temporal scales relevant to conservation of the species.

2.1.5. Conservation Recommendations

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. The following recommendations are discretionary measures that NMFS believes are consistent with this obligation and therefore should be carried out by the action agencies:

1. The SCNF should accelerate the environmentally sound replacement of the Shoup Bridge spanning the Salmon River and enlarge it from single lane to double lane to reduce the risk of collisions and accidental spills of heavy metals, acids, or other toxic materials into the Salmon River.

2. EPA should complete the Blackbird Mine cleanup remedial actions.

3. The EPA should work to establish a water quality criterion for protection of cold-water biota for nitrates.

Please notify NMFS if the action agencies carry out any of these recommendations so that we will be kept informed of actions that minimize or avoid adverse effects and those that benefit listed species or their designated critical habitats.

2.1.6. Reinitiation of Consultation

Reinitiation of formal consultation is required and shall be requested by the Federal agency or by NMFS where discretionary Federal involvement or control over the action has been retained or is authorized by law and: (a) If the amount or extent of taking specified in the incidental take statement is exceeded; (b) if new information reveals effects of the action that may affect listed species or designated critical habitat in a manner or to an extent not previously considered; (c) if the identified action is subsequently modified in a manner that has an effect to the listed species or designated critical habitat that was not considered in the Opinion; or (d) if a new species is listed or critical habitat is designated that may be affected by the identified action (50 CFR 402.16).

In the proposed action it was noted that the production life of the ICP may be extended beyond the currently proposed mine and mill life schedule. However, this effects analysis cannot consider an unspecified expansion of the project footprint or extension of the life span of the operation. Any expansion of the mine or extension of its lifespan could result in effects not considered in this Opinion. Therefore, NMFS must be contacted prior to any expansion or extension in order to evaluate whether reinitiation is warranted.

To reinitiate consultation, contact the Idaho State Habitat Office of NMFS and refer to NMFS Number assigned to this consultation.

2.2. Incidental Take Statement

Section 9(a)(1) of the ESA prohibits the taking of endangered species without a specific permit or exemption. Protective regulations adopted pursuant to section 4(d) extend the prohibition to threatened species. Among other things, an action that harasses, wounds, or kills an individual of a listed species or harms a species by altering habitat in a way that significantly impairs its essential behavioral patterns is a taking (50 CFR 222.102). Incidental take refers to takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR 402.02). Section 7(o)(2) exempts any taking that meets the terms and conditions of a written incidental take statement from the taking prohibition.

2.2.1. Amount or Extent of Take

Adult and juvenile Snake River spring/summer Chinook salmon and steelhead are likely to be present in the action area during implementation/operation of the Project and thus exposed to adverse effects. Because these effects may injure or kill individuals of these ESA-listed species, or may increase the likelihood that exposed individuals will be injured or killed, take is reasonably certain to occur.

Construction activity will occur alongside and in streams occupied by ESA-listed species. These actions will cause increases in turbidity and sediment in areas with high conservation value for the affected species. As previously described, elevated levels of turbidity or sediment deposition may cause take by affecting fish behavior or through degradation of spawning/rearing habitat. Generally speaking, the amount of take would increase with an increase in turbidity and sediment deposition. Take caused by temporary reduced habitat conditions cannot be accurately quantified as a number of fish. This is because the relationship between habitat-related effects and the distribution and abundance of fish in the action area is not precisely known, and no specific number of individuals can be predicted. In such circumstances, NMFS uses the causal link established between the activity and a change in habitat conditions affecting the species to describe the extent of take as a quantified indicator of habitat perturbation. The consequence of these effects will be temporary, minor reductions in water quality that will cause most fish to avoid the treatment area, as measured by turbidity plumes extending from 50 feet upstream to 300 feet downstream of each disturbance. Based on the project description and the known distributions of ESA-listed anadromous fish, NMFS anticipates sediment-related take to occur only in lower Williams Creek, lower Deep Creek, Blackbird Creek, and Panther Creek, from Blackbird Creek downstream to Deep Creek.

Similarly, the total number of fish likely to be harmed or killed by pollutant spill or discharge cannot be quantified. Take could occur in the unlikely event of a transportation-related spill of chemicals or fuel along the transportation route. However, mortality associated with a toxic spill will be highly dependent upon the toxicity of the pollutant spilled, the stability of the container it's enclosed in, whether it's liquid or dry, whether it makes it into a waterbody, and what fish and life stages are present should a spill occur. This uncertainty is confounded by the fact that the number of fish in the action area, and the location of those individual fish when effects occur will vary throughout the duration of the project. Because pollutant spills are not expected to occur, NMFS does not anticipate take from a pollutant spill, and therefore does not authorize take for any spill of a pollutant in this Opinion.

NMFS also does not expect any lethal take of ESA-listed salmonids as a result of effluent discharge provided limits established by the NPDES permit are met. Therefore, NMFS does not authorize lethal take associated with pollutant levels exceeding NPDES maximum daily or average monthly limits, or from exceeding EPAs acute or chronic water quality criteria. However, based on the effects analysis above, it is likely that harm could occur to ESA-listed fish or their prey base from effluent discharges. This harm could occur in the form of bioaccumulation in fish tissues. Because neither steelhead or Chinook salmon are believed to spawn in Big Deer Creek, harm would only be expected to occur to juvenile salmonids. However, because EPA's chronic water quality criteria are set at levels intended to avoid chronic effects, harm from bioaccumulation of metals and other pollutants should not occur. Therefore, any measureable bioaccumulation of metals or other pollutants in fish tissues will indicate the amount of take authorized has been exceeded.

The extent of affected habitat exceeding that described for the Projects' activities, a pollutant spill, and/or any bioaccumulation of metals/pollutants from the effluent are the thresholds for reinitiating consultation. Should any of these limits be exceeded during Project activities, the reinitiation provisions of this Opinion will apply.

2.2.2. Reasonable and Prudent Measures

The RPMs are nondiscretionary measures to avoid or minimize take that must be carried out by cooperators for the exemption in section 7(o)(2) to apply. The USFS and EPA have the continuing duty to regulate the activities covered in this incidental take statement where discretionary Federal involvement or control over the action has been retained or is authorized by law. The protective coverage of section 7(o)(2) will lapse if the USFS or EPA fails to exercise their discretion to require adherence to terms and conditions of the incidental take statement, or to exercise that discretion as necessary to retain the oversight to ensure compliance with these terms and conditions. Similarly, if any applicant fails to act in accordance with the terms and conditions of the incidental take statement, protective coverage will lapse.

NMFS believes that full application of conservation measures included as part of the proposed action, together with use of the RPMs and terms and conditions described below, are necessary and appropriate to minimize the likelihood of incidental take of listed species due to completion of the proposed action.

The SCNF and EPA shall:

1. Minimize incidental take from construction related activities (SCNF).

2. Minimize incidental take from water quality related effects (SCNF and EPA).

3. Ensure completion of a monitoring and reporting program to confirm that the Terms and Conditions in this Incidental Take Statement are effective in avoiding and minimizing incidental take from permitted activities. Ensure completion of monitoring and reporting sufficient to determine the amount and/or extent of take described in this Opinion is not exceeded (SCNF and EPA).

2.2.3. Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the SCNF, EPA, and their cooperators, including the applicant, if any, must fully comply with conservation measures described as part of the proposed action and the following terms and conditions that implement the RPMs described above. Partial compliance with these terms and conditions may invalidate this take exemption, result in more take than anticipated, and lead NMFS to a different conclusion regarding whether the proposed action will result in jeopardy or the destruction or adverse modification of designated critical habitats.

1. To implement RPM #1, minimizing incidental take from construction related activities, the SCNF shall ensure that:

    a. Prior to beginning work, all contractors working on-site shall be provided with a complete list of proposed BMPs, RPMs, and terms and conditions intended to minimize the amount and extent of take resulting from riparian disturbance and general construction activities.

    b. Confine construction impacts to the minimum area necessary to complete the Project, particularly where project activities affect riparian vegetation.

    c. Activity in the stream shall be kept to the minimum necessary. Equipment shall be moved to an upland location at least 150 feet from the water prior to refueling, repair, or maintenance.

d. Equipment will work from the streambank during installation of the effluent pipeline. Use work area isolation methods (complete or partial) and sediment containment measures (i.e. Sedimat™, straw bales, silt fence) to reduce the amount of sediment transported downstream during construction.

e. Project design criteria and BMPs associated with the culvert removal on North Fork Williams Creek will be submitted to NMFS for review and approval prior to beginning road reconstruction efforts on the Williams Creek Road.

f. At a minimum, culvert removal shall be conducted in accordance with project design criteria and BMPs established in NMFS R1/R4 Stream Crossing Structure Replacement and Removal Programmatic Biological Opinion (NMFS No. P/NWR/2005/06396). http://bluefin2.nmfs.noaa.gov/pls/pcts-pub/sxn7.pcts_upload.download?p_file=F18879/200506396_culvert_programmatic_08-08-2006.pdf

g. Ensure an adequate supply of sediment control materials (e.g., Sedimat™, silt fence, straw bales, etc.) is available on-site to address emergency situations should they arise.

h. Visual turbidity monitoring will be completed during construction of the effluent pipeline, construction of the cable car, and during road construction activities by observing any sediment plumes that might be caused by project activities. If the sediment plumes are visible more than 300 feet downstream, the USFS shall immediately notify NMFS to determine if reinitiation of consultation is necessary.

i. Specific to the culvert removal action on North Fork Williams Creek, a Fisheries Biologist or Hydrologist will monitor turbidity levels downstream from construction activities to ensure that levels do not exceed State standards of 50 nephelometric turbidity units (NTUs) above baseline (instantaneous), or > 25 NTUs above baseline (chronic). If available, a NMFS representative shall be present on-site during dewatering and re-watering of the North Fork.

j. Toxic materials do not enter live water during construction activities. Equipment used in construction of the cable car crossing on Panther Creek and the construction of the NPDES outfall on Big Deer Creek shall be clean and free of fuel and lubricant leaks prior to beginning work, and shall be inspected daily once beginning work.

k. Prior to and during the equipment crossing of Panther Creek for the cable car installation, a fish biologist shall be present on-site. The fish biologist shall walk alongside the stream and look for adult steelhead and/or steelhead redds within 350 feet of the construction site. If either adult or redd are observed, activity will cease until the fish biologist has coordinated with NMFS to determine the best approach to proceed while avoiding effects to adult steelhead and/or their redds.

l. As outlined in Section 1.2.15 of this Opinion, NMFS shall participate on the Interagency Oversight Task Force to assist with oversight of the ICP. NMFS shall review and provide input where appropriate to all project design reports, documents, and annual reports. At a minimum, as an active member of this group, NMFS expects to be able to review and provide input to the following elements described in the proposed action:

   i. The engineering design for the water treatment system;

   ii. Final design of the TWSF;

   iii. Final road construction/reconstruction plan;

   iv. Final stormwater pollution prevention plan;

   v. Final geochemical amendment plan;

   vi. Final design for the outfall diffuser;

   vii. Post-mining groundwater/surface water capture plan;

   viii. Weed control plan;

   ix. Wetland monitoring plan;

   x. Copper loading demonstration plan;

   xi. Methylmercury study plan and report; and,

   xii. Final reclamation and closure plan.

2. To implement RPM #2, minimizing incidental take from effects to water quality:

   a. The EPA shall modify the draft NPDES permit to limit the effluents maximum daily concentration for levels of nitrate + nitrite to <10 mg/L at the end-of-pipe to prevent nutrient enrichment of habitat in Big Deer and Panther Creeks.

   b. The SCNF shall work with the applicant to prioritize, schedule, and complete road reconstruction/improvements to ensure that all road segments with environmental or safety concerns are addressed in Phase I. This includes all road segments in RHCAs or draining directly into perennial or intermittent streams. The SCNF will work with the Interagency Oversight Task Force to identify and prioritize the road segments of concern.

c. The SCNF shall ensure that an appropriate native seed mix is used to mulch and seed all cuts and fills of roads, and disturbed areas from road maintenance. As described in the Mitigations section of the ICP DEIS (page 2-55), Item 3/c), disturbed areas will be treated during the same years as the construction/ disturbance activity. If vegetation is not adequately established for erosion control the mulch and seed will be applied in subsequent years until natural vegetation is established.

d. The SCNF shall require the FCC implement the following process (Table 17) to screen new reagents/formulas before changing the manufacturer, the formula, or adding a chemical not considered in the BA.

   i. **Toxicity -** If the new material is considered highly or very highly toxic with a 96 hr LC_{50} < 1,000 µg/L for fish species or aquatic invertebrates the material needs to be carefully reviewed regardless of accident probability or spill risk. If the toxicity of the proposed new material is below this threshold then the spill risk and accident probability need to be considered and evaluated in coordination with NMFS.

   ii. **Screen for Probability of Accident –** If the accident probability as described in the BA indicates that accidents near a stream are not likely to occur in more than 100 years (equates to < 59 trips/year), and toxicity is rated Moderate or lower, then no additional analysis will be required regardless of spill risk. However, if the accident rate predicts that accidents near a stream would occur in less than 100 years (>59 trips/year), toxicity is rated Moderate or High, and spill risk is rated High, additional analysis will need to be completed in coordination with NMFS.

Table 17. Screen to identify when additional toxic effects analysis is needed.

# of Trips/ year (accident probability near streams)	Spill Risk	Toxicity	Coordination Needed?
Any	Any	very highly toxic or highly toxic (Fish 96 hr LC_{50} < 1,000 µg/L)	Additional Analysis and Coordination with NMFS Necessary.
>59 (less than 100 yrs between accidents)	High or Moderate	moderately toxic (Fish 96 hr LC_{50} < 10,000 µg/L)	Additional Analysis and Coordination with NMFS Necessary.
<59 (more than 100 yrs between accidents)	Low, Moderate, or High	moderately toxic, slightly toxic, not acutely toxic	No Coordination with NMFS Necessary.

The risk of a material spill happening in case of an accident is determined based on the material packaging. Containerized solid = **Low Risk**, Containerized liquids in small containers (<100 gallons) = **Moderate Risk**, Bulk liquids = **High Risk**.

3. To implement RPM #3, monitoring and reporting:

   a. The SCNF shall monitor and report compliance with the project's proposed effects minimization measures. Ensure completion of a monitoring and reporting program to confirm that the amount and/or extent of take anticipated in this Opinion is not exceeded and that the project is implemented as proposed.

      i. Annually report on the compliance with and implementation of the RPMs and Terms and Conditions.

      ii. Adhere to the proposed monitoring as described in the ICP BA, ROD, and Supplemental Reports.

   b. The EPA shall work with FCC to develop a tissue sampling protocol and sampling scheme for salmonids in Big Deer Creek. The protocol and sampling scheme must be approved by NMFS prior to first effluent discharge. A baseline study shall be conducted prior to first effluent discharge, and annually for 3 years following, conduct tissue sampling of non-ESA listed resident salmonids in Big Deer Creek collected downstream from effluent and upstream from the falls for:

      i. Bioaccumulation of aluminum, arsenic, cadmium, cobalt, lead, manganese, mercury, nickel, selenium, thallium, and zinc;

   Measureable bioaccumulation of these metals and pollutants will indicate the amount of take authorized has been exceeded. If resident fish are not collected in numbers suitable for tissue sampling purposes, coordinate with NMFS to develop an alternative sampling protocol.

   c. The EPA shall work with FCC to develop an aquatic invertebrate sampling scheme and protocol in Big Deer Creek. The protocol and sampling scheme must be approved by NMFS prior to first effluent discharge. Prior to first effluent discharge, and annually for 3 years following, conduct sampling of aquatic invertebrates in Big Deer Creek to assess the potential for bioaccumulation of pollutants and/or changes in community structure. Measureable bioaccumulation of metals/pollutants identified in Term and Condition 3.b.i. and/or changes in community structure will indicate the amount of take authorized has been exceeded.

   d. The SCNF and EPA will annually report monitoring results as described in the ICP BA, ROD, Supplemental Reports, and this Opinion. The report shall identify in separate sections: (1) any results indicating adverse habitat modification or other adverse effects of the action on spring/summer Chinook salmon, steelhead, or sockeye salmon; (2) persistence of adverse conditions that could be improved through modification of the proposed action, or through additional actions; and

(3) recommended remedies to address the problems identified in items 1 and 2. NMFS shall work with the SCNF and EPA to determine any corrective actions, which the applicant must implement.

e. The SCNF and EPA shall submit reports and annual monitoring results noted in the BA, ROD, Supplemental Reports, and this Opinion to: NMFS, Attn: David Mabe, 10095 W Emerald, Boise, Idaho 83704.

f. NOTICE: If a sick, injured or dead specimen of a threatened or endangered species is found in the project area, the finder must notify NMFS through the contact person identified in the transmittal letter for this Opinion, or through Idaho State Habitat Office of NMFS Law Enforcement at (208) 321-2956, and follow any instructions. If the proposed action may worsen the fish's condition before NMFS can be contacted, the finder should attempt to move the fish to a suitable location near the capture site while keeping the fish in the water and reducing its stress as much as possible. Do not disturb the fish after it has been moved. If the fish is dead, or dies while being captured or moved, report the following information: (1) NMFS consultation number; (2) the date, time, and location of discovery; (3) a brief description of circumstances and any information that may show the cause of death; and (4) photographs of the fish and where it was found. NMFS also suggests that the finder coordinate with local biologists to recover any tags or other relevant research information. If the specimen is not needed by local biologists for tag recovery or by NMFS for analysis, the specimen should be returned to the water in which it was found, or otherwise discarded.

3. MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT

The consultation requirement of section 305(b) of the MSA directs Federal agencies to consult with NMFS on all actions, or proposed actions that may adversely affect EFH. Adverse effects include the direct or indirect physical, chemical, or biological alterations of the waters or substrate and loss of, or injury to, benthic organisms, prey species and their habitat, and other ecosystem components, if such modifications reduce the quality or quantity of EFH. Adverse effects to EFH may result from actions occurring within EFH or outside EFH, and may include site-specific or EFH-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810). Section 305(b) also requires NMFS to recommend measures that may be taken by the action agency to conserve EFH.

