BIOLOGICAL EVALUATION
FOR THE NPDES PERMIT FOR THE HECLA MINING COMPANY
GROUSE CREEK UNIT

NPDES Permit No.: ID0026468

U.S. Environmental Protection Agency
Region 10
Seattle, Washington

March 2016

BIOLOGICAL EVALUATION
FOR THE NPDES PERMIT FOR THE GROUSE CREEK UNIT

EXECUTIVE SUMMARY

The U.S. Environmental Protection Agency (USEPA) conducted a biological evaluation to identify potential impacts to federally listed Endangered or Threatened species that could result from the reissuance of the National Pollutant Discharge Elimination System (NPDES) permit to the Grouse Creek Unit, operated by Hecla Mining Company. The Grouse Creek Unit is an inactive gold mine and mill located in Custer County Idaho, approximately 19 miles northeast of Stanley.

The proposed NPDES permit authorizes the discharge of mine water from the Sunbeam adit, seepage and runoff from the waste rock dump, storm water from some portions of the facility, and wastewater from the tailings impoundment under-drains. The wastewater is treated and discharged through outfalls 002 and 003 into Jordan Creek and Yankee Fork Creek, respectively. Jordan Creek flows into the Yankee Fork of the Salmon River approximately four miles from the mine site. The NPDES permit places effluent limits on the discharges for which there is reasonable potential for exceedances of state standards including metals (cadmium, copper, lead, zinc, and mercury), weak acid dissociable (WAD) cyanide, whole effluent toxicity (WET), and total suspended solids (TSS), and pH. Sources of wastewater for both outfalls will be the same.

The U.S. Fish and Wildlife Service and NOAA have identified the following Endangered and Threatened species in this portion of Idaho. This list was compiled from the ESA pages of the Service's websites and FWS letter:

Endangered Species:	Sockeye salmon (*Oncorhynchus nerka*)
Threatened Species:	Spring/summer Chinook salmon (*O. tshawytscha*) Steelhead (*O. mykiss*) Bull trout (*Salvelinus confluentus*)

USEPA has determined that issuance of the permit for the Grouse Creek Unit ***is not likely to adversely affect*** sockeye salmon. USEPA has determined that the permitted discharge ***is likely to adversely affect*** the Chinook salmon, steelhead, and bull trout.

Table of Contents

I. INTRODUCTION—LIST OF SPECIES 8

II. PROJECT DESCRIPTION 10

A. OUTFALL DESCRIPTION 12
B. RECEIVING WATER 12
C. PERMIT LIMITS 14

III. SPECIES DESCRIPTIONS-INVENTORIES AND SURVEYS 18

A. AVAILABLE FISH MONITORING DATA 19
B. BULL TROUT (*SALVELINUS CONFLUENTUS*) THREATENED 19
C. SNAKE RIVER SOCKEYE SALMON (*ONCORHYNCHUS NERKA*)-ENDANGERED 23
D. SNAKE RIVER CHINOOK SALMON (*ONCORHYNCHUS TSHAWYTSCHA*) - THREATENED 23
E. SNAKE RIVER STEELHEAD (*ONCORHYNCHUS MYKISS*) - THREATENED 25

IV. ENVIRONMENTAL BASELINE 27

A. DEFINE ACTION AREA 27
B. WATER QUALITY 303B LIST 28
C. RECEIVING WATER CHEMICAL CONCENTRATIONS 28
D. CONDITIONS RELATED TO HISTORICAL MINING 28
E. HABITAT CHARACTERISTICS OF THE RECEIVING WATERS 29
F. SUMMARY OF HABITAT CONDITIONS 33

V. DESCRIPTION OF PARAMETERS OF CONCERN AND POTENTIAL EFFECTS . 33

A. METALS--INTRODUCTION 34
B. METALS WITH EFFLUENT LIMITS 34
C. METALS WITHOUT EFFLUENT LIMITS 38
D. AVOIDANCE OF METALS 41
E. AVOIDANCE BEHAVIOR ASSOCIATED WITH THE MIXTURES OF METALS 44
F. COMBINED EFFECTS OF METALS ON TOXICITY 45
G. NON METALS WITH EFFLUENT LIMITS (CYANIDE, TSS, WET, PH) 46
H. NON METALS WITHOUT EFFLUENT LIMITS: 50

VI. EFFECTS DETERMINATIONS 58

A. METHODS FOR DETERMINATION OF EFFECTS 58
B. EFFECTS DETERMINATION FOR SPECIES WITH LIMITED EXPOSURE 60
C. EFFECT DETERMINATION FOR TSS 62
D. EFFECT DETERMINATION FOR PH 64
E. EFFECT DETERMINATION FOR METALS AND WAD CYANIDE AT/BEYOND THE EDGE OF MIXING ZONE 66
F. EFFECT DETERMINATION FOR METALS AND WAD CYANIDE WITHIN THE MIXING ZONE 67
G. EFFECT DETERMINATION FOR WET AND MIXTURES 81
H. EFFECTS RELATED TO BIOACCUMULATION –SELENIUM AND MERCURY 89
I. BIOMONITORING 96

VII. CUMULATIVE EFFECTS 100

VIII. MANAGEMENT ACTIONS RELATED TO THE SPECIES 100

A. INCORPORATION OF 2003 BIOLOGICAL OPINION RPA AND RPMS 101
B. INCORPORATION OF RPAS IN THE IDAHO WATER QUALITY STANDARDS BOS 101
C. INCORPORATION OF RPMS AND TERMS AND CONDITIONS FROM NMFS BO ON IDAHO WQS 103
DD. OTHER MANAGEMENT ACTIONS 106

IX. ISSUES OF TAKE 106

X. SUMMARY OF DETERMINATIONS 107

A. SUMMARY OF EFFECTS OF THE ACTION ON ESSENTIAL ELEMENTS OF CRITICAL HABITAT 108

XI. ESSENTIAL FISH HABITAT 110

A. BACKGROUND 110
B. DESCRIPTION OF THE PROJECT/PROPOSED ACTIVITY 110
C. POTENTIAL ADVERSE EFFECTS OF PROPOSED PROJECT 111
D. EFH CONSERVATION MEASURES AND CONCLUSION 111

XII. REFERENCES 112

XIII. APPENDICES 130

Table of Figures

Figure 1. Bull trout abundance estimates for the Yankee Fork from electrofishing surveys conducted 1998, 1999, and 2001 (USFS 2004). 22
Figure 2. Spring Chinook redds survey data by IDFG, Yankee Fork 1987-2013 (Source: Streamnet 2015, query #44212). 25
Figure 3. Steelhead captures at two Jordan Creek locations (Source: USFS 1992, CEC 1999b). 27
Figure 4. Steelhead redd counts in the Salmon River between Pahsimeroi River and Warm Springs Creek confluences 1990-1997 (Source: Streamnet database). 27
Figure 5. Yankee Fork mean monthly flows based on 2003-2008 data (Source Brian Nickel, USEPA)..... 31
Figure 6. Summary of Freshwater Laboratory Studies, Copper Only Exposure LOEC's from the literature (taken from Shepard and Zodrow 2009). 43
Figure 7. Ammonia median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013 (Source: Helca monitoring data). 52
Figure 8. Ammonia median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013 (Source: Helca Limited)..... 52
Figure 9. Nitrate+nitrite median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013(Source: Helca Limited)..... 54
Figure 10. Outfall 002 monthly discharge based on data 2012 through winter 2016 (Source: Cercla data provided by Brian Nickel, USEPA)..... 61
Figure 11. Outfall 003 monthly discharge based on data 2012 through winter 2016 (Source: Cercla data provided by Brian Nickel, USEPA)..... 61
Figure 12. TSS median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013 (Source: Helca Limited)..... 63
Figure 13. TSS median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013 (Source: Helca Limited)..... 63
Figure 14. PH median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013 (Source: Helca Limited)..... 65
Figure 15. PH median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013(Source: Helca Limited)..... 66
Figure 16. Acute mixing zone for cadmium, with Jordan Creek flows less than 30 CFS. 69
Figure 17. Acute mixing zone for copper, with Jordan Creek flows less than 30 CFS (14.8 µg/L or 12.69 µg/L excess above background of 2.11 µg/L). 69
Figure 18. Acute mixing zone for zinc, with Jordan Creek flows less than 30 CFS (103.5 µg/L or 96.8 µg/L excess above background of 6.70 µg/L). 70
Figure 19. Acute mixing zone for cadmium, with Jordan Creek flows greater than or equal to 30 CFS. 70
Figure 20. Acute mixing zone for copper, with Jordan Creek flows greater than or equal to 30 CFS (23.6 µg/L or 21.49 µg/L above background of 2.11 µg/L). 71
Figure 21. Acute mixing zone for zinc, with Jordan Creek flows greater than or equal to 30 CFS (140 µg/L or 133.3 µg/L above background of 6.7 µg/L). 71
Figure 22. Dilution isoline for outfall 002, with flows less than 30 CFS. The isoline for 3:1 dilution corresponds to the 25% mixing zone for all parameters except WET. 72
Figure 23. Dilution isolines for outfall 002, with flows greater than or equal to 30 CFS. The

isoline for 1.4:1 dilution corresponds to the mixing zone for copper. The isoline for 1.64:1 dilution corresponds to the mixing zone for zinc. The isoline for 3:1 dilution corresponds to the 25% mixing zone for all parameters except copper, zinc, and WET. 73
Figure 24. Avoidance threshold concentration isoline for zinc, with Jordan Creek flows less than 30 CFS (21 µg/L, or 18 µg/L excess above upstream concentration of 3 µg/L). 74
Figure 25. Avoidance threshold concentration isoline for zinc, with Jordan Creek flows greater than 30 CFS (21 µg/L, or 18 µg/L excess above upstream concentration of 3 µg/L). 74
Figure 26. Avoidance threshold concentration isoline for copper, with Jordan Creek flows less than 30 CFS (11.8 µg/L or 11.3 µg/L excess above upstream concentration of 0.5 µg/L). 75
Figure 27. Avoidance threshold concentration isoline for copper, with Jordan Creek flows greater than 30 CFS (11.8 µg/L or 11.3 µg/L excess above upstream concentration of 0.5 µg/L). 75
Figure 28. Dilution isolines for outfall 003, with flows greater than or equal to 45 CFS. The isoline for 3.92:1 dilution corresponds to the mixing zone for copper. The isoline for 5.26:1 dilution corresponds to the mixing zone for cadmium. The isoline for 6.61:1 dilution corresponds to the 25% mixing zone for all parameters except copper, cadmium, and WET. 78
Figure 29. Temperature median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013(Source: Hecla Limited)..... 80
Figure 30. Temperature maximum values for outfall 002 and Jordan Creek monitoring sites, 2003-2013 (Source: Hecla Limited). 80
Figure 31. Temperature median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013(Source: Hecla Limited)..... 81
Figure 32. Temperature maximum values for outfall 003 and Yankee Fork monitoring sites, 2003-2013(Source: Hecla Limited). 81
Figure 33. Concentration (1.0 TUc) isoline for outfall 002 for WET, with flows less than or equal to 30 CFS. 82
Figure 34. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 002 for WET, with flows less than or equal to 30 CFS. 83
Figure 35. Concentration (1.0 TUc) isoline for outfall 002 for WET, with flows greater than or equal to 30 CFS..... 83
Figure 36. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 002 for WET, with flows greater than 30 CFS. 84
Figure 37. Concentration (1.0 TUc) isoline for outfall 003 for WET, with flows < 15 CFS. 85
Figure 38. Concentration (1.0 TUc) isoline for outfall 003 for WET, with flows from 15 – 45 CFS. 86
Figure 39. Concentration (1.0 TUc) isoline for outfall 003 for WET, with flows greater than 45 CFS. 86
Figure 40. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 003 for WET, with flows < 15 CFS. 87
Figure 41. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 003 for WET, with flows from 15 – 45 CFS. 87
Figure 42. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 003 for WET, with flows > 45 CFS. 88

Table of Appendices

Appendix A. Fish survey results..... 131

Appendix B. Summary statistics of water quality monitoring data collected at both outfalls and 4 monitoring sites 2003-2013 (excluding 2011). (source: Hecla Limited)..... 132

Table of Maps

Map 1. Map of outfalls and monitoring sites for Grouse Creek. 11

Table of Tables

Table 1. Summary of the listing history for salmon species of the Snake River basin in vicinity of Grouse Creek Action Area. 8
Table 2. Listed non-salmon species and critical habitat in the vicinity of the Grouse Creek Action Area (source: FWS letter to USEPA, Idaho Fish and Wildlife Office, 02/24/2015). 9
Table 3. Idaho Water Quality Criteria Applicable to the Grouse Creek Permit. 13
Table 4. Authorized mixing zones by parameter and flow tier. 16
Table 5. Effluent limits for outfall 002. 17
Table 6. Effluent Limits for outfall 003. 18
Table 7. Stream monitoring locations. 18
Table 8. Jordan Creek fish monitoring data 1997-2014 (GEI 2014b, GEI 2015). 21
Table 9. Physical characteristics of various sites of Jordan Creek (Source CEC 2000a). 30
Table 10. Habitat condition ratings in Jordan Creek from 1994 survey data (USFS 1994). 30
Table 11. Yankee Fork Water Temperatures (taken from USDA Forest Service 2007). 33
Table 12. Hardness based criteria for outfall 002 (from Table B-5 of the 2016 Fact Sheet). 34
Table 13. Hardness based criteria for outfall 003 (from Table B-6 of the 2016 Fact Sheet). 34
Table 14. Zinc concentrations associated with fish avoidance (Taken from IDEQ 2000). 44
Table 15. Overview of likely exposure by life history stage of listed salmonid species at each outfall. 61
Table 16. Acute mixing zones for cadmium, copper and zinc for Outfall 002. 68
Table 17. Distances and plume travel times to avoidance thresholds for outfall 003 79
Table 18. Chronic WET results of outfall 003(source: Helca Mining Co. provided to Brian Nickel, EPA). 88
Table 19. Summary of mean concentrations of selenium and mercury in sediment (mg/kg dw) collected from two sites in Yankee Fork (Source: GEI 2014c). 91
Table 20. Selenium and mercury mean concentrations in benthic macroinvertebrates collected from two sites in Yankee Fork (Source: GEI 2007, from Table 5). 93
Table 21. Selenium and mercury concentration in whole-body whitefish tissue, Yankee Fork (Source: GEI 2014c). 94
Table 22. Selenium and mercury concentration in whitefish gonadal tissue, Yankee Fork (Source: GEI 2014c). 95
Table 23. Available IDEQ BURP monitoring data (1995-2011) from action area. 97
Table 24. Results of multimetric analysis of benthic macroinvertebrates in Yankee Fork. 98
Table 25. Results of multimetric index analyses for benthic macroinvertebrate (SMI) and fish (SFI) assemblages in Jordan Creek. 99
Table 26. Effects determinations for the four listed species addressed in this BE. 108

I. INTRODUCTION—LIST OF SPECIES

USEPA is proposing to reissue the National Pollutant Discharge Elimination System (NPDES) permit for the Grouse Creek Unit (NPDES Permit No. ID0026468), inactive gold mine and mill. The Grouse Creek Unit is located in Custer County, Idaho, approximately 19 miles northeast of Stanley. The NPDES permit will authorize discharge from one outfall discharging to Jordan Creek, designated as Outfall 002 and one outfall discharging to Yankee Fork, designated as Outfall 003. The Endangered Species Act (ESA) requires federal agencies to consult with the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS) if the federal agency's actions could beneficially or adversely affect any threatened and endangered species. This Biological Evaluation (BE) identifies the endangered, threatened, and proposed species in the project area and assesses potential effects to these species that may result from the discharge of mine effluent to streams as authorized in the Grouse Creek Unit NPDES permit.

This BE also considers Critical Habitat for the listed species. Critical Habitat designations are a regulatory process intended to provide protection for specific habitats of listed species to help ensure their recovery. The NMFS and USFWS have identified the following Threatened and Endangered species and their Critical Habitat in the vicinity of the Grouse Creek Unit (Table 1 and Table 2).

Table 1. Summary of the listing history for salmon species of the Snake River basin in vicinity of Grouse Creek Action Area.

Salmonid Species	**ESU/DPS Name**	**Original Listing**	**Revised Listing(s)**	**Critical Habitat**
Sockeye Salmon (O. nerka)	Snake River Sockeye Salmon	FR Notice: 56 FR 58619 Date: 11/20/1991 Endangered	FR Notice: 70 FR 37160 Date: 6/28/2005 Endangered	12/28/93 (58FR68543) mainstem Salmon River and several lakes within the Stanley Basin. Excludes Yankee / Jordan Creeks. Reaffirmed 10/25/99 (64FR57399)
Chinook Salmon (O. tshawytscha)	Snake River Spring/ Summer Chinook salmon	FR Notice: 57 FR 34639 Date: 4/22/1992 Threatened	FR Notice: 70 FR 37160 Date: 6/28/2005 Threatened	10/25/99 (64FR57399)
Steelhead (O. mykiss)	Snake River Basin Steelhead	FR Notice: 62 FR 43937 Date: 8/18/1997 Threatened	FR Notice: 71 FR 834 Date: 1/5/2006 Threatened	September 2, 2005 (70FR52630). Includes upper Salmon River Basin (including Jordan and Yankee Fork)

Table 2. Listed non-salmon species and critical habitat in the vicinity of the Grouse Creek Action Area (source: FWS letter to USEPA, Idaho Fish and Wildlife Office, 02/24/2015).

Species	Population	ESA Status	Critical Habitat
Bull trout *Salvelinus confluentus*	Columbia River DPS	***Threatened***- 6/10/98 (62FR32268).	75 FR no200 pg. 63898, 10/18/2010, Designation of Critical Habitat for bull trout (including Jordan Creek and Yankee Fork Creek)
Whitebark pine *Pinus albicaulis*		Candidate-7/18/2011 (50CFR17)	N/A

In addition to bull trout and whitebark pine, the FWS website lists Canada lynx (threatened), yellow billed cuckoo *Coccyzus americanus* (Threatened), and Greater sage-grouse *Centrocercus urophasianus* (not warranted as of 10/2/15) in Custer County, Idaho. Possible effects to the terrestrial species identified by the USFWS are not associated with the reissuance of NPDES permits for the Grouse Creek Mine because there are no routes of exposure to the constituents of the mine effluent. Therefore, ***the whitebark pine receives a No Effect determination and this species is not addressed further in this analysis.*** Likewise, the other terrestrial species (those not specified in the FWS letter), greater sage-grouse, yellow billed cuckoo and Canada lynx have no exposure pathway to the effluent constituents of concern in this BE. These terrestrial species are not addressed further.

This BE also analyzes whether the project would result in habitat destruction or adverse modification to designated Critical Habitat, as well as potential effects of these project actions on essential fish habitat (EFH) As defined in Section 16 U.S.C. 1802(10) of the Magnuson Act, EFH includes the waters and substrate necessary for (managed) marine and anadromous fish populations to spawn, breed, feed, or grow to maturity. The Upper Snake Basin, including the project area, is EFH for Chinook salmon.

Analysis in this BE is driven by the Section 7 Consultation Provision of the Endangered Species Act (1973). The act requires all Federal Agencies to ensure that actions authorized, funded, or carried out by those agencies are not likely to jeopardize the continued existence of any Threatened, Endangered, or Proposed (TES) species, or result in the destruction or adverse modification of their critical habitat.

This BE adheres to the recommended contents of biological assessments for submission to USFWS and NMFS (USFWS and NMFS 1998). Only minor changes to organization have been made to accommodate the particular subject matter of discharge of pollutants from outfalls to streams. This report is organized into the following major sections. First, the permit action is described. Second, listed species are described with attention to their distribution and life history. This section includes information regarding the presence of the species in the project area and abundance trends. Third, general information regarding each of the pollutants in the permit is provided. Fourth, the effect of the permit on the listed species is evaluated. The effects determination is made by evaluating the potential for effects at the edge of the mixing zones and within the mixing zones. A discussion of cumulative effects, management actions related to the species, and issues of take follow the effects determinations.

II. PROJECT DESCRIPTION

The Grouse Creek Unit (GCU) consists of an inactive open pit gold mine and mill and support facilities. The facility is owned and operated by the Hecla Mining Company (Hecla). The facility operated from December 1994 through April 1997 and is currently undergoing closure.

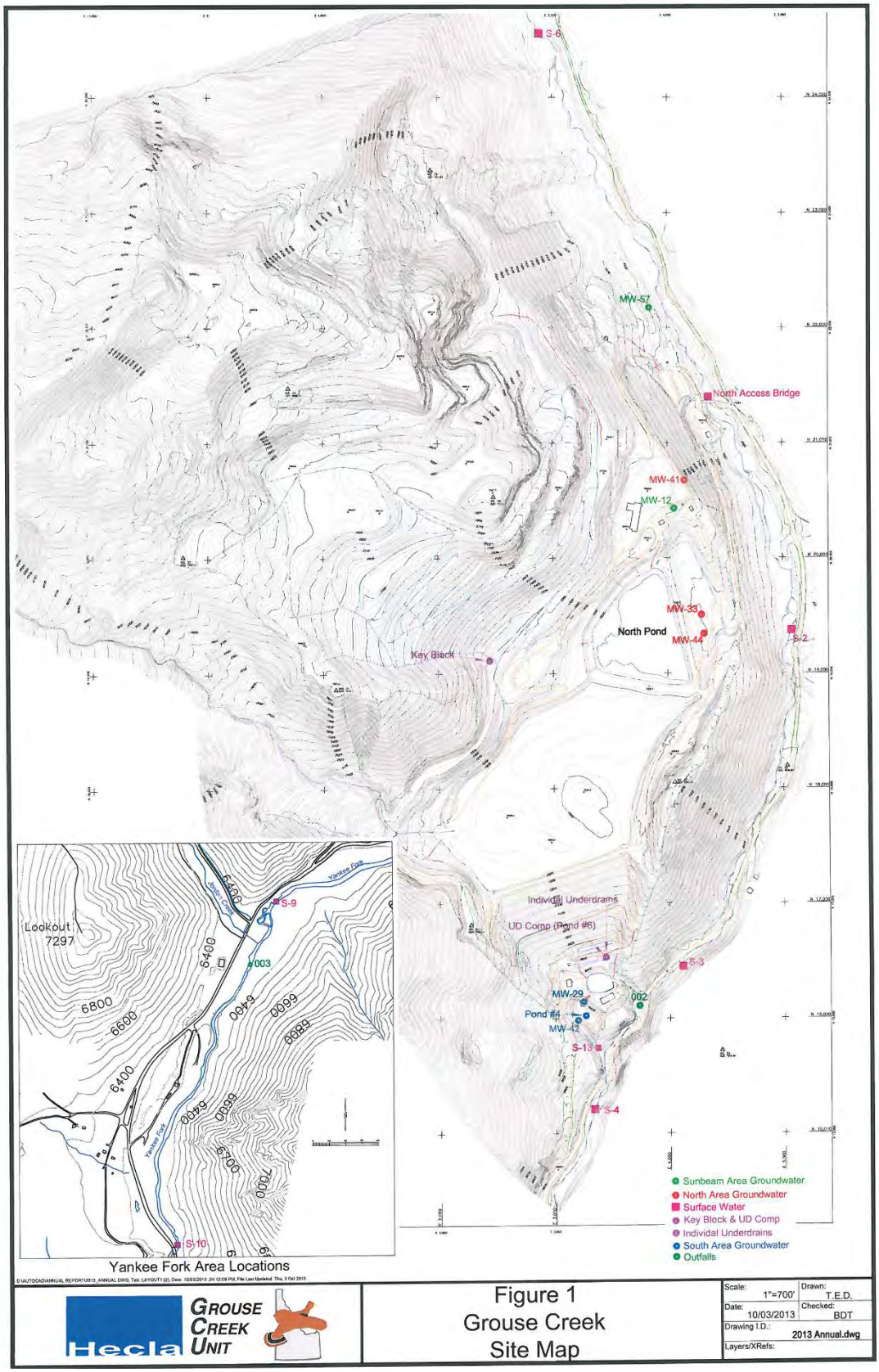
The GCU covers approximately 590 acres on both private lands and federal lands. The federal land area is managed by the U.S. Forest Service (Salmon-Challis National Forest). The mine facilities are located in the Grouse Creek, Pinyon Creek, Washout Creek, and Jordan Creek drainages. Grouse, Pinyon, and Washout creeks are tributaries to Jordan Creek which flows into Yankee Fork approximately 4 miles downstream of the mine site. Yankee Fork Creek flows into the Salmon River approximately 7.5 miles downstream from its confluence with Jordan Creek. The Jordan Creek and Yankee Fork watersheds are within the Upper Salmon River Basin (HUC # 17060201). Components of the facility that result in the generation of wastewater include mined areas and other disturbed areas, the Sunbeam mine adit, the waste rock storage area, and the tailings impoundment. A general description of these components is in the 2015 Permit Fact Sheet.

The first NPDES permit for the GCU was issued effective on November 5, 1992 and expired on November 5, 1997. This permit was administratively extended and remained effective and enforceable until the permit was re-issued in 2002. The 2002 permit expired in February 2007 but was also administratively extended and remains effective and enforceable as Hecla submitted a timely renewal application (submitted August 2006).

USEPA proposes to reissue a NPDES permit which may include revisions based upon comments received during the public comment period and the final Clean Water Act (CWA) Section 401 certification from the State of Idaho Department of Environmental Quality (IDEQ). The conditions in the draft permit are based on non-operating conditions. If Hecla decides to reopen the mine, they will need to apply for a new permit. The permit will be finalized following completion of this consultation.

The proposed NPDES permit authorizes the discharge from existing Outfall 002 to Jordan Creek. The permit application requests authorization of a new outfall, outfall 003, which discharges to the Yankee Fork just downstream from the Jordon Creek confluence with their Yankee Fork. Although the EPA is proposing to authorize discharges from outfall 003 under an NPDES permit for the first time, discharges from outfall 003 had previously been authorized under the Comprehensive Environmental Response Compensation and Liability Act (CERCLA), commonly known as Superfund.

Both of these outfalls are subject to effluent limits that ensure compliance with water quality standards and technology-based requirements, as well as monitoring, and other conditions specified in the permit. The following sections provide a description of the outfalls and the effluent that will be discharged to the streams.



Map 1. Map of outfalls and monitoring sites for Grouse Creek.

A. Outfall Description

This section provides a summary description of the outfalls. The locations of the outfalls and the monitoring sites are shown on Map 1. A detailed description of the outfalls and their location is provided in the 2015 Fact Sheet for the draft NPDES permit.

Outfall 002: outfall 002 discharges to Jordan Creek approximately 3.2 miles upstream of its confluence with the Yankee Fork. The stream flow is continuous yet variable as influenced by precipitation and snow melt. Peak flows occur during May and June. Effluent discharge to Jordan Creek consists of wastewater from the following sources:

- mine drainage water from the Sunbeam underground mine adit
- seepage and runoff from the waste rock dump
- seepage and groundwater collected from tailings impoundment under-drains
- storm-water runoff from the impoundment embankment

These wastewater sources are currently routed to collection ponds and treated prior to discharge from outfall 002. Treatment consists of carbon adsorption to reduce the cyanide contained in the tailings impoundment under-drains followed by hydroxide and sulfide precipitation of all the wastewaters to reduce the metals. The wastewater then flows to a settling pond prior to discharge through outfall 002.

Outfall 003: outfall 003 discharges to Yankee Fork between Jordan Creek and the Salmon River. The outfall is located approximately eight miles downstream of the Jordan/Yankee Fork confluence and approximately 7.5 miles upstream of the Yankee Fork/Salmon River confluence. Effluent discharge to Yankee Fork, consists of wastewater from the same types of sources and subject to the same treatment as listed above for outfall 002.

B. Receiving Water

Outfall 002: Jordan Creek is not designated for specific uses in Idaho's water quality standards. However, IDAPA 58.01.02.101.01 designates all undesignated waters for cold water aquatic life and primary and secondary contact recreation uses.

Outfall 003: Yankee Fork Creek is designated for the uses of cold water aquatic life, salmonid spawning, primary contact recreation, and domestic water supply (IDAPA 58.01.02.130.03). These are surface waters that are recognized as needing intensive protection; a) to preserve outstanding or unique characteristics; or b) to maintain current beneficial use.

In addition, all waters of the State of Idaho are designated for industrial and agricultural water supply, wildlife habitats, and aesthetics (IDAPA 58.01.02.100).

The Idaho water quality criteria applicable to these waters are based on these use designations. For each water quality parameter, water quality-based effluent limits must be based on the most stringent water quality criterion applicable to the receiving water, in order to ensure that all of the uses are protected. The applicable water quality criteria, are listed in Table 3 (from Appendix B of the 2016 Fact Sheet).

Table 3. Idaho Water Quality Criteria Applicable to the Grouse Creek Permit.

Idaho Water Quality Criteria Applicable to the Grouse Creek Permit[3]			
Parameter	**Criteria**		**Uses**
	Jordan Creek	**Yankee Fork Creek**	
Ammonia[2]	Acute: 8.31 mg/L Chronic: 3.24 mg/L	Acute: 10.3 mg/L Chronic: 3.74 mg/L	Cold Water Aquatic Life
Arsenic[4]	10 µg/L		Human Health
Cadmium[1]	Dependent upon hardness		Cold Water Aquatic Life
Copper[1]	Dependent upon hardness.		Cold Water Aquatic Life
Cyanide	Acute: 22 µg/L Chronic: 5.2 µg/L		Cold Water Aquatic Life
Lead[1]	Dependent upon hardness		Cold Water Aquatic Life
Mercury, Water Column	Acute: 2.1 µg/L Chronic: 0.012 µg/L		Cold Water Aquatic Life
Methyl Mercury, Fish Tissue	0.3 mg/kg		Human Health (consumption of fish)
Nitrate + Nitrite (Yankee Fork Creek only)	10 mg/L		Domestic Water Supply
Nitrate + Nitrite (Statewide)	100 mg/L		Agricultural Water Supply
pH	6.5 – 9.0 standard units		Aquatic Life
Sediment	Narrative criterion		Various uses
Selenium	Acute: 20 µg/L Chronic: 5 µg/L		Cold Water Aquatic Life
Silver[1]	Dependent upon hardness.		Cold Water Aquatic Life
Temperature[5]	Max. daily avg.: 19 °C Instantaneous max.: 22 °C	Max. daily avg.: 9 °C Instantaneous max.: 13 °C	Cold Water Aquatic Life (Jordan Creek) Salmonid Spawning (Yankee Fork Creek)
Whole Effluent Toxicity	"Surface waters of the state shall be free from toxic substances in concentrations that impair designated beneficial uses."		Cold Water Aquatic Life, other designated uses.
Zinc[1]	Dependent upon hardness.		Cold Water Aquatic Life

1. Applicable criteria for five of the metals are hardness dependent. See Fact Sheets for details.
2. Applicable criteria for ammonia are dependent upon pH and temperature. See the Fact Sheets for details.
3. Water Quality Criteria for the parameters copper, cyanide, mercury, selenium and zinc received a Jeopardy/Adverse Modification conclusion in the Idaho Toxics Biological Opinion (NMFS 2014 and USFWS 2015). Compliance with the Reasonable and Prudent Alternatives listed in the B.O. s are discussed in this BE in Section VIII.B.
4. There are also aquatic life criteria applicable to arsenic: an acute criterion of 340 µg/L and a chronic criterion of 150 µg/L. Since the human health criterion of 10 µg/L is an order of magnitude more stringent, it will be the basis for water quality-based controls on arsenic in nearly all cases.
5. The temperature criteria listed for Yankee Fork Creek are for the salmonid spawning use. These criteria apply in areas used for spawning and during the time spawning and incubation occurs. Otherwise, the cold water aquatic life criteria (19/22 °C) apply.

C. Permit Limits

NPDES permits can include both technology-based and water quality-based permit limits. Technology-based limits are based on section 301(b)(1)(A) and 301(b)(2) of the CWA and are designed to assure that all industries throughout the country install a baseline level of treatment for their wastewaters. Water quality-based limits are based on section 301(b)(1)(C) of the CWA and are intended to ensure that effluent from facilities do not adversely affect the waterbodies into which they discharge. The implementing regulations at 40 CFR 122.44(d) require that permits contain limits for all pollutants or parameters which "are or may be discharged at a level which will cause, have the reasonable potential to cause, or contribute to an excursion above any state water quality standard, including state narrative criteria for water quality." This permit uses technology-based limits for total suspended solids (TSS) only as the USEPA determined that the technology-based effluent limit is stringent enough to ensure compliance with water quality standards. All other effluent limits included in the permit are water quality-based.

Water quality-based limits: To determine whether water quality-based limits for a particular discharge are needed, USEPA follows guidance in its *Technical Support Document for Water Quality-based Toxics Control* (USEPA 1991) from here on referred to as the "TSD." USEPA evaluated the outfall 002 and 003 discharges to determine if "reasonable potential" to exceed the criteria in the receiving water exists. Effluent limits were developed for those pollutants where there was "reasonable potential". These pollutants are pH, metals (cadmium, copper, lead, mercury, and zinc), whole effluent toxicity (WET), and, for outfall 002, cyanide. Effluent limits are not needed for those parameters that did not exhibit reasonable potential. A description of the reasonable potential evaluation for the draft final permit is included in Appendix B of the 2016 Fact Sheet and in Appendix C of the 2015 Fact Sheet.

Although some constituents of the discharge may not have exhibited reasonable potential to exceed water quality standards, these still require scrutiny to ensure protection of listed species. These constituents are aluminum, arsenic, selenium, silver, ammonia, and nitrate + nitrite.

For those parameters that require water quality-based effluent limits, the effluent limits are developed by converting the applicable water quality criteria into permit limits. USEPA converts the criteria into effluent limits following the procedures in the TSD. Factors that influence the development of effluent limits include: effluent flow, receiving water critical low flows, effluent variability, water quality upstream of the discharge, and water hardness (for metals with hardness-dependent criteria). Reasonable worst case estimates of each of these factors were used to develop the effluent limits to ensure that they are protective of the water quality criteria under critical conditions. Each of these factors is discussed in the Fact Sheets.

The toxicity of certain metals is influenced by water hardness. Idaho's aquatic life criteria for several of the metals of concern are calculated as a function of hardness measured in mg/L as calcium carbonate ($CaCO_3$). As the hardness of the receiving water increases, the toxicity of the metals decreases and the numerical values of the criteria increase. For the purposes of calculating a conservative value for metals criteria, the USEPA uses the 5th percentile of measured hardness values. In the case of the Grouse Creek Unit, the hardness that is used to calculate the value of the water quality criteria for cadmium, copper, lead, silver, and zinc (for the purpose of

calculating the various effluent limits) varies to be consistent with the varying conditions of receiving water flow and dilution. For parameters with "concave down" criteria (i.e., the slope of the criteria, when plotted against hardness, decreases with increasing hardness) such as cadmium, copper, and zinc, the EPA used a calculated hardness value expected to occur at the edge of the mixing zone, based on conservative values (minimum or 5th percentile) for the measured effluent and upstream measured hardness and the dilution factor associated with the authorized mixing zone. For parameters with "concave up" criteria (i.e., the slope of the criteria, when plotted against hardness, increases with increasing hardness) such as lead and silver, the EPA used a conservative (minimum or 5th percentile) hardness measured downstream from the appropriate outfall. Refer to the 2015 Fact Sheet Appendix C and the 2016 Fact Sheet Appendix B for explanation of hardness calculations used to calculate the effluent limits.

Idaho water quality criteria for mercury and selenium are expressed as total recoverable and are independent of hardness. Arsenic criteria for the protection of aquatic life is also independent of hardness, and has a conversion factor of one (1), meaning that the dissolved and total recoverable criteria are the same. Water quality criteria for these parameters are the same for waters throughout Idaho unless a site-specific criterion is in effect.

Mixing zones

Under Idaho's water quality standards, mixing zones may be authorized for discharges to meet water quality standards. Mixing zones are limited areas or volumes of receiving water where wastewater mixes with the receiving water and where water quality standards may be exceeded. The dilution provided by mixing zones was factored into the development of effluent limits for some constituents of the GCU outfalls. Mixing zones are authorized for ammonia, arsenic, cadmium, copper, cyanide, lead, mercury, silver, selenium, WET and zinc for both outfalls (Table 4) (IDEQ 2015). Mixing zone dilution was also used for the determination of reasonable potential for nitrate + nitrite, for outfall 002.

Table 4. Authorized mixing zones by parameter and flow tier.

Parameter	Jordan Creek (002)				Yankee Fork (003)					
	Flow < 30 cfs		**Flow ≥ 30 cfs**		**Flow < 15 cfs**		**Flow ≥ 15 and < 45cfs**		**Flow ≥ 45 cfs**	
	Auth. Fraction of Critical Flow	Modeled Down-stream Distance (m)	Auth. Fraction of Critical Flow	Modeled Down-stream Distance (m)	Auth. Fraction of Critical Flow	Modeled Down-stream Distance (m)	Auth. Fraction of Critical Flow	Modeled Down-stream Distance (m)	Auth. Fraction of Critical Flow	Modeled Down-stream Distance (m)
Ammonia	25%	25	25%	24.7	25%	0.76	25%	0.99	25%	35.2
Arsenic	25%	25	25%	24.7	25%	0.76	25%	0.99	25%	35.2
Cadmium	25%	25	25%	24.7	9%	0.10	18%	0.53	19%	1.2
Copper	25%	25	5%	2.3	13%	0.21	25%	0.99	13%	0.6
Cyanide	25%	25	25%	24.7	25%	0.76	25%	0.99	25%	35.2
Lead	25%	25	25%	24.7	25%	0.76	25%	0.99	25%	35.2
Mercury	25%	25	25%	24.7	25%	0.76	25%	0.99	25%	35.2
Nitrate+ Nitrite	25%	25	25%	24.7	N/A	N/A	N/A	N/A	N/A	N/A
Selenium	25%	25	25%	24.7	25%	0.76	25%	0.99	25%	35.2
Silver	25%	25	25%	24.7	25%	0.76	25%	0.99	25%	35.2
WET[1]	50%	25	50%	24.9	75%	200	75%	341	75%	687
Zinc	25%	25	8%	24.1	23%	0.64	25%	0.99	25%	35.2

Notes:
1. For WET, the modeled downstream distance is the distance at which the receiving water concentration falls below 1.0 chronic toxic units (TUc), if the effluent WET is equal to the average monthly limit.

Sources: IDEQ 2015 and Nickel 2015.

Effluent limits

For outfall 002 (Table 5), two sets of effluent limits were developed to allow for variability of the effluent and receiving water (Jordan Creek) flows. For outfall 003 (Table 6), three sets of limits were developed, to allow for variability of the effluent and receiving water (Yankee Fork Creek) flows. In addition to changes in dilution ratios, the hardness varies with flow. Therefore, effluent limits for the flow tiers are different for those parameters with water quality criteria based on hardness. The bases for these effluent limits and how they were developed are detailed in the Fact Sheets.

The effluent limits are expressed in terms of concentration (e.g., µg/L). In order to ensure that total loadings to the receiving water are controlled, the permit also proposes effluent limits for flow, and, for outfall 002, the dilution ratio (ratio of receiving water flow to effluent flow) was also established as an effluent limit. Limits were added for whole effluent toxicity (WET) based on Idaho state water quality standards that require surface waters to be free from toxic substances in concentrations that impair designated or existing beneficial uses of the receiving water (IDAPA 58.01.02.200.02). Idaho's narrative water quality criterion for toxic substances was interpreted using recommendations in the TSD (Section 2.3.3).

Table 5. Effluent limits for outfall 002.

Effluent Limits for outfall 002								
Parameter and Units	**Draft Permit**				**2002 Permit**			
	Jordan Creek Flow < 30 CFS		Jordan Creek Flow ≥ 30 CFS		Jordan Creek Flow < 30 CFS		Jordan Creek Flow ≥ 30 CFS	
	Average Monthly Limit	Max. Daily Limit	Average Monthly Limit	Max. Daily Limit	Average Monthly Limit	Max. Daily Limit	Average Monthly Limit	Max. Daily Limit
Effluent Flow, CFS	—	5.57	—	5.57	No limits. Monitor and report only.			
Cadmium, total recoverable (TR), µg/L	1.44	2.72	1.32	2.50	3.7	7.5	2.2	4.4
Copper, TR, µg/L	18.6	41.9	14.9	33.5	14	35	5.6	14
Cyanide, weak acid dissociable (WAD), µg/L	7.47	21.3	7.47	21.3	21	47	21	47
Dilution Ratio	8:1 (minimum)				8:1 (minimum)			
Lead, TR, µg/L	1.80	4.84	0.84	2.28	9.5	19	4.0	8.1
Mercury, Total, µg/L	0.022	0.057	0.022	0.057	0.088	0.18	0.088	0.18
pH, standard units	6.5 – 9.0				6.5 – 9.0			
Selenium, µg/L	No limits. Monitor and report only.				No limits. Monitor and report only.			
Silver, TR	No limits. Monitor and report only.				1.8	3.6	0.60	1.1
Total Suspended Solids (TSS), mg/L	20	30	20	30	20	30	20	30
Zinc, TR, µg/L	141	304	107	230	110	250	50	110
WET, chronic, TUc	3.3	9.2	3.3	9.2	9.8	16	9.8	16

Note that some permit limits in outfall 002 for copper and zinc are less restricted than in the past permit. The draft permit also proposes to delete the prior permit's effluent limits for silver. One reason for this is that the prior permit limits were based on limited receiving stream water quality data where USEPA assumed a conservative value for water hardness. Hardness is a measure of the concentration of divalent metal cations (mostly calcium and magnesium) in the water. Some metals are less toxic to aquatic life in hard water than in soft water, therefore, the water quality criteria become less stringent (i.e. numerically greater) in harder waters. Currently, there are five years of water quality data indicating that the effluent water is considerably harder than the receiving stream water than had to be assumed for the first permit. Thus, the toxicity of some metals (e.g. copper) is reduced because of the effect of higher water hardness, allowing for higher end-of-pipe concentrations. In addition, effluent concentrations of silver have declined over time due to the cleanup and closure of the site, such that the discharges no longer have the reasonable potential to cause or contribute to excursions above water quality standards for silver. The revised and deleted effluent limits are consistent with the State of Idaho's antidegradation policy and implementation methods (IDAPA 58.01.02.051 – 052).

Table 6. Effluent Limits for outfall 003.

Effluent Limits for outfall 003						
Parameter and Units	**Yankee Fork Creek Flow < 15 CFS**		**Yankee Fork Creek Flow ≥ 15 and < 45 CFS**		**Yankee Fork Creek Flow ≥ 45 CFS**	
	Average Monthly Limit	Max. Daily Limit	Average Monthly Limit	Max. Daily Limit	Average Monthly Limit	Max. Daily Limit
Flow (CFS)	—	0.668	—	1.11	—	2.01
Cadmium, TR, µg/L	2.22	4.08	2.50	4.59	2.96	5.42
Copper, TR, µg/L	21.6	39.8	21.8	40.3	20.8	38.5
Lead, TR, µg/L	1.40	4.84	0.75	2.60	0.96	3.32
Mercury, total, µg/L	0.026	0.053	0.025	0.050	0.035	0.069
pH, standard units	6.5 – 9.0 at all times					
TSS	20	30	20	30	20	30
Zinc, TR, µg/L	158	344	147	319	167	364
WET, chronic, TUc	10	20	9.1	18	15	29

III. SPECIES DESCRIPTIONS-INVENTORIES AND SURVEYS

This section describes the threatened and endangered species that occur in the Action Area (Jordan Creek, Yankee Fork, of the upper Salmon River). The life history and distribution of each species within the action area are discussed. The information is based on sampling conducted by US Forest Service (USFS), Hecla Inc., Idaho Department of Fish and Game (IDFG), and the Shoshone-Bannock Tribe.

Much of the relevant monitoring data used in this BE were collected from established stream locations listed in Table 7. These will be referred to throughout this BE. These sites are shown on Map 1.

Table 7. Stream monitoring locations.

Site name	Location	Location relative to outfalls	Relevance to BE
S-6 (old site F4)	Jordan Creek: Just below Grouse Cr. Confluence	Approx. 8 km upstream of outfall 002	Primary upstream monitoring site, outfall 002. This is the background surface water site as it is located above any site activity
S-4	Jordan Creek: 0.15km downstream of Washout Creek confluence	0.3 km downstream of outfall 002	Primary downstream monitoring site, outfall 002
S-9	Yankee Fork: 0.5km upstream of Jordan Cr. confluence	1.0 km upstream of outfall 003	Primary upstream monitoring site, outfall 003. This is the background surface water site for outfall 003
S-10	Yankee Fork: 1.5km downstream of Jordan Cr. confluence	1.0 km downstream of outfall 003	Primary downstream monitoring site, outfall 003

A. Available Fish Monitoring Data

Jordan Creek

Fish have been sampled in Jordan Creek sporadically in the 1980s and 1990s and more consistently through the 2000s. In the 1980s, fish abundance data was collected by Hydrometrics and Ralston and Associates for the Sunbeam Mining Company (Hydrometrics 1981, Ralston and Associates 1984 and 1989 as stated in USFS 1992). In the 1990s, October sampling was conducted in Jordan Creek by Ralston and Associates (1993) and Chadwick Ecological Consultants (CEC) (1998-2000). In 1999, the sites were also sampled in August using electrofishing and snorkeling gear. Snorkeling data were collected by the Shoshone-Bannock Tribe 1996-2000 at six locations including reaches both above and below outfall 002 (Unpublished Data, Shoshone-Bannock Tribe). The most consistent fish sampling data are from the post mining era. Late summer surveys were conducted annually from the Jordan Creek upstream reference site S-6, located below the Grouse Creek confluence (located approximately 8 km upstream of outfall 002) and the downstream reference site S-4, located below Washout Creek confluence (located 0.3 km downstream of outfall 002). These data were collected by GEI using electrofishing (GEI consultants Inc. 2014b) for Hecla Mining Company. Monitoring of Chinook redds and electrofishing for detection of juvenile salmonid use in Jordan Creek has been conducted in some years by the Shoshone Bannock Tribe.