The Pacific Fishery Management Council (PFMC) designated EFH for groundfish (PFMC 1998a), coastal pelagic species (PFMC 1998b), Chinook salmon, coho salmon, and Puget Sound pink salmon (PFMC 1999). The proposed action and action area for this consultation are described in the Introduction to this document. The action area includes areas designated as EFH for various life-history stages of Chinook salmon.

Based on information provided in the BA and the analysis of effects presented in the ESA portion of this document, NMFS concludes that proposed action will have the following adverse effects on EFH designated for Snake River spring/summer Chinook salmon:

1. Localized effects to instream habitat – increased turbidity, sediment deposition, and/or riparian disturbance/streambank alteration, as described in Sections 2.1.3.1 and 2.1.3.2 of this Opinion;

2. Localized effects to water quality – increased chemical contamination, as described in Sections 2.1.3.1 and 2.1.3.2 of this Opinion.

3.1. EFH Conservation Recommendations

NMFS believes that the following two conservation measures are necessary to avoid, mitigate, or offset the impact of the proposed action on EFH. NMFS believes that RPMs 1 and 2 of this Opinion and their implementing terms and conditions are necessary to avoid, mitigate, or offset the impact of the proposed action on EFH. These conservation recommendations are a non-identical subset of the ESA RPMs and Terms and Conditions.

3.2. Statutory Response Requirement

Federal agencies are required to provide a detailed written response to NMFS' EFH conservation recommendations within 30 days of receipt of these recommendations [50 CFR 600.920(j)(1)]. The response must include a description of measures proposed to avoid, mitigate, or offset the adverse affects of the activity on EFH. If the response is inconsistent with the EFH conservation recommendations, the response must explain the reasons for not following the recommendations. The reasons must include the scientific justification for any disagreements over the anticipated effects of the proposed action and the measures needed to avoid, minimize, mitigate, or offset such effects.

In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many conservation recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, in your statutory reply to the EFH portion of this consultation, we ask that you clearly identify the number of conservation recommendations accepted.

3.3. Supplemental Consultation

The USFS and EPA must reinitiate EFH consultation with NMFS if the proposed action is substantially revised in a way that may adversely affect EFH, or if new information becomes available that affects the basis for NMFS' EFH conservation recommendations [50 CFR 600.920(k)].

4. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

Section 515 of the Treasury and General Government Appropriations Act of 2001 (Public Law 106-554) (Data Quality Act [DQA]) specifies three components contributing to the quality of a document. They are utility, integrity, and objectivity. This section of the Opinion addresses these DQA components, documents compliance with the DQA, and certifies that this Opinion has undergone pre-dissemination review.

Utility: Utility principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users.

This ESA consultation concludes that the proposed approval of FCC's POO for the ICP and the EPA's issuance of an NPDES permit for discharging treated mine wastewater into Big Deer Creek will not jeopardize the affected listed species. Therefore, the SCNF can authorize this action in accordance with its authority under the General Mining Law of 1872 and FLPMA. The EPA can authorize the wastewater discharge in accordance with its authority under the Clean Water Act. The intended users are the SCNF, EPA, and FCC.

Individual copies were provided to the above-listed entities. This consultation will be posted on NMFS Northwest Region website (http://www.nwr.noaa.gov). The format and naming adheres to conventional standards for style.

Integrity: This consultation was completed on a computer system managed by NMFS in accordance with relevant information technology security policies and standards set out in Appendix III, 'Security of Automated Information Resources,' Office of Management and Budget Circular A-130; the Computer Security Act; and the Government Information Security Reform Act.

Objectivity:

Information Product Category: Natural Resource Plan.

Standards: This consultation and supporting documents are clear, concise, complete, and unbiased; and were developed using commonly accepted scientific research methods. They adhere to published standards including NMFS ESA Consultation Handbook, ESA Regulations, 50 CFR 402.01, *et seq.*, and the MSA implementing regulations regarding EFH, 50 CFR 600.920(j).

Best Available Information: This consultation and supporting documents use the best available information, as referenced in the Literature Cited section. The analyses in this Opinion/EFH consultation contain more background on information sources and quality.

Referencing: All supporting materials, information, data and analyses are properly referenced, consistent with standard scientific referencing style.

Review Process: This consultation was drafted by NMFS staff with training in ESA and MSA implementation, and reviewed in accordance with Northwest Region ESA quality control and assurance processes.

5. LITERATURE CITED

Allin, C.J., and R.W. Wilson. 1999. Behavioral and metabolic effects of chronic exposure to sublethal aluminum in acidic soft water in juvenile rainbow trout (*Oncorhynchus mykiss*). Can. J. Fish. Aquat. Sci. 56: 670-678.

Alsop, D.H., J.C. McGeer, D.G. McDonald, and C.M. Wood. 1999. Costs of chronic waterborne zinc exposure and the consequences of zinc acclimation on the gill/zinc interactions of rainbow trout in hard and soft water. Environmental Toxicology and Chemistry 18(5): 1014-1025.

Andersen, O. 1997. Accumulation of waterborne iron and expression of ferritin and transferring in early developmental stages of brown trout (*Salmo trutta*). Fish Physiology and Biochemistry 16: 223-231.

ATSDR. (Agency for Toxic Substances and Disease Registry). 1992. Toxicological profile for thallium. Agency for Toxic Substances and Disease Registry, Atlanta, Georgia. 114 pages. http://www.atsdr.cdc.gov/toxprofiles/tp54.pdf

ATSDR. 1995. ToxFAQs for Thallium. Agency for Toxic Substances and Disease Registry, Atlanta, Georgia. http://www.atsdr.cdc.gov/tfacts54.html

Baldwin, D. H., J. F. Sandahl, J. S. Labenia, and N. L. Scholz. 2003. Sublethal effects of copper on coho salmon: Impacts on nonoverlapping receptor pathways in the peripheral olfactory nervous system. Environ. Toxicol. Chem. 22:2266–2274.

Barron, M.G., and S. Albeke. 2000. Calcium control of zinc uptake in rainbow trout. Aquatic Toxicology 20: 257-264.

Barrows, M.E., S.R. Petrocelli, and K.J. Macek. 1978. Bioconcentration and elimination of selected water pollutants by bluegill sunfish. In: Haque R., ed. Dynamics, exposure, and hazard assessment of toxic chemical. Ann Arbor, Michigan. Ann Arbor Science Publishers, Inc. pp. 379-392.

Bash, J., C. Cerman, and S. Bolton. 2001. Effects of Turbidity and Suspended Solids on Salmonids. Center for Streamside Studies, University of Washington. 74 pgs.

BC Research Inc. 1998. Benda mines sulphate and molybdenum toxicity testing. Prepared for Noranda Mining and Exploration Inc., Brenda Mines Div Project No: 2-11-825/826 (as cited in Singleton H. 2000)

Behnke, R.J. 2002. Native trout of western North America. American Fisheries Society Monograph 6.

Beiningen, K.T. 1976. Fish runs. Pages E-1-E-65 in Investigative reports of Columbia River fisheries project. Pacific Northwest Regional Commission, Vancouver, Washington.

Beltman, D.J., W.H. Clements, J. Lipton, and D. Cacela. 1999. Benthic invertebrate exposure, accumulation, and community-level effects downstream from a hard-rock mine site. Environmental Toxicology and Chemistry 18(2): 229-307.

Benoit, D.A., E.N. Leonard, G.M. Christensen, and J.T. Fiandt. 1976. Toxic effects of cadmium on three generations of brook trout (*Salvelinus fontinalis*). Transactions of the American Fisheries Society 105: 550-560.

Berg, L., and T. G. Northcote. 1985. Changes in territorial, gill-flaring, and feeding behavior in juvenile coho salmon (*Oncorhynchus kisutch*) following short-term pulses of suspended sediment. Canadian Journal of Fisheries and Aquatic Science 42: 1410-1417.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research 14:1011-1016.

Beven, D., J. Harville, P. Bergman, T. Bjornn, J. Crutchfield, P. Klingeman, and J. Litchfield. 1994. Snake River Salmon Recovery Team: Final Recommendations to National Marine Fisheries Service (NMFS). National Marine Fisheries Service, Seattle, Washington. (May 1994).

Bisson, P. A., and R. E. Bilby. 1982. Avoidance of suspended sediment by juvenile coho salmon. North American Journal Fisheries Management 4: 371-374.

Bjornn, T.C. 1991. Spawning and Development. *The Wildlife Series: Trout.* Harrisburg, PA. pp. 57-64.

Bowen, L., I. Werner, and M.L. Johnson. 2006. Physiological and behavioral effects of zinc and temperature on coho salmon (*Oncorhynchus kisutch*). Hydrobiologia 559: 161-168.

Brix, K.V., J. Keithly, D.K. DeForest, and J. Laughlin. 2004. Acute and chronic toxicity of nickel to rainbow trout (*Oncorhynchus mykiss*). Environmental Toxicology and Chemistry 23(9): 2221-2228.

BRT (Biological Review Team). 1998. Status Review Update for West Coast Chinook Salmon (*Oncorhynchus tshawytscha*) From Puget Sound, Lower Columbia River, Upper Willamette River, and UCR Spring-Run ESUs. West Coast Chinook Salmon BRT. Seattle, Washington.

BRT. 2003. Updated status of federally listed ESUs of West Coast salmon and steelhead. U. S. Department of Commerce, National Marine Fisheries Service, Northwest Fisheries Science Center and Southwest Fisheries Science Center, Seattle, Washington (July 2003). http://www.nwr.noaa.gov/AlseaResponse/20040528/brtusr.html

Budy, P., G.P. Thiede, N. Bouwes, C.E. Petrosky, and H. Schaller. 2002. Evidence linking delayed mortality of Snake River salmon to their earlier hydrosystem experience. N. Amer. J. Fish. Mgmt. 22:35-51.

Burkhalter, D.E. and Kaya, C.M.. 1977. Effects of prolonged exposure to ammonia on fertilized eggs and sac fry of rainbow trout (*Salmo gairdneri*). Transactions of the American Fisheries Society 106: 470-475.

Busby, P.J., T.C. Wainwright, G.J. Bryant, L. Lierheimer, R.S. Waples, F.W. Waknitz, and I.V. Lagomarsino. 1996. Status review of west coast steelhead from Washington, Idaho, Oregon, and California. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-27, 261p.

Camargo, J.A., A. Alonso, and A. Salamanca. 2005. Nitrate toxicity to aquatic animals: a review with new data for freshwater invertebrates. Chemosphere 58: 1255-1267.

Cataldo, D.A. and R.E. Wildung. 1983. The role of soil and plan metabolic processes in controlling trace element behavior and bioavailability to animals. Science of the Total Environment 28: 159-168.

Cederholm, C.J., and L.M. Reid. 1987. Impact of forest management on coho salmon (*Oncorhynchus kisutch*) populations of the Clearwater River, Washington: a project summary. In *Streamside Management: Forestry and Fishery Interactions*; E.O. Salo and T.W. Cundy, eds. Pgs. 373-398. Contribution 57, University of Washington, Institute of Forest Resources. Seattle, WA.

Cederholm, C.J., M.D. Kunze, T. Murota, and A. Sibatani. 1999. Pacific salmon carcasses: Essential contributions of nutrients and energy for aquatic and terrestrial ecosystems. Fisheries 24(10): 6-15.

Chambers, P.A., M. Guy, E.S. Roberts, M.N. Charlton, R. Kent, C. Gagnon, G. Grove and N. Foster. 2001. Nutrients and their Impact on the Canadian Environment. Agriculture and Agri-Food Canada , Environment Canada , Fisheries and Oceans Canada , Health Canada , and Natural Resources Canada . En21-205/2001E. http://www.nwri.ca/nutrients-nutritifs/pdfs/nutrients-nutritifs-e.pdf

Chapman, D.J. and B.E. Julius. 2005. The use of preventative projects as compensatory restoration. *Journal of Coastal Research* 40:120-131.

Chapman, D.W., W.S. Platts, E. Park, and M. Hill. 1990. Status of Snake River sockeye salmon. Final Report for Pacific Northwest Utilities Conference Committee. June 26, 1990. 90 pages.

Chapman, G. A. 1978. Toxicities of cadmium, copper, and zinc to four juvenile stages of Chinook salmon and steelhead. Transactions of the American Fisheries Society 107(6): 841–847.

Chapman, G. A. 1982. Chinook salmon early life stage tests with cadmium, copper, and zinc. Letter of December 6, 1982 to Charles Stephan, EPA Environmental Research Laboratory, Duluth. Environmental Protection Agency, Corvallis, Oregon. Online at http://www.epa.gov/ow/docket.html.

Chapman, G. A., and D. G. Stevens. 1978. Acutely lethal levels of cadmium, copper, and zinc to adult male coho salmon and steelhead. Transactions of the American Fisheries Society 107(6):837–840.

CHEVRON. 2005. Material Safety Data Sheet (MSDS): Chevron and Texaco Regular Unleaded Gasoline's.

CITGO. 2007. MSDS: Citgo Gasolines, All Grades, Unleaded.

Cleveland, L., D.R. Buckler, and W.G. Brumaugh. 1991. Residue dynamics and effects of aluminum on growth and mortality in brook trout. Environmental Toxicology and Chemistry 10: 243-248.

Cline, C. 1973. The effects of forest fertilization on the Tahuya River, Kitsap Peninsula, Washington. Washington State Department of Ecology. 55pp.

Cockell, K.A. and J. W. Hilton. 1988. Preliminary investigations on the comparative chronic toxicity of four dietary arsenicals to juvenile rainbow trout (*Salmo gairdneri R.*). Aquatic Toxicology 12(1): 73-82.

CONOCO. 2000. MSDS: No. 1 Diesel Fuel, No. 1 Fuel Oil.

COE (U.S. Army Corps of Engineers). 1999. Draft Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement. U.S. Army Corps of Engineers, Walla Walla, Washington.

COE. 2005. Natural resource management section: fish counts. Portland District, U.S. Army Engineers. https://www.nwp.usace.army.mil/op/fishdata/

Cooney, T. 2004. Updated trend data sets for Interior Columbia basin ESUs. October. http://www.salmonrecovery.gov/remand/analysis_reports/updated_interior_brt_trend_data.pdf

Coutant, C.C. 1973. Effects of thermal shock on vulnerability of juvenile salmonids to predation. J. Fish. Res. Bd. Canada 30:965-973.

CRFC (Columbia River Fisheries Council). 1979. Rationale for instream flows for fisheries in the Columbia and Snake Rivers. CRFC, Portland, Oregon.

CYTEC. 2001. MSDS: SUPERFLOC™ 330 Flocculant. MSDS No. 01922.

CYTEC. 2006. MSDS: SUPERFLOC® 577 Flocculant. MSDS No. 0035607.

Dahlberg, M.L., D.L. Shumway, and P. Doudoroff. 1968. Influence of dissolved oxygen concentration on swimming performance of largemouth bass and coho salmon. J. Fish. Res. Board Canada 25: 49-70.

Davies, P.H., J.P. Goettl, Jr., J.R. Sinley, and N.F. Smith. 1976. Acute and chronic toxicity of lead to rainbow trout (*salmo gairdneri*), in hard and soft water. Water Research 10(3): 199-206.

Davies, T.D. 2007. Sulphate toxicity to the aquatic moss, *Fontinalis antipyretica*. Chemosphere 66: 444-451.

Davies, T.D., and K.J. Hall. 2007. Importance of calcium in modifying the acute toxicity of sodium sulphate to *Hyalella azteca* and *Daphnia magna*. Environmental Toxicology and Chemistry 26(6): 1243-1247.

De Schamphelaere, K.A.C., and C.R. Janssen. 2004. Bioavailability and chronic toxicity of zinc to juvenile rainbow trout (*Oncorhynchus mykiss*): Comparison with other fish species and development of a biotic ligand model. Environmental Science and Technology 38(23): 6201-6209.

Dorn, M.W. 1989. A conditional logistic regression model for the onset of riverine salmon migrations. M.S. thesis. University of Washington, Seattle.

Dussault, È.B., R.C. Playle, D.G. Dixon, and R.S. McKinley. 2004. Effects of chronic aluminum exposure on swimming and cardiac performance in rainbow trout (*Oncorhynchus mykiss*). Fish Physiology and Biochemistry 30: 137-148.

Ecometrix. 2006. Biomonitoring Study: Panther Creek Watershed September 2005 Report prepared for: Blackbird Mine Site Group, Salmon, Idaho. EcoMetrix Inc., Brampton, Ontario.

Eisler, R. 1985a. Cadmium hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service, Biological Report 85(1.2), Contaminant Hazard Review Report Number 2. 30 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_2_Cadmium.pdf

Eisler, R. 1985b. Selenium hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service, Biological Report 85(1.5), Contaminant Hazard Reviews Report Number 5. 41 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_5_Selenium.pdf

Eisler, R. 1988a. Arsenic hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service, Biological Report 85(1.12), Contaminant Hazards Review Report Number 12. 65 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_12_Arsenic.pdf

Eisler, R. 1988b. Lead hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service, Biological Report 85(1.14), Contaminant Hazards Review Report Number 14. 94 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_14_Lead.pdf

Eisler, R. 1993. Zinc hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service, Biological Report 10, Contaminant Hazards Review Report Number 10. 126 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_26_Zinc.pdf

Eisler, R. 1997. Mercury hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service, Biological Report 85(1.10), Contaminant Hazards Review Report Number 10. 63 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_10_Mercury.pdf

Eisler, R. 1998a. Copper hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Geological Survey, Contaminant Hazard Reviews, Biological Report 33. 120 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_33_Copper.pdf

Eisler, R. 1998b. Nickel hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service, Biological Report 85(1.14), Contaminant Hazards Review Report Number 34. 95 pages. http://www.pwrc.usgs.gov/infobase/eisler/CHR_34_Nickel.PDF

El-Mowafi, A.F.A., and A. Maage. 1998. Magnesium requirement of Atlantic salmon (*Salmo salar* L.) parr in seawater-treated fresh water. Aquaculture Nutrition 4: 31-38.

Environmental Protection Agency (EPA). 1986. Quality Criteria for Water 1986. U.S. Environmental Protection Agency, EPA 440/5-86-001. Washington D.C.

EPA. 1999. 1999 Update of Ambient Water Quality Criteria for Ammonia. U.S. Environmental Protection Agency, EPA-822-R-99-014. Washington D.C.. 153 pp.

EPA. 2000. Ambient water quality criteria recommendations, rivers and streams in nutrient ecoregion II - western forested mountains. U.S. Environmental Protection Agency, EPA 822-B-00-015, Washington, DC. 39 pp. http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/rivers/index.html.

EPA. 2003a. Biological evaluation for the Potlatch Mill NPDES Permit (NPDES Permit no. ID0001163). U.S. Environmental Protection Agency, Seattle, WA. 438 pp.

EPA. 2003a. EPA Region 10 guidance for Pacific Northwest state and tribal temperature water quality standards. EPA 910-B-03-002, U.S. Environmental Protection Agency, Region 10, Office of Water, Seattle, Washington. 49 p.

EPA. 2003b. EPA National primary drinking water standards. EPA 816-F-03-016. EPA Office of Water, Washington, D.C.

EPA. 2004. Draft selenium aquatic life criterion: Fact sheet (December 2004). U.S. Environmental Protection Agency, Seattle, Washington. http://www.epa.gov/waterscience/criteria/selenium/fs.htm

EPA. 2007a. Draft NPDES Permit for Formation Capital Corporation, Idaho Cobalt Project, ID-002832-1. U.S. Environmental Protection Agency, Seattle, Washington. http://yosemite.epa.gov/r10/water.nsf/95537302e2c56cea8825688200708c9a/d99f1f33c25fe20388256d9d005f5116/$FILE/ID0028321DP.pdf

EPA. 2007b. Draft NPDES Permit fact sheet for Formation Capital Corporation, Idaho Cobalt Project, ID-002832-1. U.S. Environmental Protection Agency, Seattle, Washington. http://yosemite.epa.gov/r10/water.nsf/95537302e2c56cea8825688200708c9a/d99f1f33c25fe20388256d9d005f5116/$FILE/ID0028321FS.pdf

EPA. 2007c. 2007 updated aquatic life copper criteria. U.S. Environmental Protection Agency, Washington, D.C. (April 11, 2007). http://www.epa.gov/waterscience/criteria/copper/2007/index.htm

EPA. 2007d. Toxicity profiles. U.S. Environmental Protection Agency, Region 5 Superfund, Chicago, Illinois. 9 pages. http://www.epa.gov/R5Super/ecology/html/toxprofiles.htm

Exley, C. 2000. Avoidance of aluminum by rainbow trout. Environmental Toxicology and Chemistry 19(4): 933-939.

Ferreira, V., V. Gulis, and M.A.S. Graça. 2006. Whole-stream nitrate addition affects litter decomposition and associated fungi but not invertebrates. Oecologia 149: 718-729.

Finlayson, B.J., and K.M. Verrue. 1982. Toxicities of copper, zinc, and cadmium mixtures to juvenile Chinook salmon. Transactions of the American Fisheries Society 111: 645-650.

Fish Passage Center. 2002. Smolt data, current and historical. Fish Passage Center, 2501 SW First Avenue, Suite 230, Portland, Oregon 97201-4752. 52. http://www.fpc.org/smolt_home.html

Fish Passage Center. 2003. Adult data, current and historical. Fish Passage Center, 2501 SW First Avenue, Suite 230, Portland, Oregon 97201-4752. http://www.fpc.org/adultsalmon/adultqueries/Adult_Table_Submit.html

Fish Passage Center. 2004. Adult salmon passage at Lower Granite Dam. Fish Passage Center, 2501 SW First Avenue, Suite 230, Portland, Oregon 97201-4752. http://www.fpc.org/adultsalmon/adulthistory/YTD-LGR.html

Fish Passage Center. 2007. Adult Returns for Columbia and Snake River Dams. Fish Passage Center. Portland, Oregon. http://www.fpc.org/adultsalmon/adultqueries/Adult_Annual_Totals_Query_Results.asp

Flanagan, D.C. and S.J. Livingston. 1995. WEPP User Summary. National Soil Erosion Research Laboratory Report No. 11. W. Lafayette, Indiana.

Formation Capital Corporation. 2002. Idaho Cobalt Project Spill Prevention and Response Plan.

Frahm, J.P. 1975. Toxicity tolerance studies utilizing periphyton (Toxitoleranzversuche an Wassermoosen). Gewasser Abwasser (57/58), 59-66 (GER). As cited in Davies 2007.

Fuller, S.A., J.P. Henne, and G.J. Carmichael. 2003. Toxicity of ammonia and nitrate to the Gila Trout. North American Journal of Aquaculture 65: 162-164.

Furniss, M.J., T.D. Roelofs, and C.S. Yee. 1991. Road construction and maintenance. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*; W.R. Meehan, ed. Pgs. 297-324. American Fisheries Society Special Pub. 19. Bethesda, MD.

Galvez, F., N. Webb, C. Hogstrand, and C.M. Wood. 1998. Zinc binding to the gills of rainbow trout: the effect of long-term exposure to sublethal zinc. Journal of Fish Biology 52: 1089-1104.

Gardner, R.B. 1979. Some environmental and economic effects of alternative forest road designs. Transactions of the American Society of Agricultural Engineers 22:63-68.

Giattina, J.D., R.R. Garton, and D.G. Stevens. 1982. Avoidance of copper and nickel by rainbow trout as monitored by a computer-based data acquisition system. Transactions of the American Fisheries Society. 111: 491-504.

Goldstein, J.N., D.F. Woodward, and A.M. Farang. 1999. Movements of adult Chinook salmon during spawning migration in a metals-contaminated system, Coeur d'Alene River, Idaho. Transactions of the American Fisheries Society 128: 121-129.

Golterman, H.L. 1975. Physiological limnology: An approach to the physiology of lake exosystems. Elsevier Scientific Publishing Company. New York, New York. 489pp. As cited in USDA Forest Service 2007.

Good, T.P., R.S. Waples, and P. Adams (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Dept. Commerce., NOAA Tech. Memo. NMFS-NWFSC-66, 598 p. http://santacruz.nmfs.noaa.gov/files/pubs/00749.pdf

Gregory, R. S., and T. S. Northcote. 1993. Surface, planktonic, and benthic foraging by juvenile Chinook salmon (*Oncorhynchus tshawytscha*) in turbid laboratory conditions. Canadian Journal of Fisheries and Aquatic Sciences 50: 223-240.

Gregory, S.V. and P.A. Bisson. 1997. Degradation and loss of anadromous salmonid habitat in the Pacific Northwest. *In:* D.J. Stouder, P.A. Bisson and R.J. Naimon (eds.). Pacific salmon and their ecosystems: Status and future options, pp. 277-314. Chapman and Hall, New York.

Groot, C. and L. Margolis. 1991. Pacific Salmon Life Histories. University of British Columbia Press, Vancouver, British Columbia, Canada. 564 pages.

Hale, J.G. 1977. Toxicity of Metal Mining Wastes. Bulletin of Environmental Contamination and Toxicology 17(1): 66-73.

Hamilton, S.J. and P.M. Mehrle. 1987. Evaluation of metallothionein measurement as a biological indicator of stress from cadmium in brook trout. Transactions of the American Fisheries Society 166: 551-560.

Hansen, J. A., J. D. Rose, R. A. Jenkins, K. G. Gerow, and H. L. Bergman. 1999a. Chinook salmon (*Oncorhynchus tshawytscha*) and rainbow trout (*Oncorhynchus mykiss*) exposed to copper: Neurophysiological and histological effects on the olfactory system. Environ. Toxicol. Chem. 18:1979–1991.

Hansen, J. A., J.C.A. Marr, J. Lipton, D. Cacela, and H.L. Bergman. 1999b. Differences in neurobehavioral responses of Chinook salmon (*Oncorhynchus tshawytscha*) and rainbow trout (*Oncorhynchus mykiss*) exposed to copper and cobalt: Behavioral avoidance. Environmental Toxicology and Chemistry 18(9): 1972-1978.

Hansen, J. A., D.F. Woodward, E.E. Little, A.J. DeLonay, and H.L. Bergman. 1999c. Behavioral avoidance: Possible mechanism for explaining abundance and distribution of trout species in a metal-impacted river. Environ. Toxicol. Chem. 18(2): 313-317.

Hauge, C.J., M.J. Furniss, and F.D. Euphrat. 1979. Soil erosion in California's coast forest district. California Geology (June):120-129.

Haupt, H.F. 1959. Road and slope characteristics affecting sediment movement from logging roads. Journal of Forestry 57:329-332.

Hebdon, J.L., P. Kline, D. Taki, and T.A. Flagg. 2004. Evaluating reintroduction strategies for Redfish Lake sockeye salmon captive broodstock strategy. Pages 401-413 *in* M. J. Nickum, P.M. Mazik, J. G. Nickum, and D.D. MacKinlay, editors. Propagated fish in resource management. American Fisheries Society, Symposium 44, American Fisheries Society, Bethesda, Maryland.

Hecht, S.A., D.H. Baldwin, C.A. Mebane, T. Hawkes, S.J. Gross, and N.L. Scholz. 2007. An overview of sensory effects on juvenile salmonids exposed to dissolved copper: Applying a benchmark concentration approach to evaluate sublethal neurobehavioral toxicity. NOAA Tech. Memo. NMFS-NWFSC-83. 25 pp. + Appendices. Seattle, Washington.

Heit M. and C.S. Klusek. 1985. Trace element concentrations in the dorsal muscle of white suckers and brown bullheads from two acidic Adirondack lakes. Water, Air and Soil Pollution 25:87-96.

Hicks, B.J., J.D. Hall, P.A. Bisson, and J.R. Sedell. 1991. Responses of salmonids to habitat changes. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*; W.R. Meehan, ed. Pgs. 297-324. American Fisheries Society Special Pub. 19. Bethesda, MD.

Holcombe, G.W., D.A. Benoit, and E.N. Leonard. 1979. Long-term effects of zinc exposures on brook trout (*Salvelinus fontinalis*). Transactions of the American Fisheries Society 108: 76-87.

Holcombe, G.W., D.A. Benoit, E.N. Leonard, and J.M. McKim. 1976. Long-term effects of lead exposure on three generations of brook trout (*Salvelinus fontinalis*). J. Fish. Res. Board Can. 33:1731-1741.

Hydrometrics, Inc. 2006. Water resources technical report for the Idaho Cobalt Project. Prepared for Salmon National Forest. March 2006, Revised December 2006.

IDEQ. (Idaho Department of Environmental Quality). 2001. Middle Salmon River-Panther Creek Subbasin Assessment and TMDL. Boise, Idaho. http://www.deq.state.id.us/water/data_reports/surface_water/tmdls/salmon_river_middle_panther/salmon_panther_pref.pdf

IDEQ. 2003. 2002 Integrated Water Quality Monitoring and Assessment Report. Submitted to the U.S. Environmental Protection Agency. Approved by EPA December 2005.

IDFG. (Idaho Department of Fish and Game). 1965. Inventory of Idaho streams containing anadromous fish, including recommendations for improving production of salmon and steelhead. Prepared for U.S.Fish and Wildlife Service, Boise, ID.

IDFG. 2001. Fisheries Management Plan 2001-2006. Boise, Idaho.

IDL. 1992. (Idaho Department of Lands). Best Management Practices for Mining in Idaho. Boise, Idaho. http://www.idl.idaho.gov/Bureau/Minerals/bmp_manual1992/bmp_index.htm

Illiopoulou-Georgudaki, J. and N. Kotsanis. 2001. Toxic effects of cadmium and mercury in rainbow trout (*Oncorhynchus mykiss*): A short-term bioassay. Bull. Environ. Contam. Toxicol. 66: 77-85.

Interior Columbia Basin Technical Recovery Team (ICBTRT). 2003. Independent Populations of Chinook, Steelhead, and Sockeye for Listed Evolutionarily Significant Units within the Interior Columbia River Domain. Working Draft. July 2003. http://www.nwfsc.noaa.gov/trt/trt_columbia.htm

ICBTRT. 2005. Interior Columbia Basin TRT: Viability Criteria for Application to Interior Columbia Basin Salmonid ESUs.

Karr, M.H., J.K. Fryer, and P.R. Mundy. 1998. Snake River water temperature control project. Phase II. Methods for managing and monitoring water temperatures in relation to salmon in the lower Snake River. Columbia River Inter-Tribal Fish Commission, Prepared for U.S. Environmental Protection Agency, Region 10, Contract No. X-990375-01-1. 209 p.

Keithly, J., J.A. Brooker, D.K. DeForest, B.K. Wu, and K.V. Brix. 2004. Acute and chronic toxicity of nickel to a cladoceran (*Ceriodaphnia dubia*) and an amphipod (*Hyalella azteca*). Environmental Toxicology and Chemistry. 23(3): 691-696.

Kennedy, C.J., L.E. McDonald, R. Loveridge, and M.M. Strosher. 2000. The effect of bioaccumulated selenium on mortalities and deformities in the eggs, larvae, and fry of a wild population of cutthroat trout (*Oncorhynchus clarki lewisi*). Arch. Environ. Contam. Toxicol. 39: 46-52.

Kenter, M.L., C.A. Peery, and M.J. Heinrich. 2007. Temperature-mediated *en route* migration mortality and travel times of endangered Snake River sockeye salmon. *Ecology of Freshwater Fish 2007*:1-10. (OnlineEarly Articles, http://www.Blackwell-synergy.com/doi/abs/10.1111/j.1600-0633.2007.00267)

Kincheloe, J.W., G.A. Wedemeyer, D. L. Koch. 1979. Tolerance of developing salmonid eggs and fry to nitrate exposure. Bull. Environ. Contam. Toxicol. 23: 575-578.

Köck G., and F. Bucher. 1997. Accumulation of zinc in rainbow trout (*Oncorhynchus mykiss*) after waterborne and dietary exposure. Bull. Environ. Contam. Toxicol. 58: 305-310.

Kroupova, H., J. Machova, and Z. Svobodova. 2005. Nitrite influence on fish: a review. Veterinary Medecine – Czech, 50(11): 461-471. As cited in USDA Forest Service 2007.

Kuzis, K. 2004. Proposed Idaho Cobalt Mine, Aquatic Baseline Condition Draft Technical Report. Prepared for Formation Capital, Salmon, ID. October 2004.

LeBlanc, G.A. and J.W. Dean. 1984. Antimony and thallium toxicity to embryos and larvae of fathead minnows (*Pimephales promelas*). Bulletin on Environmental Contamination and Toxicology *32*(5):565-569.

Leland, H. V., and J. L. Carter. 1985. Effects of copper on production of periphyton, nitrogen-fixation and processing of leaf litter in a Sierra-Nevada, California, stream. Freshw. Biol. 15(2):155–173.

Leland, H. V., S. V. Fend, T. L. Dudley, and J. L. Carter. 1989. Effects of copper on species composition of benthic insects in a Sierra-Nevada, California stream. Freshw. Biol. 21(2):163–179.

Lewis, M. 1978. Acute toxicity of copper, zinc, and manganese in single and mixed salt solutions to juvenile longfin dace, *Agosia chryogaster*. Journal Fish Biology 13: 695-700.

Lewis Jr., W.M., and D.P. Morris. 1986. Toxicity of Nitrate to Fish: A Review. Transactions of the American Fisheries Society 115(2): 183-195.

Linbo, A.O., C.M. Stehr, J.P. Incardona, and N.L. Scholz. 2006. Dissolved copper triggers cell death in the peripheral mechanosensory system of larval fish. Environ. Toxicol. Chem. 25: 597-603. As cited in Hecht et al. 2007.

Logerwell, E.A., N. Mantua, P.W. Larson, R.C. Francis, and V.N. Agostini. 2003. Tracking environmental processes in the coastal zone for understanding and predicting Oregon coho (*Oncorhynchus kisutch*) marine survival. *Fisheries Oceanography* 12:554-568.

Lorz, H. W., and B. P. McPherson. 1976. Effects of copper or zinc in freshwater on the adaptation to sea water and ATPase activity, and the effects of copper on migratory disposition of the coho salmon (*Oncorhynchus kisutch*). J. Fish. Res. Board Can. 33:2023–2030.

Lorz, H. W., and B. P. McPherson. 1977. Effects of copper and zinc on smoltification of coho salmon. EPA 600/3-77-032. Oregon Department of Fish and Wildlife and U.S. EPA Environmental Research Laboratory, Corvallis, Oregon.

Lorz, H.W., R.H. Williams, and C.A. Fustish. 1978. Effects of several metals on smolting of coho salmon. Corvallis Environmental Research Laboratory, U.S. EPA Office of Research and Development. Corvallis, Oregon. EPA-600/3-78-090. 84pp.

Luce, C.H., B.E. Rieman, J.B. Dunham, J.L. Clayton, J.G. King, and T.A. Black. 2001. Incorporating Aquatic Ecology into Decisions on Prioritization of Road Decommissioning. Water Resources IMPACT Vol.3, Number 3, pp 8-14.

MacDonald, J.S. 2000. Mortality during the migration of Fraser River sockeye salmon (*Oncorhynchus nerka*): a study of the effect of ocean and river environmental conditions in 1997. Can. Tech. Rep. Fish. Aquat. Sci. No. 2315.

MacDonald, L.H., A.W. Smart, and R.C. Wissmar. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. U.S. Environmental Protection Agency, EPA/910/9-91-001.

Marr, J. C. A., J. Lipton, D. Cacela, J. A. Hansen, H. L. Bergman, J. S. Meyer, and C. Hogstrand. 1996. Relationship between copper exposure duration, tissue copper concentration, and rainbow trout growth. Aquatic Toxicology 36:17–30.

Marr, J.C.A., J. A. Hansen, J.S. Meyer, D. Cacela, T. Podrabsky, J. Lipton, and H.L. Bergman. 1998. Toxicity of cobalt and copper to rainbow trout: application of a mechanistic model for predicting survival. Aquatic Toxicology 43: 225-238.