Yankee Fork

Yankee Fork is within the Upper Salmon River Major Population Group (MPG) of the Snake River Spring/Summer Chinook ESU. This population is being monitored for adult and juvenile productivity, spatial distribution and species diversity (NMFS 2009). The IDFG has been conducting redd surveys since 1957. The SBT conducts electrofishing surveys and intensive snorkel surveys (since 1982) for juvenile production estimates as well as redd surveys for adult abundance. Bull trout monitoring has also been conducted by the USFS in some years. Data relevant to each listed species is discussed in the following sections.

B. Bull Trout (*Salvelinus confluentus*) Threatened

Bull trout are a char species, unique to western North America (Cavender 1978). Bull trout populations are known to exhibit two distinct life history forms: 1) resident bull trout that spend their entire life cycle in the same (or nearby) streams in which they emerged, and 2) migratory bull trout which can exhibit either a fluvial life history—spawning in tributary streams where the young rear from one to four years before migrating to a river, or an adfluvial form—spawning in tributary streams where the young rear before migrating to a lake (Fraley and Shepard 1989).

Bull trout generally mature at between 5 and 7 years of age (Fraley and Shepard 1989; Goetz 1989; Leathe and Enk 1985). Spawning occurs from August through November (Armstrong and Murrow 1980; Brown 1994; McPhail and Murray 1979). Embryos incubate over winter and hatch in late winter or early spring (Weaver and White 1985). Emergence has been observed over a relatively short period of time after a peak in stream discharge from early April through May (Rieman and McIntyre 1993).

Bull trout growth, survival and long-term population persistence are correlated with stream habitat conditions such as cover, channel stability, substrate composition, temperature, and migratory corridors (Rieman and McIntyre 1993). These habitat features are often impaired as the result of land management activities such as forest harvest, road building, hydropower development, irrigation diversions, and grazing. Mining has altered stream channel morphology, increased sediment transport and deposition, decreased vegetative cover, and contributed to acidic water discharge and heavy metal water pollution (Chapman et al. 1991). Mining has increased potential for chemical/petroleum spills from transportation, water use/withdrawals, and the discharge of contaminated effluent to streams (USFWS 2002).

Bull trout of Jordan Creek and the Yankee Fork are within the Upper Salmon River Core Area, of the Snake River Bull Trout Recovery Unit of the USFWS Recovery Plan (USFWS 2002). The Bull Trout Recovery plan cites livestock degrading riparian areas, widespread irrigation withdrawal/diversion and roading in valley bottoms and historic mining/logging roads as significant human disturbance in this recovery unit. The plan also describes surface mining in the Salmon River basin as widespread and impacts to tributary streams are considered significant by USFWS (USFWS 2001 as cited in USFWS 2002). In the Upper Salmon River Core Area, historical patented mining and associated roads continue to deliver sediment to upper Salmon River headwater streams (Overton et al. 1999). Historic dredge mining has left unconsolidated dredge tailings in the lower Yankee Fork River (USRITAT 1998; Overton et al. 1999).

Declines in bull trout have been associated with introductions of nonnative fish such as brook trout (Rieman and McIntyre 1993). In the Salmon River basin, however, the decline in bull trout abundance accompanied by an increase in rainbow trout abundance is likely due to high stream temperatures and other aspects of habitat degradation selecting against bull trout rather than interspecific competition from rainbow trout (USFWS 2002). In the Upper Salmon River these hybrids have been found in Valley Creek. Competitive and predator-prey relations among bull trout, rainbow trout (*Oncorhynchus mykiss*), and cutthroat trout (*O. clarki*) and the resultant effects on bull trout populations in the recovery unit are not specifically known.

Critical habitat was designated for all bull trout subpopulations. Approximately 18,975 miles of streams and 488,252 acres of lakes and reservoirs in Idaho, Oregon, Washington, Montana and Nevada are designated as Critical Habitat for this species.

Bull trout of Jordan Creek

Jordan Creek supports a resident population of bull trout, and is designated as feeding, migrating, and overwintering bull trout critical habitat in the action area (USFWS 2010). This designated feeding, migration and overwintering critical habitat extends from the Jordan Creek/Yankee Fork confluence upstream 6.4 km (4.0 mi.) (see pg. 780 of USFWS 2010). In 1985, the density of age 1+ and older bull trout was estimated as >1 per m^2 of pool habitat (Konopacky et al. 1986). Based on sampling 1981-2013, bull trout are not abundant in Jordan Creek and their distribution is limited to the lower reaches of the stream. During annual electrofishing sampling 1997-2013, the capture of bull trout has continued to be infrequent and differences between captures above and below outfall 002 are not substantial (Table 8). The bull trout present are either juveniles or resident adults with size ranging from 88 to 282mm (GEI 2014b). The USFS estimated bull trout density in Jordan Creek as 0.9 fish per $100m^2$ during baseline monitoring (USFS 2004).

Table 8. Jordan Creek fish monitoring data 1997-2014 (GEI 2014b, GEI 2015).

	Site S-6: (above outfall 002)[1]			Site S-4 (below outfall 002)[2]			
Year	**bull trout**	**shorthead sculpin**	**cutthroat/ rainbow hybrid**	**bull trout**	**shorthead sculpin**	**cutthroat/ rainbow hybrid**	**Stocked rainbow**
1997	0	10	8	0	110	11	0
1998	0	38	30	0	163	35	0
1999	0	16	31	0	67	16	0
2000	1	58	30	0	42	16	0
2001	2	24	42	3	38	28	0
2002	0	69	64	1	130	33	3
2003	0	36	45	0	129	29	6
2004	2	43	49	0	187	21	0
2005	2	27	44	2	60	59	0
2006	1	14	36	0	58	37	0
2007	0	63	46	2	67	15	4
2008	1	65	59	0	198	34	0
2009	0	29	62	1	194	98	0
2010	1	62	108	0	115	28	0
2011	0	41	65	0	202	42	0
2012	1	38	207	3	222	24	0
2013	5	63	214	1	247	37	0
2014	1	61	38	2	164	24	0

1=S-6 located above outfall 002 near Grouse Creek confluence.
2=S-4 located below outfall 002, just below Washout Creek confluence

Bull trout of the Yankee Fork

Fluvial bull trout are distributed in the Salmon River. Yankee Fork from its confluence with the Salmon River upstream 14.4 km (9.0 mi) to its confluence with Jordan Creek is designated as feeding, migrating, and overwintering bull trout critical habitat in the action area (USFWS 2010). It is believed that the reach of the Salmon River near the Yankee Fork is used only sporadically for spawning with fluvial bull trout migrating through this reach to spawning habitat in tributary streams (USFS, Tom Curet, IDFG, Pers. Com. 1999). Also fluvial and adfluvial bull trout migrate through the reach of the Salmon River near the confluence of the Yankee Fork on their way to upstream lentic and lotic spawning areas. Monitoring data from the USFS show use of the Yankee Fork by bull trout during all three years that surveys were conducted (USFS 2004) (Figure 1). More recent yet limited data on bull trout abundance are available. The Salmon – Challis National Forest has monitoring sites for bull trout. One of these sites, McKay Creek site is located in the Yankee Fork watershed. Estimated bull trout densities at this site are as follows: 4.1, 3.8, 2.3, 6.0 fish/100m^2 for years 2007-2010 (Bureau of Recreation 2012).

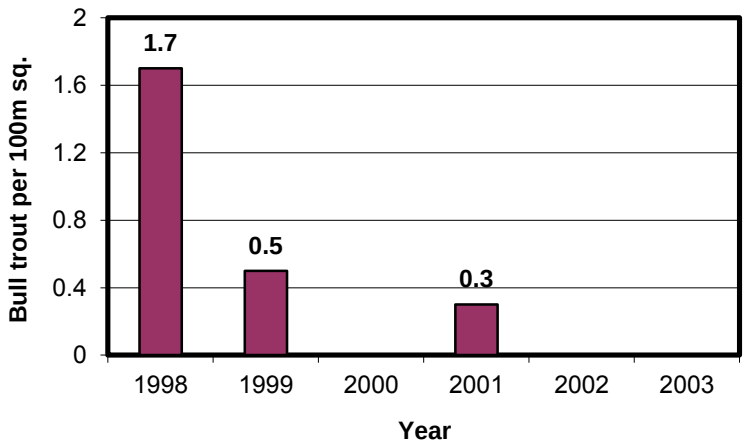


Figure 1. Bull trout abundance estimates for the Yankee Fork from electrofishing surveys conducted 1998, 1999, and 2001 (USFS 2004).

Bull Trout Biological Requirements in the Action Area

The biological requirements of the Action Area related to listed species are those physical or biological features that are essential to conservation of the species. NMFS-FWS regulations state that federal agencies must consider those physical and biological features that are essential to the conservation of a given species (FR vol.71, no.229, 69060). These features of Critical Habitat are called Primary Constituent Elements (PCEs) and are described by NMFS-USFWS for each listed fish species. These are from the 2010 rule (75 FR 63973, October 18, 2010):

1. Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia.

2. Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats, including, but not limited to, permanent, partial, intermittent, or seasonal barriers.

3. An abundant food base, including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish.

4. Complex river, stream, lake, reservoir, and marine shoreline aquatic environments and processes that establish and maintain these aquatic environments, with features such as large wood, side channels, pools, undercut banks and unembedded substrates, to provide a variety of depths, gradients, velocities, and structure.

5. Water temperatures ranging from 2 to 15 °C (36 to 59 °F), with adequate thermal refugia available for temperatures that exceed the upper end of this range. Specific temperatures within this range will depend on bull trout life-history stage and form; geography; elevation; diurnal and seasonal variation; shading, such as that provided by riparian habitat; streamflow; and local groundwater influence.

6. In spawning and rearing areas, substrate of sufficient amount, size, and composition to ensure

success of egg and embryo overwinter survival, fry emergence, and young-of-the-year and juvenile survival. A minimal amount of fine sediment, generally ranging in size from silt to coarse sand, embedded in larger substrates, is characteristic of these conditions. The size and amounts of fine sediment suitable to bull trout will likely vary from system to system.

7. A natural hydrograph, including peak, high, low, and base flows within historic and seasonal ranges or, if flows are controlled, minimal flow departures from a natural hydrograph.

8. Sufficient water quality and quantity such that normal reproduction, growth, and survival are not inhibited.

9. Sufficiently low levels of occurrence of nonnative predatory (e.g., lake trout, walleye, northern pike, smallmouth bass); interbreeding (e.g., brook trout); or competing (e.g., brown trout) species that, if present, are adequately temporally and spatially isolated from bull trout.

C. Snake River Sockeye Salmon (*Oncorhynchus nerka*)-Endangered

Snake River sockeye population uses the mainstem Salmon River as a migration corridor between freshwater spawning and rearing areas in the Stanley basin of the Sawtooth National Forest and saltwater feeding grounds. This population is distinctive because of its long freshwater migration (approximately 1500km) and high elevation spawning (2000m) relative to other sockeye populations. Both resident and anadromous forms reside in Redfish Lake. Sockeye spawn along the lake's shoals in October and November while, resident sockeye spawn in a tributary of Redfish Lake during August and September (NMFS 1991).

Juveniles rear in a lake for one or two years and smolts out-migrate in spring from April through June. Ocean residency is two to three years. Juvenile and adult migration corridors include the lakes above inlets and outlet creeks, Alturas Lake Creek, Valley Creek between Stanley Lake Creek and the Salmon River, the main fork of the Salmon River, the Snake River, and the Columbia River to the Pacific Ocean (NMFS 1993).

The designated habitat for Snake River sockeye salmon consists of reaches of the Columbia, Snake, and Salmon rivers, Alturas Lake Creek and Valley Creek and Stanley, Redfish, Yellow Belly, Pettit, and Alturas lakes (including their inlet and outlet creeks). The mainstem Salmon River is used by migrating adults and by smolts outmigrating to the ocean.

Snake River sockeye are considered in this BE because the portion of the Salmon River that drains the Yankee Fork watershed is within the range of sockeye. This river reach is used as a migration corridor for spawning adult and outmigrating juvenile sockeye. However, this species does not use Jordan Creek or the Yankee Fork at any life history stage (USFS 1992).

D. Snake River Chinook Salmon (*Oncorhynchus tshawytscha*) - Threatened

The Snake River basin supports the Snake River spring/summer Chinook salmon ESU and the Snake River fall Chinook salmon ESU. Based on NOAA ESU distribution maps (available at http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/esa/chin

ook/web_pdfs_srfr_chinook.pdf) Snake River fall Chinook salmon do not use the upper Salmon River above the Little South Fork Salmon River confluence and are not addressed in this BE. The Yankee Fork population is a spring run and is one of eight extant populations in the Upper Salmon River Major Population Group (MPG).

Migrating adult spring Chinook salmon enter the Columbia River between February and May, and summer Chinook enter in June and July (Bevan et al. 1994). Both spring and summer Chinook spawn in high elevation tributaries from August through September and juveniles rear in streams for one year before migrating to the ocean in the spring (April through June). Ocean residency varies but is generally one to four years. In the Yankee Fork, adults stage July through mid-August and spawning peaks in mid-August. The peak emergence period of fry is mid-March in most years.

Critical habitat, which includes all river reaches presently or historically accessible, has been designated for this threatened species by the NMFS (58 Fed. Reg. 68543). These reaches are the Columbia, Snake, and Salmon rivers and all tributaries of the Snake and Salmon rivers except the Clearwater River. Areas not included as critical habitat for Snake River spring/summer Chinook are those reaches above impassable natural falls and Dworshak and Hells Canyon dams.

Historically, the Yankee Fork drainage produced large runs of Chinook (NMFS 1997). These fish were an important subsistence fishery and are a ceremonial fish for the Shoshone-Bannock Indian Tribe as well as a sport fishery. Spawning is distributed broadly through the available stream reaches, extending from about one mile upstream of the Yankee Fork /Salmon River confluence to the headwaters area and the West Fork, Yankee Fork (ICTRT 2007).

The IDFG has conducted annual spawner index counts since 1957 in the Yankee Fork. In the 1980s and 90s, the number of fish produced in the Yankee Fork watershed has been dramatically reduced (NMFS 1997). In the 1960s through the early 1970s, redd counts averaged 400 and exceeded 600 in the late 1960s (Konopacky et al. 1986). In the 1980s, counts were reduced to approximately 10 redds. Redd counts remained low in the Yankee Fork through the 2000's Chinook abundance was considered to be variable (ICTRT 2007) with a 10-year geometric mean number of natural spawners was 13 and smolt to adult ratio (SAR) for the 20 year period (1978-1997) has a geometric mean of 0.80 (ICTRT 2007). Recent data are also variable and do not shift the trend from a low and variable condition for Yankee Fork. Data from the Yankee Fork red surveys from mile 9.0 to mile 21.1 are shown in
Figure 2. No data were reported for 2005, 2010, 2012, and 2013 (
Figure 2).

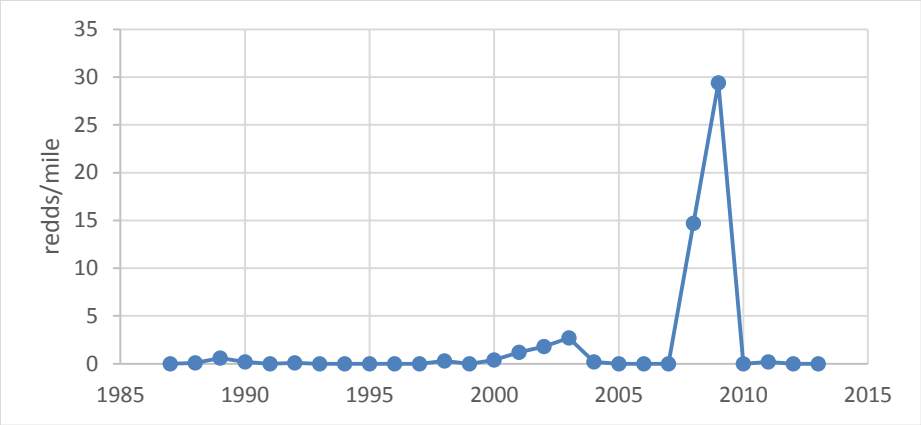


Figure 2. Spring Chinook redds survey data by IDFG, Yankee Fork 1987-2013 (Source: Streamnet 2015, query #44212).

In Jordan Creek, observations of Chinook have been rare and sporadic. Chinook have not been captured at any of the sample locations during the electrofishing surveys conducted 1981-2000 by Ralston and Associates (see Appendix E). In 1984, 1985, and 1988, the Shoshone-Bannock Tribe observed young-of- year Chinook in the lower reaches of Jordan Creek (Konopacky et al. 1986, USFS 1992). In 1998, four young-of-year Chinook (50-80mm) were observed in Jordan Creek during snorkel surveys 200m above the Yankee Fork confluence (Shoshone-Bannock Tribe, Unpublished data). Documentation of Chinook spawning in Jordan Creek has not been found and IDFG does not perform redd counts in this drainage (Pers. Comm. Tom Curet, IDFG, Dec. 2000). It is likely that juveniles hatched or stocked in the Yankee Fork move to Jordan Creek to exploit available rearing habitat. Based on current accessibility and stream gradient (<4%), the lower six miles of Jordan Creek are considered habitat for rearing Chinook (USFS 1994) (see also Section IV. of this BE). Therefore, it is possible that the mixing zone of outfall 002 is within the range of rearing Chinook.

Habitat features for spring/summer Chinook and summer steelhead consists of (1) spawning and juvenile rearing areas; (2) juvenile migration corridors; (3) areas for growth and development to adulthood, and (4) adult migration corridors (58 FR 68543). Essential features of these habitats include adequate substrate (especially spawning gravel), water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and suitable migration conditions. Primary constituent elements (PCEs) consist of the physical and biological elements identified as essential to support one or more life stages of salmon or steelhead, and therefore are essential to the conservation of the species (NMFS 2005). Discharge from the outfalls is relevant to the PCEs of water quality and substrate. These are used in this BE to evaluate critical habitat for Snake River spring Chinook.

E. Snake River Steelhead (*Oncorhynchus mykiss*) - Threatened

Steelhead have the most complex life histories of any Pacific salmon species. These fish have variable run timing and degree of anadromy and are capable of more than one spawning cycle. In the Snake River subbasin, steelhead are 'stream-maturing' as they enter freshwater in a sexually immature state and require several months in freshwater before they mature then spawn. These steelhead are referred to as 'summer run' based on the time that they enter freshwater. Snake

River steelhead enter freshwater from June to October and spawn the following spring from March to June (Thurow 1987). Snake River steelhead comprise two groups, an A and a B-run, based on migration timing, ocean-age and adult size. A-run steelhead are thought to be predominately age-1-ocean, while B-run steelhead are thought to be predominately age-2-ocean (Busby et al. 1996). Emergence occurs by early June in low elevation streams and as late as mid-July at higher elevations.

In the upper Salmon River basin, which includes the waters of the Action Area, only A-run summer steelhead are present. Adults begin freshwater migration in the summer and overwinter in the Salmon River mainstem. During spring runoff, they continue their upstream migration and then spawn. Eggs incubate for 1.5 to 4 months, and time of emergence ranges from mid-May through July. After spending typically two winters in freshwater, juveniles migrate to the ocean during spring runoff. Snake River steelhead usually smolt at age-2 or age-3. Steelhead typically reside in marine waters for 1 to 3 years prior to returning to their natal stream to spawn at 4 or 5 years of age. Although steelhead are iteroparous, it is rare for steelhead to spawn more than twice before dying and most that do so are females (NMFS 1999). Like Snake River spring/summer Chinook salmon, Snake River steelhead are vulnerable to small-scale habitat changes due to their long freshwater residence.

Steelhead can have various life histories in terms of the degree of anadromy. The anadromous form that migrates between the ocean and freshwater are termed 'steelhead', while the non-anadromous or 'resident' form does not migrate and is called 'rainbow trout.' Like steelhead, rainbow trout spawn in winter/spring and emerge in spring/early summer. Both anadromous and non-anadromous forms commonly co-occur in streams of the upper Snake River basin. Although both forms are classified as the same species taxonomically, the relationship of the two forms in a given area is typically unclear. The migratory and resident forms of this species may be eco-phenotypes within a common gene pool or they may be distinct due to reproductive isolation (Zimmerman and Reeves 2000). Without specific information on the genetic relation of Jordan Creek steelhead and rainbow trout and on the degree of their reproductive isolation (e.g. difference in spawning timing, differences in habitat selection), it is not possible to distinguish between small steelhead and rainbow trout.

Snake River steelhead are known to spawn and rear in all tributaries used by Snake River spring/summer Chinook salmon. Steelhead use many additional tributaries, some of which are steeper and much smaller than those used by spring/summer Chinook salmon. The Yankee Fork is well known spawning and rearing habitat for steelhead (SBT 2008). Critical Habitat has been designated for O. mykiss of the Snake River by NMFS (NMFS 2000). The Critical Habitat area includes all river reaches accessible to listed steelhead in the Snake River and its tributaries. This designation includes the Yankee Fork and Jordan Creek, which are within the Action Area. The PCEs for Snake River steelhead are similar to those of the other listed salmonids (50 CFR 226.212). These salmonids require high water quality at all life phases. In Jordan Creek, hybrid trout are present in both upper reaches and downstream of the Washout Creek confluence (Figure 3). Steelhead/rainbow trout have been sampled below outfall 002 several times over the 1981-2006 monitoring period (see Appendix A). Steelhead/rainbow trout fish are not abundant, with the number of fish captured during electrofishing surveys ranging from 1-18 fish in each of the downstream sample reaches (Figure 3). Above outfall 002, 1-4 steelhead individuals have been

captured over the monitoring period (above Grouse Creek confluence). Snorkel surveys obtained similar results (see Appendix A). Age 1+/2+ steelhead were observed in 1996-2000 except for 1999 (Unpublished data, Shoshone-Bannock Tribe). In April of 2007, adult steelhead were observed spawning in Jordan Creek approximately two miles from the confluence with the Yankee Fork (USFS 2007). These data confirm use of the Action Area by steelhead/rainbow trout. Finally, use of the upper Salmon River Mainstem by migrating and spawning adults and out-migrating juveniles is well known (Figure 4). Currently the Shoshone-Bannock tribe is using hatchery production to re-introduce and recover native steelhead to 'critically under-seeded habitats' within the Yankee Fork until remaining populations can become self-sustaining through natural reproduction.

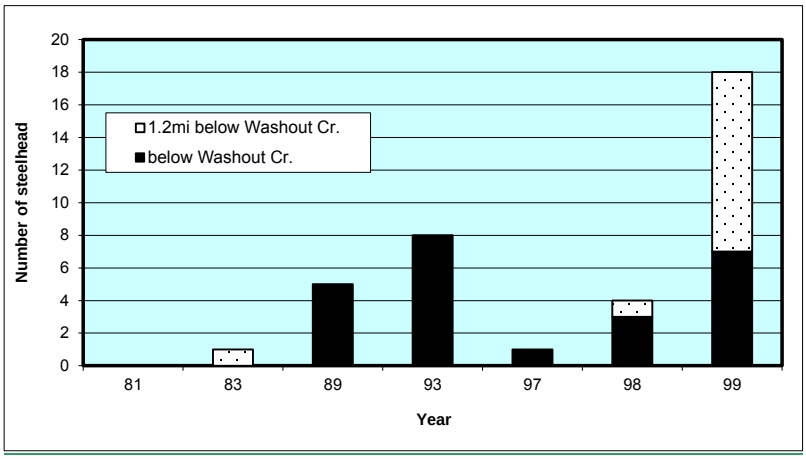


Figure 3. Steelhead captures at two Jordan Creek locations (Source: USFS 1992, CEC 1999b).

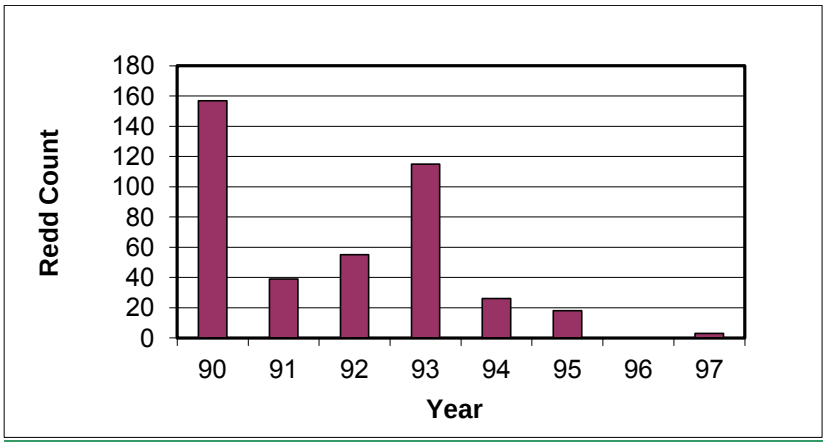


Figure 4. Steelhead redd counts in the Salmon River between Pahsimeroi River and Warm Springs Creek confluences 1990-1997 (Source: Streamnet database).

IV. ENVIRONMENTAL BASELINE

A. Define Action Area

The Action Area is within the Yankee Fork subwatershed, and consists of the stream reaches of Jordan Creek from outfall 002 downstream, the Yankee Fork of the Salmon River below the

confluence of Jordan Creek to the confluence of the Yankee Fork with the Salmon River (See Map 1).

B. Water Quality 303b List

A water quality limited segment is any waterbody, or definable portion of a waterbody, where it is known that water quality does not meet applicable water quality standards, and/or is not expected to meet applicable water quality standards. In accordance with section 303(d) of the Clean Water Act, States must identify waters not achieving water quality standards in spite of the application of technology-based controls in National Pollutant Discharge Elimination System (NPDES) permits for point sources. Such waterbodies are known as water quality limited segments (WQLSs), and the list of such waterbodies is called the "303(d) list." Once a water body is identified as a WQLS, the States are required under the Clean Water Act to develop a total maximum daily load (TMDL). A TMDL is a determination of the amount of a pollutant, or property of a pollutant, from point, nonpoint, and natural background sources (including a margin of safety) that may be discharged to a water body without causing the water body to exceed the water quality criterion for that pollutant.

Streams of the Action Area are in the Upper Salmon HUC #17060201. In Idaho's 2012 303(d)/305(b) integrated report, the segments of Jordan Creek and Yankee Fork Creek that receive discharges from the Grouse Creek mine were fully supporting their designated uses (IDEQ 2014, see interactive map at https://mapcase.deq.idaho.gov/wq2012/). Note, in previous Idaho 303(d)/305(b) integrated reports the Yankee Fork received a Category 4c designating this waterbody as impaired due to physical habitat alterations, but has now been delisted.

C. Receiving Water Chemical Concentrations

Monitoring data are available for sites both upstream and downstream of the two outfalls. Summary statistics for available parameters, both conventional and toxics, are in Appendix B. Summary of water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), **Site S-9 (Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003).** The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) and temperature is collected daily at outfalls 002 and 003 when discharging.

Appendix B. These data are used as supporting data for the effluent evaluation and its effects to listed species addressed later in this BE.

D. Conditions Related to Historical Mining

This area has a long history of both hard rock and placer mining beginning in the 1870s for gold, silver, lead, and zinc. Alluvial areas of Yankee fork from Polecamp Flat (several miles downstream of Jordan Creek confluence) to about 1.5 mile of above the Jordan Creek confluence and lower one mile of Jordan Creek were intensively dredged for gold (Frost and Box 2009). Numerous hard rock mines operated in the area.

Historic placer mining of the alluvial reaches of the action area resulting in the clearing of riparian vegetation and piling of dredged material along the banks. These activities have caused alteration of channel floodplain interactions including disconnecting tributaries, increasing flow velocities, altering channel confinement, and reducing function of riparian canopy. Aquatic habitat has been negatively impacted by reducing high water refuge areas, reducing salmonid spawning habitat, and reducing off-channel habitat.

Historic hardrock mining in the area has also resulted in a long-term signal of Se and Hg in stream sediments. A study of the stream sediment geochemisty of the Yankee Fork drainages conducted by USGS (Frost and Box 2009) concluded that historic hard rock mining has enriched the levels of metals (Hg, Se, As, Cu, Pb) in stream sediments of the Yankee Fork. Selenium is a common element in mineral deposits that were mined and the high Se content they found in the Yankee Fork sediments are likely a result of the high background levels in these mineral veins. Frost and Box (2009) concluded that the disturbance from mining has increased the amount of Se-containing rock, thus exposing it to weathering, resulting in high concentrations of Se in the Yankee Fork waters. Historic mining likely resulted in numerous releases of Hg, since it is used in the processing to recover silver. Stream areas in proximity to these early mill locations show spikes of Hg in stream sediment.

E. Habitat Characteristics of the Receiving Waters

The following description is based on information from Forest Service habitat surveys and information in the Yankee Fork Watershed BA (USFS 2006). Generally, the Lower Yankee Fork subwatershed area from the confluence with Jordan Creek to the mouth (confluence with the Salmon River) consists of deeply dissected and youthful topography. Soil development in the area is generally poor due to the youthful nature of the topography and the cool, arid climate. The granitic soils of the area tend to be thin, very permeable, nutrient poor, and acidic. Organic soils in the project area are rare. Ridges reach concordant levels at 9,000 to 10,000 feet above sea level.

Jordan Creek habitat

Jordan Creek is located in a deep, narrow, V-shaped valley characterized by shallow soil with steep topography and sparse vegetation. This small second-order stream flows north to south through a moderate-gradient canyon. The headwaters are at 2,755 m (9039 ft.) elevation and the stream drains into the Yankee Fork, 12.7 km downstream at an elevation of 1,945 m (6,380 ft.). Stream gradient ranges from 13% in the headwaters to 1% (mean 3%) (CEC 1999a, USFS 1992). Late season low flows were 10.71 cfs and 2.86 cfs in 1984 and 1985, respectively (USFS 1992). Stream temperatures are cold. In 1985, USFS recorded temperatures of 0.5 to 14.4 °C during August through mid-September (Konopacky et al. 1986).

Jordan Creek width is 3-8 m in both the upper and lower reaches (August 1981 data). Average pool depth is approximately 0.6 m and although pool habitat is very rare (USFS 1992), upper Jordan Creek has a well-defined riffle-pool sequence with substrate composed of gravel, cobble, and boulders. The lower 1.5 miles was used for dredge mining in the past and has substrate consisting of tailings (unconsolidated cobbles). General physical characteristics are provided in Table 9. Based on habitat surveys and other data, the USFS (1994) rated the quality of physical habitat features in Jordan Creek as summarized in Table 10. Although the habitat of Jordan Creek is in generally poor condition, there are no apparent physical habitat conditions that would preclude fish species from using the habitat in the vicinity of outfall 002.

Table 9. Physical characteristics of various sites of Jordan Creek (Source CEC 2000a).

Location	**% gradient**	**Substrate**	**Channel Form**	**Channel Condition**	**In channel Mining**
near Grouse Creek	4.6 - 3.5	bedrock/cobble	pool/riffle	mature/stable	no
1.6 km below Grouse Creek	3.3	cobble/boulder	pool/riffle	mature/stable	no
0.2 km above outfall 002	3.9	cobble/boulder	—	mature/stable	no
0.3 km below outfall 002	4.6	cobble/boulder	—	mature/stable	yes
0.6 km below outfall 002	2.7	cobble/boulder	—	mature/stable	yes
2.3 km below outfall 002	2	cobble/boulder	riffle dominated	unstable	yes
4.7 km below outfall 002	2.7	cobble/boulder	riffle dominated	unstable	yes

Table 10. Habitat condition ratings in Jordan Creek from 1994 survey data (USFS 1994).

Habitat Feature	**Rating**	**Comment**
water temperature	good	<14°C based on limited data. High altitude stream expected to be cool. Modified riparian zone, road likely have reduced stream shading.
instream cover	poor	All cover sources except for some large sized substrate (pools, LWD, bank associated cover) are low. Pocket water and turbidity are considered the most common cover type.
riparian vegetation	poor	Dredge spoils and stream adjacent roads have reduced riparian vegetation as well as reducing its interaction with the stream. Reduced riparian functions are: contribution to bank stability, moderation of stream temperature, providing cover (overhanging vegetation, roots, and undercuts).
pool frequency	poor	Pools with sufficient depth and cover to provide habitat diversity are extremely rare.
width/depth ratio	poor	Abundant coarse unconsolidated material from past instream mining has reduce depth. There is probably Insufficient stream power and lack of large roughness elements for the stream to work this material into a sequence of deeper pools and shallower riffles
bank stability	poor	Eroding cutbanks contribute fine sediment.
fine sediment	poor	Bank instability, surface erosion from roads and episodic mass wasting contribute fine sediment to stream. Generally low except during storm events.

Yankee Fork habitat

Stream Flow and Hydrology: Higher flows (>45cfs) occur during spring runoff in April, peaking during May and June (
Figure 5). Flows decline through the summer with mean flow <45 cfs in August. Base flow lasts from late summer through February. Flows during spring runoff periods may be 20 times the base flows. Stream flows can vary daily as precipitation and climatic conditions change, especially as a result from high intensity storm activity. These periodic storms in the Yankee Fork drainage can lead to severe turbidity and related mud/debris flows, especially noted in recent fire-burned areas.

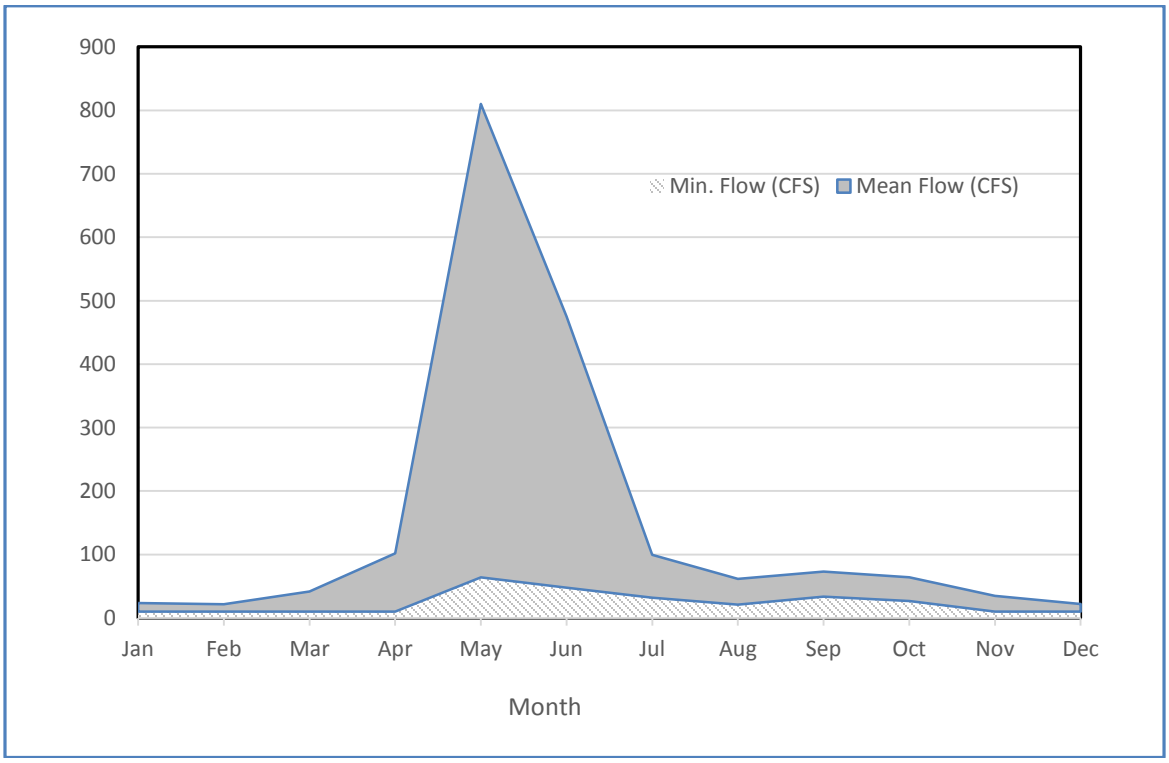


Figure 5. Yankee Fork mean monthly flows based on 2003-2008 data (Source Brian Nickel, USEPA).

Stream gradients in the Yankee Fork watershed range from <1 to >8 %. Channel types in the dredged portion of the Yankee Fork, downstream of Jordan Creek confluence, are classified as Rosgen C (low gradient, meandering, point-bar, riffle/pool, alluvial channels with broad well-defined floodplain) based on Rosgen (1994). The un-dredged portion from Polecamp Creek (located approximately 6 miles downstream of Jordan Creek) to the mouth of Yankee Fork, it is classified as Rosgen B (i.e., moderately entrenched, moderate gradient, riffle dominated channel, with infrequently spaced pools, stable profile and banks).

Mean width to maximum depth ratios in the dredged portion of the Yankee Fork (below Jordan Creek Confluence) ranged from 16.1 to 18.7 (Forest Service, 2002). The PACFISH RMO ratio is <10, indicating that this habitat element is a limiting factor in this area. Farther downstream from

Polecamp Creek to the mouth, ratios ranged from 10.3 to 10.8. In the Yankee Fork upstream of Jordan Creek, the width/depth ratios are generally within PACFISH standards. In the dredged portion of the Yankee Fork, approximately 16 pools per mile were recorded. The PACFISH RMO for this stream size is approximately 56 pools per mile. In the un-dredged portions of the river approximately 41 pools/mile were recorded (Forest Service 2002).

Stream bank stability in the Yankee Fork downstream of Jordan Creek was evaluated by the USFS (2002). For each habitat type, the mean percent stable banks were greater than 90%. Approximately 4% of the banks were considered unstable and about 3% were considered to have undercut banks. These stability values are largely due to the confining tailings rock. The channel within the dredged reaches has been disassociated from its floodplain as the bed elevation was lowered and confined by dredge piles from mining activities. Salmonid access to the floodplain is present in most reaches of the Yankee Fork; however, connectivity in the dredged area is limited by the confining tailings.

Substrate in the dredged portion of the Yankee Fork, between Jordan Creek confluence and Polecamp Flat, is 50% of cobble size fraction. The remaining 50% are gravel sizes or smaller, of which about 15% is composed of fines. From Polecamp Flat to the mouth of Yankee Fork, approximately 23% of the substrate is gravel sized or smaller, of which 4% are fines (USFS 2002). In the Yankee Fork upstream of Jordan Creek, the habitat is less disturbed with less fines and a mixed substrate of boulders and cobbles. The R4 Forest Plan standard for fine material is <20%. Although the surface fines are not considered a limiting factor, recent core sampling data indicates 14-28% fines at depth, and an increased risk of fine sediment material to be imported from the fire-burn areas and other erosional sources in the watershed.

Large woody material (LWM) is an important component of salmonid habitat. The PACFISH Riparian Management Objectives (RMO) for LWM are wood pieces >12 inches in diameter, >35 feet in length, and >20 pieces per mile. Based on survey data collected by the Forest Service in 2001, only 1 piece of LWM occurred per mile in the dredged portion of Yankee Fork. In the undredged portions of Yankee Fork, approximately 10 pieces per mile of LWM were calculated.

Dominant riparian vegetation along the dredged portion of the Yankee Fork includes scattered willows and alder, along with forbs, grasses, and sedges growing in the tailings and along the banks. Approximately 25-30 % of the riparian community is unvegetated (plant cover < 10%). Riparian vegetation is critical for maintaining cold water temperature, food production, and instream cover. Therefore, riparian condition is considered a limiting factor for salmonids in the project area (USFS 2007).

The 7-day average maximum temperatures (June through October) between the years 2000 and 2003 at Mid Yankee Fork above Jordan Creek and at Lower Yankee Fork (below Bonanza) show summer temperatures in the low 60s (near 18ºC) (Table 11). INFISH identifies maximum water temperatures below 59˚ F within adult holding habitat and below 48˚ F within spawning and rearing habitat for bull trout. It is likely that bull trout are using areas of cooler temperatures found in tributaries and springs.

Table 11. Yankee Fork Water Temperatures (taken from USDA Forest Service 2007).

Location	7-Day Ave. Max. Temperature (˚F) (June – October)			
	2000	2001	2002	2003
Mid Yankee Fork above Jordan Cr.	64.2	65.4	64.3	64.1
Lower Yankee Fork below Bonanza	61.5	63.2	53.2	65

There are no data to indicate that the stream reach in the vicinity of outfall 003 is either particularly poor or particularly extraordinary habitat for fish. During examination of diffuser area by NOAA Fisheries (2004), a Chinook redd was noted approximately 350ft below the diffuser. The habitat is the immediate area of the diffuser was described as not exceptional. NOAA Fisheries (2004) stated "No cover or velocity shelters were observed in the shallow waters immediately below the diffuser. Therefore, in the case of the low flow samples, it is doubtful that many listed fish would hold for extended periods in the immediate vicinity (<4 feet) from the diffuser. Thus the comparisons of chronic (long-term) criteria and effects data to what is likely a short-term exposure situation is conservative."

F. Summary of Habitat Conditions

The information in this section support the same conclusions made in Bureau of Reclamations Yankee Fork Tributary assessment (Bureau of Reclamation 2012):

- Historic placer mining activities in the Jordan Creek and Yankee Fork sub-watersheds have negatively impacted stream processes.
- Riparian conditions are at natural levels except where dredging has occurred.
- Contamination from historical mining affects water quality.
- Fish passage is not hindered by man-made barriers.

V. DESCRIPTION OF PARAMETERS OF CONCERN AND POTENTIAL EFFECTS

This section presents information for the parameters of concern and discusses draft permit limits in relation to potential effects to the listed species. The permit contains discharge limits for these potential pollutants:

- 5 metallic elements: cadmium, copper, lead, zinc, and mercury
- selenium
- cyanide (weak acid dissociable) (**outfall 002 only**)
- total suspended solids(TSS)
- Whole Effluent Toxicity (WET)
- pH

For the toxic constituents of the discharge, effects of chronic and acute toxicity are described. For the conventional pollutants (e.g. TSS, pH) the potential effects to aquatic biota or other beneficial uses are described. The description of toxicity of metal constituents is based largely on the USEPA Toxics Biological Assessment (BA) for Idaho's numeric water quality criteria (USEPA 1999). Table 12 and Table 13 list the water quality criteria for the constituents that

require adjustment due to variable toxicity caused by water hardness. Note that metals with a mixing zone are cadmium, copper, lead, silver, and zinc.

Table 12. Hardness based criteria for outfall 002 (from Table B-5 of the 2016 Fact Sheet).

Parameter	Flow <30 CFS		Flow ≥ 30 CFS	
	Acute Criterion (µg/L)	**Chronic Criterion (µg/L)**	**Acute Criterion (µg/L)**	**Chronic Criterion (µg/L)**
Cadmium	1.19	0.52	1.06	0.48
Copper	14.8	10.0	23.6	15.3
Zinc	103	104	140	141
Lead	24.1	0.938	13.9	0.541
Silver	0.74	N/A	0.318	N/A

Table 13. Hardness based criteria for outfall 003 (from Table B-6 of the 2016 Fact Sheet).

Parameter	<15 CFS		>45 CFS		15-45 CFS	
	Acute Criterion (µg/L)	**Chronic Criterion (µg/L)**	**Acute Criterion (µg/L)**	**Chronic Criterion (µg/L)**	**Acute Criterion (µg/L)**	**Chronic Criterion (µg/L)**
Cadmium	1.98	1.02	1.03	0.622	1.43	0.797
Copper	14.3	9.69	10.9	7.581	10.4	7.255
Lead	22.2	0.865	13.9	0.54	15.1	0.59
Silver	0.653	N/A	0.32	N/A	0.363	N/A
Zinc	79.4	80.1	57.8	58.2	75.2	75.8

Several other toxics that may be of concern for ESA species are present in the effluent. Limits were not included in the draft permit for arsenic (a metalloid), cyanide (except at outfall 002), selenium, and aluminum because the water quality-based analysis indicated that there was not 'reasonable potential' for these constituents to cause or contribute to an exceedance of state water quality criteria. Although the concentrations of these toxics are small they are discussed in terms of their potential effects.

A. Metals—Introduction

This section provides a general description of each of the six metals with permit limits. The metals arsenic (a metalloid) and aluminum, were found to be possible constituents of the outfalls but in concentrations well below the state standards. Calculations in the Fact Sheets show these do not have reasonable potential to exceed state water quality criteria. Descriptions of these metals are included. Other concerns related to metals concentrations (mixtures of metals, avoidance) are discussed at the end of this section.

B. Metals with Effluent Limits

1. Cadmium

Cadmium naturally occurs in the aquatic environment. It has no known biological use and is considered one of the most toxic metals. While cadmium is released through natural processes, anthropogenic cadmium emissions have greatly increased cadmium presence in the environment. In aquatic systems, cadmium quickly partitions to sediment, but is readily remobilized through a variety of chemical and biological processes (Currie et al. 1997). Toxicity of cadmium to aquatic

organisms varies with the type and life stage of organisms, presence of other toxicants, and the duration of exposure to cadmium.

Cadmium does not bioconcentrate (an increase in concentration in relation to the ambient concentration) significantly in fish species, but does tend to accumulate more readily in invertebrates. Omnivorous and insectivorous predators tend to accumulate cadmium in their tissues more than piscivorous predators (Scheuhammer 1991). Saiki et al. (1995) found no evidence of biomagnification in steelhead on the Upper Sacramento River. Eisler (1985a) also maintains that evidence for cadmium biomagnification suggests that only the lower trophic levels exhibit biomagnification.

2. Copper

Copper occurs naturally in the environment and is an essential element for most organisms as a component of some oxidative enzymes. While copper may form complexes with suspended organic matter, it will ultimately settle out of the water column and deposit in the sediment (USEPA 1984). Concentrations of copper associated with unpolluted freshwater systems are estimated to range between 0.5-1.0 µg/L (Moore and Ramamoorthy 1984, Groth 1971). The toxicity of copper to aquatic organisms is dependent on the speciation of the chemical itself, water hardness, and the type and life stage of the exposed organisms.