Marr, J. C. A., J. Lipton, D. Cacela, J. A. Hansen, J. S. Meyer, and H. L. Bergman. 1999. Bioavailability and acute toxicity of copper to rainbow trout (*Oncorhynchus mykiss*) in the presence of organic acids simulating natural dissolved organic carbon. Can. J. Fish. Aquat. Sci. 56(8):1471–1483.

Matthews, G. M. and R. S. Waples. 1991. Status Review for Snake River Spring and Summer Chinook Salmon. NMFS F/NWC-200. Available from NMFS, Northwest Fisheries Science Center, Coastal Zone and Estuaries Studies Division, 2725 Montlake Blvd. E., Seattle, Washington 98112-2097. 75 pages. http://www.nwfsc.noaa.gov/publications/techmemos/tm200/tm200.htm

McClure, M.B., E.E. Holmes, B.L. Sanderson, and C.E. Jordan. 2003. A large-scale multispecies status assessment: anadromous salmonids in the Columbia River Basin. *Ecological Applications* 13(4):964-989.

McCormick, S.D., Hansen, L.P., Quinn, T.P., and R.L. Saunders. 1998. Movement, migration, and smolting of Atlantic salmon (*Salmo salar*). Can. J. Fish. Aquat. Sci. 55 (Suppl. 1): 77-92.

McCullough, D.A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to chinook salmon. Water Resource Assessment, Columbia River Inter-Tribal Fish Commmission, Portland, OR. EPA 910-R-99-010. 291 pp. http://www.krisweb.com/biblio/gen_usepa_mccullough_1999.pdf

McCullough, D.A., Spalding, S., Sturdevant, D., and Hicks, M. 2001. Issue paper 5: summary of technical literature examining physiological effects of temperature on salmonids. Region 10 Temperature Water Quality Criteria Guidance Development Project. EPA-910-D-01-005. May 2001. 114 pp.

McDonald, D.G., C.M. Wood, R.G. Rhem, M.E. Mueller, D.R. Mount, and H.L. Bergman. 1991. Nature and time course of acclimation to aluminum in juvenile brook trout (*Salvelinus fontinalis*). I. Physiology. Can. J. Fish. Aquat. Sci. 48: 2006-2015.

McElhany, P., M. Ruckleshaus, M. J. Ford, T. Wainwright, and E. Bjorkstedt. 2000. Viable Salmon Populations and the Recovery of Evolutionarily Significant Units. U. S. Department of Commerce, National Marine Fisheries Service, Northwest Fisheries Science Center, NOAA Technical Memorandum NMFS-NWFSC-42. 156 p. http://www.nwfsc.noaa.gov/publications/techmemos/tm42/tm42.pdf

McGeachy, S.M., and D.G. Dixon. 1992. Whole-body arsenic concentrations in rainbow trout during acute exposure to arsenate. Ecotoxicology and Environmental Safety. Vol. 24 (3): 301-308.

McGeer, J.C., C. Szebedinszky, D. Gordon McDonald, and C.M. Wood. 2000. Effects of chronic sublethal exposure to waterborne Cu, Cd, or Zn in rainbow trout: tissue specific metal accumulation. Aquatic Toxicology 50: 245-256.

McGurk, M.D., F. Landry, A. Tang, and C.C. Hanks. 2006. Acute and chronic toxicity of nitrate to early life stages of lake trout (*Salvelinus namaycush*) and lake whitefish (*Coregonus clupeaformis*). Environmental Toxicology and Chemistry 25(8): 2187-2196.

Mebane, C. A. 2000. Evaluation of proposed new point source discharges to a special resource water and mixing zone determinations: Thompson Creek Mine, upper Salmon River subbasin, Idaho. Idaho Department of Environmental Quality, Boise, ID.

Mebane, C.A. 2006. Cadmium risks to freshwater life: Derivation and validation of low-effect criteria values using laboratory and field studies. U.S. Geological Survey, Scientific Investigations Report 2006-52345, Version 1.1, Decemeber 2006, Boise, Idaho. 130 pages. http://pubs.usgs.gov/sir/2006/5245/pdf/sir20065245.pdf

Mebane, C.A. 2007. An evaluation of the protectiveness of a site-specific criteria for cobalt in the Panther Creek watershed, Idaho for Chinook salmon and steelhead. Prepared for National Marine Fisheries Service, Idaho State Habitat Office. USGS, Boise, Idaho. 22 pp., + Appendix.

Meehan, W.R. 1991. Introduction and Overview. American Fisheries Society Special Publication 19: 1-15.

Meeings, C.C. and R.T. Lackey. 2005. Estimating the size of historic Oregon salmon runs. Reviews in Fisheries Science 13:51-66.

Megahan, W.F., and W.J. Kidd. 1972. Effect of logging roads on sediment production rates in the Idaho Batholith. Unpublished Report, USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah. 16 p.

Miller, L.L., F. Wang, V.P. Palace, and A. Hontela. 2007. Effects of acute and subchronic exposures to waterborne selenite on the physiological stress response and oxidative stress indicators in juvenile rainbow trout. Aquatic Toxicology. 83: 263-271.

Mitchell, M.K. and W.B. Stapp. 1992. Field Manual for Water Quality Monitoring. Thomason-Shore, Inc., Dexter, MI.

Mount, D.R., D.D. Gulley, J. Russel Hockett, T.D. Garrison, and J.M. Evans. 1997. Statistical models to predict the toxicity of major ions to *Ceriodaphnia dubia*, *Daphnia magna*, and *Pimephales promelas* (Fathead minnows). Environmental Toxicology and Chemistry 16(10): 2009-2019.

Mudge, J. E., T.E. Northstrom, G.S. Jeane, W. Davis, and J. L. Hickam. 1993. Effect of varying environmental conditions on the toxicity of copper to salmon. *In* J. W. Gorsuch, F. J. Dwyer, C. G. Ingersoll, and T. W. La Point, (eds.), Environmental toxicology and risk assessment, ASTM STP 1216, p. 19–33. American Society for Testing and Materials, Philadelphia, PA.

Mueller, M.E., D.A. Sanchez, H.L. Bergman, D.G. McDonald, R.G. Rhem, and C.M. Wood. 1991. Nature and time course of acclimation to aluminum in juvenile brook trout (*Salvelinus fontinalis*). II. Gill Histology. Can. J. Fish. Aquat. Sci. 48: 2016-2027.

Murphy, M.L. 1995. Forestry impacts on freshwater habitat of anadromous salmonids in the Pacific Northwest and Alaska -- requirements for protection and restoration. NOAA Coastal Ocean Program Decision Analysis Series No. 7. NOAA Coastal Ocean Office, Silver Spring, MD. 156 pp.

Myers, J. M., R. G. Kope, G. J. Bryant, D. Teel, L. J. Lierheimer, T. C. Wainwright, W. S. Grant, F. W. Waknitz, K. Neely, S. T. Lindley, and R. S. Waples. 1998. Status Review of Chinook Salmon From Washington, Idaho, Oregon, and California. NMFS-NWFSC-35. Available from National Marine Fisheries Service, Northwest Fisheries Science Center, Coastal Zone and Estuaries Studies Division, Seattle, Washington. 443 p. http://www.nwfsc.noaa.gov/publications/techmemos/tm35/index.htm

Nagpal, N.K. 1987. Water quality guidelines for lead. Ministry of Environment and Parks, Province of British Columbia, Canada. Vol. 2, Technical Appendix, 103 pages. http://www.env.gov.bc.ca/wat/wq/BCguidelines/lead/leadtech.pdf

Nagpal, N.K. 1989. Ambient water quality criteria for mercury. Ministry of Environment, Province of British Columbia, Canada. Vol. 2, Technical Appendix, 105 pages. http://www.env.gov.bc.ca/wat/wq/BCguidelines/mercury/mercurytech.pdf

Nagpal, N.K. 2004. Water quality guidelines for cobalt. Ministry of Water, Land, and Air Protection, Victoria, British Columbia, Canada. Technical Report, 59 pages. http://www.env.gov.bc.ca/wat/wq/Bcguidelines/cobalt/cobalt_tech.pdf

NAS. 1972. Water quality criteria. Freshwater aquatic life and wildlife. Appendix II. EPA Ecol. Res. Series EPA-R3-73-033, U.S. Environmental Protection Agency, Washingtion, D.C. 594 p.

Nebeker, A.V., C. Savonen, D.G. Stevens. 1985. Sensitivity of rainbow trout to early life stages of nickel chloride. Environ. Toxicol. Chem. 4: 233-239. As cited in Brix et al. 2004.

Neff, J.M. 1985. Polycyclic aromatic hydrocarbons. *In*: Fundamentals of aquatic toxicology, G.M. Rand, and S.R. Petrocelli (eds.), pp. 416-454. Hemisphere Publishing, Washington, D.C.

Nelson, R.L., M.L. McHenry, and W.S. Platts. 1991. Mining. American Fisheries Society Special Publication 19:425-457.

Newcombe, C. P., and J. O. Jensen. 1996. Channel Suspended Sediment and Fisheries: A Synthesis for Quantitative Assessment of Risk and Impact. North American Journal of Fisheries Management 16(4): 693-727.

NMFS (National Marine Fisheries Service). 1995. Endangered Species Act Section 7 Consultation Biological Opinion on the Effects of Land and Resource Management Plans for the Boise, Challis, Nez Perce, Payette, Salmon, Sawtooth, Umatilla, and Wallowa-Whitman National Forests. Northwest Region, Habitat Conservation Division, Portland, Oregon (March 1, 1995).

NMFS. 1996. Making Endangered Species Act Determinations of Effect for Individual and Grouped Actions at the Watershed Scale. Habitat Conservation Program, Portland, OR.

NMFS. 1999. Approval of Oregon water quality standards for dissolved oxygen, temperature, and pH. Northwest Region, Habitat Conservation Division, Oregon State Branch: OSB1999-0146. 7 July.

NMFS. 2000. Endangered Species Act – Section 7 Consultation, Biological Opinion; Reinitiation of consultation on operation of the Federal Columbia River Power System, including the juvenile fish transportation program, and 19 Bureau of Reclamation projects in the Columbia basin. National Marine Fisheries Service, Northwest Region, Portland, Oregon. December 21. http://www.nwr.noaa.gov/1hydrop/hydroweb/docs/Final/2000Biop.html

NMFS. 2004a. Endangered and Threatened Species; Designation of Critical Habitat for 13 Evolutionarily Significant Units of Pacific Salmon (*Oncorhynchus* spp.) and Steelhead (*O. mykiss*) in Washington, Oregon, and Idaho. Federal Register, 69 (74572). (December 14, 2004).

NMFS. 2004b. Consultation on Remand for Operation of the Columbia River Power System and 19 Bureau of Reclamation Projects in the Columbia Basin (Revised and reissued pursuant to court order, NWF v. NMFS, Civ. No. CV 01-640-RE (D. Oregon). NMFS, Northwest Region, Seattle, Washington.

NMFS. 2005. Appendix I: CHART assessment for the Snake River Basin steelhead. August. http://www.nwr.noaa.gov/Salmon-Habitat/Critical-Habitat/2005-Biological-Teams-Report.cfm.

NMFS. 2006a. Snake River Salmon and Steelhead Recovery Plan (Draft): Snake River sockeye salmon (Chapter 9). National Marine Fisheries Service, Idaho State Habitat Office, Boise, Idaho (May 2006). http://www.idahosalmonrecovery.net/recoveryplans/srsockeye.html

NMFS. 2006b. Snake River Salmon and Steelhead Recovery Plan (Draft): Snake River spring/summer Chinook salmon (Chapters 4 and 5). National Marine Fisheries Service, Idaho State Habitat Office, Boise, Idaho (May 2006). http://www.idahosalmonrecovery.net/recoverplans/spsumchinook.html

NMFS. 2006c. Snake River Salmon and Steelhead Recovery Plan (Draft): Snake River Basin steelhead (Chapters 6 and 7). National Marine Fisheries Service, Idaho State Habitat Office, Boise, Idaho (May 2006). http://www.idahosalmonrecovery.net/recoverplans/srsteelhead.html

NRC (National Research Council). 1996. Upstream: salmon and society in the Pacific Northwest. National Academy Press, Washington, D.C. 452 pages.

NPPC (Northwest Power Planning Council). 1986. Compilation of information on salmon and steelhead losses in the Columbia River basin. Appendix D of the 1987 Columbia River Basin Fish and Wildlife Program. Portland, OR. 252 pp.

NPPC. 1991. Northwest Power Planning Council 1991/Chinook Salmon Presence/Absence files for the Upper Salmon.

ODEQ. (Oregon Department of Environmental Quality). 1995. Final Issue Papers 1992 - 1994 Water Quality Standards Review. http://www.deq.state.or.us/pubs/reports.htm#WQ

Ott, D. S. and T. R. Maret. 2003. Aquatic assemblages and their relation to temperature variables of least-disturbed streams in the Salmon River basin, Idaho, 2001. U.S. Geological Survey, Water-Resources Investigative Report 03-4076, Boise, Idaho. 45 pp. http://id.water.usgs.gov/public/reports.html

Pacific EcoRisk. 2005. An evaluation of the acute toxicity of cobalt in Panther Creek Water to three resident invertebrate species (*Brachycentrus americanus*, *Centroptilum conturbatum*, and Serratella tibialis) and the acute and chronic toxicity of cobalt in Panther Creek water to *Chironomus tentans* and *Oncorhynchus mykiss*. Martinez, California. 83 pp. + Appendix.

Palace, V.P., C. Baron, R.E. Evans, J. Holm, S. Kollar, K. Wautier, J. Werner, P. Siwik, G. Sterling, and C.F. Johnson. 2004. An assessment of the potential for selenium to impair reproduction in bull trout, *Salvelinus confluentus*, from an area of active coal mining. Environmental Biology of Fishes 70: 169-174.

Pane, E.F., J.G. Richards, and C.M. Wood. 2003. Acute waterborne nickel toxicity in the rainbow trout (*Oncorhynchus mykiss*) occurs by a respiratory rather than ionoregulatory mechanism. Aquatic Toxicology 63: 65-82.

Pane, E.F., A. Haque, G.G. Goss, and C.M. Wood. 2004. The physiological consequences of exposure to chronic, sublethal waterborne nickel in rainbow trout (*Oncorhynchus mykiss*): exercise *vs* resting physiology. The Journal of Experimental Biology 207: 1249-1261.

Pedlar, R. M. and J. F. Klaverkamp. 2002. Accumulation and distribution of dietary arsenic in lake whitefish (Coregonus clupeaformis). Aquatic Toxicology 57(3): 153-166.

Pesticide Action Network (PAN) Pesticides Database. 2008. Toxicity Studies for Magnesium chloride on All Organism Groups - Toxicology studies from the primary scientific literature on aquatic organisms. http://www.pesticideinfo.org/List_AquireAll.jsp?Rec_Id=PC32916

Pickard, J., R. Yang, B. Duncan, C.A. McDevitt, and C. Eickhoff. 2001. Acute and sublethal toxicity of thallium to aquatic organisms. Bull. Environ. Contam. Toxicol. 66: 94-101.

Platts, W.S., S.B. Martin, and E.R. Primbs. 1979. Water Quality in Idaho Streams Degraded by Acid Mine Waters. General Technical Report INT-67. USDA Forest Service.

PFMC (Pacific Fishery Management Council). 1998a. Final Environmental Assessment/ Regulatory Review for Amendment 11 to the Pacific Coast Groundfish Fishery Management Plan. Pacific Fishery Management Council, Portland, Oregon (October 1998). http://www.pcouncil.org/groundfish/gffmp/gfa11.html

PFMC (Pacific Fishery Management Council). 1998b. The Coastal Pelagic Species Fishery Management Plan: Amendment 8. Pacific Fishery Management Council, Portland, Oregon (December 1998). http://www.pcouncil.org/cps/cpsfmp.html

PFMC (Pacific Fishery Management Council). 1999. Amendment 14 to the Pacific Coast Salmon Plan. Appendix A: Description and Identification of Essential Fish Habitat, Adverse Impacts and Recommended Conservation Measures for Salmon. Pacific Fishery Management Council, Portland, Oregon (March 1999). http://www.pcouncil.org/salmon/salfmp/a14.html

Ploskey, G.R. 1983. Review of the effects of water-level changes on reservoir fisheries and recommendations for improved management. Technical Report VES/TR/E-83-3. U.S. Army, Engineering and Water ways Experiment Station, Vicksburg, Mississippi.

Quinn, T.P., S. Hodgson, and C. Peven. 1997. Temperature, flow, and the migration of adult sockeye salmon (*Oncorhynchus nerka*) in the Columbia River. Can. J. Fish. Aquat. Sci. 54(6): 1349-1360.

Rand, G.M., and S.R. Petrocelli. 1985. Fundamentals of aquatic toxicology: methods and applications. Hemisphere Publishing Corp.

Rankin, M.G. and D.G. Dixon. 1994. Acute and chronic toxicity of waterborne arsenite to rainbow trout (*Oncorhynchus mykiss*). Can. J. Fish. Aquati. Sci. 51: 372-380.

Raymond, H.L. 1979. Effects of dams and impoundments on migrations of juvenile Chinook salmon and steelhead from the Snake River, 1966 to 1975. Transactions of the American Fisheries Society 108:505-529.

Regetz, J. 2003. Landscape-level restraints on recruitment of Chinook salmon (*Oncorhynchus tshawytscha*) in the Columbia River Basin, USA. Aquatic Conservation: Marine and Freshwater Ecosystems 13:35-49.

Reid, L.M., and T. Dunne. 1984. Sediment production from forest road surfaces. Water Resources Research 20:1753-1761.

Rieman, B. E., D.L. Myers, and R.L. Nielsen. 1994. Use of otolith microchemistry to discriminate Oncorhynchus nerka of resident and anadromous origin. Can. J. Fish. Aquat. Sci. 51:68-77.

Reimer, P.S. 1999. Environmental effects of manganese and proposed freshwater guidelines to protect aquatic life in British Columbia. Masters Thesis, University of British Columbia. 1988. http://www.env.gov.bc.ca/wat/wq/BCguidelines/manganese/index.html

Ribeyre, F. and A. Boudou. 1984. Bioaccumulation et repartition tissulaire du mercure-$HgCl_2$ et $Ch_3HgCl_2^-$ chez *Salmo Gairdneri* après contamination par voie directed. Waer Air Sol Pollut. 23: 169-186. As cited in Eisler 1987.

Riddell, D.J., J. M. Culp, and D.J. Baird. 2005. Sublethal effects of cadmium on prey choice and capture efficiency in juvenile brook trout (*Salvelinus fontinalis*). Environmental Toxicology and Chemistry 24(7): 1751-1758.

Ruggerone, G.T. 2000. Differential survival of juvenile sockeye and coho salmon exposed to low dissolved oxygen during winter. J. Fish Biology 56(4):1013-1016.

Sandahl, J. F., D. H. Baldwin, J. J. Jenkins, and N. L. Scholz. 2004. Odor-evoked field potentials as indicators of sublethal neurotoxicity in juvenile coho salmon (*Oncorhynchus kisutch*) exposed to copper, chlorpyrifos, or esfenvalerate. Can. J. Fish. Aquat. Sci. 61(3):404–413.

Sandahl, J. F., D. H. Baldwin, J. J. Jenkins, and N. L. Scholz. 2007. A sensory system at the interface between urban storm water runoff and salmon survival. Environ. Sci. Technol. 41:2998–3004.

Sauter, S., K.S. Buxton, K.J. Macek, and S.R. Petrocelli. 1976. Effects of exposure to heavy metals on selected freshwater fish. Toxicity of copper, cadmium, and lead to eggs and fry of seven fish species. U.S. Environmental Protection Agency, EPA-600/3-76-195. 74pp. As cited in Nagpal 1987

Saucier, D., L. Astic, and P. Rioux. 1991. The effects of early chronic exposure to sublethal copper on the olfactory discrimination of rainbow trout, *Oncorhynchus mykiss*. Environ. Biol. Fishes 30:345–351.

Scott, G. R., K. A. Sloman, C. Rouleau, and C.M. Wood. 2003. Cadmium disrupts behavioural and physiological responses to alarm substance in juvenile rainbow trout (*Oncorhynchus mykiss*). The Journal of Experimental Biology 206: 1779-1790.

Servizi, J. A., and D. W. Martens. 1987. Some effects of suspended Fraser River sediments on sockeye salmon (*Oncorhynchus nerka*), pp. 254-264. *In* H. D. Smith, L. Margolis, and C. C. Wood *eds.* Sockeye salmon (*Oncorhynchus nerka*) population biology and future management. Canadian Special Publications of Fisheries and Aquatic Sciences 96.

Servizi, J. A. and D. W. Martens. 1991. Effect of temperature, season, and fish size on acute lethality of suspended sediments to coho salmon (Oncorhynchus kisutch). Canadian Journal of Aquatic Sciences 48:493-497.

Servizi, J.A. and D.W. Martens. 1992. Sublethal responses of coho salmon (Oncorhynchus kisutch) to suspended sediments. Can. J. Fish. Aquatic. Sci. 49: 1389-1395.

Shearer, K.D., and T. Åsgård. 1992. The effect of water-borne magnesium on the dietary magnesium requirement of the rainbow trout (Oncorhynchus mykiss). Fish Physiology and Biochemistry 9(5/6): 387-392.

Shingles, A., D.J. McKenzie, E.W. Taylor, A. Moretti, P.J. Butler, and S. Ceradini. 2001. Effects of sublethal ammonia exposure on swimming performance in rainbow trout (*Oncorhynchus mykiss*). The Journal of Experimental Biology 204: 2691-2698.

Singleton, H. 2000. Ambient Water Quality Guidelines for Sulphate. Ministry of Environment, Lands, and Parks, Province of British Columbia. Technical Appendix. Vitoria, British Columbia. http://www.env.gov.bc.ca/wat/wq/BCguidelines/sulphate/index.html

Sinley, J.R., J.P. Goetti Jr., and P.H. Davies. 1974. The effects of zinc on rainbow trout (*Salmo gairdneri*) in hard and soft water. Bulletin of Environmental Contamination and Toxicology 12: 193-201.

Skidmore, J.F. 1970. Respiration and osmoregulation in rainbow trout with gills damaged by zinc sulphate. J. Exp. Biol. 52: 481-494.

Sloman, K.A., D.W. Baker, C.G. Ho, D.G. McDonald, and C.M. Wood. 2003. The effects of trace metal exposure on agonistic encounters in juvenile rainbow trout, *Oncorhynchus mykiss*. Aquatic Toxicology 63: 187-196.

Smith, E.J., and J.L. Sykora. 1976. Early developmental effects of lime-neutralized iron hydroxide suspensions on brook trout and coho salmon. Transactions of the American Fisheries Society 105(2):308-312.

Smith, M.A. and Hubert, W.A. 2003. Simulated thermal tempering versus sudden temperature change and short-term survival of fingerling rainbow trout. N. Am. J. Aquacult. Vol. 65(1): 67-69.

Soucek, D.J. 2007. Comparison of hardness- and chloride-regulated acute effects of sodium sulfate on two freshwater crustaceans. Environmental Toxicology and Chemistry 26(4): 773-779.

Soucek, D.J., and A.J. Kennedy. 2005. Effects of hardness, chloride, and acclimation on the acute toxicity of sulfate to freshwater invertebrates. Environmental Toxicology and Chemistry 24(5): 1204-1210.

Spence, B.C. 1995. Geographic variation in timing of fry emergence and smolt migration of coho salmon (*Oncorhynchus kisutch*). Doctoral dissertation. Oregon State University, Corvallis.

Spence, B.C, G.A. Lomnicky, R.M. Hughes, R.P. Novitzki. 1996. An ecosystem approach to salmonid conservation. TR-4501-96-6057. ManTech Environmental Research Services Corp., Corvallis, Oregon. (December 1996). http://www.nwr.noaa.gov/1habcon/habweb/habguide/ManTech/front.htm

Sprague, J., P. Elson, and R. Saunders. 1965. Sublethal copper-zinc pollution in a salmon river—A field and laboratory study. Int. J. Air Water Pollut. 9:531–543.

Stantec. 2004. Biomonitoring study, Panther Creek Watershed. Prepared for Blackbird Mine Site Group. Salmon, Idaho. September 2003.

Staples, C.A, Williams J.B., Craig G.R., Roberts K.M. 2001. Fate, effects and potential environmental risks of ethylene glycol: a review. Chemosphere. Volume 43, Number 3, April 2001, pp. 377-383(7).

Stockner, J.G. and E.A. LMacisaac. 1996. British Columbia lake enrichment programme: two decades of habitat enhancement for sockeye salmon. Regulated Rivers: Research and Management 12(4-5): 547-561. As cited in USDA Forest Service 2007.

StreamNet. 2007. Data Query. StreamNet, Portland, Oregon. http://www.streamnet.org/online_data.html

Stubblefield, W.A., S.F. Brinkman, P.H. Davies, T.D. Garrison, J.R. Hockett and M.W. McIntrye. 1997. Effects Of Water Hardness On The Toxicity Of Manganese To Developing Brown Trout (*Salmo Trutta*). Environmental Toxicology and Chemistry. 16(10): 2082-2089.

Stubblefield, W.A., B.L. Steadman, T. W. La Point, and H. L. Bergman. 1999. Acclimation-induced changes in the toxicity of zinc and cadmium to rainbow trout. Environmental Toxicology and Chemistry 18(12): 2875-2881.

Sullivan, K., T.E. Lisle, C.A. Dolloff, G.E. Grant, and L.M. Reid. 1987. Stream channels: the link between forests and fishes. In *Streamside Management: Forestry and Fishery Interactions*; E.O. Salo and T.W. Cundy, eds. Pgs. 191-232. Contribution 57, University of Washington, Institute of Forest Resources. Seattle, WA.

Svecevičius, G. 1999. Fish avoidance response to heavy metals and their mixtures. Acta Zoologica Lituanica Hydrobiologia 9(2): 103-113.

Swanson, F.J., and C.T. Dyrness. 1975. Impact of clear-cutting and road construction on soil erosion by landslides in the western Cascade Range, Oregon. Geology 3:393-396.

Swanston, D.N. and F.J. Swanson. 1976. Timber harvesting, mass erosion, and steepland forest geomorphology in the Pacific Northwest. In *Geomorphology and Engineering*; D.R. Coates, ed. Pgs. 199-221. Dowden, Hutchinson, and Ross. Stroudsburg, PA.

Swanston, D.N. 1991. Natural processes. In *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*; W.R. Meehan, ed. Pgs. 139-179. American Fisheries Society Special Pub. 19. Bethesda, MD.

Sylvester, J.R. 1972. Effect of thermal stress on predator avoidance in sockeye salmon. J. Fish. Res. Bd. Canada 29:601-603.

Szebedinszky, C., J.C. McGeer, D.G. McDonald, and C.M. Wood. 2001 Effects of chronic Cd exposure via the diet or water on internal organ-specific distribution and subsequent gill Cd uptake kinetics in juvenile rainbow trout (*Oncorhynchus mykiss*). Environmental Toxicology and Chemistry 20(3): 597-607.

Tang, J., M.D. Bryant, and E.L. Brannon. 1987. Effect of temperature extremes on the mortality and development rates of coho salmon embryos and alevins. Prog. Fish-Cult. 49(3):167-174.

Telesto Solutions, Inc. (Telesto). 2006. 2006 Water Monitoring Plan for the Idaho Cobalt Project. Prepared for Formation Capital Corporation.

Telesto. 2007. Operational water monitoring plan for the Idaho Cobalt Project. Telesto Solutions Incorporated, Fort Collins, Colorado. Prepared for Formation Capital Corporation, Salmon, Idaho. 27 pages + 11 tables + 5 figures + 3 appendices (September 2007).

Thurston, R.V., R.C. Russo, C.M. Fetterolf, Jr., T. A. Edsalol, and Y.M. Barber, Jr., (Eds). 1979. A review of the EPA Red Book: Quality criteria for water. Water Quality Section, American Fisheries Society, Bethesda, MD.

TTE. (Turkey Tracks Enterprises, Inc.). 2006. Idaho Cobalt Project, Transportation Baseline Report and Transportation Plan. Revision 4. Prepared for Formation Capital Corporation. August, 2006.

Twiss, M.R., B.S. Twining, and N.S. Fisher. 2004. Bioconcentration of inorganic and organic thallium by freshwater phytoplankton. Environmental Toxicology and Chemistry 23(4): 968-973.

U.S. Department of Transportation, Federal Highway Administration. 2003. Standard specifications for construction of roads and bridges on Federal highway projects. FP-03.

USDA Forest Service. 1993. Biological Assessment for Proposed and Ongoing Actions on Snake River Spring/Summer Chinook Salmon and Snake River Sockeye Salmon. Panther Creek Watershed. Salmon National Forest.

USDA Forest Service. 1997. Amendment to the Biological Assessment for the Panther Creek Watershed (In response to litigation regarding the Beartrack Mine Project, Unites States Court Order C94-1576R).

USDA Forest Service. 2002. Biological Assessment for the Gabe Creek and Pete Creek Culvert Replacements. West Fork Ranger District, Bitterroot National Forest, Hamilton, Montana.

USDA Forest Service. 2005. Biological Assessment for Proposed and Ongoing Actions within the Panther Creek Drainage on Snake River Basin Spring/Summer Chinook Salmon, Snake River Basin Steelhead, Columbia River Bull Trout and Snake River Sockeye Salmon. Salmon Challis National Forest.

USDA Forest Service. 2007. Biological Assessment and Biological Evaluation of the Effects of the Proposed Idaho Cobalt Project on Federally Listed and Regionally Sensitive Aquatic Species. Salmon-Challis National Forest, Salmon, Idaho. September 2007.

Van Kirk, R.W., and S.L. Hill. 2007. Demographic model predicts trout population response to selenium based on individual-level toxicity. Ecological Modelling. 206: 407-420.

Vincoli, J.W. 1997. Risk Management for Hazardous Chemicals, vols. 1 & II. CRC Lewis Publishers. Boca Raton, Florida.

Wallwork-Barber, M.K., K. Lyall, and R.W. Ferenbaugh. 1985. Thallium movement in a simple aquatic environment. Journal of Environmental Science and Health 20:689-700.

Waples, R.S. 1991. Definition of 'Species' Under the Endangered Species Act: Application to Pacific Salmon. U.S. Department of Commerce, National Marine Fisheries Service, Northwest Fisheries Science Center, NOAA Technical Memorandum NMFS- F/NWC-194. http://www.nwfsc.noaa.gov/publications/techmemos/tm194/waples.htm

Warren, C.E., Doudoroff, P., Shumway, D.L. 1973. Development of dissolved oxygen criteria for freshwater fish. U.S. Environmental Protection Agency, Ecological Research Series Report. EPA-R3-73-019. Washington, D.C. 121 p. As cited in ODEQ 1995.

Wedemeyer, G. 1973. Some physiological aspects of sublethal heat stress in the juvenile steelhead trout (*Salmo gairdneri*) and coho salmon (*Oncorhynchus kisutch*). J. Fish. Res. Board Can. 30:831-834.

Wedemeyer, G.A., Barton, B.A., and McLeay, D.J. 1990. Stress and acclimation. Pages 451-489 in C. B. Schreck and P. B. Moyle, editors. Methods for fish biology. Amer. Fish. Soc., Bethesda, Maryland.

Welsh, P.G., J. Lipton, C.A. Mebane, and J.C.A. Marr. 2008. Influence of flow-through and renewal exposures on the toxicity of copper to rainbow trout. Ecotoxicology and Environmental Safety 69: 199-208.

Wemple, B.C., J. A. Jones, and G.E. Grant. 1996. Channel network extension by logging roads in two basins, western Cascades, Oregon. Water Res. Bull. 32(6):1-13.

Westin, D.T. 1974. Nitrate and nitrite toxicity to salmonid fishes. The Progressive Fish Culturist 36(2): 86-89.

Wicks, B.J., R. Joensen, Q. Tang, and D.J. Randall. 2002. Swimming and ammonia toxicity in salmonids: the effect of sub lethal ammonia exposure on the swimming performance of coho salmon and the acute toxicity of ammonia in swimming and resting rainbow trout. Aquatic Toxicology 59: 55-69.

Wilber, D.H. and Clarke, D.G. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. N. Amer. J. Fish. Mgmt. 21: 855-875.

Wilkie, M.P., Brobbel, M.A., Davidson, K., Forsyth, L., and Tufts, B.L. 1997. Influences of temperature upon the postexercise physiology of Atlantic salmon (*Salmo salar*). Can. J. Fish. Aquat. Sci. 54(3): 503-511.

Winter, A.R., J.W. Nichols, and R.C. Playle. 2005. Influence of acidic to basic water pH and natural organic matter on aluminum accumulation by gills of rainbow trout (*Oncorhynchus mykiss*). Can. J. Fish. Aquat. Sci. 62: 2303-2311.

Woodward, D.F., W.G. Brumbaugh, A.J. DeLonay, E.E. Little. 1994. Effects on rainbow trout fry of a metals-contaminated diet of benthic invertebrates from the Clark Fork River, Montana. Transactions of the American Fisheries Society 123: 51-62.

Woodward, D.F., A.M. Farag, H.L. Bergman, A.J. DeLonay, E.E. Little, C.E. Smith, and F.T. Barrows. 1995a. Metals-contaminated benthic invertebrates in the Clark Fork River Montana: Effects on age-0 brown trout and rainbow trout. Can. J. Fish. Aquat. Sci. 52: 1994-2004.

Woodward, D.F., J.A. Hansen, H.L. Bergman, E.E. Little, and A.J. DeLonay. 1995b. Brown trout avoidance of metals in water characteristic of the Clark Fork River Montana. Can. J. Fish. Aquat. Sci. 52: 2031-2037.

Woodward, D.F., J. N. Goldstein, and A.M. Farag. 1997. Cutthroat trout avoidance of metals and conditions characteristic of a mining waste site: Coeur d'Alene River, Idaho. Transactions of the American Fisheries Society 126: 699-706.

World Health Organization (WHO). 1996. Thallium and thallium compounds health and safety guide. IPCS International Programme on Chemical Safety, Health and Safety Guide Number 102. United National Environment Programme, World Health Organization, Geneva, Switzerland. http://www.inchem.org/documents/hsg/hsg/hsg102.htm#SectionNumber:2.7

Zeleznock, D. and M. Twiss. 2004. Toxicity of thallium to aquatic organisms of the North American Great Lakes. Sixth Annual Symposium of Undergraduate Research Experiences, July 29, 2004, Report No. CU-122. Clarkson University, Potsdam, New York. Pages 9-12. http://www.clarkson.edu/reu/archives/2004/abstracts-04/Dave-abstract.pdf

Zitko, V. and W.V. Carson. 1975. Accumulation of thallium in clams and mussels. Bulletin of Environmental Contamination and Toxicology. 13(1): 23-30.

Zitko, V., W.V. Carson, and W.G. Carson. 1975. Thallium: Occurrence in the environment and toxicity to fish. Bulletin of Environmental Contamination and Toxicology. 14(5): 530-533

Appendix A

(Appendix to Appendix F - FEIS)

Figures and Tables

Figure 1: Transportation access route proposed mitigation.