Copper is not strongly bioconcentrated in vertebrates, but is more strongly bioconcentrated in invertebrates. Bioconcentration factors (BCF's) reported in the USEPA water quality criteria document for copper (USEPA 1984) ranged from zero in bluegill (*Lepomis macrochirus*) to 22,600 in Asiatic clams (*Corbicula fluminea*). There is little information available concerning biomagnification of copper in aquatic food chains. Also, since the literature resources describing the effects of copper on birds or mammals are minimal, there is little information from which to quantify the biomagnification of copper. Baudo (1983), Wren et al. (1983) and Mance (1987) have all concluded that copper, along with zinc and cadmium do not biomagnify in the aquatic environment.

3. Lead

Lead is a naturally occurring, ubiquitous compound that can be found in rocks, soils, water, plants, animals, and air. It is soluble in water and its bioavailability increases in environments with low pH, low organic content, and low metal salt content (Eisler 1988). 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 (Eisler 1988; Hamir et al. 1982). Species that are sensitive to lead are affected more strongly by dissolved rather than total lead. Likewise, the toxicity of lead is increased when it forms organolead compounds and when environmental conditions consist of high temperature and low pH.

Lead has been shown to bioconcentrate in aquatic species. Invertebrates tend to have higher Biological Concentration Factors (BCFs) than vertebrates. Inorganic lead is poorly accumulated in fish. Larger organic lead compounds such as tetraalkyllead are more toxic than smaller compounds such as trialkyllead. This may be due to the rapid accumulation of tetraalkyllead by fish (Hodson et al. 1984a). In vertebrates, lead concentrations tend to increase with age and

localize in hard tissues such as bone or teeth. Lead residues have been shown to be greater in older birds, sexually mature females, and in birds that have ingested lead shot pellets. While lead has been shown to concentrate in aquatic species, there is little evidence for biomagnification (Eisler 1988).

4. Mercury

Mercury (Hg) is a naturally occurring metal that is one of the most toxic found in the environment. Natural sources of mercury include geological mercury deposits, rock weathering, forest fires and other wood burning. Mercury is used in dental amalgams, exterior paints, thermometers, barometers, and electrical products such as dry-cell batteries, and fluorescent lights. Anthropogenic sources of mercury include: metal smelting, coal burning power plants and industrial incineration of wastes. Significant amounts of mercury enter ecosystems through anthropogenic emissions, reemissions and discharges. The current aquatic life criteria for Clean Water Act purposes in Idaho for water column mercury are 2.1 µg/L (acute) and 0.012 µg/L (chronic). The criteria are expressed as total recoverable (USEPA 1985a). Also, Idaho adopted USEPA's fish tissue methylmercury criterion of 0.3 mg/kg (300ng/g) to protect the health of individuals who eat fish from Idaho surface waters (IDAPA 58.0102.210). The purpose of this criterion is to protect human health and is based on adult consumption rate of 17.5 grams (g) of fish per day (USEPA 2001). The USEPA's national recommended water quality criteria for mercury in freshwater are 1.4 µg/L acute and 0.77 µg/L chronic total Hg.

Mercury is cycled through the environment through an atmospheric-oceanic exchange. This cycling is facilitated by the volatility of the metallic form of mercury. Bacterial transformation of mercury results in stable, lipid soluble, alkylated compounds such as methylmercury (Beijer and Jennelov 1979). Methylmercury is highly toxic to mammals and can interfere with thiol metabolism resulting in mitotic disturbances. This compound can also irreversibly destroy the neurons of the central nervous system (Clarkson et al. 1984). While mercury does occur naturally in small amounts in aquatic environments, the cycling of mercury prolongs the influence of human-caused mercury compounds (Hudson et al. 1995). In sediments, mercury is usually found in its inorganic forms, but aquatic environments are a major source of bioavailable methylmercury (USEPA 1985a). Many of the other prevalent forms of mercury are volatile (e.g. dimethylmercury and elemental mercury) and/or are not readily bioavailable (e.g. particulate or sediment bound inorganic mercury (Beckvar et al. 1996).

Mercury has been shown to accumulate in a variety of aquatic organisms. Of primary concern is the bioaccumulation of mercury by upper trophic level fish species. Fish have been shown to concentrate mercury as methylmercury even when they are exposed to inorganic mercury. Fish typically accumulate only small amounts of methylmercury through gill tissue and directly from the water column (Spry and Wiener 1991). The majority of the accumulated mercury is acquired through the food web. Methylmercury accumulation in microbial communities (e.g. bacteria and phytoplankton) from the water column occurs through passive diffusion (e.g. ionic movement from an area with high concentration to one with low concentration). This mercury is then accumulated through consumption by the higher trophic levels and biomagnifies at concentrations orders of magnitude greater (e.g. largemouth bass, terrestrial wildlife) than those observed in the biotic sources for contamination (e.g. phytoplankton; Beckvar et al. 1996). Aquatic predators face the greatest danger of bioconcentrating mercury, and thus their tissue

concentrations best reflect the amount of mercury available to aquatic organisms in the environment. Greater than 90% of the mercury found in tissues of upper trophic levels (e.g. top predators such as largemouth bass) takes the form of methylmercury. Human health effects related to methylmercury bioaccumulation occur at lower concentrations than toxic concentrations that affect freshwater biota.

Site conditions influence the availability of methylmercury in freshwater systems. Inorganic mercury is typically converted to methylmercury as a byproduct of microbial respiration. The rate of this transformation from inorganic to methylmercury is influenced by factors such as sediment quantity, organic content of sediment, temperature, presence of anoxic or sub-oxic conditions, and presence of bacteria, factors that are typically site-specific.

In April 2005, Idaho adopted USEPA's recommended fish tissue methylmercury criterion to protect the health of individuals who eat fish from Idaho surface waters (IDAPA 58.0102.210). This criterion of 0.3 mg/kg (or 300 ng/g) of fresh weight fish is based on protecting an adult consumer who eats an average of 17.5 grams (g) of fish per day—about one 8-ounce meal every other week (USEPA 2001).

Idaho does not have an inorganic Hg aquatic life criterion stating that the methylmercury criterion will be protective of aquatic life in most situations. However, the aquatic life total mercury water column criteria that had been previously adopted by the State of Idaho remain in effect for Clean Water Act purposes. These are an acute criterion of 2.1 µg/L and a chronic criterion of 0.012 µg/L.

5. Zinc

Zinc is naturally introduced into aquatic systems, usually via leaching from igneous rocks. Most of this naturally introduced zinc is adsorbed to sediments. However, a small amount remains in the water, predominantly in the form of the free Zn^{2+} ion. Release of zinc from sediment is enhanced by the combination of high dissolved oxygen, low salinity, and low pH (Eisler 1993). All life forms require zinc as an essential element, however aquatic animals tend to accumulate excess zinc which can result in growth retardation, hyperchromic anemia, and defective bone mineralization. Zinc primarily affects zinc-dependent enzymes regulating RNA and DNA. Zinc also increases the numbers of metallothioneins, low molecular weight proteins involved in zinc homeostasis. In mammals and birds, the pancreas and bone seem to be the primary targets of zinc toxicity, whereas in fish it is the gill epithelium (Eisler 1993). Concentrations of zinc associated with unpolluted freshwater systems are estimated to range between 0.5-15 µg/L (Moore and Ramamoorthy 1984, Groth 1971).

Because zinc combines with biomolecules in target species and most of these species accumulate more than they need for normal metabolism, data showing bioconcentration factors for target receptors may be misleading. Bioconcentration also depends on the target organism of interest. BCFs reported in the USEPA water quality criteria for zinc (USEPA 1987b) ranged from 51 in Atlantic salmon (*Salmo salar*) to 1,130 for the mayfly (*Ephemerella grandis*).

There is little to no evidence for the successive biomagnification of zinc in tissues of fish and avian receptors. This assumption is based on several factors. First, existing BCF data (USEPA

1987b) shows that the greatest BCF was seen in mayflies while the least was found in Atlantic salmon. This trend was also seen in Elder and Collins (1991) who showed that mollusks accumulated more zinc than the fish that consume these mollusks. Further, zinc toxicity data for birds are predominantly based on force feeding studies of zinc shot or dietary supplements (Eisler 1993).

C. Metals without Effluent Limits

1. Arsenic

Arsenic occurs naturally in aquatic environments in trace amounts. Typical concentrations for background freshwater streams and rivers are less than 1 µg/L As (Moore and Ramamoorthy, 1984: in USEPA 1999). Literature reviewed in CCME (2001a report lowest estimates of toxicity for fish ranged from a 28-d LC_{50} of 550 µg/L for rainbow trout (O. mykiss) (Birge et al. 1979) to a 72-h LOEC (survival) of 970 µg/L for climbing perch (A. testudineus) (Jana and Sahana 1989). The estimated threshold for sublethal, chronic (121d) toxicity of arsenic (as arsenite, As3+) to juvenile rainbow trout *(Oncorhynchus mykiss)* is 4,900 µg/l (Rankin and Dixon 1994). The current Idaho criteria for protecting human health is 10 µg/l(IDAPA 58.01.02). USEPA's National Drinking Water maximum contaminant level (MCL) is 10 µg/L based on the 2006 revision.

The toxicity of arsenic can be altered by a number of factors including pH, Eh (redox potential), organic matter, phosphate content, suspended solids, presence of other toxicants, speciation of the chemical itself, and the duration of exposure to arsenic. Temperature has been shown to alter the toxicity of arsenic. In fish, tolerance of arsenic appears to increase with temperature; (McGeachy and Dixon 1990a and 1990b: in USEPA 1999) whereas the opposite is true for invertebrates (Bryant et al. 1985: in USEPA 1999). Inorganic forms of arsenic are typically more toxic to aquatic species, particularly the more sensitive early life stages (Eisler 1988a: in USEPA 1999).

Arsenic does not readily bioconcentrate in aquatic species. It is typically water soluble and does not combine with proteins. Planktivorous fish are more likely to concentrate arsenic than omnivorous or piscivorous fishes (Hunter 1981: in USEPA 1999). Robinson et al. (1995: in USEPA 1999) found no evidence of arsenic uptake or accumulation from water in both rainbow and brown trout. Eisler (1988) also found no evidence that biomagnification (a progressive increase in concentration from one trophic level to the next higher level) occurs in aquatic food chains. Aquatic invertebrates accumulate arsenic more readily than fish, also an indication that biomagnification is unlikely (Spehar et al. 1980).

2. Aluminum

Aluminum is the most abundant metal and the third most-abundant element in the Earth's crust, but it has little or no known biological function (ATSDR 2008, Gensemer and Playle 1999). As such, it is commonly present in surface waters. The concentration of aluminum in natural waters is generally below 100 µg/L (ATSDR 2008). The USGS measured total aluminum in streams in the Upper Salmon watershed (HUC 17060201) between 1978 and 1980 (11 samples). The quantified concentrations (6 samples) ranged from 100 – 400 µg/L. In the five remaining samples, the presence of aluminum was verified but not quantified (remark code "M"). Using

maximum likelihood estimation to estimate the unquantified concentrations, the USEPA estimates that the average aluminum concentration is 141 µg/L, with a standard deviation of 120 µg/L. Seven aluminum samples are available for the Grouse Creek Mine effluent (outfalls 002 and 003), taken between November 2000 and June 2010. The average concentration is 393 µg/L, with a standard deviation of 268 µg/L, and the maximum was 880 µg/L.

Aluminum is a gill toxicant to fish (Gensemer and Playle 1999, Alstada et al. 2005). The mechanism of toxicity at the gills can be ionoregulatory, respiratory, or a combination of both. The pH, hardness of the water, and the magnitude of the pH change of water irrigating the gills determine which mechanism is most important (Sparling and Lowe 1996). In acidic waters, (pH< 5) ionoregulatory effects predominate, and are thought to be the same mechanism as the toxicity caused by low pH alone. The calcium in hard water competes with Al^{3+} binding to fish gills, thus reducing toxicity. In moderately acidic water (pH of about 5.0 – 6.0), respiratory effects are dominant. Dissolved organic matter and fluoride reduce the toxicity of aluminum by forming complexes with Al, thus reducing the amount of dissolved Al available to bind to the gills. Historically, most studies have evaluated aluminum toxicity in acidic waters, with very few studies evaluating the effects of aluminum at a pH > 8.0 (Sparling and Lowe 1996). Fish can acclimate to aluminum, but with some metabolic cost (Gensemer and Playle 1999).

The effluents and receiving waters tend toward neutral conditions; the median pH of outfalls 002 and 003 is 7.1 and 7.6, respectively, and the median pH of the receiving waters upstream from outfalls 002 and 003 is 7.3, for both outfalls. Thus, studies of aluminum toxicity in waters of neutral pH are most relevant to this action.

In a study of the effects of aluminum to Atlantic salmon (*Salmo salar*), survival was 100% after three weeks of exposure to 347 – 362 µg/L total aluminum at pH levels of 9.5 and 7.4, whereas all of the fish died at a pH of 5.8. Elevated blood glucose and hematocrit, and lowered plasma Cl^- were observed in fish exposed to elevated Al at a pH of 9.5. The authors concluded that Al toxicity to salmonids is of limited importance under alkaline conditions (Poléo and Hytterød 2003).

In a study of the effects of aluminum to juvenile rainbow trout, pH was determined to be the most important variable affecting aluminum-induced mortality in 96-hour acute toxicity tests. At near-neutral pH (7.55 – 7.64), survival was 100% after 96 hours of exposure to aluminum at concentrations ranging from 740 – 9,840 µg/L, at hardnesses ranging from 23 – 130 mg/L as $CaCO_3$. In weakly alkaline water (pH 8.10 – 8.58), the 96-hour LC_{50} of aluminum to juvenile rainbow trout ranged from 6,170 – 7,670 µg/L at hardnesses ranging from 23 – 116 mg/L as $CaCO_3$). In chronic tests, the 16-day LC_{50} was 1,940 µg/L at a hardness of 20.3 mg/L and 3,910 µg/L at a hardness of 103 µg/L (Gundersen et al. 1994).

Aluminum effects upon growth of juvenile rainbow trout were greater at near-neutral pH than at weakly alkaline pH. Hardness did not protect against effects of aluminum upon growth in 16-day tests. In weakly alkaline water (pH 7.94 – 8.14), growth was significantly different from controls only at total aluminum concentrations of at least 2,750 µg/L. In near-neutral water, (pH 7.30 – 7.35), growth was significantly different from controls for all concentrations tested, however, the lowest experimental concentration of aluminum was 890 µg/L, which is higher than the

maximum effluent concentration (Gundersen et al. 1994). Observed bioconcentration factors for aluminum in young brook trout ranged from 50 to 231 (USEPA 1988).

The USEPA-recommended water quality criteria for aluminum are a chronic criterion of 87 µg/L and an acute criterion of 750 µg/L (USEPA 1988). The State of Idaho has not adopted any water quality criteria for aluminum, however, aluminum could be regulated under the State of Idaho's narrative criterion for toxic substances (IDAPA 58.01.02.200.02) and federal regulations requiring compliance with such narrative criteria in permits (40 CFR 122.44(d)(1)(vi)). The USEPA-recommended chronic criterion was primarily based on two studies conducted at a pH of 6.5 to 6.6, one of which found that aluminum at a concentration of 88 µg/L caused a 4% reduction in weight of young brook trout in a 60-day test, and another which found no mortality of striped bass exposed to 87.2 µg/L aluminum in a 7-day test (USEPA 1988). In the striped bass test, the hardness was < 10 mg/L as $CaCO_3$ (USEPA 2015). The USEPA's national recommended water quality criteria cautions that a water effects ratio may be appropriate when applying the recommended chronic criterion because data indicate that aluminum is substantially less toxic at higher pH and hardness, and the USEPA is aware of field data which indicate that many high-quality waters in the U.S. contain more than 87 µg/L aluminum (USEPA 2009).

As explained above, the pH of the Grouse Creek effluents and receiving waters tends toward neutral conditions. The pH of the effluent and of the receiving waters upstream from the outfalls is consistently higher than 6.6. The 5th percentile pH of the effluent from outfall 002 and 003 is 6.7 and 6.9 respectively, and the 5th percentile pH of the receiving waters upstream from outfalls 002 and 003 is 6.9 and 7.66, respectively. The draft permit proposes a lower-bound pH limit of 6.5. In addition, the receiving waters are consistently harder than the very soft water used in the striped bass toxicity test upon which the recommended chronic criterion is based. The 5th percentile hardness of Jordan Creek downstream from outfall 002 is 29 mg/L as $CaCO_3$ and the 5th percentile hardness of the Yankee Fork downstream from outfall 003 is 21 mg/L as $CaCO_3$. Thus, the acidic and very soft water chemistry conditions that cause the highest potential for aluminum toxicity are unlikely to be observed in the effluents or in the receiving waters.

3. Silver

Silver occurs naturally in aquatic systems with common background concentrations of silver in unpolluted freshwater systems of <10 µg/L total silver (USEPA 1987a). Silver has no known biological function and is toxic to biota at very low concentrations (Fisher and Wang 1998). Anthropogenic discharges of silver into the environment are from industry (primarily the photographic and imaging industry), silver mining and milling, sewage discharge, and atmospheric deposition from fuel combustion (USEPA 1981).

An important characteristic of silver is its variable toxicity depending on form. The ionic forms (Ag^+ most common) are highly reactive and are the most toxic. Ionic silver will readily convert to more innocuous forms when exposed to natural chemical ligands (Purcell and Peters 1998). Because of the variable toxicity of silver, the form as well as the concentration has an important influence on the toxicity of silver in the environment.

Evidence of bioaccumulation is mixed. Trophic transfer was found to be insignificant in studies on oysters and shrimp, while others have found evidence of significant uptake in marine studies.

Uptake via food ingestion has been found to be a dominant source in freshwater biota (Fisher and Wang 1998). Once accumulated by organisms, silver can bind to proteins or sulfide forming insoluble species, which are less available to predators than other forms. Deposition as an insoluble sulfide would decrease toxicity, essentially decoupling the bioaccumulation and silver toxicity (Fisher and Wang 1998). Because of silver's tendency to combine with chemical ligands to form insoluble species, movement in the environment resulting in biomagnification is not a concern (Purcell and Peters 1998).

D. Avoidance of Metals

The behavioral response of avoidance to toxic conditions from metals is considered to have a lower severity of ill effect than chronic (sub-lethal) effects, yet, avoidance is a substantial ecological effect when it results in the decrease of an animal's ability to adapt or survive (Rand and Petrocelli 1985). The ability of fish to sense chemical changes in their environment allows them to locate beneficial habitats and to avoid less favorable conditions. Response to an adverse chemical stimulus allows fish to avoid potentially dangerous chemicals. Avoidance response can result in fish vacating affected areas and can potentially block upstream migration. Salmonid avoidance response can occur at concentrations well below the lethal thresholds for metals.

In the Grouse Creek Unit permit, avoidance behavior is an important consideration because it can have substantial effects on fish populations that must move within their aquatic system. For migratory fish (downstream outmigration of smolts and upstream movement of spawners) and for resident fish accessing different habitat types (e.g. spawning habitat), an avoidance response to metals concentration could act as a barrier to fish movement. The primary concern is that as the chronic concentration is diluted at the edge of the mixing zone, fish may avoid this outer area. This would effectively decrease the zone of passage available and interfere with fish movement.

Behavioral avoidance of metals has been demonstrated both in the laboratory and field (Sprague 1971) at very low concentrations. The avoidance response is a significant species-specific form of adaptive fish behavior occurring at sublethal concentration levels, its intensity being directly proportional to the logarithm of the concentration and independent from the toxicity of substance studied (Svecevicius 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 exposed, and the metal or mixture of metals encountered so a predicted response to a specific situation with its own characteristic water chemistry and mix of pollutants such as the Grouse Creek Unit is not exact with available information.

Avoidance response in fish has been studied for most of the metals that are of concern in the Grouse Creek action area including cadmium (Woodward et al. 1997, McNichol and Scherer 1991), copper (Hansen et al. 1999, Scherer and McNichol 1998, Woodward et al. 1997), lead (Woodward et al. 1997, Scherer and McNichol 1998), mercury (Atchison et al. 1987), and zinc (Hansen et al. 1999, Woodward et al. 1997, Scherer and McNichol 1998). Most avoidance studies have been conducted in laboratories. Because the motivations of fish are much different in the laboratory than under natural conditions, laboratory experiments can only approximate the actual response (Atchison et al. 1987). Except for copper and zinc, the literature on avoidance

response of inland fish species to metals concentrations is limited. Salmonid avoidance studies/data are not available or very limited for several of the constituents that are in the Grouse Creek Unit effluents, including arsenic, chromium, silver, and selenium. Avoidance is not a consideration for cyanide as other more serious sub-lethal effects occur at very low concentrations. Avoidance behavior associated with other metals of the effluent are as follows.

1. Cadmium

Cadmium has been reported to be toxic at concentrations lower than fish can detect and avoid (Atchison et al. 1987). Juvenile cutthroat trout did not avoid cadmium concentration of 0.66 µg/L (Woodward et al. 1997). Lake whitefish *(Coregonus clupeaformis)* showed a neutral response to cadmium at 0.2, 1, and 5µg/L (McNichol and Sherer 1993). A concentration of 8 µg/L was considered the avoidance threshold of concern for cadmium. Cadmium may contribute to avoidance behavior as a constituent in a metals mixture. This is addressed in the next section on metals mixtures.

2. Copper

Copper has been well documented to cause avoidance with salmonids and other fishes in laboratory and field conditions. The USEPA has conducted an extensive literature review (Shepard and Zodrow 2009) on copper and behavioral effects. In this review behavior was defined following Henry and Atchison (1991) "Behavior is the organismal level manifestation of the motivational, biochemical, physiological, and environmentally influenced state of the organism." This definition does not include olfaction alone as an organism level endpoint or a behavior. The literature found that copper is unusual in that low concentrations are avoided, fish are attracted to high concentrations, and fish show no preference at intermediate concentrations. For rainbow trout geometric mean LOECs for avoidance (N = 8), no preference (N = 3) and attraction (N = 5) are 11.8, 540 and 799 µg/L (Figure 6). The literature review summarized the studies for copper only exposure as follows;

- Of the 132 studies reviewed, 129 (97.7%) had LOECs greater than hardness normalized USEPA chronic copper criterion at study hardness
- The eight lowest LOECs are all salmonid avoidance studies. Salmonid avoidance appears to be the most sensitive behavior, and the only one where LOECs are lower than chronic copper criterion.
- Three studies with LOECs below USEPA chronic criterion (LOECs normalized to 100 mg/L hardness) 0.11 µg/L rainbow trout avoidance (Folmar 1976), 1.6 µg/L coho salmon altered freeze response, a type of predator avoidance (Sandahl et al. 2007) and 5.2 µg/L rainbow trout avoidance (Hansen et al. 1999)

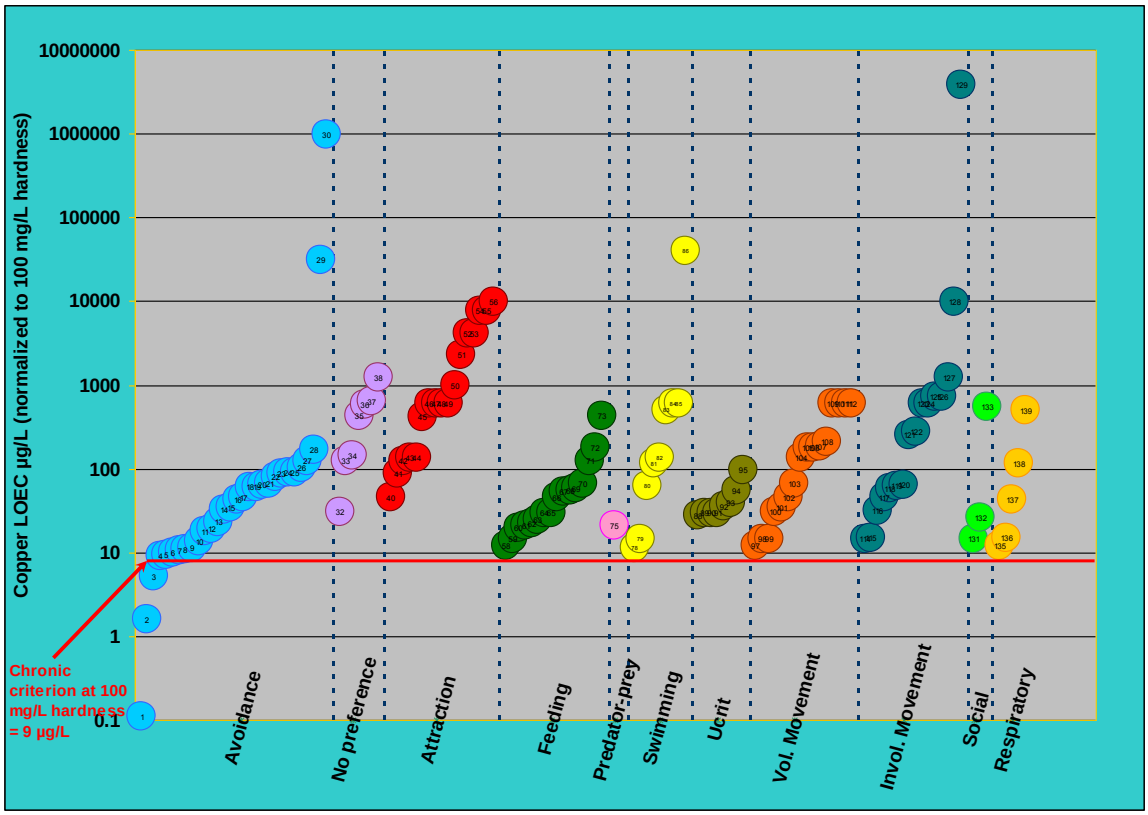


Figure 6. Summary of Freshwater Laboratory Studies, Copper Only Exposure LOEC's from the literature (taken from Shepard and Zodrow 2009).

For rainbow trout there were eight studies with avoidance behavior LOECs available. These LOECs range between 0.1 and 74 µg/L. The geometric mean of the avoidance LOECs after normalization to 100 mg/L hardness of these studies is 11.8 µg/L. This value is higher than the USEPA chronic copper criterion of 9.0 µg/L (hardness 100mg/L). The body of evidence supports that the chronic criteria should be protective of salmonids including behavioral response. Several considerations: 1) other salmonid species such as Chinook salmon may be more sensitive than the more common test species, rainbow trout, 2) copper acts additively with some metals and synergistically with zinc in eliciting avoidance behavior (addressed in next section), 3) laboratory studies must be considered with caution as actual avoidance behavior to copper by fish in natural habitat has been found to be quite different, and 4) Acclimation, defined as compensatory adjustments by an organism to alterations of environmental conditions, to low concentrations can require exposure to higher concentration before altered behavior occurs. For example, Chinook salmon that initially avoided 2.8 µg/L, no longer avoided up to 21 µg/L after 25 day acclimation to 2.2 µg/L (Hansen et al. 1999).

3. Lead

Avoidance studies of lead were rare. Giattina and Garton (1983) determined an avoidance threshold of 26 µg/L Pb for rainbow trout in laboratory experiments. No significant avoidance behavior was observed in Snake River cutthroat trout with exposed to 1.3 µg/L lead (Woodard et al. 1997). Sherer and McNicol (1998) reported 10 µg/l lead was avoided by lake whitefish. One other behavior study found that salmonids (juvenile rainbow trout) social activity/aggressive

behavior was not significantly affected by sublethal concentrations of lead (Sloman et al. 2003). The avoidance behavior values of these studies are generally higher than chronic criteria of 0.6-2.5 µg/L depending variation in hardness.

4. Mercury

Studies of mercury associated with avoidance behavior were very limited. Rehnberg and Schreck (1986) reported coho salmon avoided 20 µg/l and lost olfactory function to detect amino acids which are potent odors related to chemoreception, homing, and pairing. As stated in the effects section, mercury can cause adverse effects at very low concentrations and it is likely that avoidance behavior is not the most sensitive endpoint for mercury.

5. Zinc

Several studies document avoidance behavior by salmonids exposed to zinc. Literature on avoidance of zinc by salmonids was review by IDEQ (2000) and is listed in Table 14. They determined an avoidance level for salmonids between 14 µg/L to 28 µg/L for zinc. Zinc has been shown to act synergistically with other metals which could affect avoidance behavior. This is described in the next section.

Table 14. Zinc concentrations associated with fish avoidance (Taken from IDEQ 2000).

Zinc Conc.(µg/L)	Test Organism	Reference
6.5	Avoidance threshold with coho salmon under laboratory conditions (measured concentrations not reported nominal concentration listed)	Rehnberg and Schreck 1986
10	Avoidance threshold with lake whitefish when given no choice for light or shade	Sherer and McNicol 1998
41	Lowest field adjusted species mean avoidance threshold (SMAT)	CEC 2000b
47	94% avoidance by rainbow trout	Black and Birge 1980
53	Avoidance threshold with juvenile Atlantic salmon	Sprague et al 1965
210	Avoidance observed in the field with migrating adult Atlantic salmon (not a threshold)	Sprague et al 1965
284	Avoidance by male fathead minnows when zinc was the only variable	Korver and Sprague 1989
2200	Possible avoidance observed in the field with adult chinook salmon (not a threshold)	Goldstein et al 1999

E. Avoidance Behavior Associated with the Mixtures of Metals

In laboratory tests, copper and zinc mixtures have been shown to act together to cause a lower threshold of avoidance than would result from either metal alone (Giattina and Garton 1983). Hansen et al. (1999a) reported that behavioral avoidance to copper and cobalt mixtures in soft water differed greatly between the rainbow trout and Chinook salmon. Rainbow trout avoided 1.6 µg/L concentration of copper and 180 µg/L cobalt individually, but the response was significantly lower for a mixture of the two metals, 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. 1999).

Studies of avoidance behavior to the mixture of metals in the Clark Fork River 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. 1995, Hansen et al. 1999b). Cutthroat trout 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 5 µg/L zinc). 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 indicating that natural fish populations will avoid tributaries with high metals contamination.

The examination of metals mixture and avoidance literature affirms that the avoidance response of salmonids is variable due to water chemistry (pH and hardness for example), the mixture of metals considered, and the species of salmonid. In general, avoidance behavior is elicited at a higher concentration than the chronic criteria for individual metals (e.g. cadmium, chromium, mercury, and copper). For these reasons, the issue of behavioral avoidance response will be addressed as a separate issue from individual metals in the effluent of outfalls 002 and 003.

F. Combined Effects of Metals on Toxicity

Fish and other aquatic organisms will be exposed to the whole effluent with unknown combined effects of the metals. Antagonistic metal reactions, which reduce the toxicity of metals primarily through precipitation, result in solutions that are less toxic than single metal solutions. For example, calcium markedly counteracts the toxic effects of copper, lead, potassium, sodium and zinc (Rand and Petrocelli 1985). Conversely, combinations of metals can be synergistic; their joint effect being greater than the sum of the separate effects. Copper, zinc and cadmium, which occur in the Grouse Creek Unit effluent, are known to act synergistically. Also, metals within a mixture that are below their no-effect concentration have been found to contribute to the toxicity of metals mixtures on a chronic basis (Spehar and Fiandt 1986).

The potential combined effect of metals is addressed in the USEPA draft permit through the requirement for Whole Effluent Toxicity (WET) testing. The draft permit requires chronic toxicity testing using standard test organisms, *Ceriodaphnia dubia,* the fathead minnow, *Pimephales promelas*, and a green alga (*Selenastrum capricornutum*). After a screening period, the most sensitive species of the three species is to be used, with the water flea to be used if the most sensitive species cannot be identified. Chronic toxicity testing is required to be completed four times per year in outfalls 002 and 003. 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. The outcome of the test is interpreted in chronic toxicity units, TU_C. TU_C are chronic toxicity units that are equal to the reciprocal of the effluent concentration that causes no observable effect in chronic toxicity tests. WET testing is discussed below. In addition to chronic testing, acute WET testing using rainbow trout is required annually in outfalls 002 and 003.

G. Non Metals with Effluent Limits (cyanide, TSS, WET, pH)

1. Weak Acid Dissociable (WAD) Cyanide

The current Idaho water quality standards establish acute and chronic cyanide (CN^-) criteria of 22.0 µg/L and 5.2 µg/L, respectively (see Table 3) measured as weak acid dissociable (WAD) cyanide. WAD cyanide analysis reports ionic cyanide as well as cyanide weakly bound to metals such as copper, nickel, and zinc. As a rule, cyanide complexed with metal is less toxic than free cyanide. Free cyanide measurements are a more reliable indicator of toxicity to aquatic life than total cyanide because the latter measurement includes the relatively stable organic cyanides and metallocyanides. The cyanide measured by WAD analysis will encompass the most toxic forms along with those that are less toxic (USEPA, 1985b).

Cyanide occurs naturally in the environment via production by a variety of plant species. Background levels in freshwater systems average 0.9 µg/L cyanide (Eisler 1991). It is toxic to most living organisms and primarily occurs in aquatic environments as free cyanide (the concentration of HCN and CN^-). Relative concentrations of hydrocyanide (the more toxic form) and cyanide ion (CN^-) are dependent on pH and temperature, and the toxicity of cyanide may increase with decreasing pH and temperature. Other forms of cyanide that may occur are simple cyanides and metallocyanide complexes. Accumulation of metallocyanide complexes in sediment is not likely because dissociation occurs easily at pH values lower than 8. The mechanism of cyanide toxicity involves inhibiting cytochrome oxidase, the terminal oxidative enzyme of the mitochondrial electron transport chain, thus blocking aerobic adenosine triphosphate (ATP) synthesis. The result of this mechanism of toxicity is that cyanide is a rapid and potent asphyxiant (Eisler, 1991). Sarkar (1990) observed that the toxicity of cyanide increased with increasing temperature for fish, mollusks, insects, and plankton, although this relationship was least strong for the mollusk species.

Bioconcentration of cyanide is considered to be negligible in fish because the compound is easily metabolized. As reported by USEPA (1985b) the existing literature does not provide evidence for cyanide biomagnification. This is likely due to the fact that vertebrate species, such as fish, may readily metabolize cyanide, thus removing the cyanide from the food chain at that level.

2. Total Suspended Solids

Total suspended solids (TSS) measures both settleable and suspended sediment and organic matter in a discharge. The effluent limits for TSS are based on USEPA's technology-based effluent guidelines for the mining industry (see Appendix C of the 2015 Fact Sheet for more details regarding the technology-based guidelines). These guidelines specify the following effluent limits: monthly average of 20 mg/L TSS and a daily maximum of 30 mg/L TSS. TSS measures particles sized typically <0.1mm. Following discharge, the size of particles entrained in the receiving water varies with flow characteristics (e.g. velocity, gradient, and turbulence). Deposition of suspended sediment is related to particle size and diminished flow. The very fine particle fraction (<0.06 mm) tends to stay in suspension for the length of the fluvial system.

Because TSS only affects species within the aquatic environment, most literature focuses on effects to fish, with salmonid species being the most frequently studied. 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 affect tolerance to TSS by further stressing the animal (Servizi and Martens 1991). Also, the availability of refuge will influence the ability to avoid exposure (Bisson and Bilby 1982).

Vulnerability to TSS effects varies with life history phase. Juvenile and larval salmonids have been found to be more susceptible to TSS effects than adults (Servizi and Martens 1991). Pre-emergent larvae and eggs are considered the most susceptible, resulting in reduced survival and hatching.

A review by Newcombe and Jensen (1996) provides a summary of studies conducted on the effects of TSS exposure to fish at different life history phases. Most studies assessed higher exposure concentrations over shorter durations than those allowed by limits proposed in this permit. Truncating this meta-data set to review only studies that evaluate exposures near the effluent limits concentrations yields the following summary:

- For the adult life history stage, most of the effects resulting from exposure concentrations <80 mg/L were behavioral or sublethal effects. Of the six studies with concentrations <30 mg/L, lethal effects were reported by Peters (1967) (reduced abundance of brown and rainbow trout over a 30-day duration).
- For the juvenile life history stage, one study on juvenile Chinook (MacKinlay et al. 1987) indicates paralethal effects (reduced growth rate) at concentrations <20 mg/L at durations similar to the high flow period (approximately 60 days) experienced in Jordan Creek (Hecla 2000).
- Several studies indicate paralethal effects to juvenile salmonids at relatively low exposure concentrations but with exposure duration exceeding the high flow period expected in the permit area. Of these studies, one showed effects at 20 mg/L (growth rate declined in brook trout (Sykora et al. 1972).
- Several studies indicate behavioral effects at relatively low exposure concentration and low duration. One study found effects at 20 mg/L and 24 hour duration (arctic grayling avoided parts of stream) (Birtwell et al. 1984).

Using available past study results, Newcombe and Jensen (1996) developed matrices of TSS dose and response. For juvenile and adult salmonids, their results suggest behavioral level effects are likely at 20 mg/L TSS over short durations (1 to 144 hours). Over longer durations, the severity of ill effect increases, but does not exceed a sublethal level of effects at this exposure concentration. For larval salmonid life history stages, effects of 20 mg/L are sublethal when maintained for 1 to 144 hours. Sublethal effects include reduction in feeding rate and success and physiological stress. For the egg life history stage, effects of 20 mg/L are sublethal

(physiological stress) when maintained for 1 to 144 hours. The paralethal effects of delayed hatching and lethal effects (mortality) occur at longer durations of exposure.

A secondary effect of TSS is degradation or loss of habitat due to fine sediment deposition at diminished flow characteristics. Fine sediment deposition can result in the following effects to salmonid habitat: 1) reduction of ability of fish to build suitable redds, 2) asphyxiation of developing embryos (hinders water flow, decreasing oxygen saturation and removal of metabolic waste), 3) reduction of successful emergence of fry from redds due to burial (blockage of interstices prevents emergence of larvae), and 4) reduction of availability of habitat for juveniles by the filling of cobble interstices used for hiding and cover and filling of pools. Also, populations of macro-invertebrates, a primary food source for salmonids, can be reduced due to habitat reduction from fine sediment deposition.

3. Whole Effluent Toxicity

The Idaho water quality standards include a narrative criterion for toxicity that states: "Surface waters of the state shall be free from toxic substances in concentrations that impair designated beneficial uses." Whole effluent toxicity (WET) tests are used to determine the toxicity of wastewaters for the protection of aquatic life (Grothe et al. 1996, USEPA 1994). These tests use standard aquatic test organisms to evaluate the aggregate toxicity of effluents. WET tests focus on the sensitive life stage of the test species on the assumption that protection of this stage will protect the species as a whole. Chemical-specific effluent limits specify the amount of a particular chemical that can be discharged to a receiving water body without causing adverse effects to aquatic biota. WET tests are designed to integrate both the toxicity of the individual compounds and the interactions of these compounds in the evaluation of overall effects to the receiving water body.

Advantages of WET techniques include the following: 1) detection of toxicity caused by compounds that do not have effluent limits or are not monitored, 2) assessment of the bioavailability of the toxic constituents, and 3) measurement of the effects of interactions of constituents. The WET approach also allows for prediction of ecological impacts before they occur.

Since there are no national criteria for WET, USEPA uses recommendations in the TSD (USEPA 1991) to develop effluent limits protective of the narrative criteria, where a state criteria for toxicity is expressed as a narrative statement (as in Idaho water quality standards). USEPA's recommended magnitudes for WET are 1 chronic toxic unit (TU_c) and 0.3 acute toxic unit (TU_a) for the chronic and acute criteria, respectively, for the most sensitive test species. TU are defined as the ratio of the exposure concentration to a benchmark concentration. The magnitude of the ratio illustrates how much more or less toxic the exposure concentration is when compared to the known benchmark concentration. TU_c is equal to the reciprocal of the effluent concentration that causes no observable effect on the test organisms by the end of the chronic exposure period (i.e., 100/NOEC or $100/IC_{25}$) determined through WET testing. The NOEC is used for survival endpoints, and the IC_{25} is used for all other test endpoints. TU_a equals the reciprocal of the effluent concentration that causes 50 percent of the organisms to die by the end of the acute exposure period (i.e., $100/LC_{50}$) as determined through WET testing. The TSD (USEPA 1991) provides more information on the basis of the 1 TU_c and 0.3 TU_a values. Using the chronic and

acute criteria of 1 TU_c and 0.3 TU_a, effluent limits for WET are developed in the same manner as for the chemical criteria.

4. pH

The pH of natural waters is a measure of the acid-base equilibrium achieved by the various dissolved compounds, salts, and gases in the water and is an important factor in the chemical and biological systems of natural waters. Idaho's current aquatic life criteria for pH require that the hydrogen ion concentration (pH) values are within the range of 6.5 to 9.0 standard units (su). USEPA's technology-based effluent limits applicable to the Grouse Creek Mine discharge specify a pH range of 6.0 to 9.0 su. The effluent limits in the final permit incorporates the more stringent water quality-based minimum of 6.5 and a maximum of 9.0.

Changes in pH affect the degree of dissociation of weak acids and bases and thus directly affect the toxicity of many compounds. In addition, pH affects the solubility of metal compounds present in the water column and sediments of aquatic systems, thereby influencing the exposure dose of metals to aquatic species. In 1969, the European Inland Fisheries Advisory Commission (EIFAC 1969) concluded that pH values ranging from 5.0 to 6.0 are unlikely to harm any species unless either the concentration of free carbon dioxide exceeds 20 ppm or the water contains iron salts precipitated as ferric hydroxide, a compound of unknown toxicity; pH values ranging from 6.0 to 6.5 are unlikely to harm fish unless free carbon dioxide is present in excess of 100 ppm; and, pH values ranging from 6.5 to 9.0 are harmless to fish, although the toxicity of other compounds may be affected by changes within this range (discussed in more detail below). These and other studies evaluating the effects of pH on various fish species and macroinvertebrates (Mount 1973; Bell 1971) led USEPA (1986) to conclude that a pH range of 6.5 to 9.0 provides adequate protection for the life of freshwater fish and bottom dwelling invertebrates. Outside of this range, fish suffer adverse physiological effects, increasing in severity as the degree of deviation increases until lethal levels are reached.

Although pH itself may have toxic or deleterious effects on aquatic biota, other chemical and physical factors generally affect the biota first or more directly (e.g., dissolved oxygen, temperature, sedimentation). The following describes the pH interactions that may be applicable to discharge from outfalls.

pH activity has a significant impact on the availability and toxicity of metals. The following is summarized from Elder (1988) and Baker et al. (1990) (as mentioned in ODEQ 1995). Metal-hydroxide complexes tend to precipitate (i.e., reduced ability to remain suspended) and are quite insoluble under natural water pH conditions, thus, the metal is not able to exert a toxic effect. However, the solubility of these complexes increases sharply as pH decreases. pH activity also affects the sensitivity of organisms to a given amount of metal. 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 values. Each metal has its own range where pH and site-specific conditions become factors in the metal's bioavailability. Aluminum is the metal of greatest concern at low pH values. No adverse effects to listed species due to pH-driven changes in metal toxicity (where the metals comply with the respective metals criteria) would occur in the range of Idaho's pH criteria. The effects of low pH are also more pronounced at low concentrations of calcium. In general, increasing concentrations of calcium tend to mitigate the

toxicity of some metals like aluminum (Sparling and Lowe 1996). In summary, reductions in pH below natural levels will tend to increase metal availability and toxicity.

H. Non Metals without Effluent Limits:

1. Ammonia (NH_3)

Ammonia occurs naturally in water at low concentrations in equilibrium with other inorganic nitrogen compounds. Ammonia commonly enters the environment as a result of municipal, industrial, agricultural, and natural processes. Natural sources of ammonia include the decomposition of organic matter, gas exchange with the atmosphere, forest fires, and nitrogen fixation processes. Point sources of ammonia include emissions and effluents from industrial plants, fertilizer plants and oil refineries. Non-point sources of ammonia include agricultural, residential, municipal, and atmospheric releases.

Ammonia is highly soluble in water and its speciation is affected by a wide variety of environmental parameters including pH, temperature, and ionic strength. In aqueous solutions, an equilibrium exists between un-ionized (NH_3) and ionized (NH_4 +) ammonia species. Un-ionized ammonia refers to all forms of ammonia in water with the exception of the ammonium ion (NH_4 +). Ammonia is toxic to fish and other aquatic life when it is in the un-ionized form. It is thought that the un-ionized form is more toxic because these neutral molecules may pass through biological membranes more readily (CCME 2000).

Fish are adept at sensing and avoiding very low concentrations of ammonia. Furthermore, fish have been reported to enter waters that contain acutely toxic concentrations of ammonia without suffering any obvious long-term effects, as long as these excursions are followed by periods in which the fish are in waters that contain ammonia concentrations below acute toxicity levels (Thurston et al. 1981).

Concentrations of ammonia acutely toxic to fishes may cause loss of equilibrium, hyper-excitability, increased breathing, cardiac output, and oxygen uptake, and, in very high concentrations, 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 tissues of gills, livers, and kidneys (USEPA 1999c). Factors that have been shown to affect ammonia toxicity include dissolved oxygen concentration, temperature, pH, previous acclimation to ammonia, fluctuating or intermittent exposures, carbon dioxide concentration, salinity, and the presence of other toxicants (USEPA 1999c). Idaho WQS specify criteria for protection of early life stages of fish from chronic effects, dependent on pH and temperature. Invertebrates are generally more tolerant than fishes to the acute and toxic effects of ammonia. The following summary of toxicological test is from the Canadian Water Quality Guidelines for the Protection of Aquatic Life (CCME 2000).

Studies conducted by Thurston et al. (1984) found sensitivity to un-ionized ammonia (NH_3) concentration ranged from 0.01 to 0.07 mg/L over a period of 5 years. No correlation between ammonia concentration and number of eggs produced was observed in the parental generation. Pathological lesions in the gills and extensive tissue degradation in the kidneys were directly correlated with ammonia concentrations above 0.04 mg/L, after 4 months of exposure.