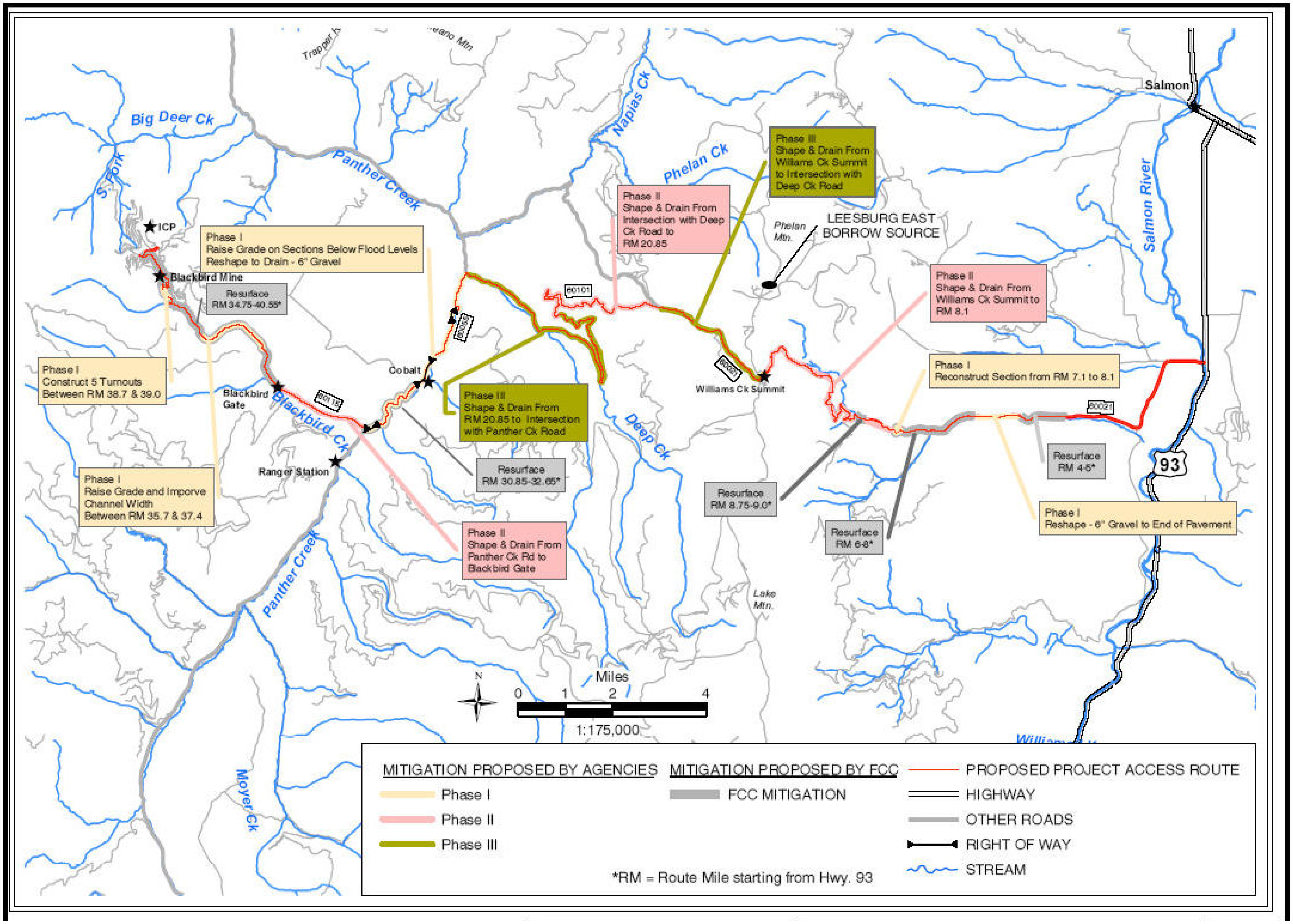
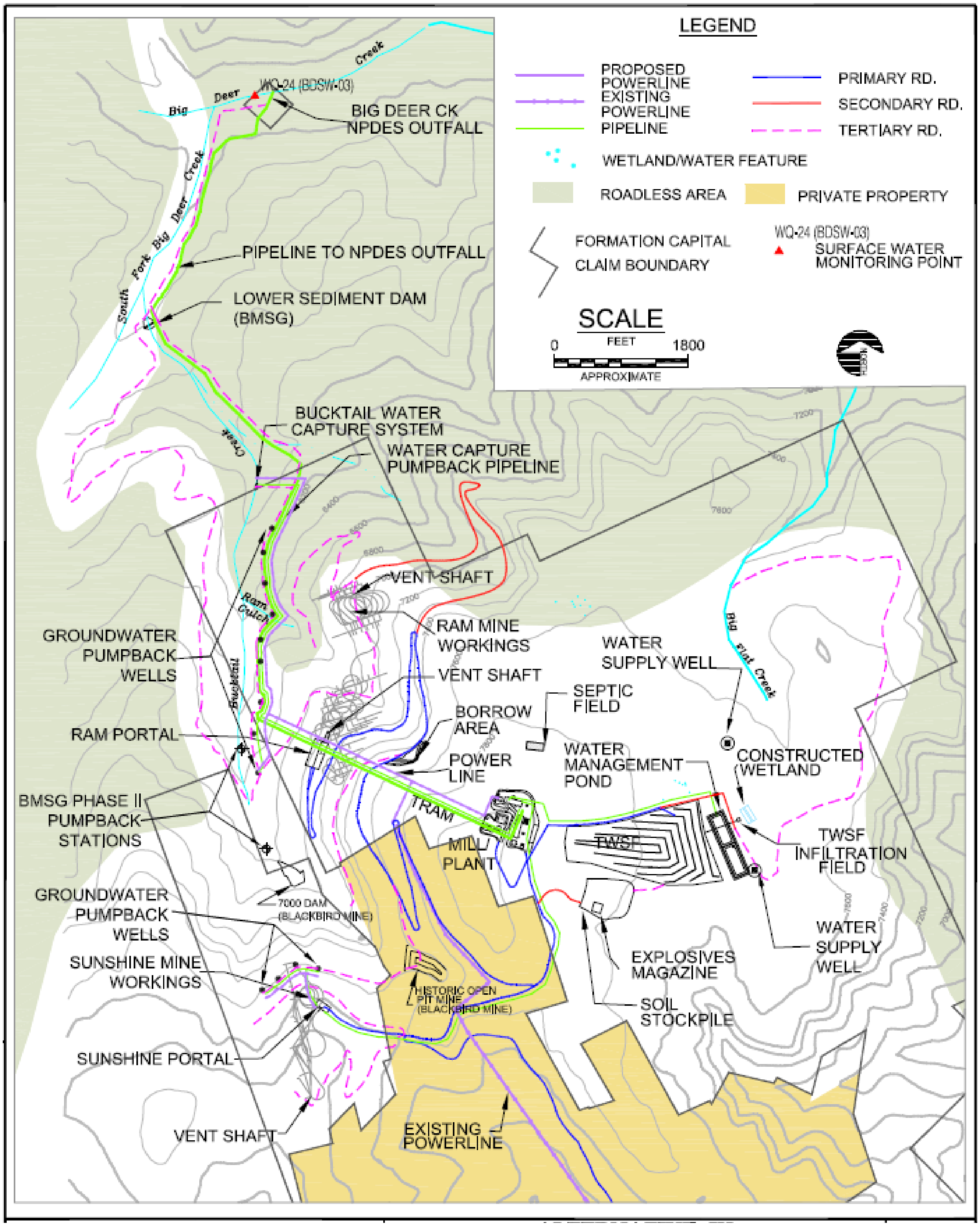
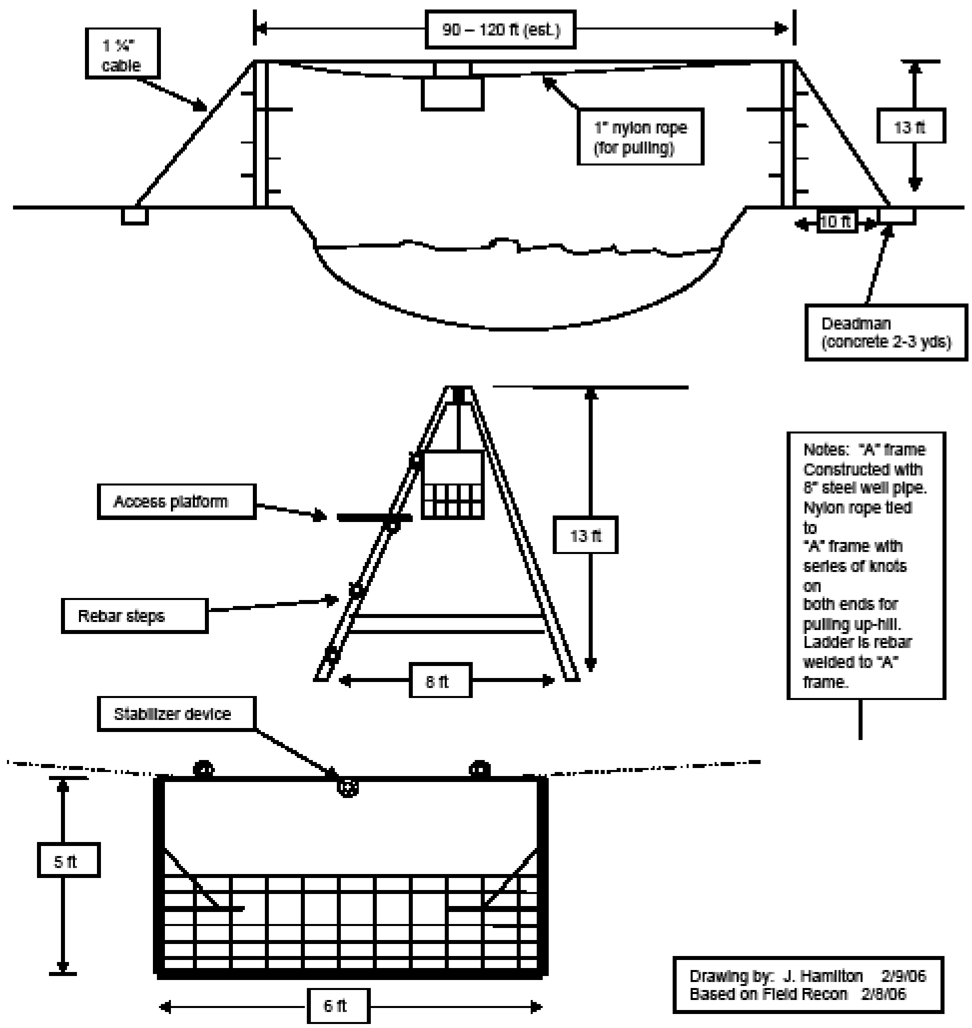


Table 1. Summary of Proposed Route Mitigation by Milepost and Phase.

Phase	Road Segment (Mileposts)		Proposed Mitigation
I	**5**	**6**	**Reshape - Gravel**
	7.1	**8.1**	**Realignment**
	27.9	**32.6**	**Reconstruct sections FR 60055 to raise road grade through sections w/in the floodplain. Construct retaining structure on the section at MP 30.6 to protect the fill slope. Shape and drain the subgrade. Place 6-inches gravel between Deep Ck. Road and Blackbird Creek Road (details follow).**
	29.07	**29.42**	**Raise grade above floodplain, drain**
	29.51	**29.63**	**Raise grade above floodplain, drain**
	29.97	**30.02**	**Raise grade above floodplain, drain**
	30.02	**30.49**	**Raise grade above floodplain, drain**
	30.5	**30.5**	**Improve ditch**
	30.6	**30.6**	**Construct retaining structure to protect fill slope**
	30.63	**30.66**	**Raise grade to drain**
	35.7	**37.4**	**Raise grade and improve channel width of Blackbird Creek**
	38.7	**39**	**Construct five turnouts**
II	**7.1**	**13.5**	**Reshape and drain**
	11.64	**12.45**	**Reinforce subgrade (geotextile)**
	16.5	**20.9**	**Reshape and drain**
	32.6	**34.8**	**Reshape and drain**
III	**13.5**	**16.5**	**Reshape and drain**
Additional Proposed Mitigation			
	4	**5**	**Resurface**
	6	**8**	
	8.75	**9**	
	30.9	**32.6**	
	34.8	**40.6**	

Figure 2: Idaho Cobalt Project proposed facility and site road layout.





Based on Noranda Cable Crossing on Panther Creek Near Big Deer

PROJECT: 222100 | TASK: 32

PREPARED BY: TELESTO

FIGURE 3
PROPOSED CABLE CROSSING DESIGN NEAR WQ-2

PREPARED FOR: Formation

Appendix B

(Appendix to Appendix F - FEIS)

An evaluation of the protectiveness of a site-specific criteria for cobalt in the Panther Creek watershed, Idaho for Chinook Salmon and Steelhead

March 1, 2007

An evaluation of the protectiveness of a site-specific criteria for cobalt in the Panther Creek watershed, Idaho for Chinook Salmon and Steelhead

Prepared by: Christopher A. Mebane, U.S. Geological Survey, Boise, Idaho

Prepared for: National Marine Fisheries Service, Idaho State Habitat Office

Contents

List of Tables

ACRONYMS

ACR	Acute-to-Chronic Ratio
BMSG	Blackbird Mine Site Group
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
FCV	Fish Chronic Value
HUC	Hydrologic Unit Code
ICV	Invertebrate Chronic Value
IDEQ	Idaho Department of Environmental Quality
LOEC	Lowest-Observed-adverse Effect Concentrations
MATC	Maximum Acceptable Toxicant Concentration
NMFS	National Marine Fisheries Service
NOEC	No-Observed-Effect Concentration
PER	Pacific EcoRisk
USGS	U.S. Geological Survey

Background and Summary

Panther Creek, Idaho (U.S. Geological Survey [USGS] hydrologic unit code [HUC] 17060203) provides habitat for Snake River Basin steelhead and Snake River spring/summer Chinook salmon, "species" listed as threatened under the Endangered Species Act (ESA). Panther Creek receives mine drainage from the Blackbird Mine. The drainage contains elevated concentrations of copper and cobalt, and has been the focus of scientific studies since the early 1990s. Substantive cleanup activities began around 1996, under supervision by the U.S. Environmental Protection Agency (EPA). Because no Idaho water quality standard or Federal recommended criteria existed for cobalt, EPA (2003c) derived a risk based cleanup level for cobalt. Using an ad hoc combination of site-specific and literature data on cobalt toxicity together with an uncertainty factor, a cleanup target of 38 µg/L for cobalt was selected for Panther Creek and portions of a tributary, Big Deer Creek[1]. The terms "criteria" or "target" are used here synonyms for benchmark or threshold of safe or low-effect concentrations, and do not attempt to match definitions used in various regulatory or environmental management schemes.

The EPA's record of decision included the sentence that "*EPA may consider a staged implementation which would allow for further cobalt toxicity analysis and biological testing, to determine if another cleanup level for cobalt is protective*" before requiring costly treatment of cobalt contaminated groundwater (EPA 2003c, at p. 12-7). On October 23, 2003, EPA completed a biological assessment of the effects of the remedial actions at the Blackbird Mine and concluded that their activities "may affect, but are not likely to adversely affect" Snake River Basin steelhead and Snake River spring/summer Chinook salmon. On November 24, 2003, the National Marine Fisheries Service (NMFS) wrote EPA concurring with assessment.

In 2004, further studies were done on the toxicity of cobalt to fish and invertebrate species that were considered representative of the Panther Creek aquatic community under representative test conditions. Rainbow trout, mottled sculpin, two mayflies, a caddisfly, and a midge were tested. The rainbow trout was considered a surrogate for the ESA-listed salmonid species. Based upon the results of the studies, EPA determined that the data supported approximately doubling the cobalt cleanup target from 38 µg/L to 86 µg/L. On November 6, 2006, EPA wrote to NMFS and the U.S. Fish and Wildlife Service, and requested concurrence with their determination that, as modified by the higher cobalt cleanup target, the remedial actions still "may affect, but are not likely to adversely affect" listed salmonids.

I have participated in and reviewed (1) earlier cobalt toxicity studies, (2) the evaluation used to develop the original 38 µg/L cobalt cleanup target, (3) the study plan and interpretation criteria for the 2004 testing, and (4) the results and interpretations of the 2004 testing. From these experiences, my view is that the 2004 test results represent the best available science on the effects of cobalt on species that are representative of the Panther Creek aquatic community. The EPA's interpretation of the test results was reasonable and conservative. I recommend concurring that the new cobalt target of 86°µg/L does not change the determination that the

[1] 38 micrograms per liter (µg/L), or parts-per-billion is equivalent to 0.038 milligrams-per-liter (mg/L) or parts per million.

remedial activities “may affect, but are not likely to adversely affect” listed Snake River Basin steelhead and Chinook salmon. Further, EPA intends to effectively use the 86°µg/L cobalt target as a water quality criteria in a new National Pollution Discharge Elimination System permit for discharges from the proposed Idaho Cobalt Project mine, located near the Blackbird Mine and discharging into Big Deer Creek. Such further application of the cleanup target would be equally supported by the data and would adequately protect listed salmonids.

Important elements of the 2004 cobalt testing program are described in more detail as follows.

Review of Methods and Decision Criteria

Little research into the toxicity of cobalt has been reported, relative to other metals such as copper, zinc, or cadmium. The few data available were difficult to interpret and apply to a watershed such as Panther Creek because of large disparities in the toxicity of cobalt reported with different species or conditions. With some freshwater crustaceans such as amphipods and daphnids, cobalt has been reported to be highly toxic with adverse effects observed at concentrations as low as 5 µg/L (Borgmann et al. 2005; Mebane 1994). In contrast, in 14-day duration tests with rainbow trout in soft-water, cobalt concentrations up to 125 µg/L caused no mortality, although sublethal effects such as reduced growth were not tested (Marr et al. 1998). Fathead minnows could tolerate up to about 290 µg/L before adverse reproductive effects were expected (Mebane 1994, citing Kimball 1978). However, with the exception of one study reporting severe impairment of mayfly development at 32 µg/L cobalt (Sodergren 1976), the tests with effect concentrations below 100 µg/L were otherwise from lentic (lake) species. The EPA (2003b) did not consider lentic species such as *Daphnia* sp. relevant to flowing water ecosystems such as Panther Creek. Instead, they focused on studies conducted on rainbow trout, a species that was likely to occur in Panther Creek and was related to other salmonid species of concern. The EPA’s (2003b) interpretation of available literature on the effects of cobalt on rainbow trout included safety factors or extrapolation factors to attempt to account for shortcomings in existing data, such as extrapolating from a chronic test that apparently used a metals-resistant strain of fish and to extrapolate acute effects to possible chronic effects. This resulted in the initial cleanup target of 38 µg/L.

The EPA’s (2003b) interpretation of a cobalt cleanup target was reasoned, addressed uncertainty conservatively, and considered the best science available to them at the time. However, because of the anticipated costs to meet the 38 µg/L cobalt target and because of the scientific uncertainty in the target, the Blackbird Mine Site Group (BMSG, the responsible parties conducting the cleanup) decided to undertake a series of aquatic toxicity tests on cobalt in Panther Creek water to make a more rigorous assessment of site-specific toxicity to resident fish, anadromous salmonids, and their prey.

Discussions among the BMSG, EPA, Idaho Department of Environmental Quality, and NMFS on study design began in fall 2003, study plans were agreed to in spring 2004 (Stantec 2004), and testing was conducted in summer and fall 2004. The study design had similarities to EPA’s resident species approach to developing site-specific criteria (Carlson et al. 1984; EPA 1994), but was modified to address concerns for ESA-listed salmonids (Table 1).

The cobalt toxicity testing workplan was the result of a cooperative effort to objectively define how test data would be used in advance of the testing. Considerable effort was made by the workgroup participants to both define in advance which tests were to be conducted and how the data would be interpreted. At the time, it was anticipated that these efforts would reduce later disagreements or misunderstandings on data interpretation. The workplan covered all major questions that the authors and reviewers could anticipate in advance of the testing. However, it did not cover every situation, question, or extenuating circumstance that resulted, and best professional judgment was still needed in some points in the project. This was particularly the case in the interpretation of a single, highly influential acute toxicity test with a mayfly.

By law, EPA has primary responsibility for administering the Blackbird Mine remediation. Therefore, the final responsibility and authority for determining cobalt cleanup values in Panther Creek rested with EPA.

Rationale for evaluating direct effects on listed species

Rainbow trout (*Oncorhynchus mykiss*) were selected as the best available single surrogate test species for ESA-listed Snake River Chinook salmon (*O. tshawytscha*), steelhead and bull trout (*Salvelinus confluentus*)[2]. Rainbow trout are the same species as listed Snake River Basin steelhead and are in the same genus as Chinook salmon and family as bull trout. In comparative toxicity testing with cadmium, copper and zinc, rainbow trout have been about as sensitive or were more sensitive than Chinook salmon or bull trout (Chapman 1978; Hansen et al. 2002a; Hansen et al. 2002b). However, inbred strains of cultured test animals or even wild strains may have wide differences in sensitivities to metals. For example, even after normalization to reduce the influence of water quality differences on metals sensitivities, rainbow trout differed in sensitivity by a factor of 6 with cobalt (EPA 2003c), a factor of 20 with copper (EPA 2003a), and a factor of 10 with cadmium (Mebane 2006). As a precaution against the risk conducting an expensive chronic rainbow trout toxicity test with an unrepresentative strain of hatchery rainbow trout, rainbow trout from the same hatchery stock were first tested for resistance to copper, a well tested metal. The assumption was that if the responses of the test trout to copper were intermediate or sensitive in comparison to previous reports, the fish would be similarly responsive to cobalt.

One feature of the data interpretation that was specific to addressing ESA concerns was the use of the no-observed-effect concentration (NOEC) statistic in the decision criteria for the rainbow trout test, rather than less protective conventional endpoints. Typically in EPA's criteria derivation process, the statistical value used to summarize chronic toxicity data and set criteria is the geometric mean of the statistical NOEC and lowest-adverse-effect-concentration (LOEC). In toxicological textbooks and literature (Rand et al. 1995) this value is often referred to as a hypothetical "maximum acceptable toxicant concentration" (MATC). Despite this reassuring

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[2] While this memo focuses on listed anadromous fish, ESA listed bull trout occur in Panther Creek and affected tributaries. Along with the listed anadromous fish, the intent of the work group was for the study design to sufficiently protect bull trout, and the entire aquatic community.

sounding term, when applied to ESA listed or other vulnerable species, this test statistic may not reflect a "maximum acceptable" concentration, since the magnitude of adverse effect may be quite high. Suter et al. (1987) determined that levels of effect associated with the MATC were 20% to 40% reductions in survival or fecundity, and questioned whether these could be considered "acceptable." Instead, for the present project, the NOEC was used. NOECs are not a perfect statistic either because of the problem that effects that may be biologically important may not be statistically significant if the test design is not robust or if the responses have high variability. For example, Suter et al. (1987) found that the average effect of a NOEC for fecundity in fish corresponded to about a 14% reduction. Conversely, biologically trivial effects could be statistically significant if tests have low variability and lots of replicates. Still, the NOEC is a much more protective statistic than is the MATC and seemed the most suitable test statistic for analyzing rainbow trout responses to cobalt, since rainbow trout were considered the direct surrogate for listed salmonids. The use of the NOEC to estimate a chronic effects value for a listed species is consistent with the national methods manual for conducting biological evaluations under section 7 of the Endangered Species Act, *"For this chronic analysis, each test should be represented by a value that estimates a very low effect level. For most tests, the NOEC will be used as this value that estimates a very low effect level. While the NOEC is not a true "effects" concentration, the NOEC may be the best parameter, for purposes of an ESA Section 7 biological evaluation"* (EPA 2003d).

Rationale for evaluating indirect effects on listed species

"Indirect effects" of an action on listed species include effects on species that interact with listed species, such as prey, predators, or competitors for food or space. Protection against indirect effects is intended to protect the conservation value of the ecosystem, avoid adverse modifications of habitat, and avoid impeding recovery.

The cobalt toxicity study plan was probably sufficient to protect against appreciable adverse indirect effects on aquatic species. In the higher elevation parts of the Panther Creek drainage, the fish assemblage consists almost entirely of salmonids and sculpin. Thus by testing rainbow trout and sculpin, test species representative of the entire fish assemblage were being tested. As Panther Creek drops in elevation and warms, minnow and sucker species become better represented in the assemblage. Previous information on the relative sensitivities of salmonids, sculpin, minnows, and suckers to cobalt and other metals suggested that either sculpins or salmonids were generally most sensitive to metals, and most data for minnows and suckers indicated they were less sensitive (Dillon and Mebane 2002; EPA 2003a; EPA 2003c; Mebane 1994). Thus, minnows and suckers would probably be sufficiently protected if sculpin and salmonids were protected.

The invertebrate testing targeted species that were usually abundant in upper sections of Panther Creek and tributaries that were unaffected by mine effluents and were representative of groups that are important in stream food webs (mayflies, caddisflies, and midges). Caddisflies and midges were well known to be resistant to acute metals toxicity. However, an implicit assumption of this approach is that it is not necessary (or feasible) to test the most sensitive resident invertebrate species, rather what is important is to test representatives from major taxonomic groups.

Extrapolating acute to chronic effects data: the acute-to-chronic ratio (ACR) issue

A conundrum for this project, other consultations and assessments of criteria, and most projects involving ecotoxicology, is that exposures in the stream are likely indefinite and long-term whereas the toxicity tests for most species are mostly conducted for 48 or 96 hours. Acute-to-chronic ratios (ACR) are a simple mathematical approach for extrapolating chronic toxicity test values to untested taxa from tested taxa. As a practical matter, the ACR approach is a necessary tool for estimating standards that are protective of chronic endpoints of untested species. The ACR approach is widely used for several reasons despite its reliance of several implicit, fundamental, and untested assumptions. Reasons for its longstanding use include:

- Conducting direct chronic toxicity tests (e.g. long-term laboratory exposures that are designed to assess endpoints important to the protection of populations and communities) on all species of concern is an impossibility, even at a discrete location. Thus some other estimation techniques are needed;

- Life history and culture requirements of many important species are poorly understood, and techniques for culturing and testing some species in laboratories may be uncertain;

- Present understanding of chronic toxicity mechanisms in response to long-term exposures to metals is inadequate to predict responses of untested (or untestable) species based upon on theoretical relationship.

Thus, the simple, empirical approach of using ACRs is considered a necessary, interim approach for protecting aquatic ecosystems from chronic toxicity until chronic toxicity mechanisms are better understood.

While seldom explicitly described in criteria documents, the rationale for the ACR rests upon several fundamental, implicit assumptions. These include:

1. Mechanisms of acute and chronic metals toxicity are related so that predictions of one as a constant proportion of the other are warranted;

2. Even if mechanisms for acute and chronic toxicity vary, the factors that make metals more or less toxic affect chronic responses in the same manner and at a constant proportion as acute toxicity. In other words, even if acute metals toxicity is due to accumulation on the gill surfaces and chronic toxicity is due to kidney failure, the influences of hardness, pH, and organic carbon may be assumed to act equally in mitigating toxicity.

3. In cases where ACRs are not constant between species, the rank order of acutely sensitive species and chronically sensitive species will generally hold.

4. The ACRs vary mostly as a function of acute sensitivity. Assuming that acute responses to a given metal are more variable than chronic responses, it follows that ACRs vary in a

predictable manner such that tests that provide lower acute LC50s will not yield proportionally lower chronic values.

5. If assumptions numbers 3 and 4 are accepted, it follows that if the acute sensitivities have been tested for a wide variety of species, and if an ACR is derived for the most acutely sensitive (or nearly so) species, then it is unlikely that a acutely less-sensitive species will be chronically more-sensitive than the most acutely-sensitive species.

These implicit assumptions may be internally conflicting, in conflict with current aquatic toxicology theory, and at the least, under tested. In practical applications such as the EPA criteria documents, these assumptions may hold in some cases but not in others. While the assumptions follow patterns sometimes seen in testing, there is little theoretical support for these assumptions, and in some cases these assumptions are known to be false. The questionable scientific bases of ACRs not withstanding (and because of the limited, practical alternative approaches), ACRs remain a useful way to extrapolate the chronic toxicity to untested species.

Important Statistical Issues

In contrast to the direct surrogate for listed salmonids (rainbow trout), other chronic effect estimates for sculpin and invertebrates were defined less conservatively with the EC20 statistic (Table 1). The EC20 statistic is the concentration causing a 20% reduction in some important biological endpoint, such as survival, growth, emergence, or fecundity. The EC20 value was accepted by a consensus among the cobalt technical advisory group of consulting and agency scientists. No analysis was attempted of why a 20% effect was expected to be acceptable to sculpin populations or sensitive invertebrates. The EC20s have been used in risk assessments and in recent EPA criteria documents as a regression-based substitute that reflects similar magnitudes of adverse effects for the MATC (EPA 1999; EPA 2003a). No biological analyses of the assumed acceptability of 20% adverse effects to stream communities is known of and none was attempted here.

To ensure that the results of the toxicity testing would be representative of Panther Creek waters during the seasons when cobalt is of greatest risk, a large volume of water was pumped from Panther Creek, upstream of mining effluents, and shipped in refrigerated trucks to the Pacific EcoRisk (PER) test facility in Martinez, California (Figure 1). Water was collected during the summer at base flow; unlike copper which is highest in the early spring snowmelt, cobalt concentrations tend to be higher and thus of most concern at base flows. Native invertebrates were collected from Panther Creek, identified and sorted on site (Figure 2), and shipped by air to the PER facility for testing. Because little information on the acute toxicity of stream insects to cobalt was available, the tests with resident invertebrates were first conducted as rangefinding tests. As the name suggests, “rangefinding” tests are relatively low precision tests that encompass a wide range of concentrations in order to hone in on a narrower range of concentrations for follow-on “definitive” testing. Definitive tests are expected to give more precise results.

No attempts were made to collect or breed resident sculpin so that the sensitive early life stages could be tested. Instead testing was conducted with a broodstock of mottled sculpin (*Cottus* *bairdi*) maintained by the USGS Columbia Ecological Research Center, Columbia, Missouri . The USGS Columbia center had recently conducted an extensive series of acute and chronic tests with mottled sculpin and cadmium, copper, and zinc (Besser et al. 2006). The Missouri mottled sculpin were considered a surrogate species for Panther Creek mottled and shorthead sculpin populations. Because of uncertainty whether the Missouri mottled sculpin stock were more or less resistant to cobalt than Panther Creek sculpin, they were tested in conjunction with hatchery rainbow trout as a "common denominator" (Appendix 1).

Chronic tests with rainbow trout and the midge *Chironomus tentans* were done using animals purchased from commercial suppliers. The chronic tests were conducted aquaria or beakers with automated refreshing of the exposure solutions (flow through tests) (Figure 3). Water chemistry was measured frequently during the chronic tests both to measure the cobalt exposures and to document and interpret water quality conditions that may affect the toxicity of the metal or the animals' responses (e.g. organic carbon, hardness, pH, alkalinity, temperature, dissolved oxygen).

Table 1. General approach for developing a site-specific chronic criterion for cobalt in Panther Creek, using resident species

Fish Chronic Value		**Invertebrate Chronic Value**	
Lower of either:	Remarks	Lower of either	Remarks
1. Rainbow trout no-observed-effect concentration (NOEC), or	Rainbow trout were considered a surrogate for ESA listed steelhead and Chinook salmon. NOEC was chosen as a low-effect measure appropriate for evaluating direct effects to a listed species. NOEC is defined as the highest tested concentration that wasn't statistically different from the control with 95% confidence.	1. The lowest chronic EC20 obtained with the midge *Chironomus tentans* (non-resident laboratory cultured population); or	The midge was selected because it was an invertebrate species for which full life cycle test methods had been developed, and midges are important stream resident insects. Midges are often metals-resistant, so the data were intended to be used in a relative sense to relate acute responses to chronic responses (acute to chronic ratios, ACRs)
2. The estimated sculpin chronic concentration adversely affecting 20% of the test population (EC20) <u>if</u> sculpin more acutely sensitive than trout	Sculpin were included because they have been shown to be sensitive to other metals.	2. The lowest estimated chronic EC20 obtained from resident invertebrate species collected from Panther Creek.	7 species were targeted for collection based on their abundance in surveys. 4 species were successfully tested.
	An EC20 was selected as an endpoint because of the assumption that adverse effects to 20% of the sculpin populations are sustainable.	Chronic EC20 values for resident invertebrates were estimated as the acute values, divided by an acute-to-chronic ratio (ACR). The ACR was defined as the geometric mean of the ACRs obtained from the rainbow trout tests and from the midge tests.	

Table 2. Summary of results of toxicity tests using species resident to Panther Creek (in milligrams per liter, mg/L, parts per million).

Fish Acute tests		**Invertebrate Acute Tests**	
Species	LC50 (mg/L)	Species	LC50 (mg/L)
Rainbow trout (acute test #1 in Panther Creek water)	0.86	Midge *Chironomus tentans*	157
Rainbow trout (acute test #2 in Panther Creek water)	0.80	Caddisfly *Brachycentrus americanas*	7,219
Rainbow trout (USGS acute test conducted in laboratory water)	1.36	Ephemerellid mayfly (*Serratella sp.*)	79
Mottled sculpin (USGS acute test conducted in laboratory water)	2.11	Baetid mayfly (*Centroptilum* sp.), rangefinding test #1	≈ 2
		Baetid mayfly (*Centroptilum* sp.), rangefinding test #2	9.4
		Baetid mayfly (*Centroptilum* sp.), definitive test	2.0
Fish chronic test	**Effect (mg/L)**	**Invertebrate chronic test**	**Lowest EC20 (mg/L)**
Rainbow trout (NOEC-no reduction in growth or survival)	0.101	Midge *Chironomus tentans* (reduced survival at 20 days)	0.237
Rainbow trout (LOEC- 5% reduction in growth, as total growth of individuals)	0.242		
Rainbow trout (20% reduction in growth or survival)	>0.971		

Table 3. Determination of an instream cobalt cleanup criteria (the lower of the fish chronic value (FCV) or the invertebrate chronic value (ICV))

Fish Chronic Value (FCV)		**Invertebrate Chronic Value (ICV)**	
Lower of either:	Remarks	Lower of either	Remarks
1. Rainbow trout no-observed-effect concentration (NOEC), or	0.101 mg/L	**3.** The lowest chronic EC20 obtained with the midge *Chironomus tentans* (non-resident laboratory cultured population); or	0.237 mg/L
2. The estimated sculpin chronic concentration adversely affecting 20% of the test population (EC20) if sculpin more acutely sensitive than trout	Sculpin were less acutely sensitive than rainbow trout.	**4.** The lowest estimated chronic EC20 obtained from resident invertebrate species collected from Panther Creek. This was estimated as the lowest acute invertebrate value, divided by an acute-to-chronic ratio (ACR). The ACR was defined as the geometric mean of the ACRs obtained from the rainbow trout tests and from the midge tests. ACR calculations for rainbow trout: ACR = Mean acute LC50/Chronic EC20 = (0.83 mg/L)/ (>0.971 mg/L) = < 0.82 (unitless) [Note: Because the >0.971 value was unquantifiable, the value was treated as 0.971 for ACR calculations] ACR calculations for midge *Chironomus*: ACR = (157 mg/L)/ (.237 mg/L) = 662 Site ACR = geometric mean of <0.82 and 662 = <23.3	
		4b. Lowest resident invertebrate LC50: mayfly *Centroptilum*.	2.0
		5. ICV = Lowest invertebrate LC50/Site ACR = (2.0 mg/L)/(<23.3)	>86 mg/L
		6. Overall site cleanup value, also called cobalt toxicity reference value is the lower of the fish or invertebrate chronic value. (FCV = 0.101 mg/L; ICV >=0.086 mg/L, ICV considered lower	0.086 mg/L (ppm) = **86 µg/L (ppb)**

Figure 1. About 113, 000 L (30,000 gallons) of Panther Creek water were collected upstream of mining influences and shipped in 6 refrigerated trucks to the Pacific EcoRisk Laboratories, Martinez, California (PER). Photos from McKee and Hansen (McKee and Hansen 2005).

Figure 2. Native macroinvertebrates were collected from Panther Creek with kick-nets, were sorted and identified on site, and selectees were shipped by air to PER for testing. Photos from McKee and Hansen (2005).

Figure 3. Chronic trout test setup (top) and chronic midge test setup (bottom). Photos from McKee and Hansen (2005).