Sockeye salmon (*O. nerka*) were exposed to total ammonia for 62 days from fertilization to hatching (Rankin 1979). Concentrations of un-ionized ammonia were calculated and ranged from 0.00097 - 4.92 mg NH3/L at 10ºC and pH 8.2 and hatchability was the measured endpoint. Hatchability was 63.3%, 49% and 0% in controls, at 0.12 mg/L, and 0.46 mg/L, respectively. An EC_{20} was calculated for this study by CCME (2000) with correction for control mortality. The reported EC_{20} was 0.057 mg/L un-ionized ammonia. Bader and Grizzle (1992) exposed catfish (*Ictalurus punctatus*) fry to ammonia in a 7-day static renewal test. An IC_{20} for fry growth was determined by CCME (2000) at 0.162 mg/L un-ionized ammonia. There was no incremental mortality up to 0.490 mg/L exposure. Smith et al. (1984) conducted a 30-day early life-stage test on bluegill sunfish (*Lepomis macrochirus*). The test exposed 28-day old embryos and monitored them to the swim-up fry life stage. No significant reduction was found in percent of hatch up to a concentration of 37 mg/L un-ionized ammonia. However, larvae were deformed and generally died within 6 days. An IC_{20} (survival and growth) of 0.060 mg/L was calculated (CCME 2000) for this study.

The total ammonia criteria for the receiving water of Jordan Creek are 6.77 mg/L acute and 2.8 mg/L chronic. Monitoring data show outfall 002 annual median ammonia levels are typically <0.2 mg/L and the maximum value over all years is 0.74 mg/L (Figure 7 and Appendix B. Summary of water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), **Site S-9 (Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003).** The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) and temperature is collected daily at outfalls 002 and 003 when discharging.

Appendix B**)**. Downstream monitoring station Site S-4 annual median concentrations are ≤0.10 mg/L. In Yankee Fork, the acute criterion is 4.42 mg/L and the chronic criterion is 2.02 mg/L. Monitoring data for outfall 003 show 2005 was the last year that either limit was exceeded. The maximum value recorded since 2005 is 1.6 mg/L. Downstream monitoring station Site S-10 annual median concentrations are low (≤0.10 mg/L) (Figure 8).

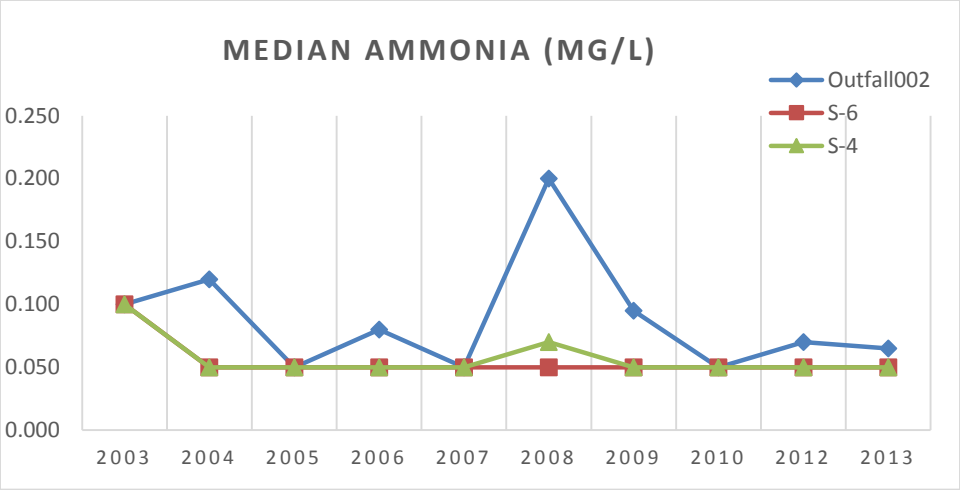


Figure 7. Ammonia median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013 (Source: Helca monitoring data).

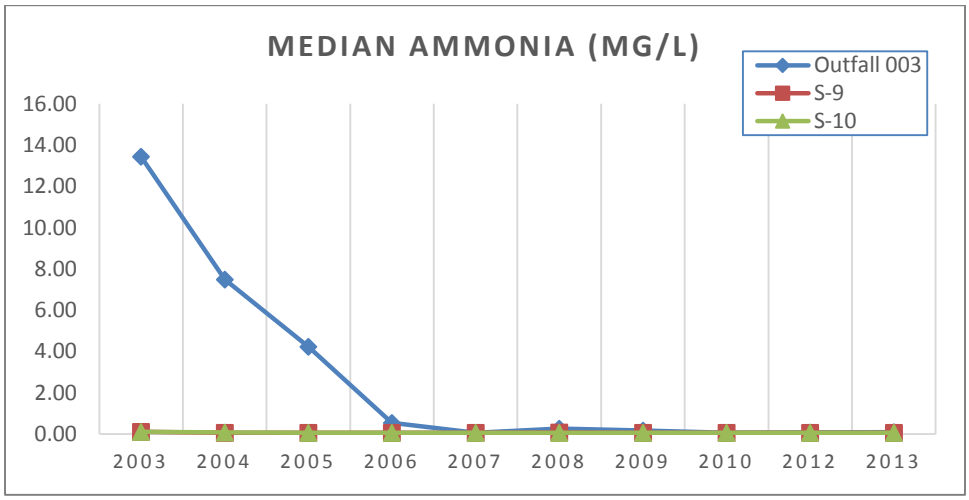


Figure 8. Ammonia median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013 (Source: Helca Limited).

2. Nitrogen (Nitrate + Nitrite)

Water quality criteria are expressed as a combined form of nitrogen as "nitrate plus nitrite" because nitrite nitrogen is relatively unstable and is easily oxidized to the nitrate (NO_3) form. 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) and Kroupova et al. (2008). Elevated ambient nitrite concentrations are a potential problem for freshwater fish since nitrite is actively taken up across the gills in competition with chloride and disrupts multiple physiological functions (Kroupova et al. 2008). However, nitrite levels in most water supplies are very low. Nitrate is more stable and can be transformed into nitrite. Thus, this section will focus on nitrate.

Nitrate is a pollutant present in mine drainage water as a residual from explosives used in underground blasting operations. Nitrates at high concentrations negatively impacts domestic water supply, livestock drinking water, and are directly toxic to aquatic organisms. Also, nitrate can cause excessive plant and algal growth that depletes oxygen levels and alters pH regimes.

For nitrate + nitrite, the EPA has interpreted the State of Idaho's narrative water quality criteria as follows: 10 mg/L for domestic drinking water supply (based on the EPA's 1986 "Gold Book") and 100 mg/L for agricultural/livestock supply (based on the EPA's 1972 "Blue Book").

Adult salmonids appear to be relatively tolerant to nitrate including the following 96-h LC_{50}s study results: 5800 mg/L for Chinook salmon (*O. tshawytscha*) (Westin 1974), 5941 mg/L for fathead minnows (*Pimephales promelas*) (Scott and Crunkilton 2000), 6000 mg/L for *O. mykiss* (Westin 1974). The CCME (2003) literature review determined that early life history phases of amphibians are the most sensitive species in the literature to nitrate. They recommend a freshwater guideline for the nitrate ion of 13 mg/L NO_3. More recent studies confirm the sensitivity of early life history phases of fish. McGurk et al. (2006) reported lowest observed effects for chronic exposures were in the range of 6.25 to 25 mg/L for lake whitefish (*Coregonus clupeaformis*) and lake trout (*Salvelinus namaycush*) embryos. Camargo et al. (2005) concluded from a nitrate literature review that safe levels below the drinking water standard of 10 mg/L are recommended for protecting the most sensitive freshwater species.

Nitrate-nitrite as a nutrient is not addressed directly by the Idaho water quality standards. A numeric criterion for nutrients is not specified as stream and river productivity varies greatly due to natural water chemistry. Instead, a narrative criterion is applied to prevent eutrophication from human sources. USEPA (2000) published nutrient criteria guidance based on eco-regional specific nutrient conditions, but these recommendations have not been widely adopted by the states (nor by Idaho). In general, the concentrations for nitrates recommended for the Rocky Mountain ecoregion are very low, in the range of 0.2 to 0.5 mg/L. As a comparison, minimally disturbed watersheds in the Salmon River basin have background nitrite + nitrate levels ranging from undetectable levels to 0.225 mg/L (Ott and Maret 2003).

Although continuous effluent discharge of nitrate extended periods 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.

Monitoring data indicate nitrate+nitrite concentrations have been below the EPA-recommended criterion of 10 mg/L for human health at outfall 002 and the Jordan Creek monitoring sites (Figure 9 and Appendix B. Summary of water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), **Site S-9 (Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003).** The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not

collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) and temperature is collected daily at outfalls 002 and 003 when discharging.

Appendix B).

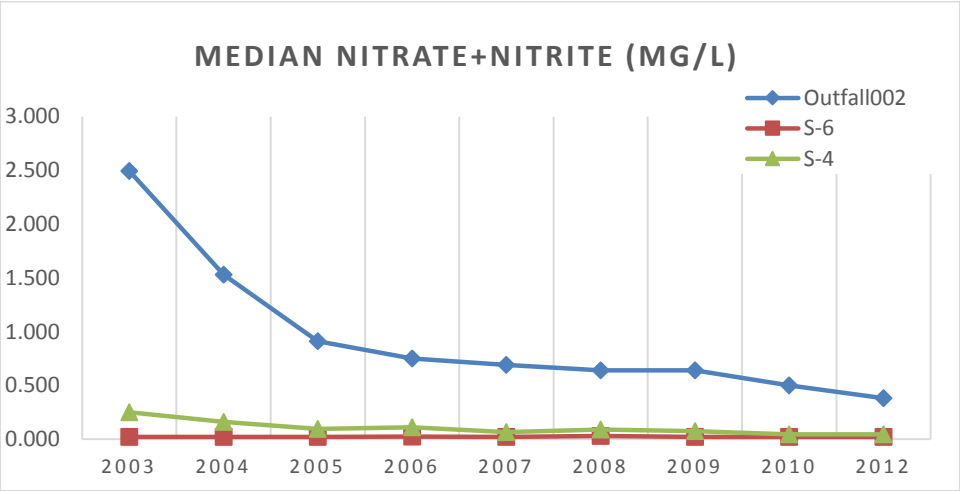


Figure 9. Nitrate+nitrite median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013(Source: Helca Limited).

3. Selenium

Selenium is a naturally occurring mineral that is distributed widely in nature in most rocks and soils. Selenium is considered a rare element because there are no large deposits anywhere in the world. In its pure form, it exists as metallic gray to black hexagonal crystals, but in nature it is usually combined with sulfide or with silver, copper, lead, and nickel minerals. It is mainly obtained as a by-product of the electrolytic refining of copper. Selenium is used in various industries (in photocells, as pigment, rubber products, and lubricants). Anthropogenic sources of selenium include fuel (coal and oil) combustion, metal industries, waste (domestic, municipal, and industrial) disposal and incineration, manufacturing processes, mining, smelting, and refining. Selenium concentrations in aquatic environments are generally low, but can vary widely. Concentrations ranging from 0.12 to 0.44 µg Se/L have been reported in drinking water from worldwide sources (Eisler 1985b).

Selenium exists as selenide (Se^{2-}), elemental Se, selenite (SeO_3^{2-}), and selenate (SeO_4^{2-}) oxidation states. Each oxidation state exhibits different chemical behavior. The concentration, speciation, and association of Se in a given environment depend upon pH and redox conditions, and the solubility of its salts (selenates are more soluble than selenites). Selenide and elemental Se occur in acidic, reducing, and organic-rich environments. Metallic selenides, Se-sulphides, and elemental Se are insoluble, and therefore, biological unavailable. For the pH and redox conditions of most soil and aquatic environments, selenite and selenate should be the dominant forms of Se (McNeal and Balistrieri 1989). A decrease in toxicity with increased water hardness

is not evident for either form of selenium. There is a relationship between selenate and sulfate where selenate is more toxic to fish species in low sulfate water than in higher sulfate water.

Fish appear to be sensitive to selenium toxicity under conditions of long-term exposure from both water and dietary sources. Waterborne selenium is depurated in fish via a passive excretion pathway, while dietary selenium is excreted more actively. The half-life of selenium is inversely proportional to dietary loading. Inorganic selenium absorbed from water is stored in fish as inorganic selenium. However, inorganic selenium absorbed from the diet is transformed by the liver to an organic form that is more toxic, but can be excreted easily (Hodson et al. 1984b). Selenium taken up from water is absorbed across the gills and taken directly to all tissues except the liver. Dietary selenium is taken up through the gut, thus passing through the liver first. The tissue distribution of selenium within fish is a function of the loading rate, but not the source of selenium (Hodson and Hilton 1983).

Due to the sensitivity of fish to long-term low concentration exposures of selenium, the indications of relative sensitivity to waterborne selenium may become reversed when comparing acute and chronic studies. For this reason, comparisons of acute and chronic sensitivities of fish to selenium should be interpreted with caution (Lemly 1985). Hermanutz et al. (1992) also suggest that the estimation of effects using studies of waterborne exposure exclusively may underestimate the danger of selenium exposure to fish. The optimum dietary selenium level in rainbow trout is estimated to be between 0.15-0.38 µg/g by Hilton et al. (1980). However, trout appear to be able to accommodate excess dietary selenium in the short term using both behavioral and physiological adaptations.

Exposure to selenium can reduce fish growth particularly weight and, to a lesser extent, length (Albers et al. 1996; Green and Albers, 1997; Hamilton et al. 1990). At selenium concentrations of 250 µg/L in water, rainbow trout fry growth was reduced following a 21-day exposure (Eisler, 1985b). Weight was reduced by 29-70% in Chinook salmon fed more than 18.2 µg/g for 90 days (Hamilton et al. 1990). Concentrations of 35.4 µg/L for 60 days and 9.6 µg/L for 90 days reduced Chinook salmon body weight and survival (Hamilton et al. 1986). Selenium exposures can also reduce red blood cell volumes and cellular blood iron content in rainbow trout juveniles at concentrations greater than or equal to 53 and 16 µg/L respectively, after 44 weeks. Hatchability of eggs was affected at concentrations as low as 16 µg/L. A slight decrease in the time to hatch was observed at 4.4 µg/L, however the results were not statistically significant when compared to controls (Hodson et al. 1980). Selenium concentrations of 13 µg/L for 6 weeks reduced smolting success of Chinook salmon (Hamilton et al. 1986).

Selenium also affects the immune responses of fish by influencing the activity of glutathione peroxidase (GPX). GPX is an antioxidant that protects cellular membranes and organelles from peroxidative damage that may be caused by superoxide radicals (Felton et al. 1990). At concentrations of 47-50 µg/L in water, selenium exposures were associated with anemia and reduced hatch of rainbow trout (Eisler, 1985b). Investigators observed reduced hatch of eyed embryos of rainbow trout with exposure concentrations of 47 µg/L over 41 days (USEPA 1980). Significant deformities resulted from exposure of rainbow trout eggs to 80 µg/L selenium (Lemly and Smith, 1987).

In chronic exposures (44 weeks) to 130 µg/L selenium caused elevated mortality rates in rainbow trout along with increased incidence of deformities at concentrations as low as 60 µg/L (Hodson et al. 1984b). Significant mortality was observed in rainbow trout eyed-eggs at concentrations greater than or equal to 25 µg/L after 44 weeks exposure (Hodson et al. 1980). In Chinook salmon fry, exposures to 17 µg/L for 30 days caused a significant increase in mortality (Hamilton et al. 1986). The 43-day LC_{50} for Chinook larvae and the 48-day LC_{50} for Chinook fry was 160 µg/L (Eisler 1985b; Lemly and Smith 1987). The 48-day LC_{50} for rainbow trout larvae was determined to be 500 µg/L and significant mortality was observed at 80 µg/L over a 12 month exposure (Lemly and Smith 1987). A literature review by CCME (2001b) summarized studies on chronic selenium toxicity to salmonids. The available data indicate that chronic toxicity of selenium was experienced by salmonids between 12 to 2900 µg/L. Younger age-classes were more affected by selenium under chronic exposure.

In acute selenium exposure tests with rainbow trout, the 96-hour and 9-day LC_{50}s were determined to be 8,100 µg/L and 6,500 µg/L, respectively (Hodson et al. 1980). Similar results were reported by (USEPA 1980) with the rainbow trout, the 9-day LC_{50} estimated to range between 5,400-7,000 µg/L. In bull trout, the LC_{50} was estimated to be 10,200 µg/L (USEPA 1980). The CCME (2001b) reviewed literature on acute selenium toxicity to salmonids. The acute toxicity to salmonids ranged from 7,200-10,000 µg/L. Hamilton and Buhl (1990) observed that selenite-Se is more toxic than selenate-Se with both coho and Chinook salmon. The swim-up fry (8-12 weeks) of coho salmon showed 50% mortality at 7.8 mg/L of selenite-Se or 32.5 mg/L of selenate-Se. These studies also indicated that the younger life stages of both coho and Chinook salmon were more sensitive to the toxic effects. Hamilton and Buhl (1990) demonstrated that, in general, mixtures of selenate and selenite in the ratios of 3.5:1, 6:1, and 14:1 have similar acute toxicity values regardless of fish age. They concluded that the joint acute toxicity of these mixtures to the fish could be characterized as strictly additive.

Lethal effects of selenium can vary among and within species. For example, comparisons of Puget Sound wild and hatchery reared coho salmon fish survival rates showed wild fish rates were 1.5 – 2.0 times higher than those of hatchery reared fish exposed to the same selenium contaminated water (Felton et al. 1990). Selenium residues were also higher in wild fish versus hatchery reared fish.

Due to the ability of fish and invertebrates to bioconcentrate selenium, fish can be exposed to harmful concentrations of selenium via diet even when water concentrations are low. Dietary concentrations as low as 13 µg/g caused elevated mortality, reduced feeding, slower growth, higher feed-to-weight gain ratios and liver paleness in trout within 4 weeks (Hilton et al. 1980). In Chinook salmon, reduced survival was observed at 35.4 µg/g dietary selenium for 60 days and > 9.6 µg/g dietary selenium for 90 days (Hamilton et al. 1990). In Chinook salmon swim-up larvae and fingerlings, 3.2 µg/g selenium in the diet adversely affected growth. Using a bioaccumulation factor (BAF) of 1,800 for aquatic invertebrates, it would be possible to obtain a dietary concentration of 3.2 µg/g at a water concentration as low as 1.8 µg/L. Lemly (1996) proposed a chronic limit of 2.0 µg/L as hazardous to the health and survival of fish. Selenium concentrations at low levels near this limit would primarily act through bioaccumulation. The current USEPA (2009) recommended freshwater water quality criteria for selenium is 5.0mg/L Criterion Continuous Concentration (CCC, an estimate of the highest concentration of a material

in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect). USEPA also has a draft selenium fish tissue criterion for protection of aquatic life of 7.91 µg/g dry weight (USEPA 2004). Note: this criterion is controversial at this time as a review conducted by personnel from other federal agencies determined that there is evidence to support that the criteria should be lower to be protective of ESA species—approximately 5µg/g (Skorupa et al. 2004).

Unlike most other effluent constituents assessed in this BE, the most sensitive endpoints for selenium toxicity in natural populations of fish and aquatic dependent wildlife are measures of reproductive success. Thus, the best tissue for risk assessment is reproductive tissue such as eggs or ovaries. Obtaining reproductive tissue is often difficult because breeding/incubation periods are seasonal and because breeding habitat is spatially limited. Whole-body tissue is used as the endpoint due to these constraints. Any criteria based on whole body concentrations have to be translated into what is relevant to the more crucial reproductive endpoint.

The USEPA is currently in the process of revising the selenium chronic criterion (USEPA 2014). Because the reproductive endpoint is so important, expressing the new criterion as both an egg-ovary concentration and two water column concentrations (one for lentic waters and one for lotic waters) is being considered. The egg-ovary concentration is the most consistent toxicity endpoint across the broadest range of fish species. The two water column concentrations will be derived using conservative translations of the egg-ovary concentration to water concentrations for lentic and lotic waters and are intended to be protective in the vast majority of cases. State adoption of new water concentrations will allow the development of permits without the need for tissue to water translations, whereas the egg-ovary concentration can be used as the basis for developing site-specific water concentration translations of the criterion if desired. Instructions on how to develop such site-specific translations will be included in the criteria document.

Because selenium does bioaccumulate, it can have a protracted effect on the aquatic environment and species beyond the area of initial dilution. Therefore, it is important to consider effects outside of the acute/chronic concentration zones and consider potential effects of selenium that may accumulate in downstream areas and the possibility of bioaccumulation by biota downstream of the Action Area. In order to assess the likelihood that the local aquatic environment will process Se in a way that is conducive to selenium bioaccumulation the selenium cycle must be considered. The following is summarized from Lemly (1999).

Once released into the aquatic system Se has three general fates: 1) absorbed or ingested by organisms 2) bind to particles or enter sediments by adsorption, complexation, and co-precipitation processes, or 3) remain in solution. Through time, most Se will be taken up by organisms or deposited in sediment. Ongoing biological, chemical and physical processes cause Se to be cycled in and out of sediments. For example, direct uptake by primary producers can remove the Se from the water column and then it can precipitate into the sediments by deposition process of dead organic matter. Selenium is usually accumulated in the top layer of sediments. Dynamic processes in the aquatic system cause sediment to act as only a temporary repository for Se. There are several processes in aquatic systems that result in mobilization of Se from sediment into the food chain (e.g. sediment oxidation and methylation by plant roots, physical perturbation and chemical oxidation, oxidation by plant photosynthesis).

Sediment is important in selenium cycling in the environment. Therefore, the physical setting of the waterbody of concern must be consider as it will influence the ability of selenium to be accumulated or to be flushed downstream. Sediments that are non-mobile, high in organics, and have high biologically activity, promote the cycling of selenium through biological pathways. Aquatic systems that accumulated selenium are shallow, low gradient waters with low flushing rates. These types of water bodies would be more likely to trap selenium and allow for direct uptake by biota. Conversely, in streams with higher gradients, little available fine sediment would not efficiently trap selenium and flushing of selenium through the system before it could be processed by biota or deposited would likely occur.

VI. EFFECTS DETERMINATIONS

This section evaluates the possible effects of the effluent limit concentrations on the listed species. The chemicals/constituents of the discharge that were found to have reasonable potential as stated in the Fact Sheets are addressed. This section is organized into the follow subsections:

A) Methodology for effect determinations
B) Species with limited exposure
C) TSS end-of-pipe criterion
D) pH end-of-pipe criterion
E) Effects of metals (cadmium, copper, lead, mercury, silver, zinc) and cyanide at and beyond the edge of the mixing zones.
F) Effects of metals and cyanide within the mixing zones.
G) WET effluent limits and mixtures
H) Effects related to bioaccumulation of selenium and mercury
I) Biomonitoring results

A. Methods for Determination of Effects

The ‘determination of effects’ was made by analyzing the distribution and mobility of the listed species in relation to the discharge and the possibility of exposure (exposure pathways). Where an exposure pathway was incomplete or did not result in significant exposure, then a “not likely to adversely affect” determination was made. For those species where there was a possibility of exposure, the effluent limits or the criteria for the edge of the mixing zone analysis were compared to published toxicity data for that species or a surrogate species. Monitoring data from receiving waters and outfalls were used as another line of evidence to evaluate effects to listed species in the action area. These steps are discussed in the following paragraphs. Both outfalls are within Critical Habitat designation for all listed salmonid species.

Besides evaluating the effect of the action on the species, the action must be considered in terms of whether or not it will alter Critical Habitat and appreciably diminish its value for both survival and recovery of the listed species (NMFS 1996). NMFS and USFWS have identified PCEs for ESUs for Chinook, steelhead and bull trout. These are listed in section III.B (Page 22) of this document. The main PCE for ESA fish habitat addressed in this BE is water quality and issues related to water quality. Water quality is considered a PCE at all life history phases for all three of these species determined to be present in the action area. High water quality, both physical

(e.g. temperature and entrainment of sediment) and chemical characteristics, is necessary for these species at all life history phases. Other critical features of fish habitat are addressed under the rubric of water quality (these are considered indirect effects). Other aspects of critical habitat affected by water quality including the following:

- Food — water quality influences fish prey populations (e.g. macroinvertebrates).
- Space — water quality can influence growth of macrophytes and algae, affecting habitat space.
- Passage conditions — water quality can influence fish instream movement by creating thermal barriers or toxic conditions.
- Substrate quality — deposition of entrained fine sediment decreases spawning habitat and rearing space.

For the species that are present and have complete exposure pathways, the potential for effects was evaluated both within the mixing zone and beyond the edge of the mixing zone. Determinations of effect were made as follows:

"May affect, and is likely to adversely affect" (LAA) means that listed resources are likely to be exposed to the action or its environmental consequences and will respond in a negative manner to the exposure. This determination was made if 1) no information was available detailing the toxicity of the chemical with regard to the species of concern or an acceptable surrogate or 2) the published toxicity data indicated adverse effects at concentrations at or below the established water quality criteria (for the edge of the mixing zone determinations) or effluent limits (for the mixing zone determinations), 3) other consequences to critical habitat besides chemical exposure were determined to be significant.

"May affect, but not likely to adversely affect" (NLAA) means that all effects are beneficial, insignificant, or discountable. This determination was made if the published toxicity data indicated adverse effects at concentrations above the established criteria or effluent limits. The effects analysis is provided in the Biological Evaluation of Idaho's Water Quality Standards for toxic substances (USEPA 1999). The EPA notes that there is not agreement on the protectiveness of these standards with NMFS (2014) and USFWS (2015). This is reviewed in Section VIII.B. Adverse effects on species were divided into sublethal and lethal effects. Sublethal effects included any measurable or observable effect on a species, not including mortality, while lethal effects consisted only of mortality. Both lethal and sublethal effects were evaluated for each criterion and limit. This determination is made if other consequences to critical habitat related to water quality (besides chemical exposure) are determined to be insignificant.

USEPA has evaluated and approved Idaho's Water Quality Standards criteria for water quality constituents that are of concern in this BE at the edge of the mixing zone and beyond. Thus, these standards have been reviewed in the context of their protectiveness to aquatic biota. These standards act as the base of the analysis of effect. Adjustments to this approach when these criteria appear to conflict with the consensus from the literature are incorporated into the effects determination. For the mixing zone determinations, indirect effects (e.g. food) and the ramifications of the loss of the habitat area within the mixing zone were also addressed. More

specific information as to how effects were determined are included in each of the following subsections.

The analysis of the potential effects of toxic pollutants on listed species included the examination of research conducted primarily with surrogate species. The surrogate species were selected as the closest related organism for which information was available. For example, little research exists describing the effects of toxic chemicals on Chinook and bull trout, but much information exists describing the effects of toxic chemicals on rainbow trout. Therefore, rainbow trout often served as a surrogate species to determine the effects of toxic pollutants on other salmonid species.

B. Effects Determination for Species with Limited Exposure

1. Species Presence and Exposure Pathways

This subsection summarizes the information presented in Section III regarding the presence of the species in the permit action area and combines this information with exposure pathways of concern to determine which species to carry forward into the effect determinations.

The timing of effluent discharge determines the exposure of particular life history stages for the fish species present. Both outfall 002 and 003 have the potential to discharge during all times of the year (Figure 10 and Figure 11). Discharge pattern is similar to the basin flow discharge characteristic of higher discharge April –June (see Figure 5). Based on information presented in Section III, the life history stages of the listed fish species present in the action area are as shown in Table 15. The EPA notes that these are based on available information that includes monitoring surveys for fish and redd estimates for population surveys coupled with habitat availability. Fish species that are exposed to the effluent have two routes of exposure from direct contact (internal and external) and dietary through prey consumption

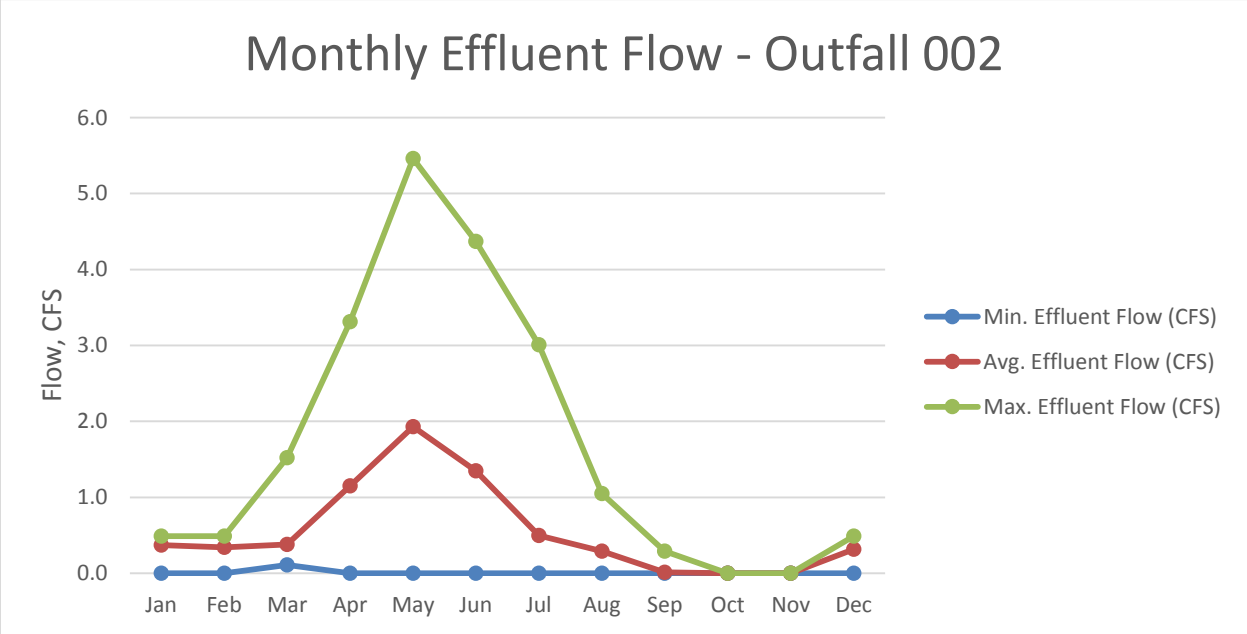


Figure 10. Outfall 002 monthly discharge based on data 2012 through February 2016 (Source: CERCLA data provided by Brian Nickel, USEPA).

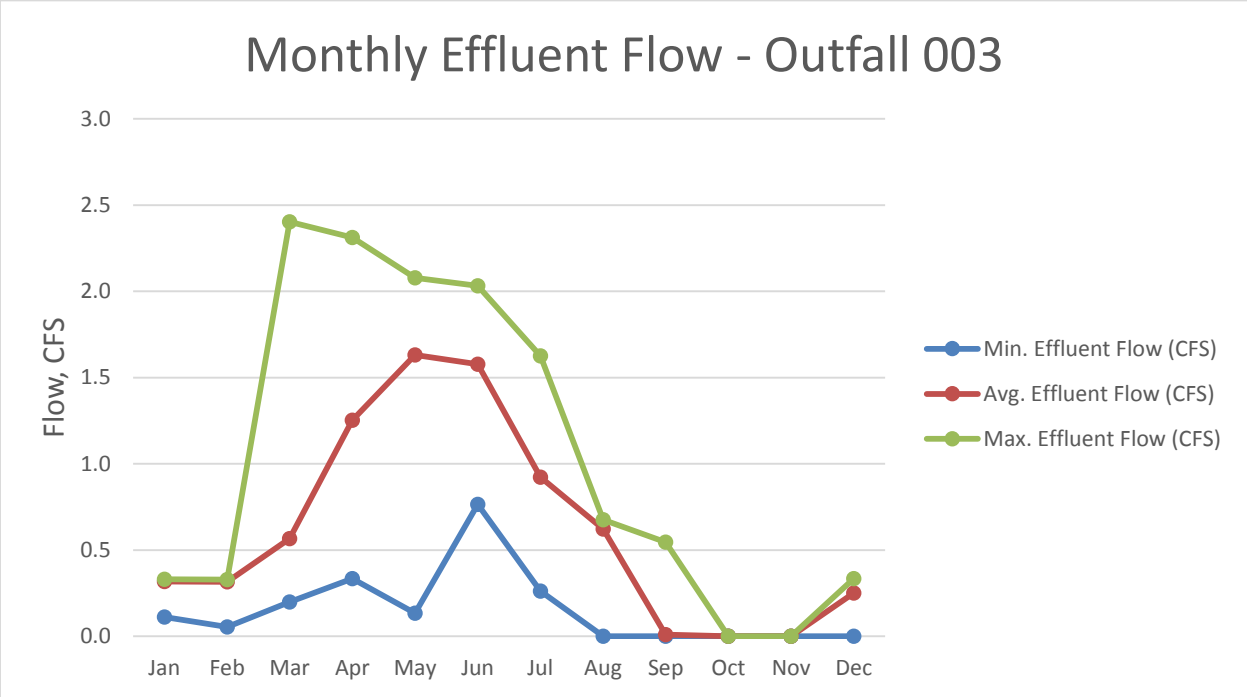


Figure 11. Outfall 003 monthly discharge based on data 2012 through February 2016 (Source: Cercla data provided by Brian Nickel, USEPA).

Table 15. Overview of likely exposure by life history stage of listed salmonid species at each outfall.

Species	Jordan Creek (002)	Yankee Fork (003)	Salmon River
Snake River sockeye	—	—	adult migration and juvenile outmigration
Snake River spring/summer Chinook salmon	juvenile rearing, spawning, and egg incubation in lower reaches near Yankee Fork confluence	adult holding, spawning, egg incubation, juvenile rearing	adult migration, holding, spawning, egg incubation, juvenile rearing, outmigration
Snake River summer steelhead	adult holding, spawning, egg incubation, juvenile rearing	adult holding, spawning, egg incubation, juvenile rearing	adult migration and holding, juvenile rearing and outmigration
Columbia River DPS bull trout	resident population, all life history stages	adult holding, spawning, egg incubation, juvenile rearing	adult migration and holding, juvenile rearing and outmigration

Snake River Sockeye are not distributed in the stream reaches that would result in exposure to effluent from outfalls 002 and 003. This population uses the mainstem of the Upper Salmon River as a migration corridor for returning adults and outmigrating juveniles but does not access Yankee Fork or Jordan Creek at any stage of their life history (as discussed in Section III.B.).

The confluence of the Upper Salmon River is well downstream of the location of the outfall 003 on the Yankee Fork (8 miles). The possibility of exposure of Snake River Sockeye to the effluent is extremely remote as these fish do not use the Action Area and the dilution that occurs over this distance rendering the effluent as not a risk of toxic effects. **From this information, the USEPA has determined that the proposed Grouse Creek Unit NPDES Permit is not likely to adversely affect the Snake River Sockeye.**

C. Effect Determination for TSS

The permit limits TSS concentrations to a monthly average of 20 mg/L TSS and a daily maximum of 30 mg/L TSS applied at both outfalls. The information that is available suggests that sublethal effects are possible at exposure concentrations ≤ 20 mg/L but generally at exposure durations ≥ 60 days for adults and ≤ 6 days for larvae (Newcombe and Jensen 1996).

The TSS concentration at outfall 002 is typically below 10 mg/L. TSS has rarely exceeded 25 mg/L over the monitoring period and the maximum TSS value reported over all of the years is 88 mg/L (Figure 12 and Appendix B. Summary of water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), **Site S-9 (Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003).** The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) and temperature is collected daily at outfalls 002 and 003 when discharging.

Appendix B). Spikes of over 20 mg/L are rare, occurring during the spring runoff period or during storm events. Outfall 003 TSS concentrations are typically ≤ 4 mg/L (Figure 13) and the maximum value reported over all sample years is 180 mg/L.

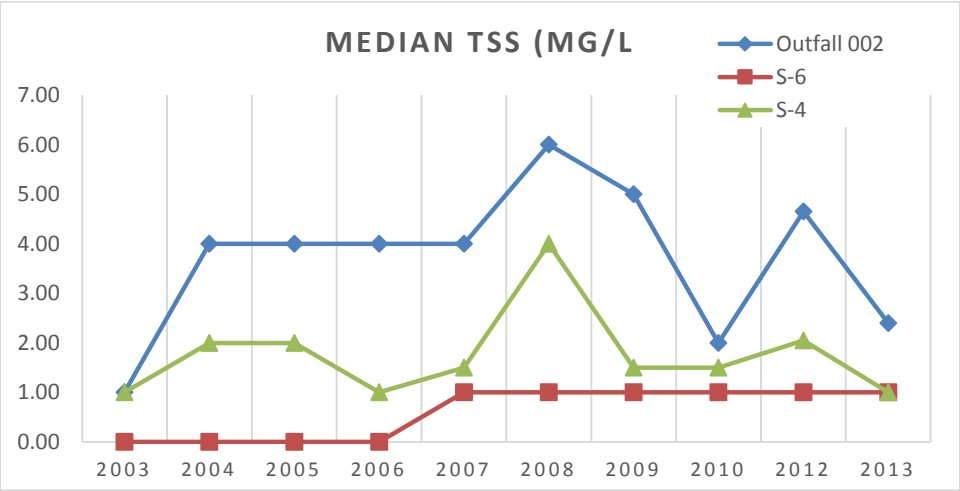


Figure 12. TSS median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013 (Source: Helca Limited).

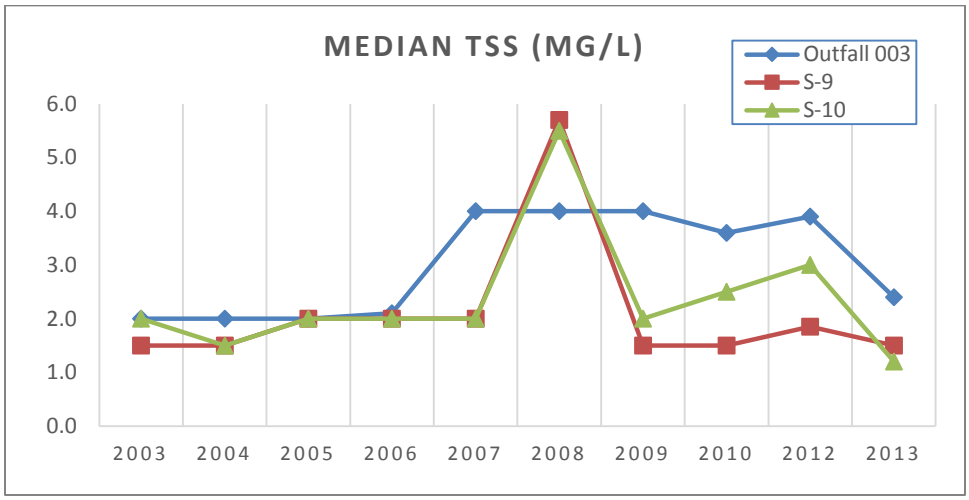


Figure 13. TSS median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013 (Source: Helca Limited).

The TSS limit (<20 mg/L monthly and 30 mg/L daily maximum) prescribed would have a negligible effect on these species as these are lower than the concentrations that may cause sublethal effects. TSS data collected at the outfalls has median and mean values ≤ 20mg/L. Exceedances have occurred during storm and runoff events resulting in short term elevated TSS. **Therefore, USEPA has determined that TSS discharge at the proposed permit limits is not likely to adversely affect Snake River Spring/Summer Chinook salmon, Snake River steelhead, and bull trout.**

Currently, deposition of fine sediment is considered a limiting factor in both Jordan Creek and the Yankee Fork (USFS 1994 and USFS 1995). This is considered a condition that is beyond a background expected condition for these streams. Therefore, the greater concern is for the deposition of fine sediment that would contribute to fine sediment in the substrate. Sediment containment at the Grouse Creek action area results in TSS levels of generally ≤ 20 mg/L. This low level of TSS contributed from the discharge to bedload sedimentation would be very minor.

Flocculants and coagulants

Related to discharge of TSS is the use of chemical coagulants and flocculants. Grouse Creek Mine states that they do not currently use coagulant as it is not considered effective in their particular treatment process. They do use a dry anionic polyacrylamide polymer flocculant product 'Z Flocc 565', prepared with site water. The Mine states the following regarding use of flocculants: 'Grouse Creek targets the optimum flocculant concentration to obtain the most efficient settling characteristics of solids in the effluent. Excess flocculant would hinder formation and settling of solids and is avoided' (Personal Communication to Brian Nickel from Brant Tritthart, Hecla Mining 11/24/15). From this description, they are using the amount to accomplish the settling. If the use of flocculants were excessive, then floc would likely be observed in the substrate in the vicinity of the outfalls. This would cause a degradation of sediment quality for T&E fishes that may use this habitat. We do not have recent observations of

the substrate to verify that a floc is not present. However, the EPA has not observed or found any reports of floc observed in the past. The EPA notes that concentration of flocculant in the effluent cannot be determined as there are no tests available to make this analysis.

D. Effect Determination for pH

The proposed permit requires pH to be within the range of 6.5 to 9.0 standard units for all outfalls. The lower limit (6.5) is based on Idaho water quality standards with no mixing zone applied. The higher limit (9.0) is based on both technology-based effluent guidelines and the Idaho water quality standards.

Although most studies looked at the effects of pH on adults, the life stages most sensitive to effects of pH are egg incubation and alevin/fry development. Data regarding the effects of pH on the aquatic biota are limited and dated. Studies on the effects of pH on salmonids are usually ancillary to other objectives of the research.

In the development of USEPA's criteria, (USEPA 1986) two bioassay references on freshwater fish showed a lower pH limit of about 6.5 for normal development (EIFAC 1969; Mount 1973). Vulnerable life stages of Chinook salmon are sensitive to pH values below 6.5 and possibly at pH values greater than 9.0 (Marshall et al. 1992). For Chinook, Rombough (1983) reported that low pH decreases egg and alevin survival, but specific values are lacking. Adult salmonids seem to be at least as sensitive as most other fish to low pH including rainbow, brook, and brown trout and Chinook salmon (ODEQ 1995). In studies of biological changes with surface water acidification, Baker et al. (1990) found that decreased reproductive success may occur for highly acid-sensitive fish species (e.g., fathead minnow, striped bass) at pH values of 6.5 to 6.0. At pH values between 6.0 and 5.5, Baker et al. (1990) found decreased reproductive success in lake trout. The lower critical pH value for rainbow trout is approximately 5.5 (Baker et al. 1990). Based on the USEPA criteria documents and Baker et al. (1990), the low-end of Idaho's pH standard of 6.5 is considered protective for salmonids.

At the higher end of the pH scale, even less is known regarding effects on fish. In USEPA's review for water quality criteria development, the upper limit of 9.0 was obtained from only one reference (EIFAC 1969). Though no recent data have been generated, studies conducted earlier in the 20th century show salmonids, including both trout and salmon species, to be sensitive to pH values in the range of 9.2 to 9.7 (ODEQ 1995). Non-salmonid fish are, with some exceptions, more tolerant of high pH, with sensitivity appearing at or over pH 10 for most species tested (EIFAC 1969). Levels of pH greater than 9.0 may adversely affect benthic invertebrate populations, thereby altering the food base for salmonids (ODEQ 1995).

Monitoring data collected at the outfalls indicate that pH is within the acceptable range for salmonids. Monitoring data from both outfalls show the effluents generally slightly elevated from the neutral value of 7.0 pH. Outfall 002 annual median pH values range from 7.0-7.3 and minimum and maximum values overall years range from 6.6-8.8 (Figure 14 and Appendix B. Summary of water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), **Site S-9**

(Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003). The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) and temperature is collected daily at outfalls 002 and 003 when discharging.

Appendix B). At outfall 003, annual median pH values range from 7.2-7.6 and minimum and maximum values overall years range from 6.3-8.9 (Figure 15 and Appendix B. Summary of water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), **Site S-9 (Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003).** The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) and temperature is collected daily at outfalls 002 and 003 when discharging.

Appendix B). Also, data from upstream and downstream monitoring sites are comparable. **Therefore, USEPA has determined that discharge within 6.5-9.0 pH permit limit in outfalls 002 and 003 is not likely to adversely affect Snake River Spring/ Summer Chinook salmon, Snake River steelhead, and bull trout.**

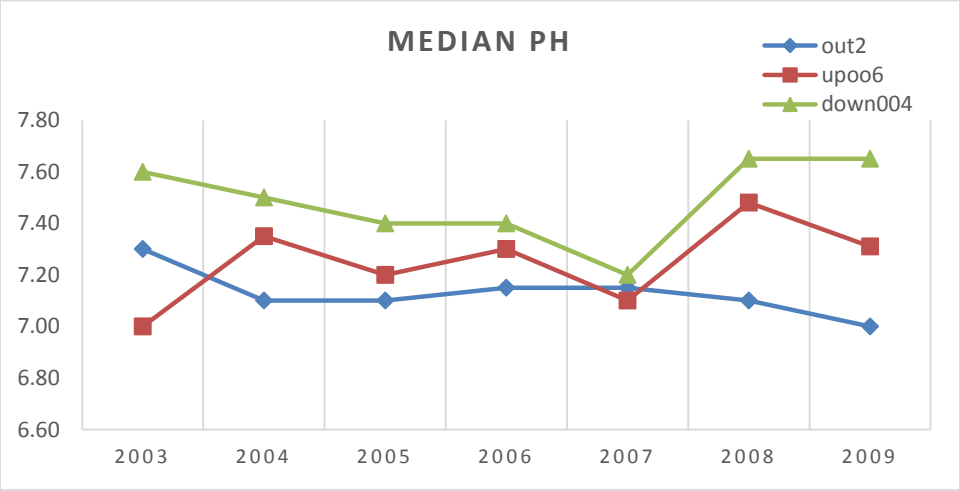


Figure 14. PH median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013 (Source: Helca Limited).