Test results and data interpretation

Rainbow trout were consistently the most acutely sensitive fish and thus results with rainbow trout could safely be assumed to be protective of sculpin (Table 2). However, interpretation of the most acutely sensitive native invertebrates were much more complicated and controversial. Of the invertebrate tests conducted, it was clear that the Baetid mayfly was most sensitive. However, in the data report for these tests, the reported LC50 (concentration lethal to 50% of the test population) was mistakenly reported as 3.9 mg/L. Upon review and recalculation, it was found that depending upon the statistical model used, adjustments for deaths that occurred in the control treatment, adjustments for larvae that emerged from the treatments prior to the end of the test and other assumptions or decisions, possible LC50 estimates for this test could be calculated ranging from about 1.6 to 4.6 mg/L. The estimates straddled a concentration that when divided by an acute-to-chronic toxicity extrapolation factor, would either influence the final cobalt cleanup criteria or would drop out of the decision criteria (Tables 1 and 3). This triggered extensive debates, statistical analyses, written exchanges and a meeting between the workgroup members on how best to calculate an LC50 for this single test.[3] In my view, valid arguments could be made supporting LC50 calculations for this test ranging from about 1.8 to 2.4 mg/L and within this narrow range, these estimates probably would not have changed the conclusion reached here that the site-specific cobalt target avoids appreciable adverse effects to listed species. The final LC50 value of 2.0 mg/L selected by EPA fit the data well, the statistical and biological assumptions seem fully appropriate, and thus it probably reflects the best interpretation of the available science for this test and species (Figure 4).

<u>Convergence of lines of evidence</u> – In sum, there were three independent lines of evidence completed to support an instream cobalt criteria for aquatic life in Panther Creek: (1) the highest cobalt concentration that did not result in direct adverse effects to rainbow trout, the surrogate species for listed salmonids; (2) the lowest chronic EC20 for the midge *Chironomus tentans*; and (3) an estimated chronic value for a sensitive, native invertebrate (the lowest definitive native invertebrate acute LC50 divided by the mean site acute-chronic ratio).

Despite the wide range of results obtained from disparate taxa, these three lines of evidence converged within a factor of 3 (86 to 237 µg/L). Since literature values for cobalt targets or safe endpoints for different species range over a factor of at least 60 (<5 to >300 µg/L), the range of the Panther Creek final endpoints is fairly narrow. Whether this convergence indicates corroboration or mere coincidence, it does offer some reassurance that the approach and results were reasonable.

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[3] Including exchanges of letters by (1) Michael Ives, Humboldt State University, March 31, 2005; (2) Russell Erickson, EPA National Health and Environmental Effects Laboratory, Duluth, of April 6, 2005; (3) Russell Erickson of April 18, 2005; (4) Frank Dillon and Shaun Roark, CH2M Hill, Seattle of April 18, 2005; (5) Steven Hansen and Paul McKee of May 6, 2005; (6) Fran Allans, EPA, Boise of May 23, 2005, (7) a meeting in Boise on August 25, 2005, and (8) a letter with further analysis from Russell Erickson on about September 5, 2005.

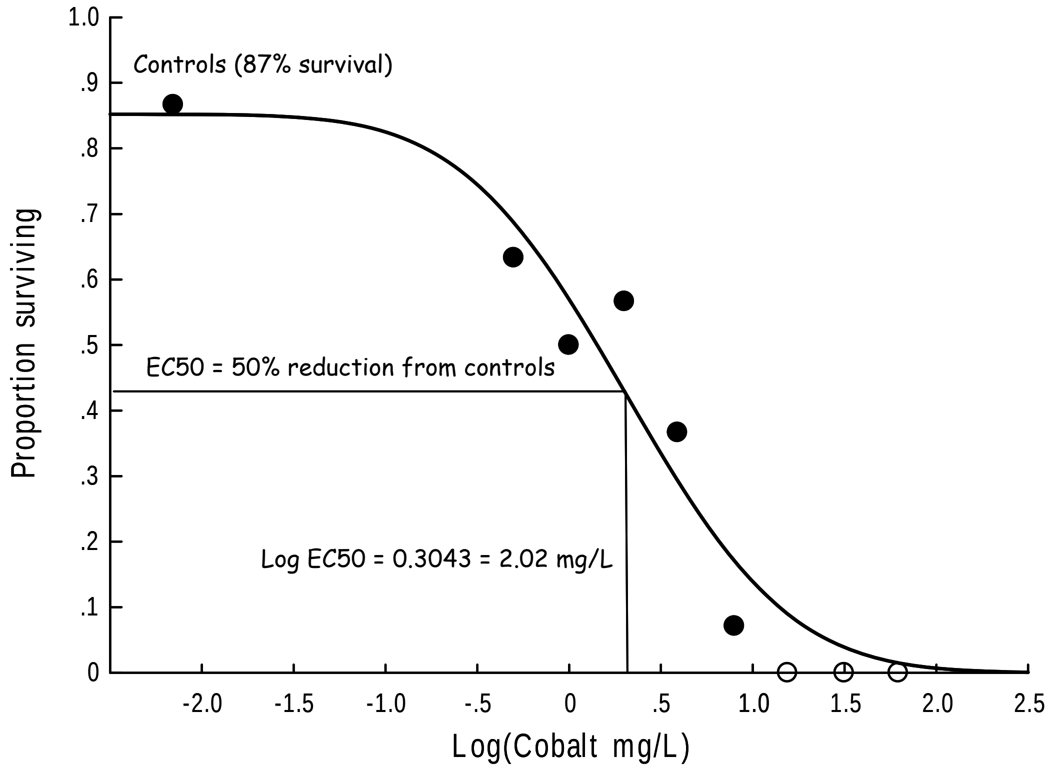

Figure 4. Illustration of best EC50 estimate from the definitive mayfly Centroptilum test (most sensitive invertebrate). Open circles indicate data points with complete mortality that were censored from the analysis to avoid right skewing the EC50 estimate.

Uncertainties in the protectiveness of the site-specific cobalt criteria for listed salmonids

Ecotoxicology and ecological risk assessment are not certain sciences, and pronounced uncertainties are unavoidable in most projects. The cobalt site-specific criteria project reviewed here was an ambitious effort to reduce one major aspect of uncertainty - the extrapolation of toxicity testing results using standard laboratory cultured test organisms in laboratory water supplies. By using site water, site organisms, and at least laboratory or hatchery stocks of the same or similar species that occur in the study area, the uncertainty associated with that extrapolation was reduced. Several other areas of scientific uncertainty were considered (Table 4). Overall, the uncertainties remaining probably represent acceptable risks. Some of the remaining uncertainties were interpreted conservatively to give the benefit of doubt to the listed species and environment. Those uncertainties that were not treated conservatively did not seem to be of such magnitude that they compromised the protectiveness of the overall target.

Table 4. Some uncertainties and their implications to the protectiveness of the Panther Creek watershed cobalt site-specific criteria.

Uncertainties	**Considerations**	**Direction of Potential Bias (C–conservative, NC– non-conservative)**
Fish value based on rainbow trout NOEC	A statistical NOEC (no-observed-effect-concentration) is not a "true" effect, but is a statistical absence of effects.	Conservative
Rainbow trout threshold of effects was subtle	The statistical LOEC (lowest-observed-effect-concentration) resulted from only a 5% reduction in total weight of individual fish in the treatment. This low level of effect suggests that there is some safety margin in the cobalt target against severe, direct adverse effects to salmonids	Conservative
ACR of <23 is highly uncertain	Ratio is the geometric mean of two numbers differing by ~700X. Viewed in isolation, averaging numbers as disparate as <1 and 662 may be biologically meaningless. But when viewed in the context of other data, the resulting ACR of <23 is not out of the range of ACRs seen with other metals for similar taxa; When applied in the decision criteria, the result was similar to the trout NOEC. While this convergence was probably serendipitous, it indicates the ACR value is biologically plausible. (Continued from right column) The use of a geometric rather than arithmetic mean lessens the conservatism (Arithmetic mean of 1 and 662 is 331, but the geometric mean is 23). Still the ACR is probably somewhat conservative.	Probably conservative, since (1) the number is calculated as "less than," and (2) acute-to-chronic ratios tend to be lower for more acutely sensitive species. The estimated sensitivity of the *Centroptilum* mayfly (LC50 2.0 mg/L) was closer to that of the rainbow trout (LC50 0.83 mg/L) than to the midge (LC50 157 mg/L). *Continued*
Most sensitive early instar life-stage of stream invertebrates doubtfully tested	But, mayflies seem to be recovering in Panther Creek under ambient Co concentrations of about 40-60 µg/L, suggesting earliest instars aren't orders of magnitude more sensitive than the target	Non-conservative. Difficult or infeasible to test earliest instars of field collected insects.
Most sensitive resident invertebrate species were doubtfully tested	But, among the invertebrates found in fast moving, mountain streams, mayflies are often more sensitive to metals than other major groups such as common caddisflies, stoneflies, and true flies. Among the mayflies, the Heptageniid and Ephemerellid groups may be generally more sensitive than Baetid mayflies. Efforts to collect and test Heptageniid mayflies failed; an	Non-conservative. This is an inherent limitation of testing resident species; difficult or infeasible to collect and test large numbers of rare taxa

Uncertainties	Considerations	Direction of Potential Bias (C–conservative, NC– non-conservative)
	Ephemerellid was tested but was less sensitive than the Baetid *Centroptilum*. The species tested include those that are abundant in site-waters and are likely important salmonid prey items. This assumes that the more important prey species are those that are relatively abundant and thus available in fish diets, not necessarilty those that are most sensitive	
LC50 value from most sensitive resident invertebrate, the mayfly *Centroptilum*	EPA's best estimate of the LC50 for this test was 2.0 mg/L; other estimates ranged from 1.6 to 4.6 mg/L.	Conservative. Although the 2.0 mg/L estimate was neither conservative or non-conservative for the definitive test, the two lower precision rangefinding test LC50s conducted earlier in the summer were about 2 mg/L and 9.4 mg/L. Together the three tests suggest that under some circumstances, the *Centroptilum* mayfly species could be less sensitive.

Uncertainties	Considerations	Direction of Potential Bias (C–conservative, NC– non-conservative)
Cobalt tested singly but always occurs with copper. Metals mixtures may be more toxic than either singly.	Previous literature reports that the toxicity of copper and cobalt mixtures is greater than the toxicity of either alone. The joint toxicity of copper -cobalt mixtures was about additive with an invertebrate (*Daphnia*) and slightly less than additive with a fathead minnow (Mebane 1994). With rainbow trout tested at mixture ratios more relevant to Panther Creek, the presence of low copper levels would increase cobalt toxicity by about 10% (Marr et al. 1998)	Non-conservative, perhaps by about a factor of 1.1 (Marr et al. 1998) or of up to 2 (Mebane 1994). The lower factor of 1.1 seems more likely, since the ratios tested were more relevant to Panther Creek conditions. (continued in middle column below)
	(Continued from above right) Recent field surveys of macroinvertebrates show about a 20% decline in taxa richness in Panther Creek sites located downstream from Blackbird Mine drainages, compared to upstream reference sites. However, overall invertebrate abundances including abundances of salmonid food items were similar between upstream and mine influenced sites. Changes in macroinvertebrate metrics associated with mine effluents were not correlated with cobalt concentrations, but were correlated with copper concentrations and with temperature. No adverse effects to Panther Creek fish communities were obvious in recent survey results (EcoMetrix 2006). Therefore, the magnitude of increased toxicity from copper-cobalt mixtures under site-specific conditions is probably not so large as to overwhelm the conservatism of other factors, nor large enough to compromise the validity of the site-specific criteria analysis or to change conclusions of the overall protectiveness of the target.	

Conclusion

Based upon (1) the review of and in some instances participation in previous studies relating the effects of cobalt on Panther Creek stream ecosystems, and (2) review of the methods and results of the most recent studies, I think the current data described here likely represents the best available science for evaluating direct and indirect effects to Snake River Basin steelhead and Snake River Chinook salmon. The EPA's interpretation of the testing design and test results was well reasoned and most major uncertainties were interpreted sufficiently conservatively as to give the benefit of doubt to the listed species. The majority of the information reviewed indicates that appreciable adverse effects to Snake River steelhead or Snake River Chinook salmon populations or their ecosystem associated with sustained cobalt concentrations up to 86 µg/L (the cleanup target) are unlikely.

References

Besser, J. M., C. A. Mebane, C. D. Ivey, J. L. Kunz, E. I. Greer, T. W. May, and C. G. Ingersoll. 2006. Relative sensitivity of mottled sculpins (*Cottus bairdi*) and rainbow trout (*Onchorhynchus mykiss*) to toxicity of metals associated with mining activities. U.S. Geological Survey, Columbia Environmental Research Laboratory, Administrative Report to U.S. Environmental Protection Agency, Project No. DW14939902-01-0, Columbia, Mo. 37 pp.

Borgmann, U., Y. Couillard, P. Doyle, and D. G. Dixon. 2005. Toxicity of sixty-three metals and metalloids to *Hyalella azteca* at two levels of water hardness. Environmental Toxicology and Chemistry. 24(3): 641–652.

Carlson, A. R., W. A. Brungs, G. A. Chapman, and D. J. Hansen. 1984. Guidelines for deriving numerical aquatic site-specific water quality criteria by modifying national criteria. U.S. Environmental Protection Agency, EPA-600/3-84-099 PB85-121101, Washington, DC.

Chapman, G. A. 1978. Toxicities of cadmium, copper, and zinc to four juvenile stages of chinook salmon and steelhead. Transactions of the American Fisheries Society. 107(6): 841-847.

Dillon, F. S. and C. A. Mebane. 2002. Development of site-specific water quality criteria for the South Fork Coeur d'Alene River, Idaho: application of site-specific water quality criteria developed in headwater reaches to downstream waters. Prepared for and in conjunction with the Idaho Department of Environmental Quality. Windward Environmental, Seattle, WA. 95 pp. http://www.deq.state.id.us/water/data_reports/surface_water/monitoring/site_specific_criteria.cfm.

EcoMetrix. 2006. Biomonitoring study, Panther Creek watershed, September 2005. Prepared by EcoMetrix Incorporated, Brampton, Ontario, for the Blackbird Mine Site Group, Salmon, Idaho. 96 pp.

EPA. 1994. Water quality standards handbook. U.S. Environmental Protection Agency, EPA-823-B-94-005a, Washington, D.C http://www.epa.gov/ost/standards/handbook/.

EPA. 1999. 1999 update of ambient water quality criteria for ammonia. U.S. Environmental Protection Agency, EPA/822/R-99-014, Washington, D.C. 147 pp.

EPA. 2003a. 2003 update of ambient water quality criteria for copper. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, EPA 822-R-03-026, Washington, DC. 86 pp. http://www.epa.gov/waterscience/criteria/copper/.

EPA. 2003b. Biological Assessment of the Effects of the Remedial Action Activities for the Blackbird Mine. Prepared by CH2M Hill for the U.S. Environmental Protection Agency, Boise, Idaho. 94 pp.

EPA. 2003c. Blackbird Mine Superfund Site, Lemhi County, Idaho Record of Decision. U.S. Environmental Protection Agency, Seattle, WA. 159 pp.

EPA. 2003d. Draft methodology for conducting biological evaluations of aquatic life criteria: analysis document - copper prototype. U.S. Environmental Protection Agency, Office of Science and Technology, EPA-823-B-03-003, Washington, DC. 63 pp.

Hansen, J. A., J. Lipton, and P. G. Welsh. 2002a. Relative sensitivity of bull trout (*Salvelinus confluentus*) and rainbow trout (*Oncorhynchus mykiss*) to acute copper toxicity. Environmental Toxicology and Chemistry. 21(3): 633–639.

Hansen, J. A., P. G. Welsh, J. Lipton, D. Cacela, and A. D. Dailey. 2002b. Relative sensitivity of bull trout *(Salvelinus confluentus)* and rainbow trout *(Oncorhynchus mykiss)* to acute exposures of cadmium and zinc. Environmental Toxicology and Chemistry. 21(1): 67–75.

Marr, J. C. A., J. A. Hansen, J. S. Meyer, D. Cacela, T. Podrabsky, J. Lipton, and H. L. Bergman. 1998. Toxicity of cobalt and copper to rainbow trout: application of a mechanistic model for predicting survival. Aquatic Toxicology. 43: 225-237.

McKee, P. and S. R. Hansen. 2005. A resident species approach to the development of a site-specific water quality criterion for cobalt in Panther Creek, Idaho. 32nd Aquatic Toxicology Workshop, Waterloo, Ontario, Canada, October 2-5, 2005.

Mebane, C. A. 1994. Blackbird Mine preliminary natural resource survey. U.S. National Oceanic and Atmospheric Administration, Hazardous Materials Assessment and Response Division, Seattle, WA. 130 pp.

Mebane, C. A. 2006. Cadmium risks to freshwater life: derivation and validation of low-effect criteria values using laboratory and field studies. U.S. Geological Survey, Scientific Investigation Report 2006-5245. 130 pp. http://pubs.water.usgs.gov/sir20065245/.

Rand, G. M., P. G. Wells, and L. S. McCarty. 1995. Introduction to aquatic toxicology. Pages 3-67 *in* G. M. Rand, editor. Fundamentals of aquatic toxicology: effects, environmental fate, and risk assessment, second edition. Taylor and Francis, Washington, D.C.

Sodergren, S. 1976. Ecological effects of heavy metal discharge in a salmon river. Institute of Freshwater Research (Drottingholm, Sweden). 99: 91-131.

Stantec. 2004. Cobalt toxicity testing workplan. Prepared by Stantec Consulting, Ltd and S.R. Hansen and Associates for the Blackbird Mine Site Group, Salmon, Idaho. April 6, 2004, Brampton, Ontario. 17 pp.

Suter, G. W., II, A. E. Rosen, E. Linder, and D. F. Parkhurst. 1987. Endpoints for responses of fish to chronic toxic exposures. Environmental Toxicology and Chemistry. 6: 793-809.

Appendix: Summary of acute toxicity testing of mottled sculpin and rainbow trout with cobalt by the U.S. Geological Survey, Columbia, Missouri

Besser, J. M. 2004. Summary of acute toxicity testing of mottled sculpin and rainbow trout with cobalt. U.S. Geological Survey, Columbia Ecological Research Center, Letter of August 24, 2004 to Chris Mebane, U.S. Geological Survey, Boise, Idaho, Columbia, Missouri. 5 pp.