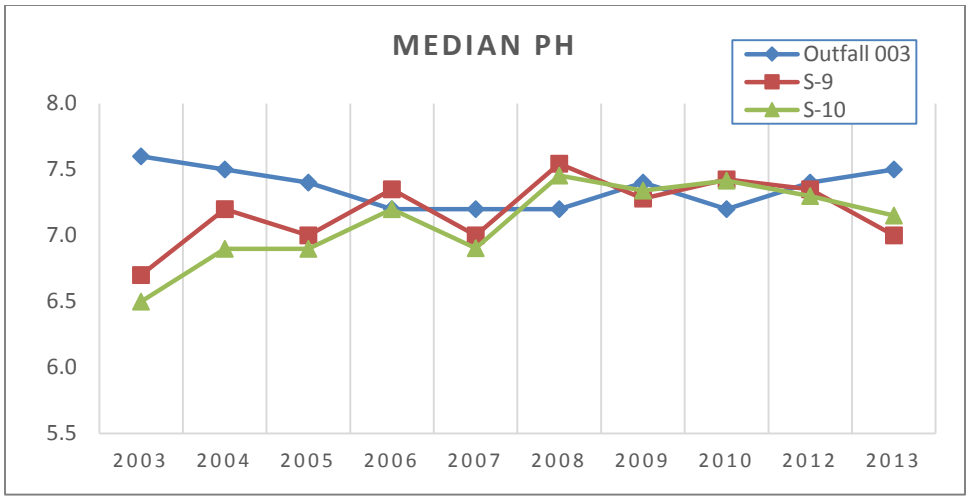


Figure 15. PH median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013(Source: Helca Limited).

E. Effect Determination for Metals and WAD Cyanide At/Beyond the Edge of Mixing Zone

Effects are determined separately within and at/beyond the mixing zone. Effluent limits are based on achieving water quality criteria at the edge of the mixing zone and in the case of metals and cyanide, water quality criteria have already been evaluated in the Idaho BA (USEPA 1999). The effluent limits for the metals cadmium, copper, lead, mercury, and zinc, WAD cyanide, and WET were calculated to ensure compliance with aquatic life water quality criteria at the edge of the mixing zone.

USEPA (1999) has made determinations regarding the potential for the metals and cyanide criteria established by the Idaho water quality standards to adversely affect threatened and endangered species as follows: 1) acute criterion were compared to published toxicity data where exposure durations were ≤ 96 hours and 2) chronic criterion were compared to published toxicity data where exposure durations were > 96 hours. Based on USEPA's analysis in the 1999 Biological Assessment, cadmium, copper, lead, mercury, silver, zinc and WAD cyanide are not likely to adversely affect Chinook salmon, steelhead, or bull trout at the concentrations set forth by Idaho DEQ in the Idaho water quality standards. Because the effluent limits are based on achieving the acute and chronic criteria at the edge of the mixing zone, **USEPA has determined that discharge at the effluent limits for cadmium, copper, lead, mercury, silver, zinc and WAD cyanide is not likely to adversely affect the Snake River Spring/Summer Chinook salmon, Snake River steelhead, and Columbia River bull trout at and beyond the edge of the mixing zones for outfall 002 and outfall 003.** NMFS (2014) and USFWS (2015) BOs on the EPA's approval of Idaho's water quality standards for toxic substances concluded that the water quality criteria for several of the parameters discharged by the Grouse Creek Unit are likely to jeopardize the continued existence of Snake River basin steelhead, Snake River Spring/Summer Chinook salmon, and bull trout. NMFS also concluded that these criteria would have an adverse effect on EFH. All of the criteria considered in the 2014 and 2015 BOs remain in effect, thus, the EPA must implement these criteria in Clean Water Act actions, including NPDES permits. The BOs include RPAs, including "interim" RPAs that apply prior to the State

of Idaho's adoption and the EPA's approval of revised criteria. The draft permit complies with the RPAs in the BOs. These are discussed in Section VIII.B.

F. Effect Determination for Metals and WAD Cyanide within the Mixing Zone

This section evaluates the effect of the mixing zones for each species of concern. Mixing zones are a long established concept in water quality regulations. Mixing zones are defined as a limited area or volume of water where the discharge plume is progressively diluted by the receiving water. Water quality criteria may be exceeded in the mixing zone, at concentrations that are specific to each discharge, as long as acutely toxic conditions are prevented from occurring and the applicable existing and designated uses of the water body are not impaired as a result of the mixing zone (USEPA 1994). Mixing zones are allowed at the discretion of the State, which authorizes the allowable size and configuration of mixing zones based on state water quality standards regulations. The State's authorization is submitted to USEPA in a CWA Section 401 certification for the NPDES permit. USEPA received this certification for the Grouse Creek Unit permit from IDEQ (IDEQ 2015), which authorized mixing zones for outfalls 002 and 003. In determining the effect of the chemicals within the areas of the mixing zones on the species of concern, the following issues are considered in the analysis:

1. Acute and chronic effects within the mixing zone from either long term exposure (e.g. developing eggs) or repeated short-term exposures (e.g., instream movement of resident fish)
2. Under an avoidance scenario, likelihood that the mixing zones result in a significant/ substantial habitat loss for listed fish species at each life history phase;
3. Likelihood that mixing zone acts as a barrier to movement due to avoidance behavior, and
4. Likelihood that mixing zone width results in necessity of exposure likely when avoidance behavior is not a factor.

Using site specific information and the methodology described in **Section VI. A**. (Methods for Determination of Effects) above, the following subsections evaluate the effect of each of the two metals and WAD cyanide mixing zones on the listed fish species.

1. Outfall 002—Description of mixing zone for metals and WAD cyanide

In general, proper placement, design and configuration of effluent discharge pipe/diffuser will result in a mixing zone that rapidly mixes with stream flow, thereby minimizing the extent of the mixing zone and maintaining an adequate zone of passage for fish. The efficiency of the mixing of the effluent with stream water is affected by several factors including: the physical dimensions of the stream, stream flows, hydrodynamics, placement of the diffuser, stream substrate, and large structural elements. Outfall 002 is straightforward in that the effluent reaches Jordan Creek via the dewatered Pinyon Creek channel. After entering Jordan Creek, the effluent plume hugs the right bank of Jordan Creek.

<u>Outfall 002 acute mixing zone</u>

The chemicals that exceed the acute toxicity criteria within the mixing zone may have the potential to affect bull trout, steelhead, and juvenile Chinook salmon and to eliminate/impair

their habitat. A discharge containing toxic substances at levels that may cause direct mortality (acute toxicity) may result in adverse effects until the discharges are diluted to below the acute criteria. Authorized mixing zones by parameter are listed in Table 4. Although IDEQ's mixing zone authorizations did not distinguish between acute and chronic mixing zones, in effect, there is no mixing zone for acute criteria for cyanide, lead, or mercury, for outfall 002, because both the maximum daily and average monthly effluent limits for these parameters are below the acute criteria.

The parameters with effluent limits that exceed the acute criteria are: cadmium, copper, and zinc. The acute mixing zone is the area in which the receiving water would exceed acute water quality criteria, if the facility were discharging at its maximum daily limit, under critical conditions for receiving water and effluent flow. Table 16, below, lists the downstream extent of the portions of the mixing zones where acute criteria are exceeded for these parameters and the plume travel times to reach that point.

Table 16. Acute mixing zones for cadmium, copper and zinc for Outfall 002.

Parameter	**Receiving Water Flow Tier**	**Downstream Extent (m)**	**Plume Travel Time (sec.)**
Cadmium	< 30 CFS	24.5	53
Cadmium	≥ 30 CFS	24.5	28
Copper	< 30 CFS	24.9	54
Copper	≥ 30 CFS	4.3	3.6
Zinc	< 30 CFS	24.9	54
Zinc	≥ 30 CFS	24.5	28

The effluent limits for cadmium, copper and zinc exceed the acute criteria, and modeling indicates that acute criteria will be achieved within 4.3 – 24.9 meters downstream from the outfall (USEPA CORMIX modeling 2015 –Brian Nickel). Model results for Jordan Creek are in Figures 16-21. Listed fish could be exposed to acutely toxic conditions in this relatively small area. Resident bull trout and steelhead moving through this reach to access habitat up or downstream could have multiple exposures. The travel times for the plume to reach the edge of the acute mixing zones are less than 1 minute. This is well within the TSD's recommendation that travel time through the acute mixing zone should be less than 15 minutes, in order to ensure that organisms are not exposed to concentrations exceeding the acute criteria for more than 1 hour (see Section 2.2.2 of the TSD). Also, this zone of acute mixing is not channel spanning and there is unaffected habit that can be used by fish moving through this portion of stream.

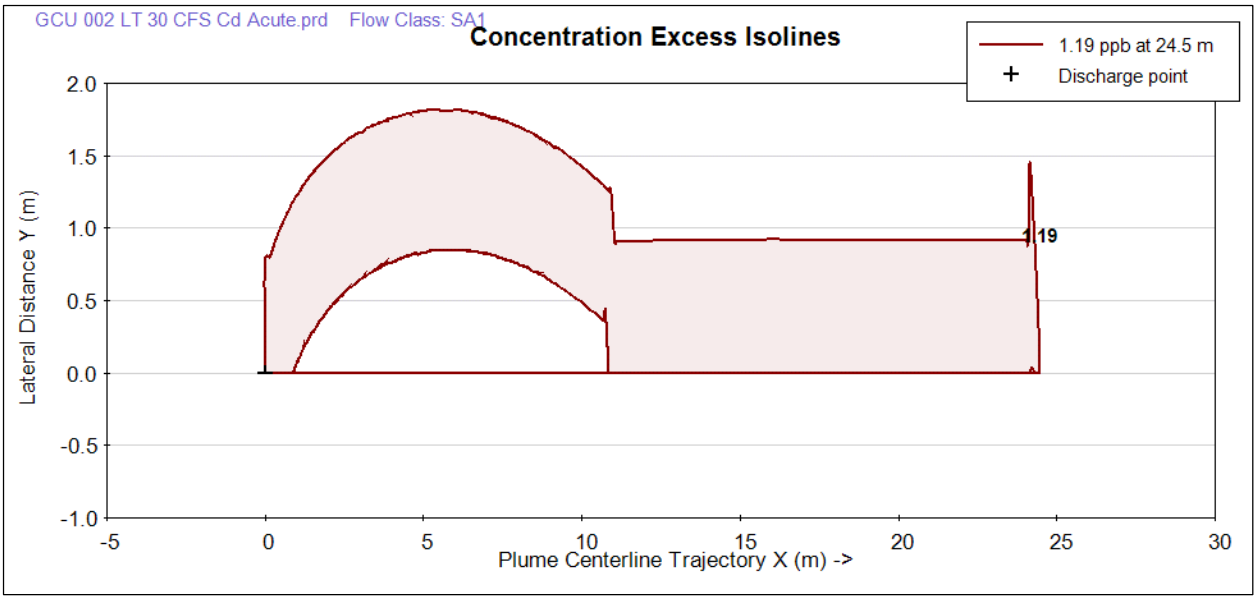


Figure 16. Acute mixing zone for cadmium, with Jordan Creek flows less than 30 CFS.

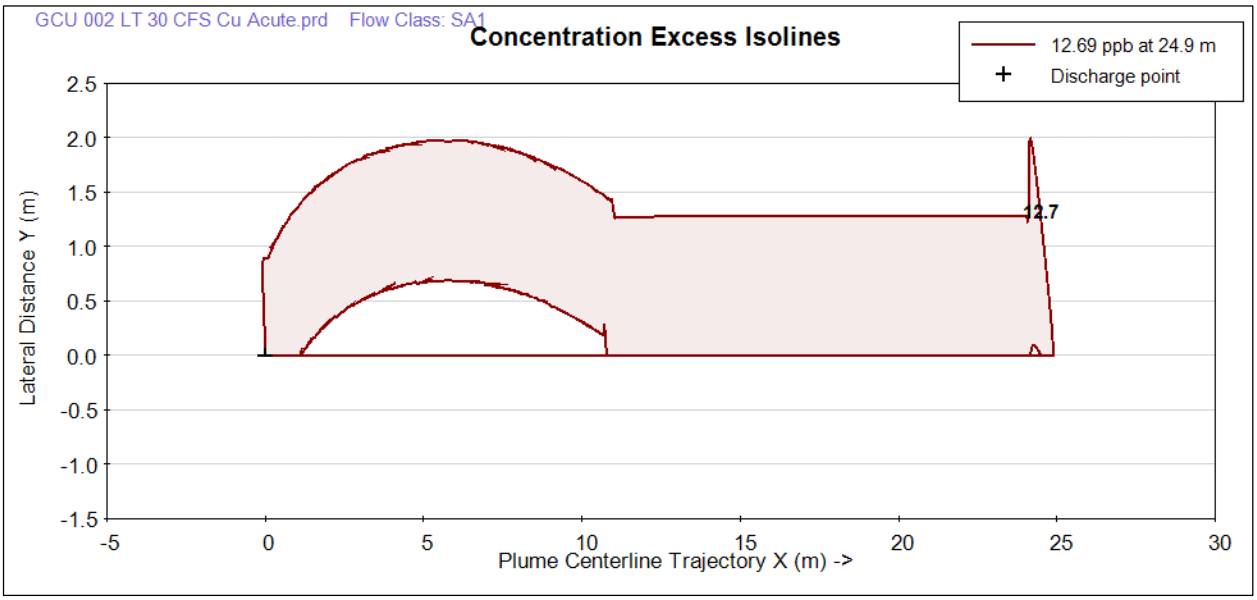


Figure 17. Acute mixing zone for copper, with Jordan Creek flows less than 30 CFS (14.8 µg/L or 12.69 µg/L excess above background of 2.11 µg/L).

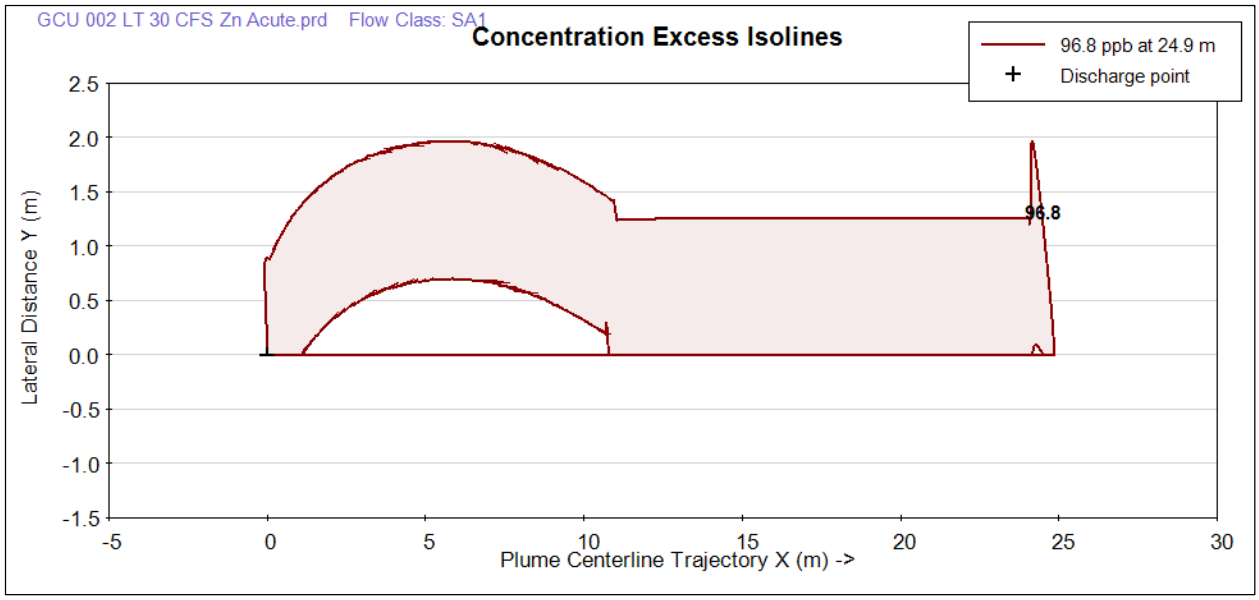


Figure 18. Acute mixing zone for zinc, with Jordan Creek flows less than 30 CFS (103.5 µg/L or 96.8 µg/L excess above background of 6.70 µg/L).

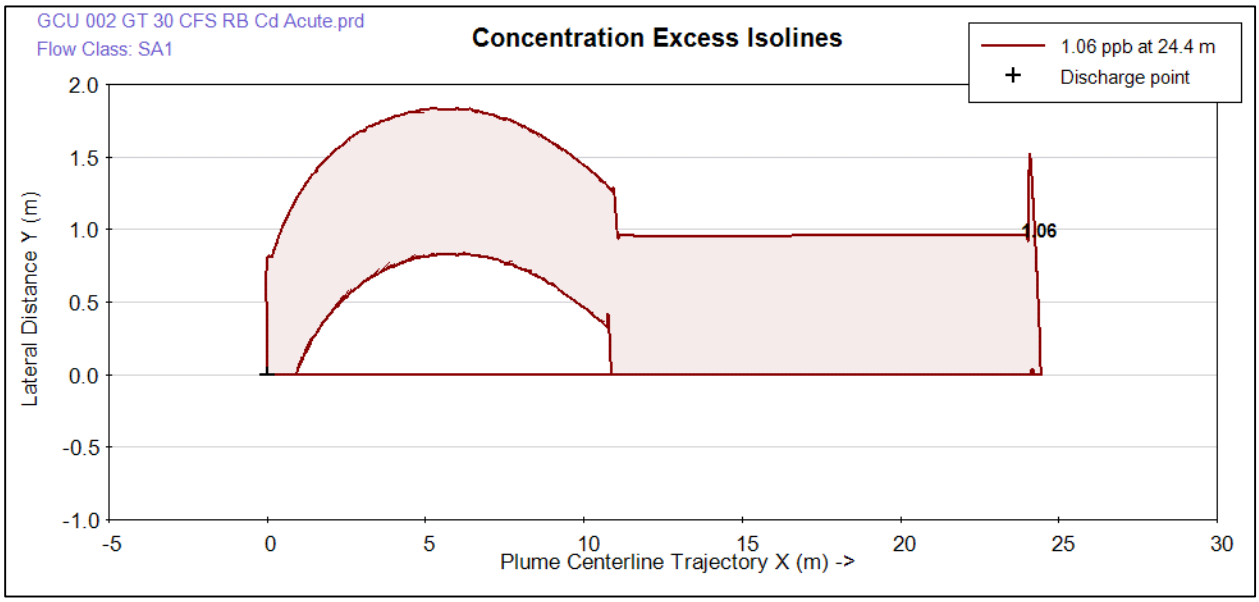


Figure 19. Acute mixing zone for cadmium, with Jordan Creek flows greater than or equal to 30 CFS.

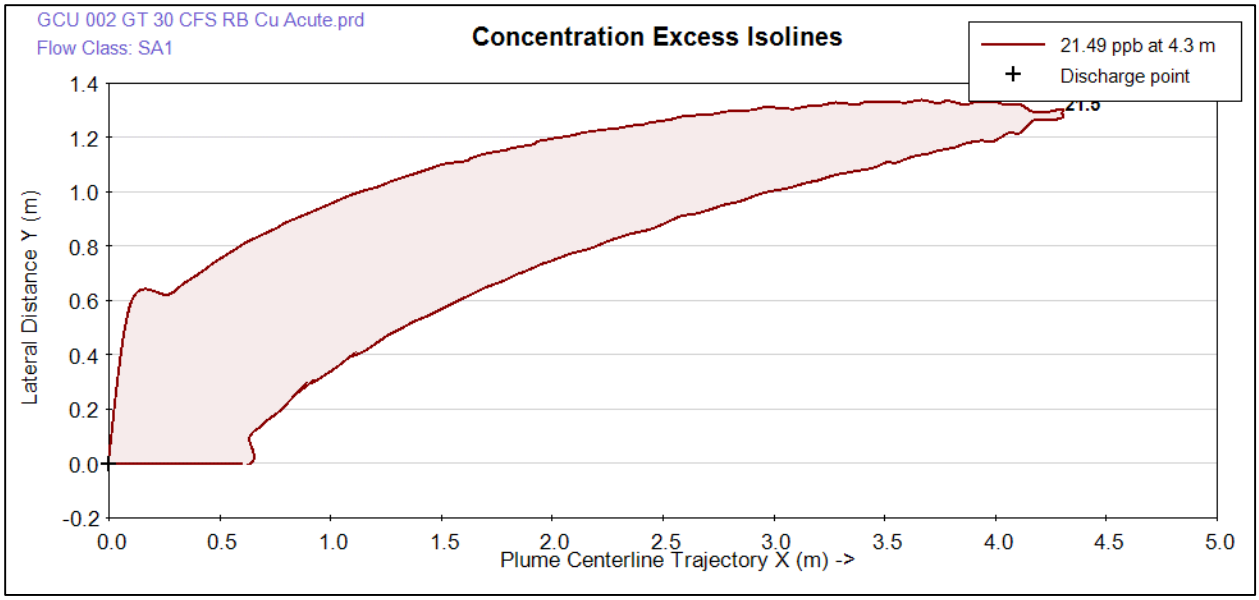


Figure 20. Acute mixing zone for copper, with Jordan Creek flows greater than or equal to 30 CFS (23.6 µg/L or 21.49 µg/L above background of 2.11 µg/L).

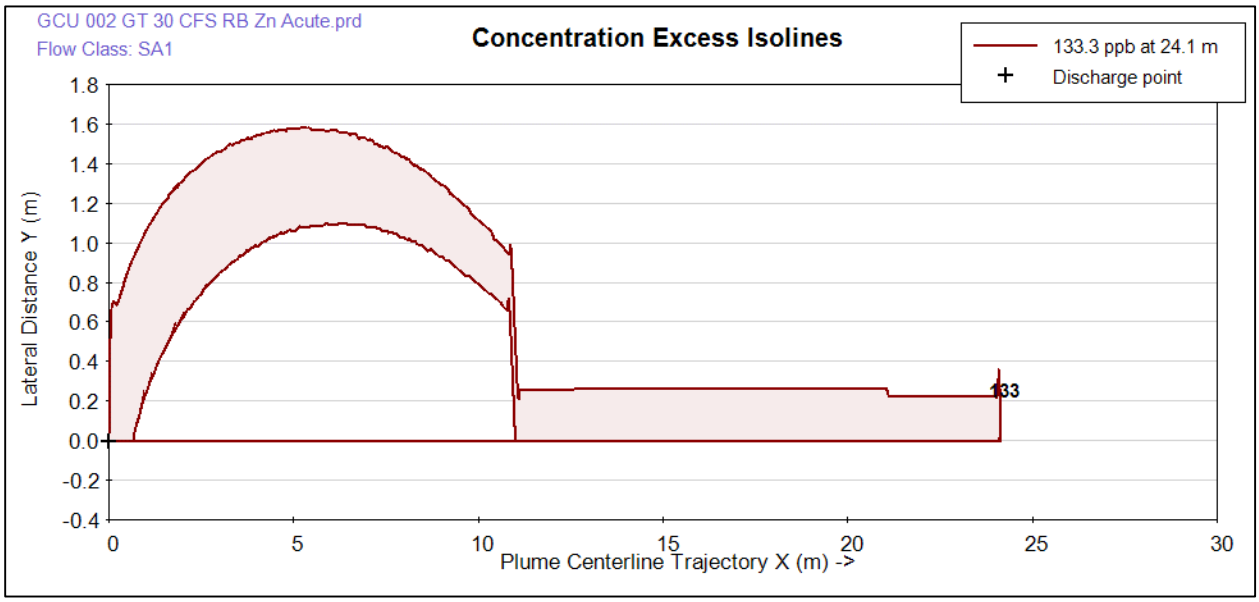


Figure 21. Acute mixing zone for zinc, with Jordan Creek flows greater than or equal to 30 CFS (140 µg/L or 133.3 µg/L above background of 6.7 µg/L).

Discharge of acutely toxic levels of metals would result in degradation of water quality of this Critical Habitat area for Columbia River bull trout, Snake River summer steelhead, and Snake River Chinook salmon. However, the area of this diminished habitat is very small and not channel spanning, and the plume travel time to the end of the acute mixing zone is less than 1 minute. Because of the spatial and temporal limitations of these acute toxic levels, bull trout, steelhead/rainbow trout, and Chinook juveniles are not expected to spend enough time within the area of acute exposure to be adversely affected. **Therefore, USEPA has determined that the**

acute mixing zone of outfall 002 for cadmium, copper and zinc is not likely to adversely affect Snake River Chinook salmon, Snake River steelhead, and Columbia River bull trout.

Outfall 002 chronic mixing zone

The metals concentrations above the chronic criteria could potentially affect spawning and rearing that may occur in the chronic mixing zone (see Table 15 for life history stages by species likely to be present in vicinity of the outfall). The mixing zones for specific chemicals span about 50% or less of the width of the stream and can extend as much as 25 meters downstream under various flow regimes (Figure 22 and Figure 23).

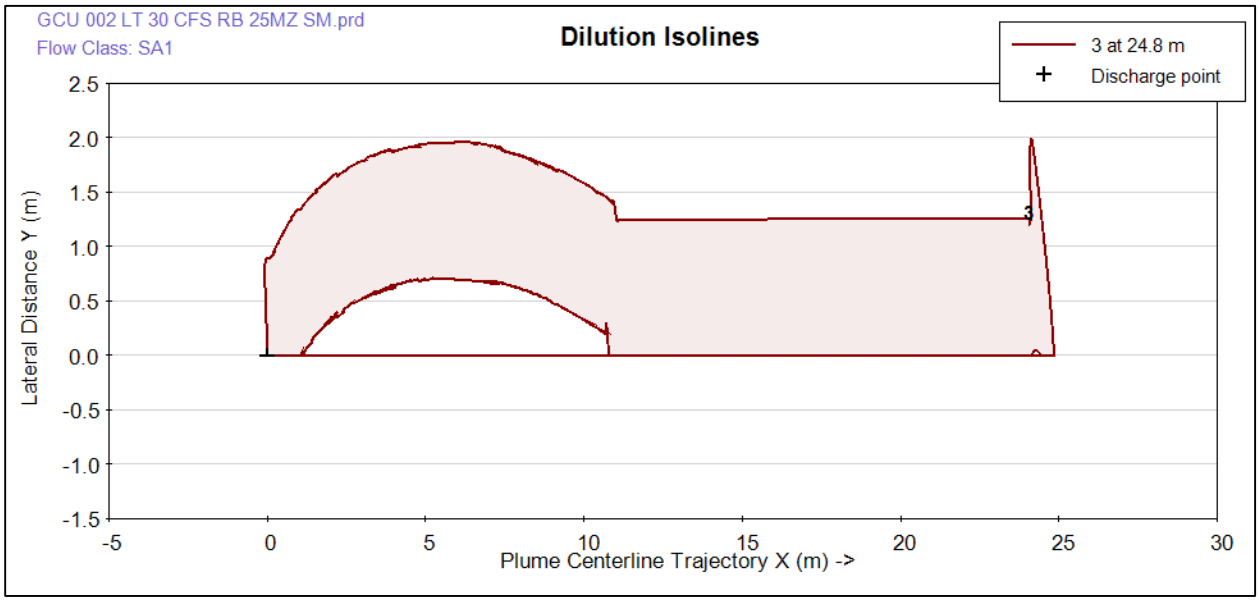


Figure 22. Dilution isoline for outfall 002, with flows less than 30 CFS. The isoline for 3:1 dilution corresponds to the 25% mixing zone for all parameters except WET.

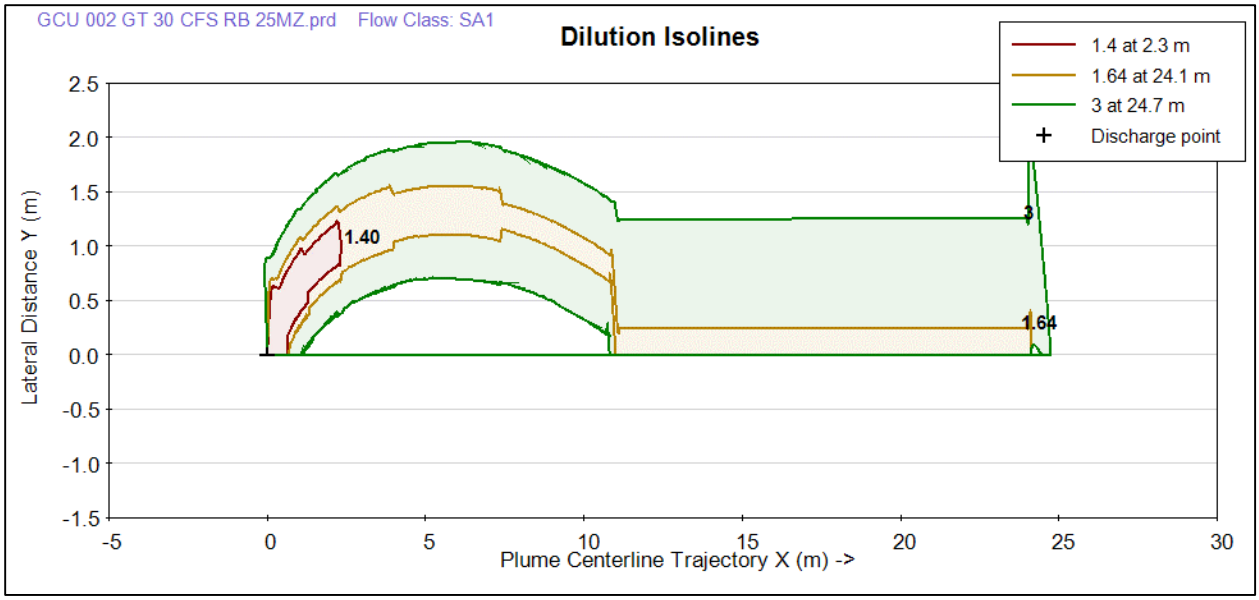


Figure 23. Dilution isolines for outfall 002, with flows greater than or equal to 30 CFS. The isoline for 1.4:1 dilution corresponds to the mixing zone for copper. The isoline for 1.64:1 dilution corresponds to the mixing zone for zinc. The isoline for 3:1 dilution corresponds to the 25% mixing zone for all parameters except copper, zinc, and WET.

<u>Outfall 002 copper and zinc avoidance thresholds</u>

As explained above, avoidance behavior is difficult to predict for mixtures of metals. However, the EPA has used the CORMIX model to estimate the areas where the receiving water may exceed the individual avoidance thresholds for zinc and copper. The EPA used the permit's average monthly limits and the median upstream concentrations of copper and zinc as modeling inputs for this purpose. For zinc, the EPA used an avoidance threshold of 21 µg/L, which is the midpoint of the range of avoidance levels (14 – 28 µg/L) determined by IDEQ (2000). For copper, the EPA used the geometric mean of the avoidance LOECs identified by Shepard and Zodrow (2009) after normalization to 100 mg/L hardness, which is 11.8 µg/L (Figure 6). EPA considers this a reasonable value to use for calculation of avoidance threshold area. In fact, more recent analysis concludes that this calculation results in a conservative threshold (Burt Shephard, EPA, Pers. Comm. 2/4/16). The following points were considered:

1) The threshold concentration for olfaction is lower than the threshold concentration resulting in adverse ecologically relevant effects on a fish. That is to say, you can't have an adverse effect until the fish can smell the contaminant.
2) Therefore any Cu criterion that is protective of olfactory effects will also be protective of olfactory–mediated behavioral effects (DeForest et al. 2011).
3) There are uncertainties in linking olfactory related responses to organism-level behavioral effects, such as avoidance, and in interpreting the ecological relevance of the 'olfaction' endpoint.

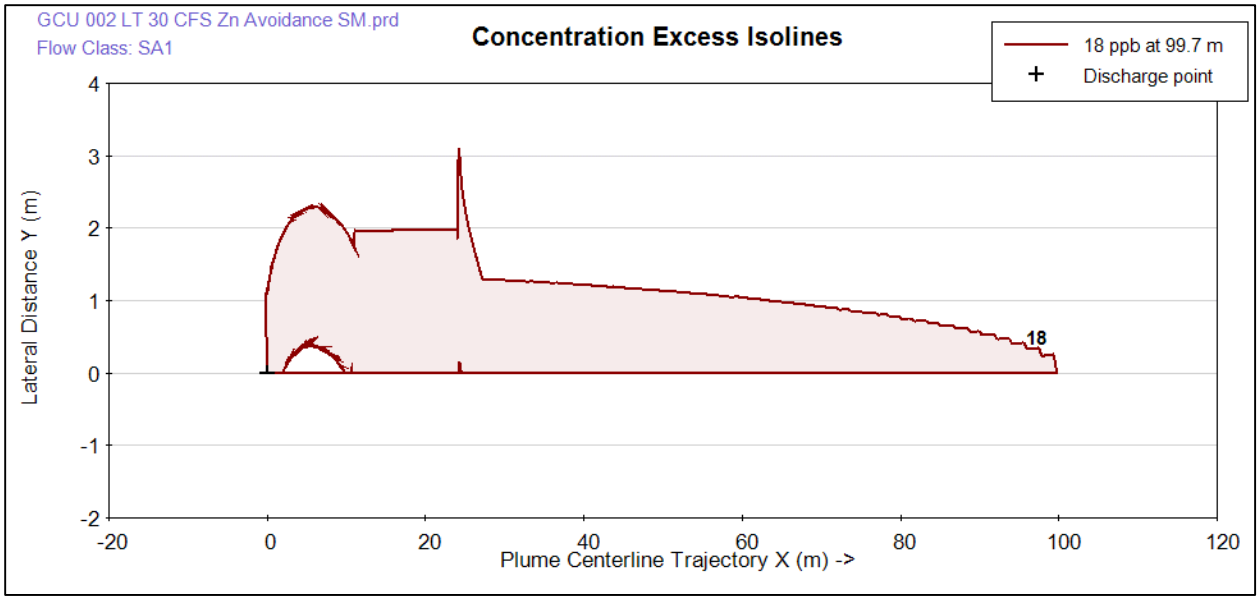


Figure 24. Avoidance threshold concentration isoline for zinc, with Jordan Creek flows less than 30 CFS (21 µg/L, or 18 µg/L excess above upstream concentration of 3 µg/L).

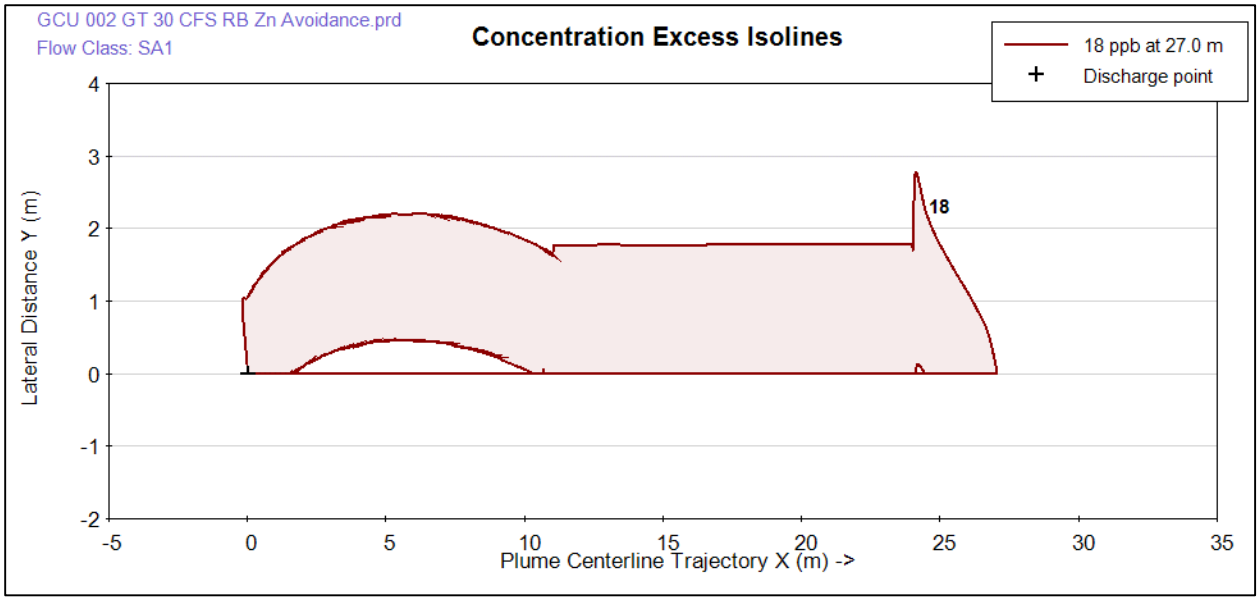


Figure 25. Avoidance threshold concentration isoline for zinc, with Jordan Creek flows greater than 30 CFS (21 µg/L, or 18 µg/L excess above upstream concentration of 3 µg/L).

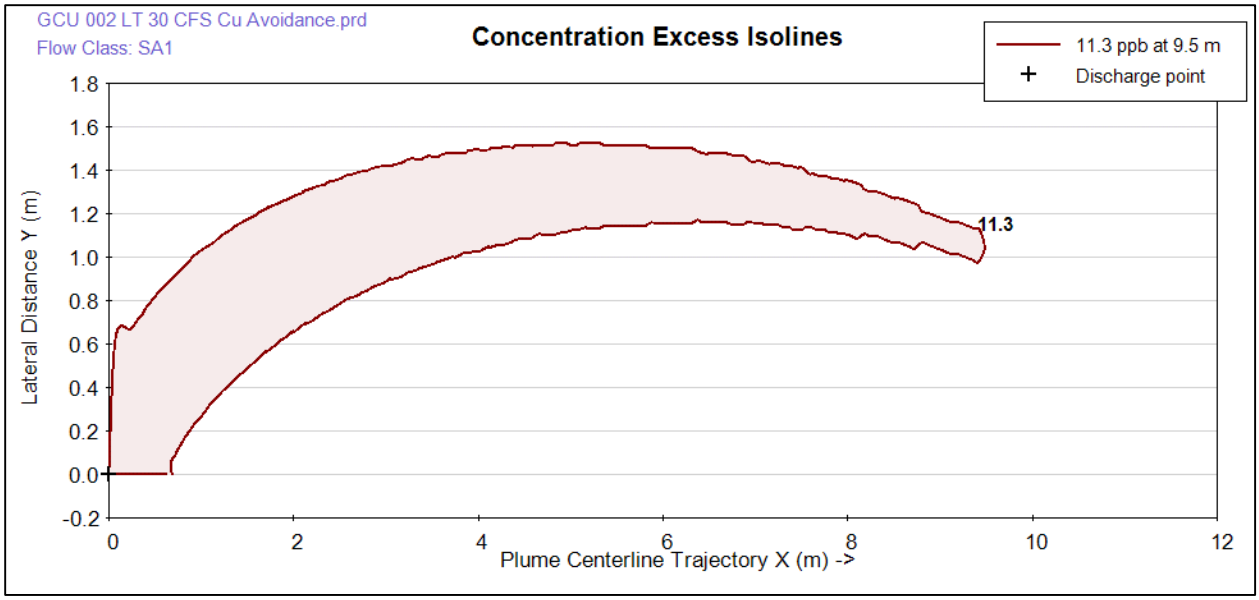


Figure 26. Avoidance threshold concentration isoline for copper, with Jordan Creek flows less than 30 CFS (11.8 µg/L or 11.3 µg/L excess above upstream concentration of 0.5 µg/L).

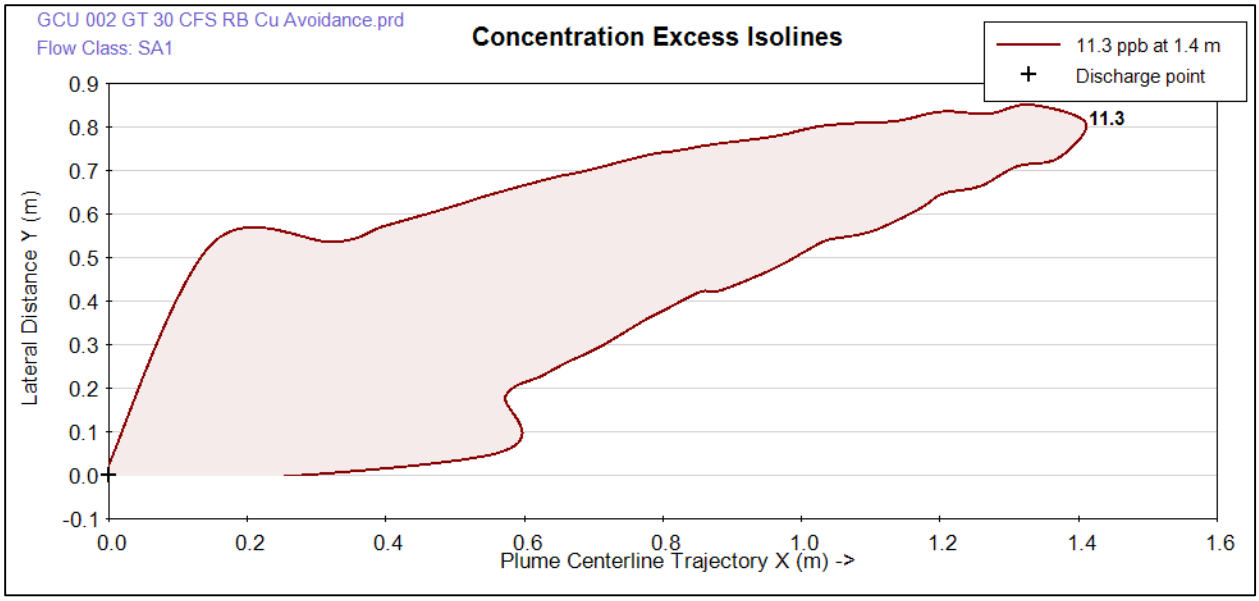


Figure 27. Avoidance threshold concentration isoline for copper, with Jordan Creek flows greater than 30 CFS (11.8 µg/L or 11.3 µg/L excess above upstream concentration of 0.5 µg/L).

The mixing zone could potentially be avoided by listed salmonids as a behavioral response to the chemicals. If fish were to avoid the mixing zone then this habitat would effectively be unavailable. However, as shown in Figures 24-27, above, the areas where copper and zinc avoidance thresholds could be exceeded are small and there is an unaffected area that acts as a zone of passage. Thus, the negative impact of the loss of this area of would be negligible.
USEPA has determined that the chronic mixing zones for metals and cyanide of outfall 002

are not likely to adversely affect Columbia River bull trout, Snake River summer steelhead, and Snake River Chinook.

Another consideration for the outfall 002 is the effect to the salmonid prey-base. The majority of the salmonid diet comes from aquatic benthic macro-invertebrates. Monitoring of benthic macroinvertebrate assemblage in the Action Area receiving streams indicates that benthic populations are stable in both Jordan Creek and the Yankee Fork. Refer to BE Section VI.I (biomonitoring) for information on the status of the benthic population in the Action Area.

The effects to the benthic macro-invertebrate portion of the population within the mixing zone are not known. As with other aspects of the acute mixing zone, the affected area is small and therefore a small proportion of the macro-invertebrate population would occupy this habitat area. As the primary diet item for salmonids, bioaccumulation of toxins in macroinvertebrates could result in accumulations of toxins in tissues of fish. This is a chronic effect that would have long-term consequences to their survival and reproduction. The likelihood of bioaccumulation due to the effluent discharge is addressed in the selenium/mercury bio-accumulation analysis for both outfalls (see Section VI.H.). Bioaccumulation monitoring studies do not show significant differences between upstream and downstream sites indicating that the discharge is not causing detectable bioaccumulation in macroinvertebrates and fish tissue.

2. Outfall 003—Description of mixing zone for metals and WAD cyanide

The effluent for outfall 003 enters the Yankee Fork via an 11 ft. long diffuser with eleven ports situated approximately mid-channel. The mixing properties are somewhat complicated in that the diffuser is situated approximately 20 feet (6 meters) upstream of a medial rocky bar with a large boulder at the lower end of the bar. At low flows the creek flow is split into a west channel (~33% flow) and an east channel (~66 %flow) at low flows. Most of the effluent (83%) mixes with the East Branch Yankee Fork (NOAA 2004).

USEPA estimated dispersal and dilution of the effluent using the CORMIX model, which uses hydrodynamic parameters to predict the mixing zone. However, the channel-form of Yankee Fork at outfall 003 violates the assumptions of the model because the channel is complex, whereas the model assumes a uniform rectangular channel. Because of the constraints of the model assumptions, NOAA had concerns that the CORMIX model would not adequately predict the toxicity of the mixing zone and the form/configuration of the mixing zone. These concerns were legitimate because of the rock island that split the flow into two separate channels during low flow periods, potentially resulting in unequal toxicity in the two channels.

In order to validate the model function, NOAA conducted a simple dye study to further the information on how the effluent would mix with the stream water to compare relative flows and effluent dilutions. NOAA also reviewed the water quality monitoring data that was collected at various points downstream of the diffuser to verify the configuration of the mixing zone and to determine how quickly the toxicity of the effluent would be diluted. NOAA reported the following results. This information is taken from NOAA 2004:

Results from high flow sampling:
1. The effluent had not yet fully mixed with the Yankee Fork at 255 feet downstream,

2. Concentrations of several pollutants not detected or were only detected at very low concentrations (e.g. cadmium, copper, lead, mercury, selenium).

3. Ammonia was the only pollutant that was consistently detected during high flows.

4. Although elevated above background, ammonia concentrations were much lower than the chronic aquatic life criteria for ammonia approximately 4 times less than the criteria.

Results from low flow sampling:
1. As with high flows concentrations of several pollutants not detected or were only detected at very low concentrations (e.g. cadmium, lead, mercury, selenium).

2. Ammonia and copper were the only pollutants that were consistently detected during low flows but concentrations were well below the aquatic life criteria (except for one copper sample).

NOAA found concentrations of chemicals in the effluent were consistently lower than concentrations which could cause adverse effects to listed salmonids. Also, NOAA considered the data used in this evaluation represents the best available science as it is actual measured concentrations from samples that were collected in and below the mixing zone. During low flows NOAA did not observe hydraulic refuges or other cover features near the diffuser and concluded that it is unlikely for listed fish to hold in close proximity to the diffuser and that exposure would likely be short term. Under higher flows the boulder at the downstream end of the medial bar would be a cover feature for holding salmon and salmon would likely move up the east channel thus exposed to the majority of the effluent flow (personnel communication to Brian Nickel, USEPA from Ed Murrell, NOAA).

A couple of cautionary statements by NOAA (personnel communication to Brian Nickel, USEPA from Ed Murrell, NOAA) are 1) this study evaluates effluent from 2003 when the outfall 003 was initiated as an emergency tailings impoundment dewatering action (US Forest Service and US USEPA 2003). If the effluent for the new permit were to have higher allowable levels of contaminants there could be unacceptable levels that exceed the aquatic life criteria. 2) this study was conducted when flows may have been elevated due to precipitation a day or two before data were collected.

Although the medial rocky bar located about 20 feet (6 meters) downstream from outfall 003 violates the assumptions of the CORMIX model, as explained below, the model will nonetheless produce useful estimates of dilution. The hydrodynamic near field region extends only 1.6 meters downstream, thus it is entirely upstream of the island, and the island will therefore not disrupt the rapid, discharge-induced mixing that occurs in the near field. At the end of the hydrodynamic near field, complete vertical mixing has been achieved, and the dilution factor is at least 5.1:1 under all three flow tiers.

Regardless of the presence of the rocky island, subsequent mixing downstream from the hydrodynamic near field would occur via diffusion, controlled by the ambient characteristics of the creek. The field evaluation concluded that a portion of the effluent plume travels down both

channels of Yankee Fork Creek (i.e., the plume is bisected by the rocky island). Under these conditions, each portion of the plume will entrain ambient creek water from on one edge (i.e., the edge closer to the stream bank) as it passes the island, resulting in entrainment of ambient water occurring along two edges of the plume, which is no different from what would occur if the plume were not bisected. Thus, the island will not significantly change the far field mixing of the effluent with Yankee Fork Creek.

Based on imagery from Google Earth Pro, EPA estimates that the island is about 20 meters long. Even if mixing was slowed by the presence of the island, mixing consistent with the assumptions of the CORMIX model would resume downstream of the island.

When flows are high enough that the island is submerged such that the stream flows through, rather than around, the rocky area, the rocks would create turbulence which would cause mixing to occur more rapidly than it would in a uniform river channel as assumed by CORMIX. Imagery from Google Earth Pro, captured on October 26, 2013, when the flow in Yankee Fork Creek was 32 CFS, show that the island is partially submerged under these conditions.

As shown in Table 4, above, in most cases, the authorized mixing zones for outfall 003 extend less than 1.5 meters downstream from the outfall. Thus, the mixing zone is entirely within the hydrodynamic near field and unaffected by the island. The only mixing zones extending more than 6 meters downstream are those authorized for WET (for all flow tiers, addressed in next section) and for chemical-specific parameters for the high receiving water flow tier (≥ 45 CFS). However, at these higher flows, the flow may not be split by the rocky bar. As explained above, CORMIX would produce conservative estimates of dilution in the hydrodynamic far field if the island is partially submerged. The maximum modeled downstream distance for metals at the high flows is 35.2 m (Table 4). Schematics of the high flow tier (>45 CFS) mixing zones for parameters other than WET are shown in Figure 28, below.

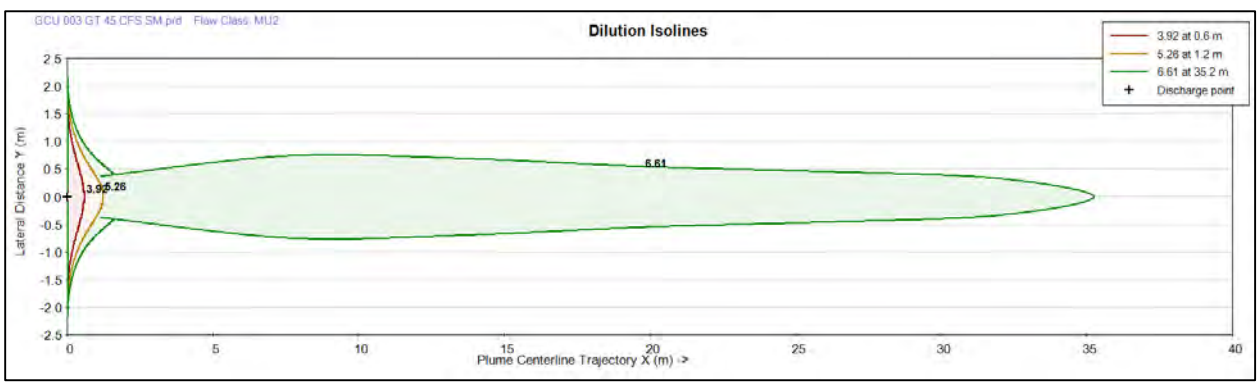


Figure 28. Dilution isolines for outfall 003, with flows greater than or equal to 45 CFS. The isoline for 3.92:1 dilution corresponds to the mixing zone for copper. The isoline for 5.26:1 dilution corresponds to the mixing zone for cadmium. The isoline for 6.61:1 dilution corresponds to the 25% mixing zone for all parameters except copper, cadmium, and WET.

<u>Outfall 003 copper and zinc avoidance thresholds</u>
Similar to outfall 002, the EPA has used the CORMIX model to estimate the areas where the receiving water may exceed the individual avoidance thresholds for zinc and copper for outfall

003. The EPA used the permit's average monthly limits and the median upstream concentrations of copper and zinc as modeling inputs for this purpose. The EPA used the same avoidance thresholds as for outfall 002: 11.8 µg/L for copper and 21 µg/L for zinc. Results are summarized in Table 17.

Table 17. Distances and plume travel times to avoidance thresholds for outfall 003

Parameter	Receiving Water Flow Tier	Downstream Extent (m)	Plume Travel Time (sec.)
Copper	< 15 CFS	0.05	0.08
Copper	15 – 45 CFS	0.08	0.09
Copper	≥ 45 CFS	0.05	0.03
Zinc	< 15 CFS	121	582
Zinc	15 – 45 CFS	461	2,082
Zinc	≥ 45 CFS	183	442

The mixing zone could potentially be avoided by listed salmonids as a behavioral response to the chemicals. If fish were to avoid the mixing zone then this habitat would effectively be unavailable. However, as shown in Table 17, above, the areas where copper and zinc avoidance thresholds could be exceeded are small and there is a zone of passage. Thus, the negative impact of the loss of this small area of would be negligible.

As stated for outfall 002, acutely toxic concentrations of metals are a condition that diminishes habitat value for species survival and recovery. Based on the mixing zone analysis of monitoring data by NOAA, acutely toxic conditions are not expected to occur except within a few feet of outfall 003. This area is small and bull trout, steelhead/rainbow trout, and Chinook salmon are unlikely to spend enough time within this is small area to be adversely affected. Discharge of acutely toxic levels of metals would result in degradation of water quality of this Critical Habitat area for Columbia River bull trout, Snake River summer steelhead, and Snake River Chinook. However, this area of diminished habitat is small. As a result, **USEPA has determined that the acute and chronic mixing zones for metals and cyanide (WAD) in Outfall 003 are not likely to adversely affect Columbia River bull trout, Snake River summer steelhead, and Snake River Chinook.**

3. Other Mixing Zone Issues

An issue raised in the past is the possibility of a temperature differential between the receiving water and the effluent resulting in thermal refuge for bull trout. This would result in bull trout being attracted to the chronic mixing zone and possibly spending an inordinate amount of time exposed to the chemicals in the effluent Monitoring data do not indicate that Outflow 002 is consistently colder than the receiving water, thus, we cannot reasonably speculate that the outfall will create a cold water refuge that would act as an attractant to fish during warm periods (Figure 29 and Figure 30). We note that to verify whether tributary flow is acting as a cold water refuge, and therefore an attractant, would require fish density data at a micro-habitat scale and more frequent temperature sampling. Likewise, we have no data to support a second scenario that the outfall 002 could act as an attractant to bull trout or possibly their prey species during winter as the water temperature of the outfall is also very cold during the winter. Also, low flows in this

riffle area would limit fish movement and feeding thus this reach probably does not get used substantially by fish in winter. A similar situation exists in the Yankee Fork where effluent temperatures are cold in the winter and summer maximums range from 16.7°C to 20.4°C through the sample years (Figure 31 and Figure 32). Note water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), Site S-9 (Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003). The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) June 1 through October 1 and temperature is collected daily at outfalls 002 and 003 when discharging.

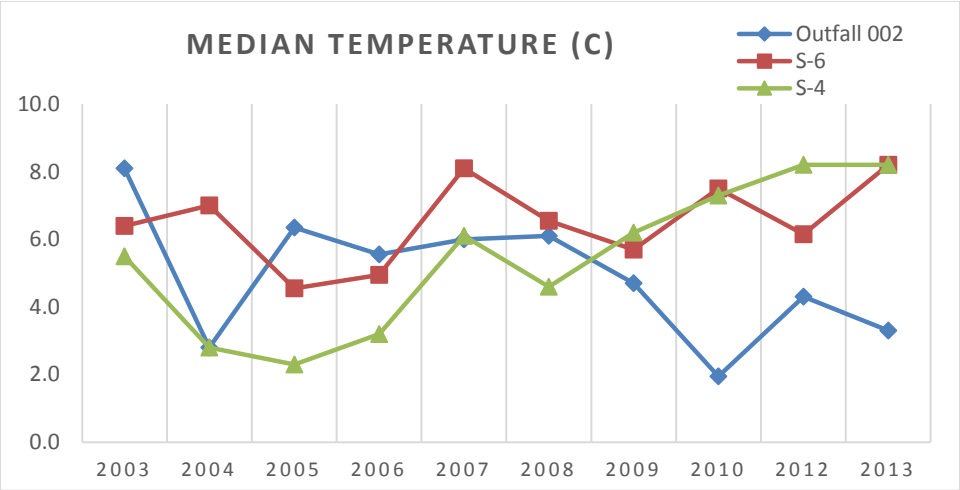


Figure 29. Temperature median values for outfall 002 and Jordan Creek monitoring sites, 2003-2013(Source: Hecla Limited).

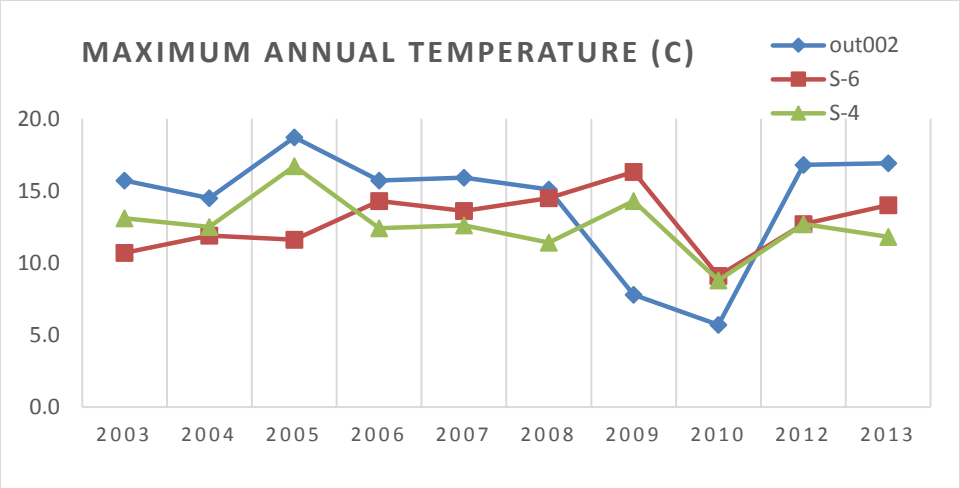


Figure 30. Temperature maximum values for outfall 002 and Jordan Creek monitoring sites,

2003-2013 (Source: Hecla Limited).

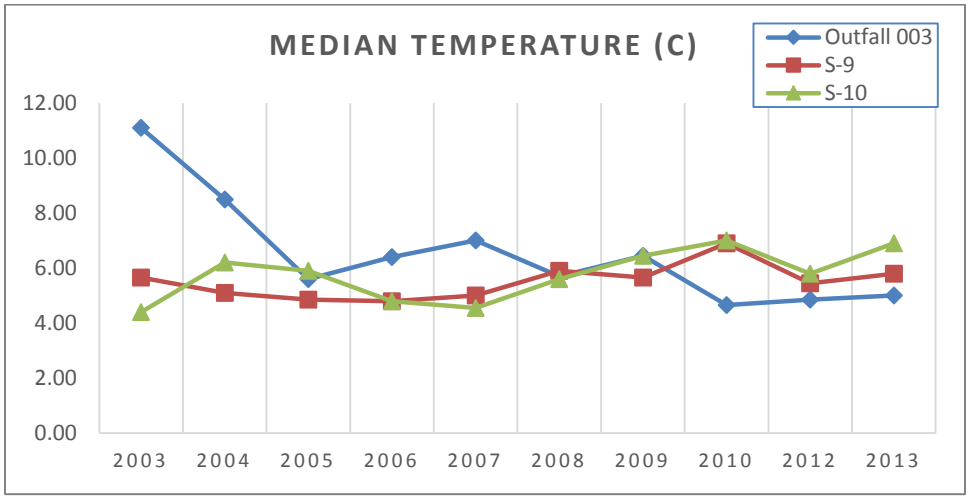


Figure 31. Temperature median values for outfall 003 and Yankee Fork monitoring sites, 2003-2013(Source: Hecla Limited).

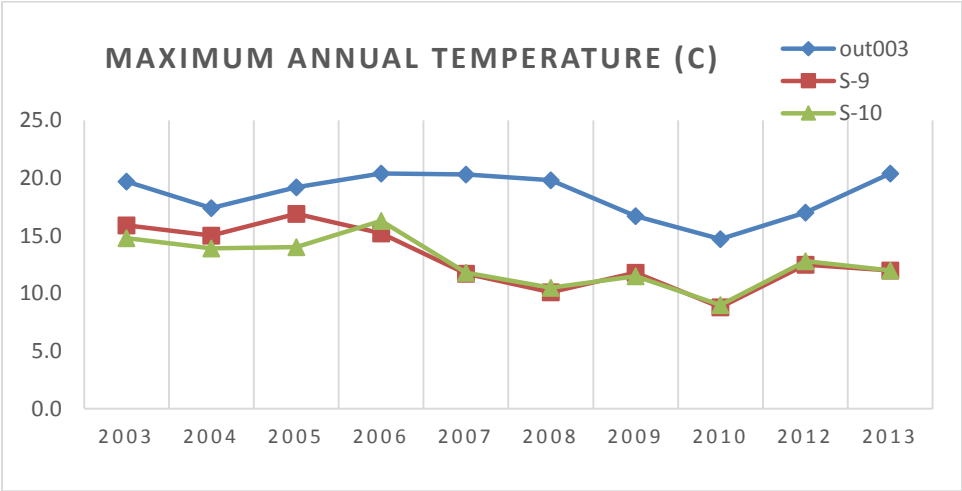


Figure 32. Temperature maximum values for outfall 003 and Yankee Fork monitoring sites, 2003-2013(Source: Hecla Limited).

G. Effect Determination for WET and Mixtures

Whole Effluent Toxicity (WET)

The proposed permit limits for WET are in Table 5 and Table 6. As discussed in Section V.G.3., the limits were calculated based upon the criteria of 1 TU_c for the chronic criterion and 0.3 TU_a (or, equivalently, 3.0 TUc, based on an acute-to-chronic ratio of 10:1) for the acute criterion. The limits are greater than the criteria, because a mixing zone was factored into the calculation (see Section V.F.).

Outfall 002

For outfall 002, with Jordan Creek flows less than 30 CFS, the chronic mixing zone for WET (where WET exceeds 1.0 TU_c, if the facility is discharging at its average monthly limit) spans about 50% of the width of Jordan Creek and continues 25 meters downstream, and the acute mixing zone for WET (where WET exceeds 3.0 TUc, or equivalently, 0.3 TUa, if the facility is discharging at its maximum daily limit) spans about 50% of the width of Jordan Creek and extends 24.88 meters downstream (**CORMIX modeling 2015 – Brian Nickel**). With Jordan Creek flows greater than 30 CFS, the chronic mixing zone spans about 1/3rd of the width of Jordan Creek and continues 687 meters downstream, and the acute mixing zone for WET spans about 50% of the width of Jordan Creek and extends 24.8 meters downstream. Travel times to reach the end of the acute mixing zone are 28 seconds when the flow is greater than 30 CFS and 54 seconds when the flow is less than 30 CFS. See Figure 33,Figure 34Figure 35, and Figure 36 below.

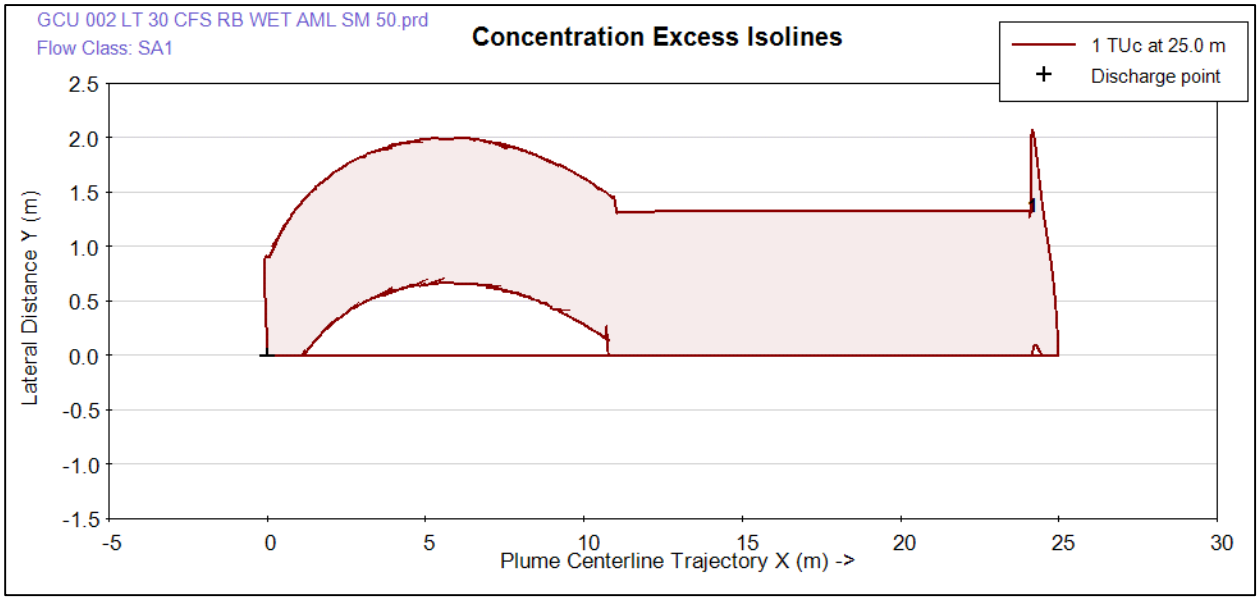


Figure 33. Concentration (1.0 TUc) isoline for outfall 002 for WET, with flows less than or equal to 30 CFS.

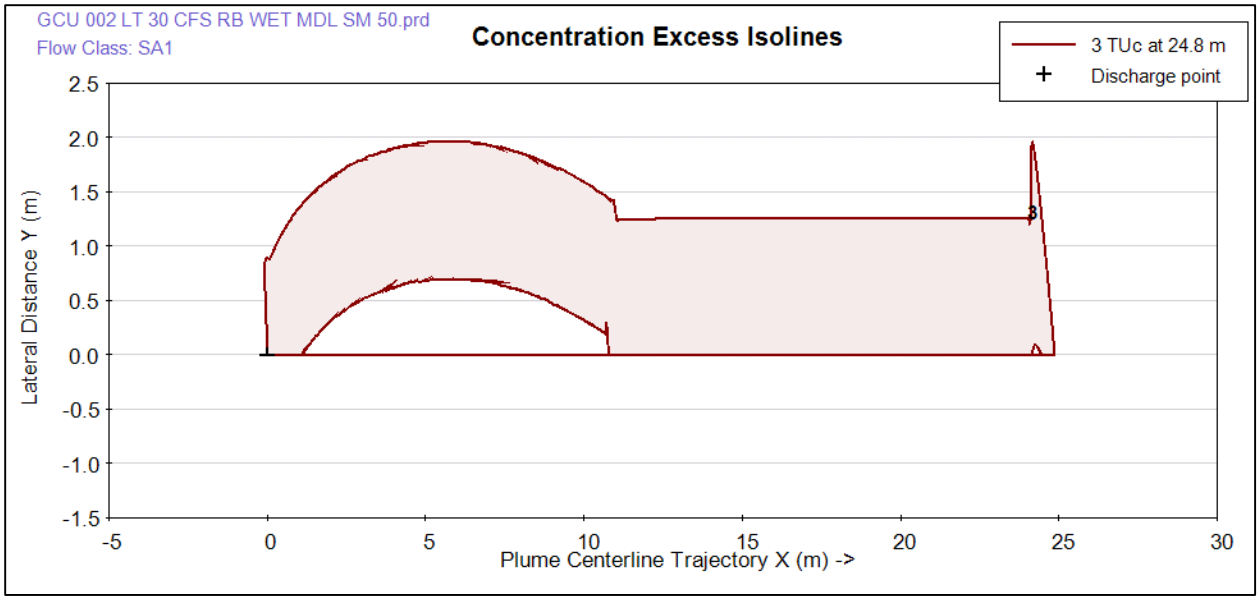


Figure 34. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 002 for WET, with flows less than or equal to 30 CFS.

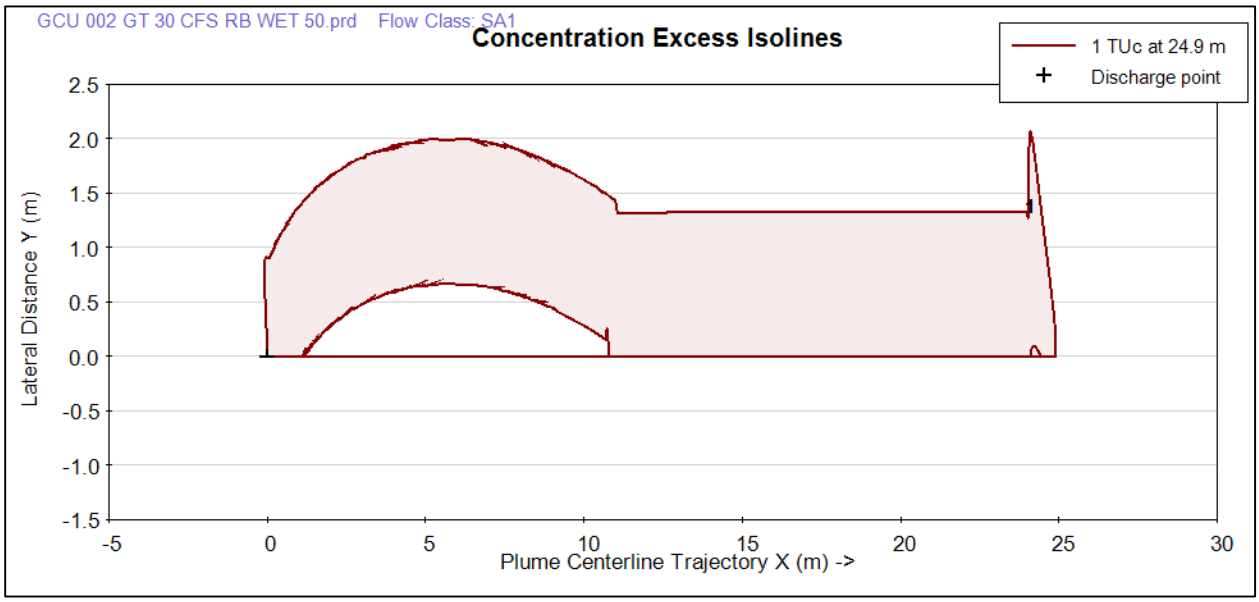


Figure 35. Concentration (1.0 TUc) isoline for outfall 002 for WET, with flows greater than or equal to 30 CFS.

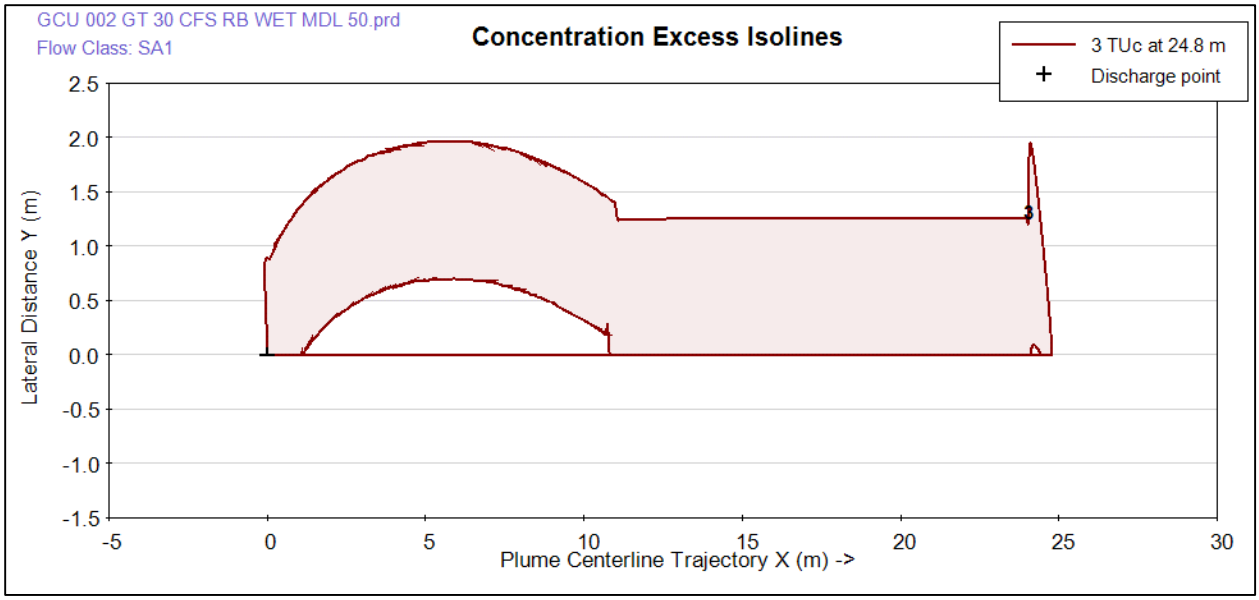


Figure 36. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 002 for WET, with flows greater than 30 CFS.

The chronic WET value of 1.0 TU_c is intended to prevent any chronic toxicity. However, for endpoints other than survival, the IC_{25} is used for determining chronic toxicity. The IC_{25} is a point estimate of the toxicant concentration, expressed in percent effluent, that causes a 25% reduction in a non-quantal biological measurement (e.g., reproduction or growth) calculated from a continuous model. Further details on chronic toxic units are in the Draft Permit.

WET is exclusive to a facility so data from literature is not available to evaluate the protection provided by the WET permit limits. However, WET includes all the parameters for which the permit provides limits. The test species used to determine compliance with the WET effluent limits (*Pimephales* and *Ceriodaphnia*), may not have the same sensitivity to the effluent as the threatened and endangered salmonids. Acute WET testing using salmonids (*Oncorhynchus mykiss*) is required on an annual basis. The results are not used to determine compliance with the WET limits. However, results exceeding trigger levels will require additional testing and investigations to determine the cause of toxicity.

For outfall 002, the acute WET mixing zone extends 24.8 meters downstream, but does not span the width of the stream, and thus there is a zone of passage for instream movement of salmonids in Jordan Creek. The chronic WET mixing zone extends somewhat further downstream than the acute mixing zone (25 meters), but, similar to the acute mixing zone, it is not channel spanning and not likely to be a barrier to instream movement of salmonids in Jordan Creek. Because both the chronic and acute mixing zone are small, each has a zone of passage, and the plume travel time to the end of the acute mixing zone is less than 1 minute, **USEPA has determined that the permit limits for WET in outfall 002 are not likely to adversely affect the Snake River Chinook salmon, Snake River steelhead, and bull trout.**

Outfall 003

The downstream extent of the WET mixing zones for outfall 003 are substantially longer that for outfall 002. The CORMIX model predicts that the downstream extent of the mixing zone for chronic WET (i.e., the downstream distance at which the receiving water concentration falls below 1.0 chronic toxic units or TUc, if the effluent WET is equal to the average monthly limit) is 199.5, 341, and 686.7 m for the three flow tiers < 15 cfs, ≥ 15 and < 45 cfs, and ≥ 45 cfs respectively (Table 4). The width of this mixing zone is at most about 6 meters wide (about 1/3 of the stream width) and the plume travel time to reach the end of the chronic WET mixing zone is at most 28 minutes. Figure 37, Figure 38Figure 39, below, are schematics of the model results for chronic wet, for the low, middle and high flow tiers.

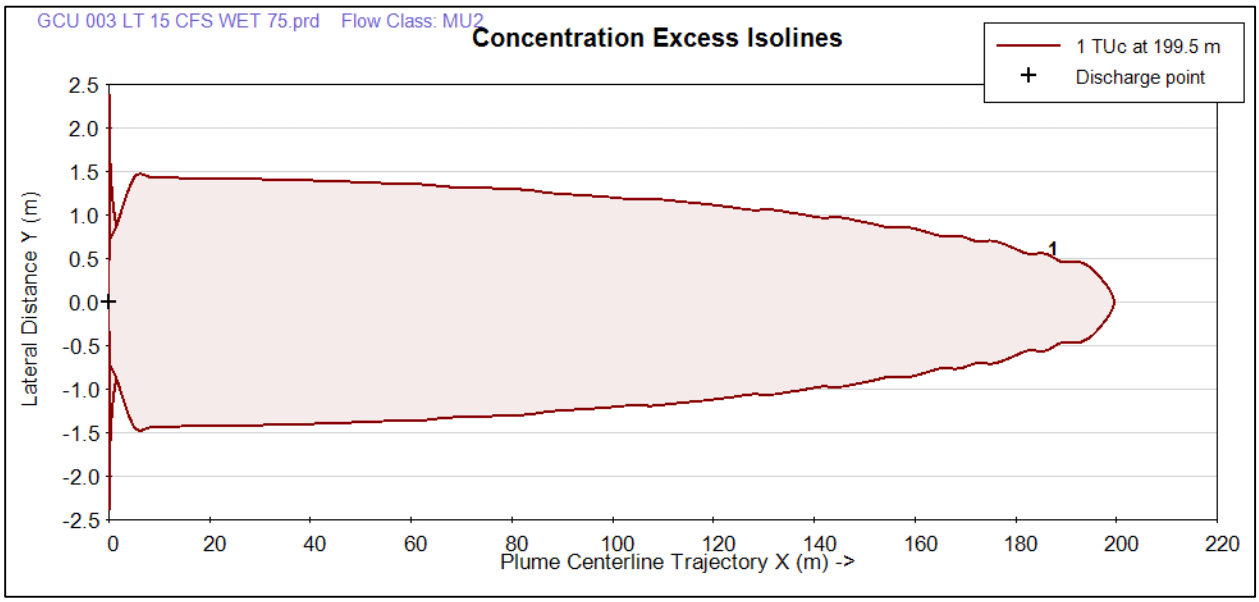


Figure 37. Concentration (1.0 TUc) isoline for outfall 003 for WET, with flows < 15 CFS.

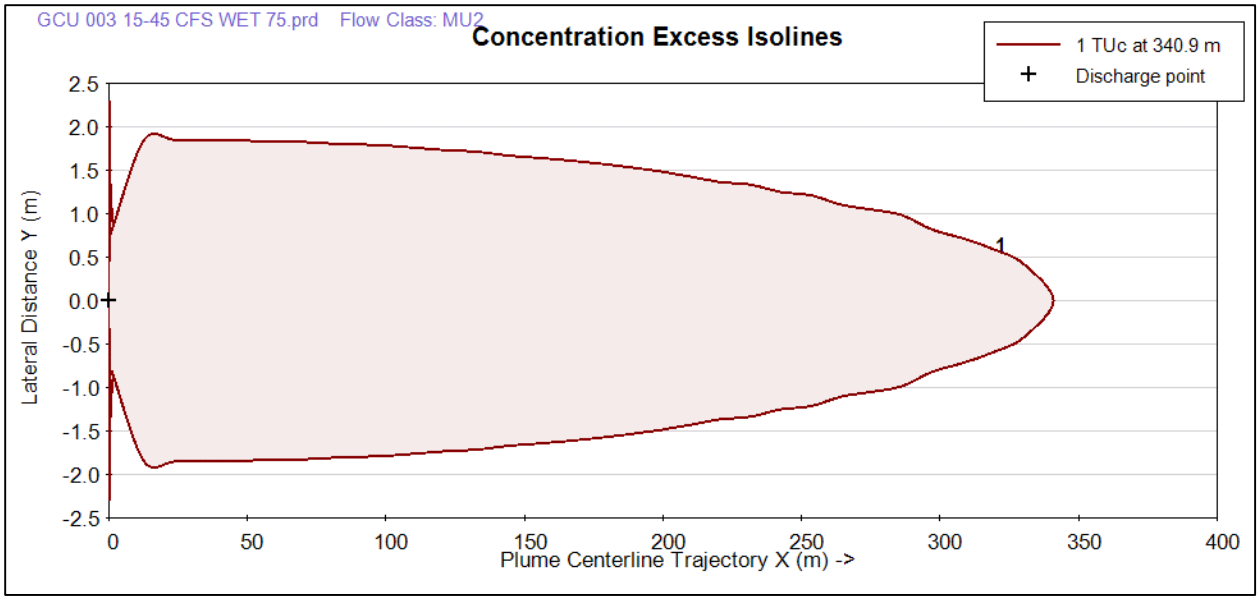


Figure 38. Concentration (1.0 TUc) isoline for outfall 003 for WET, with flows from 15 – 45 CFS.

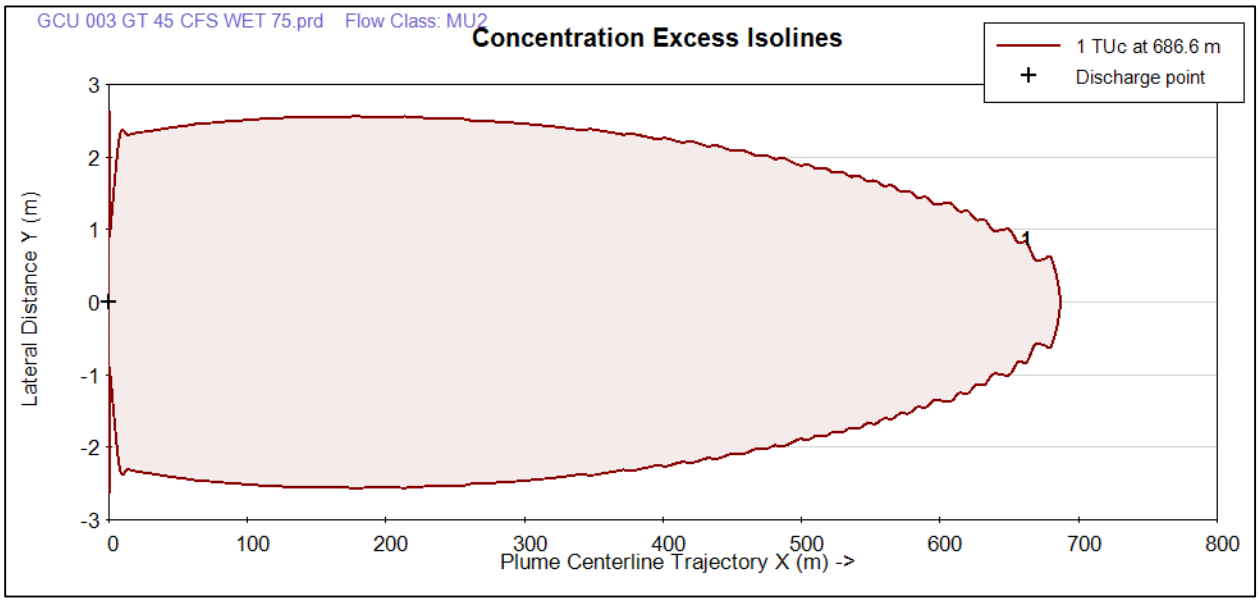


Figure 39. Concentration (1.0 TUc) isoline for outfall 003 for WET, with flows greater than 45 CFS.

The acute mixing zone is relatively small in area and there is a zone of passage where fish could avoid the acute zone. The plume travel time to reach the end of the acute mixing zone is at most 8.6 minutes. This is less than the 15 minute travel time recommended in EPA permitting guidance, in order to ensure that swimming or drifting organisms are not exposed to concentrations above the acute criterion for more than 1 hour. The acute mixing zone for outfall 003, for the middle and high flow tiers, is shown in Figure 40Figure 41, and Figure 42.

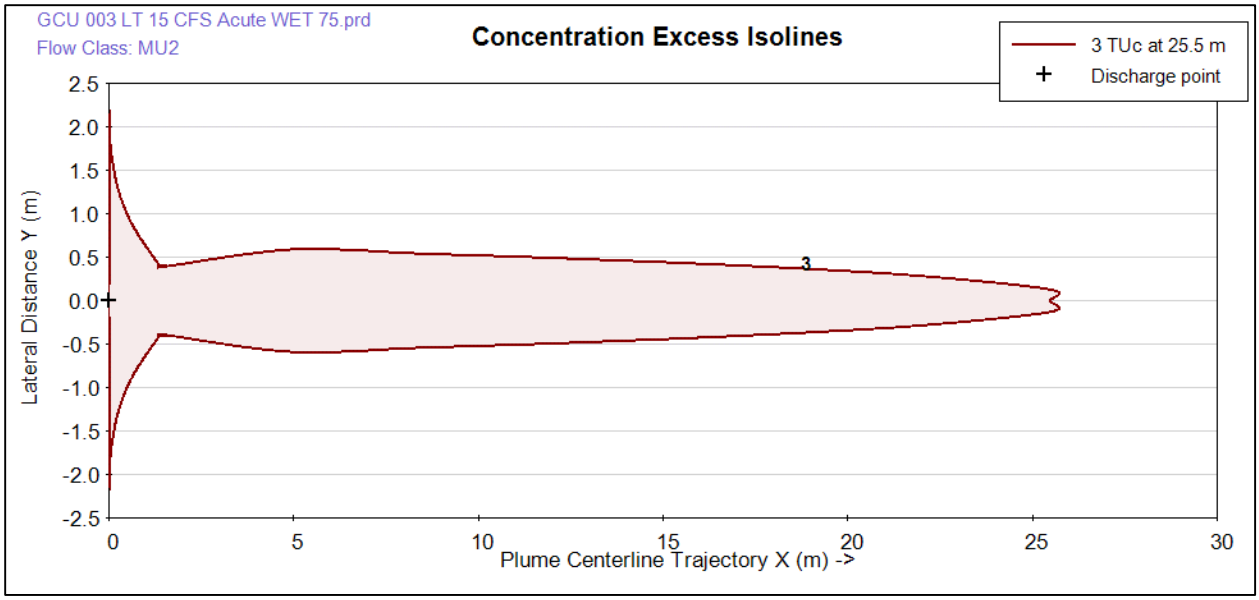


Figure 40. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 003 for WET, with flows < 15 CFS.

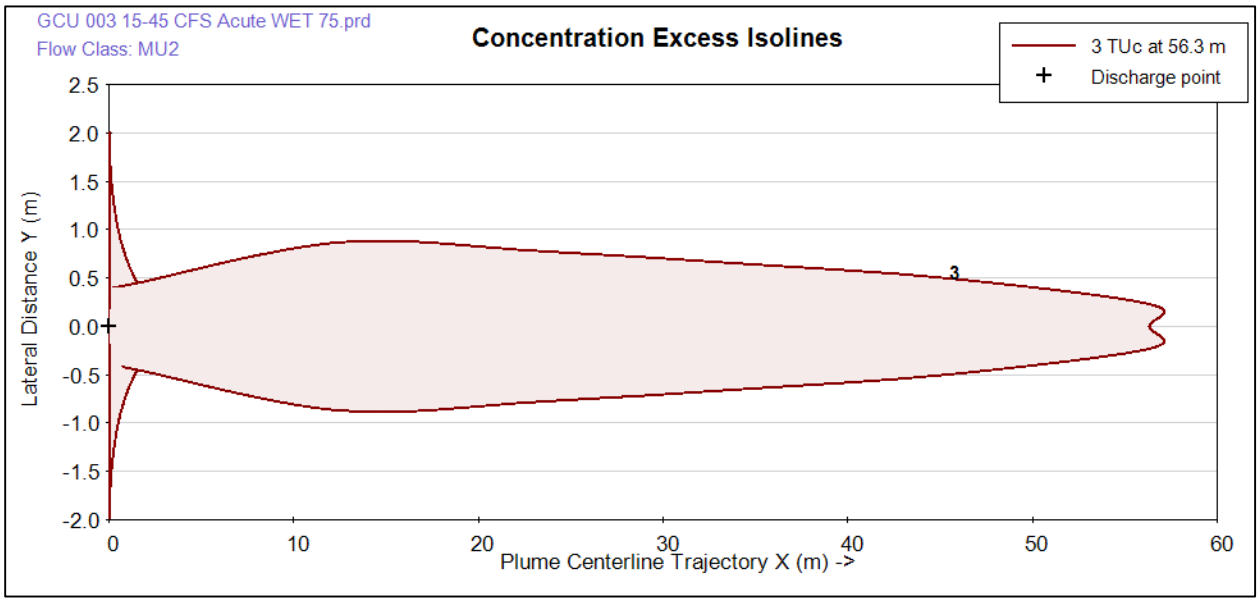


Figure 41. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 003 for WET, with flows from 15 – 45 CFS.

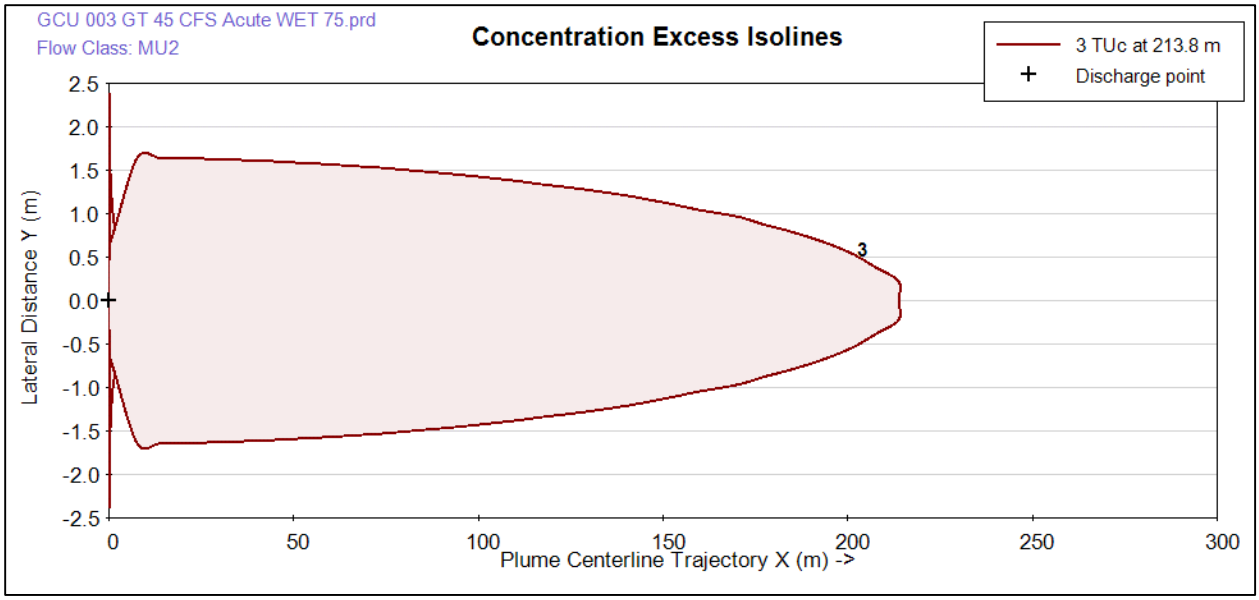


Figure 42. Concentration (3.0 TUc or 0.3 TUa) isoline for outfall 003 for WET, with flows > 45 CFS.

Factors that influence the likelihood of effects to ESA species is the substantial length of the chronic mixing zone at all flows. WET monitoring results from outfall 003 are in Table 18. **Therefore, USEPA has determined that the permit limits for WET in outfall 003 is likely to adversely affect the Snake River Chinook salmon, Snake River steelhead, and bull trout.**

Table 18. Chronic WET results of outfall 003(source: Helca Mining Co. provided to Brian Nickel, EPA).

Effluent sample date	Chronic WET value (T_{uc})
10/6/2010	<1
5/16/2011	14.4
11/14/2011	1.382
6/18/2012	<1
5/6/2013	<1
5/12/2014	19
5/5/15,5/7/15, and 5/9/15	TU_{cIC25}= 9.2 Survival NOEC=18.0 tuc

Mixtures

NMFS and USFWS issued BOs on the EPA's approval of Idaho's water quality standards for toxic substances in 2014 and 2015, respectively. These BOs concluded that the water quality criteria for several of the parameters discharged by the Grouse Creek Unit are likely to jeopardize the continued existence of Snake River basin steelhead, Snake River fall run Chinook salmon, Snake River sockeye salmon, Snake River Spring/Summer Chinook salmon, and bull trout. NMFS also concluded that these criteria would have an adverse effect on EFH.

The NMFS BO specified in its terms and conditions that the EPA shall minimize the potential for mixture toxicity in discharges. To implement this reasonable and prudent measure (RPM), the BO specifies that the EPA shall calculate the cumulative criterion units (CCU) to be allowed in

the receiving waters. CCU are defined for this purpose as CCU = Σ (C_d ÷ CCC) where C_d is the projected authorized concentration in the fully mixed receiving waters downstream of the effluent discharge, the CCC is the applicable chronic criterion concentration of each regulated constituent calculated for that location.

For the Grouse Creek Unit permit, > 1 CCU is authorized in the receiving waters for both outfalls and all flow tiers. For outfall 002, 2.8 CCU are authorized when stream flows are less than 30 CFS, and 3.1 CCU are authorized when stream flows are greater than 30 CFS. For outfall 003, 2.6 CCU are authorized when stream flows are less than 15 CFS, 3.0 CCU are authorized when stream flows are between 15 and 45 CFS, and 2.8 CCU are authorized if stream flows are greater than 45 CFS.

If the toxicity of the individual constituents of the effluent is additive, this means that the discharge could cause a toxic effect in the receiving water, even though each individual constituent is regulated such that criteria are achieved at the edges of relatively small mixing zones.

Therefore, USEPA has determined that mixtures of pollutants authorized to be discharged from outfalls 002 and 003 are likely to adversely affect the Snake River Chinook salmon, Snake River steelhead, and bull trout.

The NMFS BO states that, if discharges and the permit limits are authorized such that > 1 CCU would be calculated to be allowed in receiving waters, then WET testing and biomonitoring consistent with Appendix E to the NMFS BO shall be included in the permit provisions. The permit requires WET testing for both outfalls and biomonitoring of the receiving waters, consistent with RPM #1 and Appendix E to the NMFS BO.

H. Effects Related to Bioaccumulation –Selenium and Mercury

Selenium and mercury are the chemicals in the effluent that could potentially bioaccumulated in tissue of listed fish species. This section reviews available monitoring data for:
1) Environmental conditions for biotic uptake of these chemicals
2) Evidence of accumulation of Se and Hg in sediment
3) Evidence of accumulation of Se and Hg in tissue

1. Environmental conditions
The primary path of uptake of Se and Hg to the aquatic food web is through the water column by algae, bacteria and invertebrates (Hamilton and Lemly 1999). The presence of Se and Hg in the water column and associated water quality criteria are discussed in the previous sections. Presence of these metals in the water column is the most important source of their availability to biota and further discussion of environmental condition in no way diminishes that fact. The secondary pathway of uptake of Se and Hg is benthic-detrital processing in the sediment. As discussed in sections describing Se and Hg, stream conditions influence the bioavailability of Se and Hg (Section V). Sediment, slow flow, and shallow water are conducive to Se deposition in sediments for uptake and processing by biota. For Hg, sediments with high biological activity (bacteria) and rich in organics with anoxic or suboxic conditions convert inorganic mercury to

the bioavailable methylated form. Other factors such as temperature and nutrients that would promote microbial activity would also influence methylation.

Sediment conditions in the Action Areas do not appear favorable for selenium cycling/retention/deposition/accumulation or mercury methylation for several reasons: quantity of fine sediment is low, organic content of sediment is low, other factors that could promote deposition of Se or Hg methylation are not present, including warm water temperature, presence of aquatic macrophytes, and low gradient.

Fine sediment abundance is low in the Action Area as both stream reaches are cobble-bedded and are dominated by riffle type habitat with limited slow water areas (Chadwick 2001). Measurements of sediment quantity indicate that there is limited accumulation of fine sediment in these stream reaches. Based on pebble counts, in Jordan Creek, quantity of sediment estimated as < 1% above outfall 002 and 2 – 11% below outfall 002 (Chadwick 2000a and Chadwick 2001). The actual total area of depositional areas was low 0.5 – 1.0 % above and 2.2 – 2.8% below outfall 002). In Yankee Fork, fine sediment estimates of <1% above outfall 003 and 7% below 003 based on pebble counts (GEI 2007a). Spatial extent of fine sediment deposition was 1.4% of stream reach above and 0.8% below outfall 003. Fine sediment deposition has remained low through time. Recent monitoring data shows fines (< 2 mm diameter) ranged from 4 – 5% at the Yankee Fork monitoring sites (S-9 and S-10) 2010-2013 (GEI 2012a, 2013a, 2014a). Surface fines were between 4.5% and 11.1% in riffle and low gradient runs over this same time period.

Sediment biological activity, necessary for mercury methylation and for accumulation/processing of selenium, was estimated using total organic carbon (TOC) content. Since selenium can bind with particulate organic matter, TOC quantity provides information on the ability of the sediments to incorporate selenium. TOC can also be an indicator of the ability of sediments to contribute to methylation of Hg as methylation is promoted by decomposable organic content. Van Derveer and Canton (1997) proposes that TOC of > 5% is conducive to cycling/accumulation of Se in sediment of western streams. Jordan Creek data collected in 2000 found TOC is low in locations immediately above and below the outfall (1.34% upstream and 1.13 – 1.92% downstream) (Chadwick 2001). Values were higher at monitoring sites farther upstream and downstream (sites S6 and S4) with TOC ranging from 0.64 – 5.54%. In Yankee Fork, TOC content was low at both the upstream and downstream monitoring sites for 2000 – 2006 biannual sampling with highest measured sediment TOC of 2.7% in the upstream sample reach (GEI 2007b). The trend in TOC in Yankee for has remained consistent with percent in sediment ranging from < 0.1 – 1.7% at sites S-9 (upstream) and S10 (downstream) 2007 – 2013 (GEI 2014c).

2. Presence of Se and Hg in sediment

Jordan Creek

Jordan Creek sites were sampled for Se and Hg in April, August, and October 2000. Measurements of sediment Se in sites upstream of outfall 002 ranged from 0.4 – 1.57 mg/kg dry weight and for downstream sample sites < 0.10 – 2.10 mg/kg dry weight. In general, this is low level of Se in sediment. Sediment mercury in Jordan Creek ranges from 0.02 – 0.10 mg/kg dry weight in upstream sites and 0.06 to 1.40 mg/kg in downstream sites. Water monitoring results for 1997 – 2000 from sites upstream (reference) and just downstream of outfall 002 found values

<0.2 µg/L total Hg at both sites and values <0.2 µg/L at the actual outfall (Chadwick 2001).

Yankee Fork

In Yankee Fork, data are available for outfall 003 reference monitoring site (S-9) and downstream monitoring site (S-10) prior to discharge (2000 – 2001) and post discharge (2003 – 2013). For Selenium, values are all below 1 mg/kg dw both pre and post discharge for the upstream and downstream sites (Table 19). Differences between the median values for reference and the downstream site are not significant (GEI 2014c). Also, no trends over time at either site were evident. Downstream site values are within the range of the pre-discharge period.

A useful benchmark for evaluating sediment Hg concentration data is from MacDonald et al. (2000) which extends a threshold effects concentration (TEC) of 0.18 mg/kg as the level at which harmful effects are not likely to be observed. Comparing this value to monitoring results shows that mean sample values for the S-10 (below discharge) during the post-discharge (starting in 2003) years have not exceeded this mercury TEC. The upstream reference site had higher sediment mercury values. Median mercury concentrations were determined to be significantly different between upstream and downstream site over the sampling period, although no significant increasing or decreasing trends were detected in either site over these years (GEI 2014c).

Table 19. Summary of mean concentrations of selenium and mercury in sediment (mg/kg dw) collected from two sites in Yankee Fork (Source: GEI 2014c).

Concentrations in Sediment (mg/kg dw)					
Month	**Year**	**Selenium**		**Mercury**	
		S-9 (reference)	**S-10 (below 003)**	**S-9 (reference)**	**S-10 (below 003)**
Apr	*2000*	*NS[1]*	*<.10*	*NS*	*0.16*
Aug	*2000*	*0.50*	***0.81***	*0.21*	*0.24*
Oct	*2000*	*0.20*	*0.15*	*1.26*	*0.28*
Apr	*2001*	*0.35*	*0.38*	***1.51***	***0.35***
Aug	2003	0.06	0.09	0.11	0.07
Apr	2004	0.17	0.12	0.13	0.08
Aug	2004	0.48	0.61	0.15	0.15
Apr	2005	NS	NS	NS	NS
Aug	2005	0.41	0.36	0.36	0.10
Apr	2006	NS	NS	NS	NS
Aug	2006	0.14	0.14	0.15	0.09
May	2007	0.43	NS	0.16	NS
Aug	2007	0.23	0.18	0.10	0.12
Apr	2008	NS	NS	NS	NS
Aug	2008	0.17	0.19	0.14	0.10
May	2009	**0.90**	NS	0.18	NS
Aug	2009	0.18	0.12	0.32	0.17
May	2010	0.26	NS	0.18	NS
Aug	2010	0.33	0.40	0.27	0.17
May	2011	NS	NS	NS	NS.
Aug	2011	0.16	.013	0.12	0.07

May	2012	NS	NS	NS	NS
Aug	2012	0.11	0.10	0.16	0.07
May	2013	NS	NS	NS	NS
Aug	2013	0.31	0.25	1.14	0.02

Note: shading denotes pre-discharge data.
1. NS=not sampled

The lack of a signal between reference and downstream monitoring site is not surprising as the high overall background condition Se is characteristic of this basin. The USGS study conducted in Yankee Fork basin concluded that high Se contents in the sediments are most likely a result of high background (Frost and Box 2009). Extensive hardrock mining disturbance in Yankee Fork drainage has increased the volume of Se containing rock that is exposed to weathering.

3. Accumulation of Se and Hg in tissue

Jordan Creek

Benthic macro-invertebrate and fish tissue were sampled for Se and Hg in Jordan Creek, in August and October of 2000. Macro-invertebrate tissue Se concentration was 0.61 µg/g ww at site S-6 (reference) and 0.51 µg/g ww at site S-4(downstream) in August and 0.9 and 1.1 µg/g ww respectively in Oct. In fish, the mean concentrate of Se ranged from 0.84-1.23 ppm ww (3.36-4.92 ppm dw) in trout. Data were not significantly different between months or between sites. Measures of Se in sculpin were significantly higher from fish collected at the reference site (S-6) compared to the downstream site (S-4) (Chadwick 2001). Concentrations of Hg in trout tissue were similar at both sites and at both sample periods. All samples had concentrations below the 0.3 ppm wet weight of fish tissue criterion.

Yankee Fork

Benthic macroinvertebrates

In Yankee Fork, benthic invertebrate tissue Se concentration is higher in the downstream site compared to upstream site in several sample years (Table 20). The upstream site reference concentrations are not substantially different than the downstream sites over the sample period. Also, concentrations post-discharge are not substantially different than the pre-discharge data. There is a significant increasing trend from 2003 – 2013 at the upstream site in mean concentration but not for the downstream site (GEI 2014c). For Hg, concentrations were not significantly different between pre- and post-discharge sample events and or between the reference and downstream sample sites (Table 20). There are no significant trends over the years of sampling (GEI 2014c).

Table 20. Selenium and mercury mean concentrations in benthic macroinvertebrates collected from two sites in Yankee Fork (Source: GEI 2007, from Table 5).

Concentrations in benthic macroinvertebrates (mg/kg wet weight)					
Month	**Year**	**Selenium**		**Mercury**	
		S-9 (reference)	**S-10 (below 003)**	**S-9 (reference)**	**S-10 (below 003)**
Aug	*2000*	0.32	0.55	<.04	<.04
Oct	*2000*	0.50	0.60	**0.07**	<.04
Apr	*2001*	0.55	0.47	0.05	0.03
Apr	*2001*	**0.67**	0.46	0.06	0.03
Aug	2003	<.30	0.50	<.04	<.05
Aug	2004	<.20	<.20	<.04	<.05
Aug	2005	0.31	0.30	0.07	**0.81**
Aug	2006	NA	**0.83**	NA	<.04
Aug	2007	0.45	0.59	0.50	0.40
Aug	2008	NA	NA	NA	NA
Aug	2009	0.53	0.82	0.06	0.07
Aug	2010	0.46	0.48	0.03	0.03
Aug	2011	0.53	0.36	0.01	0.01
Aug	2012	0.57	0.45	0.02	0.02
Aug	2013	0.25	0.43	0.02	0.02

Note: shaded data are pre-discharge. outfall 003 discharge began May 2003.

Whole fish tissue

Mean selenium concentrations in whole fish tissue have been higher in the downstream site (S-10) than the upstream site (S-9) in some years. However, statistical differences are not evident (Table 21) (GEI 2014c). There are no trends in concentration at either site over the study period. The selenium mean concentrations at both sites have been below the USEPA 2014 External Peer Review Draft Criterion of 8.1 mg/kg dw in fish whole-body tissue (USEPA May 2014) in all years except 2007. However, the tissue concentration does exceed the limit of 5 µg/g suggested by personnel of other federal agencies as protective of ESA species (Skorupa et al. 2004) at both sites in most years. For mercury, values are not significantly different between the reference site and the downstream site (Table 21).

Table 21. Selenium and mercury concentration in whole-body whitefish tissue, Yankee Fork (Source: GEI 2014c).

Concentrations in whole body whitefish (mg/kg dry wt.)					
Month	**Year**	**Selenium**		**Mercury**	
		S-9 (reference)	**S-10 (below 003)**	**S-9 (reference)**	**S-10 (below 003)**
Oct	*2000*	NA	NA	NA	NA
Aug	2003	6.03	5.88	0.22	0.30
Aug	2004	5.46	7.71	<0.16	<0.16
Aug	2005	5.96	5.50	0.50	0.16
Aug	2006	6.77	7.13	0.25	0.35
Aug	2007	**9.35**	**10.38**	0.28	0.48
Aug	2008	5.27	5.43	0.25	0.31
Aug	2009	5.40	6.55	**0.60**	**0.43**
Aug	2010	5.88	7.23	0.33	0.23
Aug	2011	5.44	7.35	0.23	0.38
Aug	2012	5.61	7.53	0.44	0.38
Aug	2013	NA	6.13	NA	0.36

Note: shaded data are pre-discharge. outfall 003 discharge began May 2003.
Note: mg/kg same units as the criterion µg/g.

Fish gonadal tissue

Selenium concentration in whitefish gonadal tissue samples do not show clear trends as concentrations are higher in the downstream site (S-10) in some years and higher in the upstream site (S-9) in others (GEI 2014c). There are no increasing or decreasing trends over the years of sampling. The selenium mean concentrations in females has exceeded the USEPA 2014 External Peer Review Draft Criterion of 15.2 mg/kg dw in fish eggs or ovaries (USEPA May 2014) in all years in both the upstream and downstream sites (Table 22).

Table 22. Selenium and mercury concentration in whitefish gonadal tissue, Yankee Fork (Source: GEI 2014c).

Concentrations in whitefish gonadal tissue (mg/kg dw)					
Month	**Year**	**Selenium**		**Mercury**	
		S-9 (reference)	**S-10 (below 003)**	**S-9 (reference)**	**S-10 (below 003)**
male	2003	5.25	5.13	<0.11	<0.12
female	2003	22.80	20.15	<0.07	<0.05
Male	2004	NA	NA	NA	NA
female	2004	24.25	31.31	<0.13	<0.13
male	2005	NA	NA	NA	NA
female	2005	26.45	22.09	0.12	<0.13
male	2006	7.44	5.58	<0.15	**<0.22**
female	2006	NA	32.89	NA	<0.13
male	2007	13.01	10.66	**<0.20**	<0.19
female	2007	55.06	45.71	<0.10	<0.11
male	2008	NA	NA	NA	NA
female	2008	27.41	23.16	0.14	<0.13
male	2009	NA	NA	NA	NA
female	2009	15.41	23.97	0.17	0.16
male	2010	7.96	NA	0.14	NA
female	2010	20.04	33.96	0.08	0.04
male	2011	NA	NA	NA	NA
female	2011	21.32	29.76	.04	0.08
male	2012	NA	NA	NA	NA
female	2012	18.59	22.63	.08	0.06
male	2013	NA	NA	NA	NA
female	2013	NA	23.66	NA	0.06

Note: shading indicates female gonadal tissue data used in comparison to USEPA draft criterion.

Summary of bioaccumulation

Se and Hg bioaccumulation in trophic levels sampled in Yankee Fork do not appear to be attributable to the discharge from outfall 003. Review of the bioaccumulation monitoring studies conducted at the outfall monitoring sites does not indicate a substantial signal between the upstream monitoring station, and below the outfalls. Elevated levels of Se are related to exposure of rock to weathering from extensive mining in the area (Frost and Box 2009). Thus, the source of the Se is more of a drainage wide problem. A study by Rhea et al. (2013) of Se and Hg concentrations in various Yankee Fork trophic levels also found insignificant differences between study sites in the Yankee Fork. The following conclusions were made: 1) Se concentrations in biofilm and macro-invertebrates were not significantly different between sample sites (including sites that have similar upstream and downstream locations as the monitoring sites S-9 and S-10), 2) concentrations of Hg and Se in whole fish samples were generally consistent across sites, 3) concentrations of Hg in fish and fish tissues from Yankee Fork are similar to concentrations from other comparable areas and do not indicate unusual bioaccumulation or contamination, and 4) Se concentrations in fish tissue from all sites is high and therefore Yankee Fork fish may accumulate Se at levels that cause adverse effects.

I. Biomonitoring

Another line of evidence on the overall effect of the discharge to the ecological condition of the stream reaches is the measure of biotic integrity. According to the USFWS 2003 Biological Opinion (USFWS 2003), the USFWS consider this assemblage useable to detect affects to the bull trout population. Macro-invertebrates are an important food source for bull trout, thus effects (declines or alterations in relative abundance) to the benthic macro-invertebrate community would be indicative of adverse impacts to the bull trout population and that it can be used to infer take. They state that statistically significant decreases in invertebrate abundance, diversity or composition between upstream and downstream sample locations are indicative of adverse effects. Monitoring data on macro-invertebrates was collected from both streams to monitor possible effects to the benthic population. Data were compared to the State of Idaho screening criteria for assessment of ecological integrity. The following is a summary of conclusions from these reports. Additional, the fish assemblage was monitored in Jordan Creek with the same objectives to compare the biological integrity between sites upstream and downstream of outfall 002 to determine if the effluent is having an adverse effect.

IDEQ aquatic monitoring result in the action area

IDEQ's Beneficial Use Reconnaissance Project (BURP) collects benthic invertebrate, fish assemblage, water chemistry and habitat data to determine the quality of Idaho's waters (Grafe et al. 2002). Stream field data are evaluated using multi-metric indices for fish assemblage (SFI), habitat (SHI), and benthic macroinvertebrates (SMI) which are specific to the bioregions. Data results are compared to established bioregion specific reference condition and then assigned a rating of 3, 2, or 1 corresponding to high, medium, low relation to the reference condition. The results are used by IDEQ to evaluate the support of cold water aquatic life. Details on methodology are in (Grafe 2002a and Grafe 2002b) The IDEQ BURP database was searched for monitoring data available from the Action Area for describing stream quality in the action area and for validation of biological monitoring data collected by the mine. Data for two relevant areas were found: BURP Yankee Fork assessment area (ID17060201SL032_04) and BURP Jordan Creek assessment area (9ID17060201SL041_03) (Table 23). All data were obtained from http://www2.deq.idaho.gov/water/BurpViewer/Menu.

Table 23. Available IDEQ BURP monitoring data (1995-2011) from action area.

Burp ID	SMI score	SMI rating	SFI Score	SFI Rating	SHI Score	SHI Rating	Ave. rating	Location
Jordan Creek BURP data								
1995SIDFA039	78.9	3	80.2	2	34	1	2.0	Below 002
2002SIDFA079	78.3	3	77.7	2	69	3	2.7	Below 002
2008SIDFA144	63.5	3	—	—	43	1	2.0	Below 002
2008SIDFA152	66.4	3	78.1	2	48	1	2.0	Below 002
Yankee Fork Creek BURP data								
1995SIDFA093	83.2	3	61.9	1	30	1	1.7	Below 003
1995SIDFA091	72.7	3	51.2	1	52	1	1.7	Above 003
2008SIDFA068	No data	—	—	—	—	—	—	Below 003
2011SIDFA029	83.6	3	60.7	1	88	3	2.3	Above 003
2011SIDFA030	81.2	3	80.6	2	81	3	2.7	Above 003

Jordan Creek results collected at various locations downstream of outfall 002 give an overall 'medium' rating. Benthic invertebrate and fish assemblage scores were consistently high and medium respectively. Habitat scores were all 'low' except in 2002 (high). Overall, conditions were consistent over between these 4 sample events (1995 – 2008).

Yankee Fork BURP data are less informative. Only one downstream site available from 1995 indicating high benthic invertebrate assemblage score and low for Fish assemblage and habitat. More recent data were collected far upstream of outfall 003. These sites have a 'high' rating due to better habitat condition.

Biomonitoring data –Hecla Limited

Biomonitoring data collected by GEI for Hecla Limited are available from sites upstream and downstream of both outfalls. Benthic macroinvertebrate data were collected from both Jordan Creek and Yankee Fork and fish assemblage data were collected from Jordan. Annual data collection began pre-discharge to establish a baseline condition for comparison and continued, in most years, post discharge (2003 – 2013). Data are in annual reports and the latest synthesis of historic trends are in GEI 2014a and GEI 2014b.

Yankee Fork biomonitoring

Two monitoring sites, one above (S-9, which is 1 km above outfall 003) and one below (S-10, 1 km below outfall 003) are compared to evaluate possible effects of the discharge. Hecla Inc. used the classification and rating methods developed and used by IDEQ BURP for rivers (Grafe 2002a). These are similar to those used for the BURP data presented above but are adjusted for non-wadeable streams. Both the upstream and downstream monitoring sites were rated as high ('good') for benthic macroinvertebrate condition in the past three years (GEI 2012a, 2013a,

2014a) (Table 24). These results are comparable to those collected by IDEQ in Yankee Fork (Table 23).

Table 24. Results of multimetric analysis of benthic macroinvertebrates in Yankee Fork.

	S-9 upstream site		S-10 downstream site	
Year	**RMI score**	**Rating**	**RMI score**	**Rating**
2011	21	Good	23	Good
2012	21	Good	21	Good
2013	21	Good	21	Good

Source: GEI 2012a, 2013a, 2014a.

Three benthic invertebrate metrics, density, total taxa, and EPT taxa, were used to evaluate trends over the entire monitoring period in Yankee Fork in the vicinity of outfall 003. Data were collected pre-discharge (1999 and 2000) and post discharge 2003 – 2013 GEI (2014a). This information shows the benthic macroinvertebrate assemblage in the Yankee Fork is stable with no decreasing trends in any of these metrics. These results indicate that instream water quality and habitat conditions are not degrading and specifically do not indicate degradation associated with discharge from outfall 002. The following are a summary of findings over this monitoring period.

Benthic invertebrate density (#/m^2) has been variable at both the upstream and downstream sites. The two sites are compatible in terms of year to year variability. There is evidence that stream flow may be the driver for this variability in density (GEI 2014). There is no statistically significant increasing or decreasing trend in density for the upstream or downstream site over the monitoring years of sampling (1999 – 2013).

Total mean number of taxa has been generally higher since 2003 compared to the 1999 – 2000 pre-discharge baseline. There is no statistically significant increasing or decreasing trend in density for the downstream site. The upstream site has a significant upward trend over the monitoring years of sampling (1999 – 2013). A possible explanation for the positive trend for the upstream site is that this area has better riffle habitat that would be likely to support a more diverse macro-invertebrate assemblage.

EPT mean number of taxa has been similar or higher at both Yankee Fork monitoring sites compared to the pre-discharge baseline data (1999 – 2000). There is no statistically significant increasing or decreasing trend in EPT taxa abundance for the upstream or downstream site over the monitoring years of sampling (1999 – 2013).

<u>Jordan Creek biomonitoring—Hecla data</u>
Two monitoring sites, one above (S-6, which is 2.4 km above outfall 002) and one below (S-4, which is 0.3 km below outfall 002) are compared to evaluate possible effects of the discharge. Data were available for both benthic macroinvertebrate and fish assemblage from Hecla Inc. reports. Their analysis use the multimetric classification and rating methods developed and used by IDEQ BURP for streams (Grafe 2002b). Scoring for both assemblages over the past three years does not indicate a degraded condition for biota in this area (Table 25). These results are comparable to those collected by IDEQ in Jordan Creek (Table 23).

Table 25. Results of multimetric index analyses for benthic macroinvertebrate (SMI) and fish (SFI) assemblages in Jordan Creek.

	S-6 upstream site		S-4 downstream site	
Year	**score**	**Rating**	**score**	**Rating**
	Benthic macroinvertebrate assemblage			
2011	87	Very good	88	Very good
2012	82	Very good	77	Good
2013	79	Good	78	Good
	Fish assemblage (SFI)[1]			
2011	76	Medium	93	High
2012	81	High	94	High
2013	85	High	89	High

Source: GEI 2012b, 2013b, 2014b.
1. GEI did not apply rating convention described in Grafe 2002b. The table is populated with fish assemblage condition ratings based on cut-offs from the Grafe 2002b methodology for the Forested Bioregion. This is consistent with the procedure used in the IDEQ results presented in the previous table.

Four benthic invertebrate metrics were used to evaluate trends over the entire monitoring period 1999-2013 GEI (2014b). This information shows the benthic macroinvertebrate assemblage in Jordan Creek has stable or increasing trends. These results indicate that instream water quality and habitat conditions are not degrading and specifically do not indicate degradation associated with discharge from outfall 002. The following are a summary of findings over this monitoring period.

Benthic invertebrate density (#/m^2): There is a statistically significant increasing trend in density for both the upstream (S-6) and downstream site (S-4) over the monitoring years of sampling (1999 – 2013). Site S-4 had a significantly higher trend in density than S-6 over the same period.

Total mean number of taxa: There is a statistically significant increasing trend in mean number of taxa for both the upstream (S-6) and downstream site (S-4) over the monitoring years of sampling (1999 – 2013). Site S-6 had a significantly higher trend in number of taxa than S-4 over the same period.

EPT mean number of taxa: There is no statistically significant increasing or decreasing trend in EPT taxa abundance for the upstream or downstream site over the monitoring years of sampling (1999 – 2013). The long-term mean number of EPT taxa was significantly higher at site S-6 than at site S-4.

Shannon –Weaver diversity index: There is no statistically significant trend in the diversity index over the monitoring years of sampling. The long-term mean diversity index value at site S-6 is significantly higher than S-4.

Trends in Jordan Creek fish assemblage in the vicinity of outfall 002 are described using metrics of trout biomass, trout density, sculpins biomass, and sculpins density over the entire monitoring period 1999 – 2013 GEI (2014b). This information shows the fish assemblage in Jordan Creek is variable over time yet stable. Differences in physical habitat and elevation between the upstream and downstream sites likely influence differences in sculpins abundance. These fish assemblage metrics do not indicate degradation associated with the discharge from outfall 002. The

following are a summary of fish assemblage monitoring results.

Trout biomass: Biomass has been variable at both the upstream (S-6) and downstream(S-4) sites over the monitoring period (refer to the fish abundance section III.A.). There are no significant trends in trout biomass for either site over the sample period. The biomass trend is significantly higher at site S-4 compared to S-6.

Trout density: There is a significant increasing trend in trout density at the upstream site but no significant trend at the downstream site. Mean trout density at Site S-6 is significantly higher than at S-4 over the study period.

Sculpin biomass and density: Sculpin biomass and density over the monitoring period are both significantly higher at the downstream site (S-4) than the upstream site (S-6). No significant trends in sculpins biomass or density over the period were found.

Based on review of biomonitoring data from above and below the two outfalls, there does not appear to be adverse effects to the aquatic habitat from the effluent that would result in a detectable change in the presence, abundance, and diversity of the benthic assemblage.

VII. CUMULATIVE EFFECTS

“Cumulative effects” are defined in the Section 7 Consultation Handbook (USFWS and NMFS 1998) as “unrelated future non-federal activities that are reasonably certain to occur within the action area that are likely to affect the species.” Determining cumulative effects by this definition is typically hindered by the lack of definitive information on future State, tribal, local, or private actions.

The action area within the Jordan Creek drainage and the Yankee Fork contains little private or State land, with most land under the responsibility of the USFS (97% federal land, 3% private land). Private and State activities that occur on USFS land could however affect listed species in the area. These activities are generally limited to ongoing mining activities that have either been determined not likely to adversely affect listed salmonids or their critical habitat (small mining claims; NMFS 1997) or fall under the jurisdiction of federal ESA Section 7 consultation. USEPA is unaware of any future activities that will not require federal permits.

VIII. MANAGEMENT ACTIONS RELATED TO THE SPECIES

Besides establishing effluent limits for the outfall 002 and outfall 003 discharge, the draft final permit also incorporated the Reasonable and Prudent Alternatives (RPAs) and Reasonable and Prudent Measures (RPMs) related to the NPDES discharge as recommended by the Services in published Biological Opinions related to this action. A Biological Opinion for the last issuance of this permit in 2001 is currently not available from the NMFS. Biological and Conference Opinions (USFWS 2003) was released by USFWS in 2003. This document does not identify any Reasonable and Prudent measures for bull trout or Canada lynx. The USEPA permit also includes requirements for effluent monitoring, receiving water monitoring, preparation and implementation of a Quality Assurance Plan, and preparation and implementation of a Best

Management Practices Plan. These additional actions are described in Section VIII.B., below.

Implementation of the following actions will allow USEPA to ensure that the chemical, physical, and biological integrity of the receiving waters is not adversely impacted by the discharges and to assess whether more stringent effluent limitations or other permit conditions are needed.

A. Incorporation of 2003 Biological Opinion RPA and RPMs
The USFWS issued a Biological Opinion (BO) on the Grouse Creek Unit in December 2003 which at that time consisted of only the outfall at Jordan Creek (002) (USFWS 2003). The purpose of the BO was to determine if the Grouse Creek Unit operations were likely to affect the Bull trout or Canada lynx or affect their designated critical habitat. In the BO, USFWS states the following conclusion:

> "Although water quality in Jordan Creek may still impact individual bull trout as a result of the subject NPDES permit, the overall Columbia River DPS will not be so adversely impacted that recovery would not be possible". "It is the Services' biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Columbia River DPS of bull trout." Further, the USFWS concluded that the permit reissuance would not likely destroy or adversely modify bull trout Critical Habitat.

On the issue of take the USFWS concluded that the USEPA has implemented all practical measures to minimize take of bull trout and thus did not identify any reasonable or prudent measures (RPM) to further reduce incidental take. Since no RPMs were stated there are no related 'Terms and Conditions' that needed to be addressed by USEPA. However the USFWS did state reporting and monitoring requirements in their Biological Opinion (2003).

B. Incorporation of RPAs in the Idaho Water Quality Standards BOs
NMFS and USFWS issued BOs on the EPA's approval of Idaho's water quality standards for toxic substances in 2014 and 2015, respectively. These BOs concluded that the water quality criteria for several of the parameters discharged by the Grouse Creek Unit are likely to jeopardize the continued existence of Snake River basin steelhead, Snake River fall run Chinook salmon, Snake River sockeye salmon, Snake River Spring/Summer Chinook salmon, and bull trout. NMFS also concluded that these criteria would have an adverse effect on EFH.

All of the criteria considered in the 2014 and 2015 BOs remain in effect, thus, the EPA must implement these criteria in Clean Water Act actions, including NPDES permits. The BOs include RPAs, including "interim" RPAs that apply prior to the State of Idaho's adoption and the EPA's approval of revised criteria. The draft permit complies with the RPAs in the BOs, as described below.

Arsenic
The interim RPA for arsenic in the USFWS BO, for protection of bull trout, applies to discharges where geometric mean concentrations of arsenic are higher than 5 µg/L above background at the edge of the mixing zone. The geometric mean effluent concentration of arsenic at outfall 003 is 2.6 µg/L. Effluent data for arsenic are not available for outfall 002, however, since the sources of water for both outfalls are the same, the effluent concentration of arsenic at outfall 002 is

expected be similar for outfall 003. Because the geometric mean effluent concentration of arsenic is less than 5 µg/L, the geometric mean arsenic concentration will not be higher than 5 µg/L above background at the edge of the mixing zone. The arsenic RPA in the USFWS BO for protection of bull trout are therefore inapplicable to this discharge.

The interim RPA for arsenic in the NMFS BO is to apply the State of Idaho's 10 µg/L human health criterion. The EPA has applied this criterion in the reasonable potential analysis for outfall 003. No arsenic effluent data are available for outfall 002, however, as discussed above, the effluent concentrations of arsenic for outfall 002 are expected to be similar to outfall 003, so outfall 003 effluent arsenic data were used to perform a reasonable potential analysis for outfall 002. The discharges do not have the reasonable potential to cause or contribute to excursions above the 10 µg/L human health criterion at the edges of the authorized mixing zones under critical conditions.

Copper
The interim RPAs for copper in the USFWS and NMFS BOs, for protection of bull trout, Snake River fall run Chinook salmon, Snake River sockeye salmon, and Snake River Spring/Summer Chinook salmon, are to maintain a zone of passage sufficient to allow unimpeded passage of adult and juvenile salmonids. The BOs state that, "if the regulatory mixing zone is limited to less than or equal to 25 percent of the volume of a stream, then a sufficient zone of passage is presumed to be present."

The mixing zones for copper are less than or equal to 25% of the volume of the stream for both outfalls and all flow tiers. Furthermore, as discussed in the effect determination above, CORMIX modeling shows that the mixing zones for copper are small and do not span the stream channels.

The State of Idaho has begun negotiated rulemaking to adopt water quality criteria for copper based on the biotic ligand model, consistent with EPA recommendations. Effluent and receiving water monitoring for conductivity and dissolved organic carbon, in addition to pH, temperature and hardness, is required in the draft permit so that, when the State of Idaho adopts water quality criteria for copper based on the biotic ligand model, water quality criteria for copper can be evaluated.

Cyanide
The RPAs for cyanide in the USFWS and NMFS BOs, for protection of bull trout, Snake River fall run Chinook salmon, Snake River sockeye salmon, and Snake River Spring/Summer Chinook salmon, is to maintain a zone of passage sufficient to allow unimpeded passage of adult and juvenile salmonids. The USFWS BO states that, "if the regulatory mixing zone is limited to less than or equal to 25 percent of the volume of a stream, then a sufficient zone of passage is presumed to be present." The NOAA BO references the mixing zone limitations in the RPA for copper, which , as stated above, is to provide a zone of passage and that a sufficient zone of passage is presumed to be present if the regulatory mixing zone is less than or equal to 25 percent of the stream volume.

The mixing zones for cyanide are less than or equal to 25% of the volume of the stream for both outfalls and all flow tiers. Furthermore, as discussed in the effect determination above, CORMIX modeling shows that the mixing zones for copper are small and do not span the stream channels.

Selenium
The interim RPAs for selenium in the USFWS and NMFS BOs, for protection of bull trout, Snake River fall run Chinook salmon, Snake River sockeye salmon, and Snake River Spring/Summer Chinook salmon, are triggered when annual geometric mean concentrations of selenium at the edge of the mixing zone are higher than 2 µg/L. The geometric mean effluent concentrations of selenium at outfalls 002 and 003 are 1.6 and 1.4 µg/L, respectively. Because the geometric mean effluent concentration of selenium is less than 2 µg/L, the geometric mean selenium concentration will not be higher than 2 µg/L at the edge of the mixing zone. The fish tissue monitoring requirements of the RPAs therefore are inapplicable to the Grouse Creek Unit discharges.

Mercury
The RPAs for mercury in the USFWS and NMFS BOs, for protection of bull trout Snake River fall run Chinook salmon, Snake River sockeye salmon, and Snake River Spring/Summer Chinook salmon, is to implement the applicable human health fish tissue criterion of 0.3 mg/kg wet weight.

As explained in the Fact Sheets, the EPA followed EPA and IDEQ guidance in evaluating reasonable potential of the Grouse Creek Unit to cause or contribute to excursions above the methylmercury fish tissue criterion. The EPA found that the Grouse Creek Unit does not have the reasonable potential to cause or contribute to excursions above the methylmercury fish tissue criterion.

Zinc
The RPA for zinc in the USFWS BO, for protection of bull trout, is to ensure a zone of passage sufficient to allow unimpeded passage of adult and juvenile bull trout. The BO states that, "if the regulatory mixing zone is limited to less than or equal to 25 percent of the volume of a stream, then sufficient zone of passage is presumed to be present."

The mixing zones for zinc are less than or equal to 25% of the volume of the stream for both outfalls and all flow tiers. Furthermore, as discussed in the effect determination above, CORMIX modeling shows that the mixing zones for zinc are small and do not span the stream channels.

C. Incorporation of RPMs and Terms and Conditions from NMFS BO on Idaho WQS
The NMFS BO specified the following terms and conditions. The EPA shall:

1. Minimize the potential for mixture toxicity in discharges.
2. Minimize the potential adverse effects that occur when discharging under NPDES permits.
3. Minimize exposure of aquatic life to pentachlorophenol.
4. Use updated procedures for calculating any WERs developed for determining discharge limits.

5. Ensure completion of a monitoring and reporting program to confirm that the terms and conditions in this incidental take statement are effective in avoiding and minimizing incidental take from permitted activities and ensure the amount of incidental take is not exceeded.

To implement RPM #1, the BO specifies that the EPA shall calculate the cumulative criterion units (CCU) to be allowed in the receiving waters. CCU are defined for this purpose as CCU = Σ ($C_d \div$ CCC) where C_d is the projected authorized concentration in the fully mixed receiving waters downstream of the effluent discharge, the CCC is the applicable chronic criterion concentration of each regulated constituent calculated for that location. If discharges and the permit limits are authorized such that >1 CCU would be calculated to be allowed in receiving waters, then WET testing and biomonitoring consistent with Appendix E to the NMFS BO shall be included in the permit provisions.

For the Grouse Creek Unit permit, > 1 CCU is authorized in the receiving waters for both outfalls and all flow tiers. The permit requires WET testing for both outfalls and biomonitoring of the receiving waters, consistent with RPM #1 and Appendix E to the NMFS BO.

To implement RPM #2, the BO specifies that, for existing discharges that include silver, nickel and zinc, a zone of passage shall be ensured. If the regulatory mixing zone is limited to less than or equal to 25% of the volume of the stream, then sufficient zone of passage is presumed to be present. As discussed above, the regulatory mixing zone for all chemical specific parameters is less than or equal to 25% of the volume of the stream. Furthermore, as discussed in the effect determination above, CORMIX modeling shows that the mixing zones are small and do not span the stream channels.

The BO also specifies that, to implement RPM #2, the conservative assumptions described in Appendix D to the BO shall be applied when calculating effluent limits. In developing the effluent limits for the Grouse Creek Unit, the EPA applied conservative assumptions as described in Appendix D to the BO, including assuming only a portion of the stream and using the maximum permitted discharge volumes to calculate dilution. In general, the effluent limits in the Grouse Creek Unit are tiered based on stream flow and thus not based on critical low flows as described in Appendix D. However, discharge limits were calculated for critical flow conditions (i.e. maximum effluent flow and minimum stream flow) within each flow tier. The flow-tiered limits also provide certainty that the assumptions upon which the effluent limits were calculated are, in fact, valid.

The Grouse Creek Unit is not a source of pentachlorophenol, thus, RPM #3 is inapplicable to the Grouse Creek Unit permit.

No water effect ratios have been used in the development of the Grouse Creek Unit permit, therefore, RPM #4 is inapplicable to the Grouse Creek Unit permit.

To implement RPM #5, the EPA shall perform certain monitoring and reporting "each time a new NPDES permit is issued that discharges a toxic substance evaluated in this Opinion into waters containing Snake River salmon or steelhead or their critical habitat." Specifically, the EPA shall provide a copy of the NPDES permit issued to the facility and calculate the total area

of mixing zones granted for the new permit and for existing permits that discharge a toxic substance into waters occupied by listed salmon or steelhead. In addition, the EPA is to require biomonitoring consistent with Appendix E to the NMFS BO.

Consistent with RPM #5, the EPA has notified NMFS and USFWS of the availability of draft permits for this facility and provided copies of the draft permits to NMFS and USFWS. Mixing zones are described in detail in Section VI of this BE. The draft permit requires biomonitoring of the receiving waters consistent with Appendix E to the NMFS BO.

Appendix E to the NMFS BO states that, when biomonitoring is necessary to implement RPAs or RPMs, sampling for tissue residues of arsenic in benthic invertebrate prey organisms and selenium in juvenile salmonids and sculpins is to be performed. As explained below, the EPA does not believe that tissue monitoring for arsenic or selenium is necessary or appropriate for the Grouse Creek Unit permit.

As explained in Section VIII.B above, the interim RPA for arsenic in the NMFS BO is to implement the State of Idaho's human health criterion for arsenic (10 µg/L). The maximum measured concentration of arsenic in the effluent was 6.7 µg/L, the average effluent concentration was 2.7 µg/L, and the standard deviation was 1.3 µg/L. In the 2016 Fact Sheet, the EPA performed a reasonable potential analysis using the 99th percentile of the effluent data, which resulted in a reasonable potential multiplying factor of 2.16, and, in turn, a maximum projected effluent concentration of 14.5 µg/L. Although the maximum projected effluent concentration exceeded the 10 µg/L human health criterion, the concentration of arsenic would be less than 10 µg/L at the edges of the mixing zones for both outfalls and all flow tiers. Thus, the EPA concluded that the discharges do not have the reasonable potential to cause or contribute to excursions above Idaho's human health water quality criterion for arsenic.

In addition, in Section 3.3.2, the TSD states that, "although (the 99th percentile) does represent a measure of the upper bound of an effluent distribution, other percentiles could be selected by a regulatory agency." Using the 95th percentile effluent concentration instead of the 99th yields a reasonable potential multiplying factor of about 1.4 (see the TSD at Table 3-2), and a maximum projected effluent concentration of 9.4 µg/L. Thus, there is less than a 5% probability that the effluent concentration of arsenic will exceed 10 µg/L at the end of pipe. Because of the low concentration of arsenic in the discharges, the EPA has not required monitoring for arsenic in benthic invertebrate prey organisms.

As explained in section VIII.B above, fish tissue monitoring for selenium may be required as part of the interim RPA for selenium. However, tissue monitoring is required under the interim RPA only if concentrations of selenium at the edge of the mixing zone are higher than 2 µg/L as an annual geometric mean. In this case, the geometric mean effluent concentrations of selenium are less than 2 µg/L at the end of pipe, for both outfalls, and will be even lower at the edge of the mixing zone. Thus, fish tissue monitoring for selenium is not necessary under the interim RPA. Since fish tissue monitoring for selenium is not required under the interim RPA due to low effluent concentrations of selenium, the EPA has not required fish tissue monitoring for selenium.

D. Other Management Actions

The following actions are included in the draft final permit. Most monitoring actions are similar to those that were required in the 2002 permit. See the draft permit and Fact Sheets for more details.

Effluent monitoring: The permit requires effluent monitoring for outfalls 002 and 003 on the limited parameters. The permit includes additional monitoring for other parameters of potential concern such as arsenic for the purpose of effluent characterization (see Tables 3 and 4 of the 2016 Fact Sheet).

WET monitoring: In order to determine compliance with the WET effluent limit and determine potential toxic effects of the discharge, the draft final permit requires annual acute WET testing and quarterly chronic WET testing of effluent from outfall 002 and outfall 003.

Receiving water monitoring: for both Jordan Creek and Yankee Fork, the draft permit requires surface water monitoring four times per year for ammonia, conductivity, copper, dissolved organic carbon, lead, mercury, selenium, zinc, ammonia, nitrate+nitrite, pH, selenium, turbidity and hardness both upstream and downstream of the outfalls. Also, continuous monitoring of temperature is required upstream and downstream of the outfalls from May through October. A change from the 2002 permit is that monitoring for cadmium, cyanide, silver, dissolved oxygen, and TSS will be will be discontinued in Jordan Creek as concentrations are very low based on monitoring data (see the Fact Sheets).

Best Management Practices (BMP) Plan: Best management practices (BMPs) are measures that are intended to prevent or minimize the generation and the potential for release of pollutants from industrial facilities to waters of the U.S. The draft final permit requires Hecla to prepare and implement a BMP Plan to minimize the quantity of pollutants discharged, reduce the toxicity of the discharges to the extent practicable, prevent the entry of pollutants into waters, and minimize storm water contamination. See Section II of the permit for more details regarding the BMP requirements.

Quality Assurance Plan: The draft final permit requires the development and implementation of a Quality Assurance Project Plan (QAPP) to ensure that monitoring data is accurate and to explain data anomalies if they occur. Other provisions of the draft permit require Hecla Company to use analytical methods that can achieve method detection limits less than the effluent limits (for the effluent monitoring) and less than the water quality criteria (for the receiving water monitoring). The QAPP must specify analytical methods to achieve these method detection limits.

IX. ISSUES OF TAKE

The purpose of this section is to assess whether or not take of a listed species is likely to result from the proposed activity. “Take” as defined in Section 3(18) of the ESA means to “harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or attempt to engage in any such conduct.” The USFWS further defines “harm” as “significantly impairing behavioral patterns such as breeding, feeding, or sheltering,” and “harass” as “actions that create the likelihood of injury of listed species to such an extent as to significantly disrupt normal behavior patterns

which include, but are not limited to, breeding, feeding or sheltering." Further, the "incidental take" in Section 10(a)(1)(B) of the ESA means "any taking otherwise prohibited by Section 9(a)(1)(B) if such taking is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity." Finally, a "take" may occur only to individuals of a species, not to a species' habitat or to designated critical habitat.

These definitions are relevant to the permit discharge under the following scenarios:

1) Harm: an incidental take, in the form of harm would occur if a fish entered a mixing zone for feeding or breeding but was unable to perform these activities due to physiological alteration from exposure to metals, cyanide, or WET toxicity. This result is dependent on how long the fish remains within the mixing zone.

2) Harassment: an incidental take in the form of harassment could occur if there were developing embryos deposited within the mixing zone. Exposure to toxic levels of metals could impair embryo development or result in death. Another possible form of harassment would be if resident fish move through mixing zones resulting in multiple exposures to individuals.

Based on the monitoring results available for fish population trends, biomonitoring of macroinvertebrates, habitat quality, bioaccumulation studies, and water quality comparisons, evidence of take is not apparent.

Any take associated with monitoring activities (including fish collection to evaluate abundance, age classes, or lengths) are not addressed in this BE. Those activities are under taken by state or federal agencies or by the Mine and their contractors. These fish collection activities require state and federal permits. Therefore, this type of take is accounted for in that process.

X. SUMMARY OF DETERMINATIONS

Previous sections evaluated the water quality parameters at the concentrations that would be attained at the edge of the mixing zone. At this boundary, water quality standards are within the chronic criteria. Further dilution would result in water quality that is below the chronic criteria. This biological evaluation determined that the water quality standards attained at the edge of the mixing zone are acceptable for protecting the listed species and their critical habitat.

Based on the analysis in this BE, USEPA believes that reissuance of this **permit is likely to adversely affect Snake River Chinook salmon (spring run), Snake River steelhead, and Columbia River bull trout, and may affect but is not likely to adversely affect Snake River sockeye salmon** (Table 26). Further, USEPA has determined that the reissuance of this permit is likely to adversely affect the Critical Habitat for Snake River Chinook salmon, Snake River sockeye salmon, Snake River steelhead, and Columbia River bull trout. The primary lines of evidence on which we make these conclusion are from the monitoring data coupled with the implied protectiveness of the water quality standards. We do not have evidence of adverse effects and we considered the available data to represent a reasonable body of evidence to support these determinations. However, the EPA recognizes that uncertainty exists. Greater certainty would require macro-scale site data of actual fish use and behavior associated with the

acute and chronic mixing zones. These data do not exist and are unlikely to be collected in the future.

Table 26. Effects determinations for the four listed species addressed in this BE.

Species	Population	Species Effects Determination	Critical Habitat Effects Determination
Chinook salmon spring/summer run (*O. tshawytscha*)	Snake River ESU[a]	LAA	LAA
Sockeye salmon (*O. nerka*)	Snake River ESU	NLAA	NLAA
Snake River steelhead (*O. mykiss*)	Snake River DPS[b]	LAA	LAA
Bull trout (*Salvelinus confluentus*)	Columbia River DPS	LAA	LAA

Abbreviations of Effect Determinations: No Effect (NE), Not Likely to Adversely Affect (NLAA), Likely to Adversely Affect (LAA)
a: Evolutionarily Significant Unit,
b: Distinct Population Segment

A. Summary of Effects of the Action on Essential Elements of Critical Habitat

This action was considered in terms of the likelihood of direct or indirect alteration of the Critical Habitat which could thereby appreciably diminish its value for both survival and recovery of the listed species. The PCEs for bull trout are listed in section III-B. These include many factors: physical habitat features such as complex rearing habitat, deep pools for adult migration and spawning substrate; adequate forage base, and cold water. Similarly, the PCEs for spring Chinook and steelhead include water and substrate quality. The action would pose several possible influences to PCEs, particularly numbers 3 (abundant food), 5) water temperature, 6) spawning habitat, and 8) water quality. Physical habitat features could be affected (e.g. inputs of TSS that could result in sediment accumulations; excessive nutrient inputs that would contribute to excess growth of algae, macrophytes). The PCEs that are most relevant to this action are those that influence water quality needed to meet the physiological requirements of listed fish. In general, the discharge limits imposed by the permits restricts discharge to the water quality standards that are protective of listed fish at the edge of the mixing zone. Based on the analysis, effects of the action on PCEs is as follows:

Sediment quality: Inputs of TSS from the outfalls are unlikely to accumulate on the substrate as quantities are low and are part of the suspended load (see section V.G.2. Total Suspended Solids) Also, TSS levels are low (typically <10 mg/L) based on monitoring data (See Appendix B) and are therefore not likely to adversely affect the Critical Habitat. Information indicating the presence of flocculant in the vicinity of the outfalls, which could negatively impact substrate, have not been found.

Space: Discharge of nutrients (nitrate-nitrite) are at low levels, which do not result in excessive algae/macrophytes growth that could affect habitat (living space).

Food: Biomonitoring data indicate that macroinvertebrate assemblage does not appear to be negatively impacted by the discharges.

Temperature: Thermal inputs from these two outfall is negligible (<0.3ºC). These discharges will not influence water temperature of the two streams.

Passage conditions: No physical or thermal barriers will result from this action. The mixing zones area small (not channel spanning) and are not likely to pose a barrier to fish movement.

Toxicity: Acutely toxic conditions are restricted to small acute mixing zones within a few feet of the outfall. Acutely toxic conditions that degrade water quality are limited in extent. Also, Chronic mixing zones are relatively small. However, calculations of mixing zones related to mixtures of metals (i.e., cumulative criterion units and whole effluent toxicity) indicate the length of these zones may be considered substantial.

Bioaccumulation of toxics: Bioaccumulation of selenium may occur at levels that cause adverse effects. This appears to be a historic and basin-wide problem based on monitoring conducted upstream of the discharges and not an effect of the action.

XI. ESSENTIAL FISH HABITAT

In this section, Essential Fish Habitat (EFH) is assessed for potential adverse impacts from the issuance by USEPA of NPDES Permit No. ID0026468 for discharges of process wastewaters and storm water from the Grouse Creek Unit.

A. Background

The Magnuson-Stevens Fishery Conservation and Management Act (MSA), as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), requires federal agencies to consult with NOAA Fisheries on activities that may adversely affect EFH. According to the Magnuson-Stevens Fishery Conservation and Management Act (MSA§3), EFH means those waters and substrate necessary to fish for spawning, breeding, feeding, or growth and maturity. For the purpose of interpreting this definition of EFH: "waters" include aquatic areas and their associated physical, chemical, and biological properties that are used by fish; "substrate" includes sediment, hard bottom, structures underlying the waters, and associated biological communities; "necessary" means the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem; and "spawning, breeding, feeding, and growth to maturity" covers a species' full life cycle (50 CFR 600.01). "Adverse effect" means any impact which reduces quality and/or quantity of EFH, and may include direct (e.g. physical disruption), indirect (e.g. loss of prey), site-specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810).

Pursuant to the MSA the Pacific Fisheries Management Council (PFMC) has designated EFH for three species of federally-managed Pacific salmon: Chinook (*Oncorhynchus tshawytscha*); coho (*O. kisutch*); and Puget Sound pink salmon (*O. gorbuscha*) (PFMC 1999). Freshwater EFH for Pacific salmon includes all those streams, lakes, ponds, wetlands, and other water bodies currently, or historically accessible to salmon in Washington, Oregon, Idaho, and California, except areas upstream of certain impassable man-made barriers (as identified by PFMC 1999), and longstanding, naturally-impassable barriers (i.e. natural waterfalls in existence for several hundred years).

The objective of this EFH assessment is to determine if the proposed action may "adversely affect" designated EFH for relevant commercially or federally managed fisheries species within the proposed action area. It also describes conservation measures proposed to avoid, minimize or otherwise offset potential adverse effects to designated EFH resulting from the proposed action.

B. Description of the Project/Proposed Activity

The USEPA is proposing to reissue the National Pollutant Discharge Elimination System (NPDES) permit for the Grouse Creek Unit (NPDES Permit No. ID 002646-8) located in Custer County, Idaho. The mine is operated by the Hecla Company. The NPDES permit will authorize discharges from two outfalls, one on Jordan Creek and one on Yankee Fork a tributary to the Upper Salmon Sub-basin Hydrologic Unit Code (HUC) #17060201. The *Idaho water quality standards* (IDEQ 2009) designate these waters as protected for the following uses: cold water aquatic communities, salmonid spawning, and secondary contact recreation. In addition, the Salmon River is protected for domestic water supply.

C. Potential Adverse Effects of Proposed Project

The GCU Action Area is within the designated EFH for Chinook salmon. In the Yankee Fork, Chinook salmon are present and use this portion of the Action Area as a migration corridor, spawning habitat, rearing, and smolt out-migrate. Also, some rearing use by spring-run Chinook salmon in lower portion of Jordan Creek is known.

Water quality is an important component of EFH. The effects of authorized discharges from the Grouse Creek Unit on Chinook salmon EFH within the action area for this permit are the same as those described for fish species of concern in Sections V and VI. A summary of the determinations made for ESA listed species is found in Section X. Effluent limitations and acute/chronic surface water criteria described in the permit provide restrictions that are sufficient to prevent harm to life stages of threatened and endangered species in the action area. Monitoring data indicate that excursions beyond the criteria are rare (quantify from monitoring data). Using the information presented in Section V and VI, USEPA has determined that issuance of the Grouse Creek Mine Site permits is **likely to adversely affect** Chinook salmon EFH in the vicinity of the discharges.

D. EFH Conservation Measures and Conclusion

The Grouse Creek Mine will monitor the effluent discharges from both outfalls following NPDES requirements. The proposed permits requires the Grouse Creek Mine to continue a discharge monitoring program in order to detect changes in discharge that may be unacceptable and may require alteration of discharge operations. This work is consistent with measures that are recommended by the Pacific Fishery Management Council (PFMC 1999) to minimize the potential adverse effects to Chinook salmon EFH. USEPA concludes that the proposed action is **likely to adversely affect** EFH for Chinook salmon.

XII. REFERENCES

Agency for Toxic Substances and Disease Registry (ATSDR). 2008. Toxicological profile for aluminum. U. S. Department of Health and Human Services, Public Health Service. Division of Toxicology and Environmental Medicine/Applied Toxicology Branch. Atlanta, Georgia.

Albers, P.H., D.E. Green, and C.J. Sanderson. 1996. Diagnostic criteria for selenium toxicosis in aquatic birds: dietary exposure, tissue concentrations, and macroscopic effects. Journal of Wildlife Diseases 32:468–485.

Alstada, N. E., B.M. Kjelsberga, L.A.Vøllestada, E. Lydersenb, and A.B. Poleo. 2005. The significance of water ionic strength on aluminum toxicity in brown trout (Salmo trutta L.). Environmental Pollution 133 (2005) 333–342.

Armstrong, R. H. and J. E. Murrow. 1980. The Dolly Varden char (*Salvelinus malma*). In: Balon, E. K., (ed). Chars, Salmonid Fishes of the Genus *Salvelinus*. The Hague, Netherlands: Dr. W. Junk Publishers. pp. 99-140.

Atchison, G.J., M.G. Henry, and M.B. Sandheinrich. 1987. Effects of metals on fish behavior: a review. Environmental Biology of Fishes. 18:11-25.

Bader, J.A. and J.M. Grizzle, 1992. Effects of ammonia on growth and survival of recently hatched channel catfish. Journal of Aquatic Animal Health 4 (1992):17–23.

Baker, J. P., D. P. Bernard, S. W. Christensen, M. J. Sale, J. Freda, K. Heltcher, D. Marmorek, L. Rowe, P. Scanlon, G. Suter, W. Warren-Hicks, and P. Welbourn. 1990. Biological Effects of Changes in Surface Water Acid-Base Chemistry. NAPAP rpt. 13. In: National Acid Precipitation Assessment Program, Acidic Deposition: State of Science and Technology, Vol. II.

Baudo, R. 1983. Is analytically-defined chemical speciation the answer we need to understand trace element transfer along a trophic chain? In: Leppard, G. C. (ed) Trace element speciation in surface waters and its ecological implications. Plenum Press. New York. pp. 275-290.

Beckvar, N., J. Field, S. Salazar, and R. Hoff. 1996. Contaminants in aquatic habitats at hazardous waste sites: mercury. National Oceanic and Atmospheric Administration, Hazardous Materials Response and Assessment Division. Seattle, Washington.

Beijer, K. and A. Jennelov. 1979. Methylation of mercury in natural waters. Pages 201-210 *In* J. O. Nriagu (ed.). The biogeochemistry of mercury in the environment. Elsevier/ North Holland Biomedical Press. New York, NY.

Bell, M.C. 1971. Water Demands for Enhancement of Fisheries (Food and Growth). State of Washington Water Research Center. Olympia, Washington.

Bevan, D., J. Harville, P. Bergman, J. Crutchfield, P. Klingeman, and J. Litchfield. 1994. Snake River Salmon Recovery Team: Final Recommendations to National Marine Fisheries Service.

Birtwell, I. K., G. F. Hartman, B. Anderson, D. J. McLeay, and J. G. Malick. 1984. A brief investigation of Arctic grayling (*Thymallus arcticus*) and aquatic invertebrates in the Minto Creek drainage, Mayo, Yukon Territory: an area subjected to placer mining. Canadian Technical Report of Fisheries and Aquatic Sciences. Number 1287.

Bisson, P. A. and R. E. Bilby. 1982. Avoidance of suspended sediment by juvenile coho salmon. North American Journal of Fisheries Management 2:371-374.

Black, J.A. and W.J. Birge. 1980. An avoidance response bioassay for aquatic pollutants. NTIS PB80-180490.

Brown, L.G. 1994. On the zoogeography and life history of Washington's native char; Dolly Varden (*Salvelinus malma*) and bull trout (*Salvelinus confluentus*). Washington Department of Fish and Wildlife, Fisheries Management Division. Report #94-04. Olympia, WA.

Bureau of Reclamation. 2012. Yankee Fork Tributary Assessment, Upper Salmon Subbasin, Custer County, Idaho: Department of Interior, Bureau of Reclamation, Pacific Northwest Region, Boise, Idaho, 153 p.

Busby, P. J., T. C. Wainwright, G. J. Bryant, L. J. Lierheimer, R. S. Waples, F. W. Waknitz, and I. V. Lagomarsino. 1996. Status review of west coast steelhead from Washington, Idaho, and California. U.S. Department of Commerce, NOAA Technical Memorandum. NMFS-NWFSC-27. 261 pp.

Camargo, J.A., A. Alonso, and A. Salamanca. 2005. Nitrate toxicity to aquatic animals: a review with new data for freshwater invertebrates. Chemosphere 24:23-28.

Canadian Council of Ministers of the Environment (CCME). 2001a. Canadian Water Quality Guidelines for the Protection of Aquatic Life: Arsenic. Canadian Environmental Quality Guidelines, Canadian Council of the Environment, 1999, updated 2001. Winnipeg. Available at: http://ceqg-rcqe.ccme.ca/

Canadian Council of Ministers of the Environment (CCME). 2001b (Sept.). Water Quality Guidelines for Selenium, Technical Appendix. Prepared by: N. K. Nagpal and K. Howell. Water Protection Branch, Water, Lands and Air Protection. Victoria, British Columbia.

Canadian Council of Ministers of the Environment (CCME). 2000. Canadian water quality guidelines for the protection of aquatic life: ammonia. Canadian Council of Ministers of the Environment, Winnipeg. Available at http://www.ccme.ca

Canadian Council of Ministers of the Environment (CCME). 2003. Canadian water quality guidelines for the protection of aquatic life: nitrate ion. Available at http://www.ccme.ca

Cavender, T.M. 1978. Taxonomy and distribution of the bull trout, *Salvelinus confluentus* for the American Northwest. California Fish and Game 64(3):136-174.

Chadwick Ecological Consultants (CEC). 1999a. Biological evaluation of the effects of the Grouse Creek mine outfall 002 on Jordan Creek. Prepared for Hecla Mining Company by Chadwick Ecological Consultants, Inc., Littleton. CO.

Chadwick Ecological Consultants (CEC). 1999b. Fish population monitoring of Jordan Creek, Custer County, Idaho, 1998. Prepared for Hecla Mining Company by Chadwick Ecological Consultants, Inc., Littleton. CO.

Chadwick Ecological Consultants (CEC). 2000a. Biological evaluation of the effects of the Grouse Creek mine outfall 002 on Jordan Creek. Prepared for Hecla Mining Company by Chadwick Ecological Consultants, Inc., Littleton. CO.

Chadwick Ecological Consultants (CEC). 2000b. Review of issues related to fish avoidance in Thompson Creek. Draft Technical Memorandum. Prepared for Thompson Creek Mining Company by Chadwick Ecological Consultants. Littleton, CO. 14 pp.

Chadwick Ecological Consultants (CEC). 2001. Addendum: Technical Memorandum-mercury and selenium evaluation report (October 2000) Jordan Creek. Prepared for Hecla Mining Company by Chadwick Ecological Consultants. Littleton, CO. 20pp.

Chambers, P.A., M. Guy, E.S. Roberts, M.N. Charlton, R.A. 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. 241 pp.

Chapman, DW, A. Giorgio, M. Hill, A. Maule, S. McCutcheon, D. Park, W. Platts, K. Pratt, J. Seeb, L. Seeb, and F. Utter. 1991. Status of Snake River Chinook salmon. Prepared for Pacific Northwest Utilities Conference Committee by Don Chapman Consultants, Inc. Boise, ID.

Clarkson, T. W., Hamada, R. and L. Amin-Zaki. 1984. Mercury. Pages 285-309 in J. O. Nriagu (ed.). Changing metal cycles and human health. Springer-Verlag. Berlin, Germany.

Currie R.S., W.L. Fairchild, and D.C.G. Muir. 1997. Remobilization and Export of Cadmium from Lake Sediments by Emerging Insects. Environmental Toxicology and Chemistry 16:2333-2338.

DeForest, D.K., J.S. Meyer, R. W. Gensemer, B. K. Shephard, W.J. Adams, R.L. Dwyer, J.W. Gorsuch, and E.J. Van Genderen. 2011. Are ambient water quality criteria for copper protective of olfacotory impairment in fish. *in* Learned Discourses. Integrated Environmental Assessment and Management. 7(1):141-149)

Eisler, R. 1985a. Cadmium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. Fish and Wildlife Service, Biological Report 85(1.2).

Eisler, R. 1985b. Selenium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. Fish and Wildlife Service, Biological Report 85(1.5).

Eisler, R. 1988. Lead hazards to fish, wildlife, and Invertebrates: A Synoptic review. U.S. Fish and Wildlife Service. Biological Report 85(1.14).

Eisler, R. 1991. Cyanide Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.23). Fish and Wildlife Service Contaminant Hazard Review.

Eisler, R. 1993. Zinc Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U. S. Fish and Wildlife Service, Biological Report 10, Contaminant Hazard Reviews Report 26.

Elder, J. F. 1988. Metal Biogeochemistry in Surface-Water Systems - A Review of Principles and Concepts. U.S.G.S. Circular 1013, cited by Oregon Department of Environmental Quality (1995), Standards & Assessment Section, 1992-1994 Water Quality Standards Review, pH.

Elder, J.F., and Collins, J.J. 1991. Freshwater molluscs as indicators of bioavailability and toxicity of metals in surface-water systems. Reviews in Environment Contaminant Toxicology 122:37-79.

European Inland Fisheries Advisory Commission (EIFAC). 1969. Water Quality Criteria for European Freshwater Fish - Extreme pH Values and Inland Fisheries. In U.S. Environmental Protection Agency. 1986. Quality Criteria for Water 1986. USEPA/440/5-86-001.

Felton, S. P., Wenjuan, J. and Mathews, S.B. 1990. Selenium concentrations in coho salmon outmigrant smolts and returning adults; a comparison of wild versus hatchery-reared fish. Dis. Aquat. Org. 9:157-161.

Ferreira, V., V. Gulis, and M.A.S. Graca. 2006. Whole-stream nitrate addition affects litter decomposition and associated fungi but not invertebrates. Oecologia (2006) 149:718–729.

Fisher, N. S. and Wen-Xiong Wang. 1998. Trophic transfer of silver to marine herbivores: A review of recent studies. Environmental Toxicology and Chemistry 17:562-571.

Folmar, L.C. 1976. Overt avoidance reaction of rainbow trout fry to nine herbicides. Bull. Environ. Contaminant Toxicology 15:509-514.

Fraley, J. J. and B. B. Shepard. 1989. Life history, ecology and population status of bull trout (*Salvelinus confluentus*) in the Flathead Lake and River system, Montana. Northwest Science 63:133-143.

Frost, T.P., and S.E. Box. 2009. Streams-sediment geochemistry in mining-impacted drainages of the Yankee Fork of the Salmon River, Custer County, Idaho: U.S. Geological Survey Scientific Investigations Report 2009-5115, 49 p. and data files. Reston, VA.

Gensemer, R.W. and R.C. Playle. 1999. The Bioavailability and Toxicity of Aluminum in Aquatic Environments, Critical Reviews in Environmental Science and Technology, 29:4, 315-450, DOI: 10.1080/1064338999125924

Geotechnical Watershed Resources (GEI). 2007a. (March). Aquatic population monitoring of the Yankee Fork of the Salmon River Idaho 2006. Biomonitoring report. Submitted to Hecla Mining Company. GEI Consultants, Inc. Chadwick Ecological Division. Littleton, Colorado.

Geotechnical Watershed Resources (GEI). 2007b. Mercury and selenium bioaccumulation evaluation report Yankee Fork of the Salmon River, Idaho, 2006. Submitted to: Hecla Mining Company. GEI Consultants, Inc. Chadwick Ecological Division, Littleton. CO.

Geotechnical Environmental Water Resources Ecological (GEI). 2012a. Aquatic population monitoring of the Yankee Fork of the Salmon River, Idaho, 2011. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Geotechnical Environmental Water Resources Ecological (GEI). 2012b. Aquatic biomonitoring of Jordan Creek, Idaho, 2011. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Geotechnical Environmental Water Resources Ecological (GEI). 2013a. Aquatic population monitoring of the Yankee Fork of the Salmon River, Idaho, 2012. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Geotechnical Environmental Water Resources Ecological (GEI). 2013b. Aquatic biomonitoring of Jordan Creek, Idaho, 2012. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Geotechnical Environmental Water Resources Ecological (GEI). 2014a. Aquatic population monitoring of the Yankee Fork of the Salmon River, Idaho, 2013. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Geotechnical Environmental Water Resources Ecological (GEI). 2014b. Aquatic biomonitoring of Jordan Creek, Idaho, 2013. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Geotechnical Environmental Water Resources Ecological (GEI). 2014c. Mercury and selenium bioaccumulation evaluation report for the Yankee Fork of the Salmon River, Idaho, 2013. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Geotechnical Environmental Water Resources Ecological (GEI). 2015. Aquatic biomonitoring of Jordan Creek, Idaho, 2014. Submitted to Hecla Limited, Grouse Creek Unit. GEI Consultants, Inc. Ecological Division. Denver, Colorado.

Giattina J.D. and R.R. Garton. 1983. A review of the preference -avoidance responses of fishes to aquatic contaminants. Residue Reviews 87: 43-90.

Goetz, F. 1989. Biology of the bull trout *Salvelinus confluentus*: a literature review. U. S. Forest Service, Willamette National Forest. Eugene, Oregon.

Goldstein, J.N., D.F. Woodward, and A.M. Farag. 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.

Grafe, C.S.(ed). 2002b. Idaho Small Stream Ecological Assessment Framework: An Integrated Approach. Idaho Department of Environmental Quality; Boise, Idaho.

Grafe, C.S.(ed.). 2002a. Idaho River Ecological Assessment Framework: an Integrated Approach. Idaho Department of Environmental Quality; Boise, Idaho.

Grafe, C.S., C.A. Mebane, M.J. McIntyre, D.A. Essig, D.H. Brandt, and D.T. Mosier. 2002. The Idaho Department of Environmental Quality Water Body Assessment Guidance, Second Edition-Final. Idaho Department of Environmental Quality; Boise, Idaho.

Green, E. D. and Albers, P. H. 1997. Diagnostic Criteria for Selenium Toxicosis in Aquatic Birds: Histologic Lesions. Journal of Wildlife Diseases. 33:385-404.

Groth, P. 1971. Untersuchungen uber eihige spur en elemente in Seen. Arch. Hydrobiol. 68:305-375. as cited in Draft Biological Evaluation for Reissuance of a National Pollutant Discharge Elimination System Permit for the Potlatch Corporation, Lewiston, Idaho. 1993. Science Applications International Corporation. San Diego, CA.

Grothe, D.R., K.L. Dickson, and D.K. Reed-Jenkins, Eds. 1996. Whole Effluent Toxicity Testing: An Evaluation of Methods and Prediction of Receiving System Impacts. Society of Environmental Toxicology and Chemistry Press.

Gundersen, D.T., S. Bustaman, W.K. Seim, and L.R. Curtis. 1994. pH, hardness, and humic acid influence aluminum toxicity to rainbow trout (Oncorhynchus mykiss) in weakly alkaline waters. Can. J. Fish. Aquat. Sci. 51: 345-1355.

Hamilton, S. J. and A. D. Lemly.1999. Water-Sediment Controversy in Setting Environmental Standards for Selenium. Ecotoxicology and Environmental Safety 44(3): 227-235.

Hamilton, S. J., A. N. Palmisano, G. A. Wedemeyer and W. T. Yasutake. 1986. Impacts of selenium on early life stages and smoltification of fall chinook salmon. In: Transactions of the Fifty-first North American Wildlife and Natural Resources Conference, R. McCabe (ed). March 21-26, 1986. Wildlife Management Institute, Washington, D. C. pp. 343-356.

Hamilton, S. J., Buhl, K. J., Faerber, N. L., Wiedmeyer, R. H. and Bullard, F. A. 1990. Toxicity of Organic Selenium in the Diet to Chinook Salmon. Environmental Toxicology and Chemistry. 9:347-358.

Hamilton, S.J and K.J. Buhl. 1990. Safety Assessment of Selected Inorganic Elements to Fry of Chinook Salmon (Oncorhynchus tshawytscha). Ecotoxicology and Environmental Safety 20:307-324.

Hamir, A. N., Sullivan, N. D. and P. D. Handson. 1982. The effects of age and diet on the absorption of lead from the gastrointestinal tract of dogs. Australian Veterinary Journal 58:266-268.

Hansen, J.A., D.F. Woodward, E.E. Little, A.J. DeLonay and H.L. Bergman. 1999. Behavioral avoidance: possible mechanism for explaining abundance and distribution of trout species in a metal impacted river. Environmental Toxicology and Chemistry. 18:313-317.

Hartwell, S.I., J.H. Jin, D.S. Cherry, and J. Cairns, Jr. 1989. Toxicity Versus Avoidance Response of Golden Shiner, *Notemigonus crysoleucas*, to Five Metals. Journal of Fisheries Biology 35:447-456.

Hecla Limited. 2010. Summary of water quality 2009: Hecla Limited Grouse Creek Unit. Prepared by Hecla Limited Grouse Creek Unit, PO Box 647, Challis Idaho.

Hecla Limited. 2011. Summary of water quality 2010: Hecla Limited Grouse Creek Unit. Prepared by Hecla Limited Grouse Creek Unit, PO Box 647, Challis Idaho.

Hecla Limited. 2013. Summary of water quality 2012: Hecla Limited Grouse Creek Unit. Prepared by Hecla Limited Grouse Creek Unit, PO Box 647, Challis Idaho.

Hecla Limited. 2014. Summary of water quality 2013: Hecla Limited Grouse Creek Unit. Prepared by Hecla Limited Grouse Creek Unit, PO Box 647, Challis Idaho.

Hecla Mining Company (Hecla). 2000. Mixing zone information for discharge point 002, Grouse Creek. Hecla Mining Company, Grouse Creek Unit. Challis, Idaho.

Henry, M.G., Atchison, G.J. 1991. Metal effects on fish behavior e advances in determining the ecological significance of responses. In: Newman, M.C., McIntosh, A.W. (Eds.), Metal Ecotoxicology: Concepts and Applications. Lewis, Chelsea, MI, pp. 131e143.

Hermanutz, R. O., Allen, K. N., Roush, T. H. and Hedtke, S. F. 1992. Effects of Elevated Selenium Concentrations on Bluegills in Outdoor Experimental Streams. Environmental Toxicology and Chemistry. 11:217-224.

Hilton, J.W., Hodson, P.V., Slinger, S.J., 1980. The requirement and toxicity of selenium in rainbow trout (Salmo gairdneri). J. Nutr. 110:2527–2535.

Hodson, P. V. and Hilton, J. W. 1983. The nutritional requirements and toxicity to fish of dietary and waterborne selenium. Ecological Bulletin. 35:335.

Hodson, P. V., Spry, D. J. and Blunt, B. R. 1980. Effects on rainbow trout (*Salmo gairdneri*) of a long-term exposure to waterborne selenium. Canadian Journal of Fisheries and Aquatic Science. 37:233-240.

Hodson, P. V., Whittle, M. D. and Hallett, D. J. 1984b. Selenium contamination of the Great

Lakes and its potential effects on aquatic biota. In: J. O. Nriagu and M. S. Simmons, eds., Toxic Contaminants in the Great Lakes. John Wiley & Sons. New York, NY, pp. 371-391.

Hodson, P.V., D.M. Whittle, P.T.S. Wong, U. Borgman, R.L. Thomas, Y.K. Chau, J.O. Nriagu, and D.J. Hallet. 1984a. Lead Contamination of the Great Lakes and its Potential Effects on Aquatic Biota (chapter 16). In: J. O. Nriagu and M. S. Simmons, eds., Toxic Contaminants in the Great Lakes. John Wiley & Sons. New York, NY.

Hudson, R. J. M., S. A. Gherini, W. F. Fitzgerald and D. B. Porcella. 1995. Anthropogenic influences on the global mercury cycle: a model-based analysis. Water, Air and Soil Pollution. 80: 265-272.

Hydrometrics, Inc. 1981. Reconnaissance; terrestrial wildlife and aquatic resources, Sunbeam Mine area, Idaho. Report prepared for Sunbeam Mining Company, Stanley, ID.

Idaho Division of Environmental Quality (IDEQ). 2000. Draft. Evaluation of proposed new point source discharges to a special resource water and mixing zone determinations: Thompson Creek Mine facility, Upper Salmon River subbasin, Idaho, NPDES Permit Application ID-002540-2. Idaho Division of Environmental Quality. Boise, Idaho. 128pp.

Idaho Division of Environmental Quality (IDEQ). 2009. The Idaho water quality standards and Wastewater Treatment Requirements. Boise, ID.

Idaho Department of Environmental Quality. 2014. Idaho's 2012 Integrated Report. Boise, ID: Idaho Department of Environmental Quality. Boise, ID.

Idaho Department of Environmental Quality (IDEQ). (Feb 9) 2015. Draft 401 water quality certification for NPDES Permit Number ID0026468 Hecla Mining Company, Grouse Creek Unit. Idaho Division of Environmental Quality (IDEQ). Boise, ID.

Interior Columbia River Technical Recovery Team (ICTRT). 2007. Draft. Yankee Fork Salmon River Spring Chinook Salmon Population. Population Viability Assessment. Northwest Fisheries Science Center, Seattle. Available: www.nwfsc.noaa. gov.

Jana, S. and S. S. Sahana. 1989. Sensitivity of the freshwater fishes *Clarias batrachus* and *Anabas testudineus* to heavy metals. Environmental Ecology 7(2):265-270.

Konopacky, R.C., P.J. Cernera, and E.C. Bowles. 1986. Salmon River habitat enhancement, Annual Report FY 1984. Subproject III: Yankee Fork of the Salmon River: inventory and problem identification. Prepared for Bonneville Power Administration, Project No. 83-359.

Korver, R.M. and J.B. Sprague. 1989. Zinc avoidance by fathead minnows: computerized tracking and greater ecological relevance. Canadian Journal of Fisheries and Aquatic Science. 46:494-502

Kroupova, H., J. Machova, et al. 2008. Effects of subchronic nitrite exposure on rainbow trout (Oncorhynchus mykiss). Ecotoxicology and Environmental Safety 71(3):813-820.

Leathe, S. A. and M. D. Enk. 1985. Cumulative effects of microhydro development on the fisheries of the Swan River drainage, Montana. Volume 1. Summary report prepared for the Bonneville Power Administration, Contracts DE-A179-82BP36717 and DE-A179-83BP39802, Project 92-19.

Lemly, A. D. 1999. Selenium transport and bioaccumulation in aquatic ecosystems" a proposal for water quality criteria based on hydrological units. Ecotoxicology and Environmental Safety 42:160-156.

Lemly, A.D. 1985. Toxicology of selenium in a freshwater reservoir: Implications for environmental hazard evaluation and safety. Ecotoxicology and Environmental Safety 10:314-338.

Lewis, W.M. and D.P. Morris. 1986. Toxicity of nitrite to fish: a review. Transactions of the American Fisheries Society 115:183–195.

MacDonald, D.D., C. G. Ingersoll, and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Archives of Environmental Contamination and Toxicology 39:20-31.

MacKinlay, D. D., D. D. MacDonald, M. K. Johnson, and R. F. Fielden. 1987. Culture of Chinook salmon (*Oncorhynchus tshawtschya*) in iron-rich ground-water: Stuart pilot hatchery experiences. Canadian Manuscript Report of Fisheries and Aquatic Sciences 1944.

Mance, G. 1987. Pollution threat of heavy metals in aquatic environments. Elsevier Applied Science. New York. 372 pp.

Marshall, D. B., M. Chilcote, and H. Weeks. 1992. Sensitive Vertebrates of Oregon. Oregon Dept. of Fish and Wildlife, Portland, OR, (as cited by Oregon Department of Environmental Quality (1995), Standards & Assessment Section, 1992-1994 Water Quality Standards Review, pH).

McGeachy, S.M. and D.G. Dixon. 1990a. The impact of temperature on the acute toxicity of arsenate and arsenite to rainbow trout (*Salmo gairdneri* Richardson). Ecotoxicology, and Environmental Safety. 17:86-93.

McGeachy, S.M. and D.G. Dixon. 1990b. Effect of Temperature on the Chronic Toxicity of Arsenate to Rainbow Trout. Canadian Journal of Fisheries and Aquatic Sciences. 47:2228-2234.

McGurk et al. 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: 2187-2196.

McNichol, R.E. and E. Scherer. 1991. Behavioral responses of lake whitefish (*Conegonus clupeaformis*) to cadmium during preference-avoidance testing. Environmental Chemistry and Toxicology 10:225-234.

McNichol, R.E. and E. Sherer. 1993. Influence of cadmium pre-exposure on the preference avoidance responses of lake whitefish *(Coregonus clupeaformis)* to cadmium. Archives of Environmental Contamination and Toxicology. 25:36-40.

McPhail, J.D. and C.B. Murray, 1979. The early life history and ecology of Dolly Varden (*Salvelinus malma*) in the Upper Arrow Lakes. University of British Columbia, Department of Zoology and Institute of Animal Resources. Vancouver, BC. 133 p.

Moore, J. W. and S. Ramamoorthy. 1984. Heavy Metals in Natural Waters: Applied Monitoring and Impact Assessment. Spring-Verlag, New York, NY. as cited in Draft Biological Evaluation for Reissuance of a National Pollutant Discharge Elimination System Permit for the Potlatch Corporation, Lewiston, Idaho. 1993. Science Applications International Corporation. San Diego, CA.

Mount, D.I. 1973. Chronic Effect of Low pH on Fathead Minnow Survival, Growth, and Reproduction. In U.S. Environmental Protection Agency. 1986. Quality Criteria for Water 1986. Report USEPA/440/5-86-001. Office of Water. Washington, D.C.

National Marine Fisheries Service (NMFS). 1991. Endangered and Threatened Species; Endangered Status for Snake River Sockeye Salmon. November 20, 1991. Federal Register, Vol. 56(224):58619-58624.

National Marine Fisheries Service (NMFS). 1993. Designated Critical Habitat; Snake River Sockeye Salmon, Snake River Spring/Summer Chinook Salmon and Snake River Fall Chinook Salmon. December 28, 1993. Federal Register, Vol. 58, No. 247 pp. 68543-68553.

National Marine Fisheries Service (NMFS). 1995. Proposed Recovery Plan for Snake River Salmon. U.S. Dept of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service (NMFS). 1996. Factors for the Decline - A Supplement to the Notice of Determination for West Coast Steelhead Under the Endangered Species Act. NMFS, Portland, OR and NMFS, Long Beach, CA. 83 pp.

National Marine Fisheries Service (NMFS). 1997. Endangered Species Act - Section 7 Consultation, Biological Opinion, Hecla Mining Company--Grouse Creek Unit. U.S. Dept of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service (NMFS). 1999. An assessment of lower Snake River hydrosystem alternatives on survival and recovery of Snake River salmonids. Anadromous fish

appendix *in* Draft lower Snake River juvenile salmon migration feasibility study/environmental impact statement. U.S. Army Corps of Engineers, Walla Walla, Washington.

National Marine Fisheries Service (NMFS). 2000. Designated critical habitat: critical habitat for 19 evolutionary significant units of salmon and steelhead in Washington, Oregon, Idaho, and California. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, NMFS. February 16, 2000. Federal Register Vol. 64 No. 24 pp. 5740-5753.

National Marine Fisheries Service (NMFS). 2009. Upper Salmon River MPG-Snake River spring/summer Chinook VSP monitoring analysis. Interior Columbia Technical Recovery Team. Nation Marine Fisheries Service. Seattle, WA.

Newcombe, C. P. and J. O. T. Jensen. 1996. Channel suspended sediment and fisheries; a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16:693-727.

NOAA. (Sept) 2004. Evaluation of risks to salmonids in the mixing zone of effluent discharged from the Grouse Creek Mine into the Yankee Fork of the Salmon River, Idaho. Appendix A. Prepared by Chris Mebane, National Marine Fisheries Service, Idaho State Habitat Office. Boise, ID.

NOAA. 2005(Aug). Final Assessment of NOAA Fisheries Critical Habitat Analytical Review Teams For 12 Evolutionarily Significant Units of West Coast Salmon and Steelhead. Prepared by NOAA Fisheries Protected Resources Division. Portland, Oregon.

Oregon Department of Environmental Quality (ODEQ). 1995. Final Issue Papers 1992 - 1994 Water Quality Standards Review. Portland, OR.

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, central Idaho, 2001: U.S. Geological Survey Water-Resources Investigations Report 03-4076. 45 p.

Overton, C.K, M.A. Radko, and S.P. Wollrab. 1999. Completion report for the Yankee Fork Ranger District: watershed analysis approaches for Chinook salmon, Yankee Fork of the Salmon River, Idaho: an example. USDA Forest Service, Rocky Mountain Research Station. Boise, Idaho. 47pp.

Pacific Fisheries Management Council (PFMC). 1999. Appendix A. Identification and Description of Essential Fish Habitat, Adverse Impacts and Recommended Conservation Measures for Salmon. Amendment 14 to the Pacific Coast Salmon Plan Portland, Oregon: Pacific Fishery Management Council.

Peters, J. C. 1967. Effects on a trout stream of sediment from agricultural practices. Journal of Wildlife Management 31:805-812.

Poleo, A.B and S. Hytterød. 2003. The effect of aluminum in Atlantic salmon (Salmo salar) with special emphasis on alkaline water. Journal of Inorganic Biochemistry. 97:89–96.

Purcell, T. W. and J. J. Peters. 1998. Sources of silver in the environment. Environmental Toxicology and Chemistry 17:539-546.

Ralston and Associates. 1984. An inventory of aquatic invertebrates and fish in Jordan Creek. Report prepared for Sunbeam Mine, Stanley, Idaho.(As cited in USFS 1992).

Ralston and Associates. 1989. (As cited in USFS 1992, citation not available).

Ralston and Associates. 1993. Jordan Creek fish community survey. Report prepared for Hecla Mining Company, Challis, Idaho.

Rand, G.M. and S.R. Petrocelli. 1985. Fundamentals of aquatic toxicology: methods and applications. Hemisphere Publishing Corp. New York.

Rankin, M. G. and D. G. Dixon (1994). Acute and chronic toxicity of waterborn arsenite to rainbow trout (Oncorhynchus mykiss). Canadian Journal of Fisheries and Aquatic Sciences 51:372-380.

Rehnberg, B.C. and C.B. Schreck. 1986. Acute metal toxicology of olfaction in coho salmon: behavior, receptors, and odor-metal complexation. *Bulletin of Environmental Contamination and Toxicology.* 36:579-586.

Rhea, D.T., A.M. Farag, D.D. Harper, E. McConnell, and W.G. Brumbaugh. 2013. Mercury and selenium concentrations in biofilm, macroinvertebrates, and fish collected in the Yankee Fork of the Salmon River, Idaho, USA, and their potential effects on fish health. Arch. Environ. Contam. Toxicol. (2013) 64:130-139.

Rieman, B. E. and J. D. McIntyre. 1993. Demographic and habitat requirements of bull trout. United States Forest Service, General Technical Report INT-302. Intermountain Research Station, Boise, Idaho.

Rombough, P. J. 1983. Effects of low pH on embryos and alevins of pacific salmon. Canadian Journal of Fisheries and Aquatic Sciences. 40:1575-1582.

Saiki M. K., K. T. Castleberry, T. W. May, B. A. Martin and F. N. Bullard. 1995. Copper, Cadmium, and Zinc Concentrations in Aquatic Food Chains from the Upper Sacramento River (California) and Selected Tributaries. Archive of Environmental Contaminant Toxicology 29:484-491.

Sandahl, J.F., D.H. Baldwin, J.J. Jenkins and N.L. Scholz. 2007. A sensory system at the interface between urban stormwater runoff and salmon survival. Environmental Science and Technology 41:2998-3004.

Sarkar, S.K. 1990. Toxicity evaluation of sodium cyanide to fish and aquatic organisms: Effects of temperature. Sci. Cult. 4:165-168.

Scherer, E. and R.E. McNichol. 1998. Preference-avoidance responses of lake whitefish (*Coregonus clupeaformis*) to competing gradients of light and copper, lead, and zinc. Water Research. 32:924-929.

Scheuhammer, A. M. 1991. Effects of Acidification on the Availability of Toxic Metals and Calcium to Wild Birds and Mammals. Environmental Pollution 71:329-375.

Servizi, J. A. and D. W. Martens. 1987. Some effects of suspended Fraser River sediments on sockeye salmon (*Oncorhynchus nerka*). P 254-264 *in* H. d. Smith, L. Margolis, and C. C. Wood editors: Sockeye Salmon (*Oncorhynchus nerka*) population biology and future management. Canadian Special Publication 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.

Shephard B. and J. Zodrow. 2009. Copper Effects on Fish Behavior: A Review and Synopsis of Existing Studies. A presentation at the 30th SETAC North America Annual Meeting on November 23, 2009, New Orleans, LA. USEPA Region 10, Seattle, Washington.

Shoshone Bannock Tribes (SBT). 2008. Salmon River Habitat Enhancement. 2007 Annual Report, Prepared by T. Tsosie,, K. Bacon and P. Wadsworth, Shoshone Bannock Tribes. Project No. 94-50 Bonneville Power Administration. Portland, Oregon.

Skorupa, J.P., T.S. Presser, S.J. Hamilton, A.D. Lemly, and B. E. Sample. 2004. USEPA's draft tissue-based selenium criterion: A technical review. Unpublished report presented to USEPA , June 16, 2004. Corresponding author Joseph P. Skorupa, US Fish and Wildlife Service. Arlington Virginia.

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(2):187-196.

Sparling, D.W. and T.P. Lowe. 1996. Environmental hazards of aluminum to plants, invertebrates, fish, and wildlife: *In* G.W. Ware. (ED), Reviews of environmental contamination and toxicology. Volume 145. Springer-Verlag. New York.

Spehar, R. L. and J. T. Fiandt (1986). Acute and chronic effects of water quality criteria-based metal mixtures on three aquatic species. Environmental Toxicology and Chemistry 5: 917-931.

Spehar, R.L., J.T. Fiandt, R.L. Anderson, and D.L. DeFoe. 1980. Comparative toxicity of arsenic compounds and their accumulation in invertebrates and fish. Arch. Environ. Contam. Toxicol. 9:53-63.

Sprague, J.B., P.F. Elson, and R.L. Saunders. 1965. Sublethal copper-zinc pollution in a salmon river - a field and laboratory study. International Journal of Air and Water Pollution 9:531-543.

Sprague, J.B. 1971. Measurement of pollutant toxicity to fish—III: Sublethal effects and "safe" concentrations. Water Research.5(6):245-266.

Spry, D.J. and J.G. Wiener. 1991. Metal bioavailability and toxicity to fish in low-alkalinity lakes: A critical review. Environmental Pollution. 71(2-4):243-304.

StreamNet. 2015. Fish Distribution online database. Gladstone, Oregon. URL<http://www.streamnet.org/>.

Svecevicius, G. 1999. Acute toxicity of zinc to common freshwater fishes of Lithuania. Acta Zoologica Lituanica. Hydrobiologia, 9 (2): 114-118.

Sykora, J. L., E. J. Smith, and M. Synak. 1972. Effect of lime-neutralized iron hydroxide suspensions on juvenile brook trout (*Salvelinus fontinalis*). Water Research 6:935-950.

Thurow, R. 1987. Evaluation of the South Fork Salmon River steelhead trout fishery restoration program performed for the U.S. Department of Interior, Fish and Wildlife Service. Lower Snake River Fish and Wildlife Compensation Plan Contract No. 14-16-0001-86505.

Thurston, R.V., C. Chakoumakos, and R.C. Russo. 1981. Effect of fluctuating exposures on the acute toxicity of ammonia to rainbow trout (Salmo gairdneri) and cutthroat trout (S. clarki). Water Res. 15:911-917.

Thurston, R.V., R.C. Russo, R.J. Luedtke, C.E. Smith, E.L. Meyn, C. Chakoumakos, K.C. Wang, and C.J.D. Brown. 1984. Chronic toxicity of ammonia to rainbow trout. Trans. Am. Fish. Society. 113:56-73.

U. S. Environmental Protection Agency (USEPA). 1999. Biological assessment of the Idaho water quality standards for numeric water quality criteria for toxic pollutants for the U.S. Fish and Wildlife Service and the National Marine Fisheries Service. USEPA Region 10 Office. Seattle, WA.

U. S. Environmental Protection Agency (USEPA). 2000 (Dec.). Ambient water quality criteria recommendations information supporting the development of state and tribal nutrient criteria for rivers and streams in nutrient ecoregions II western forested streams. Report USEPA-822-B-00-015.U.S. USEPA, Office of Water, Office of Science and Technology, Health and Ecology Criteria Division, Washington D.C.

U. S. Environmental Protection Agency (USEPA). 2004. Draft: Aquatic life water quality criteria for selenium USEPA-882-D-04-001. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology. Washington, D.C. Available at: http://www.epa.gov/waterscience/criteria/aqlife.html

U. S. Environmental Protection Agency (USEPA). 2009. National recommended water quality criteria for priority pollutants. USEPA, Office of Water, Office of Science and Technology. Available at: http://www.epa.gov/ost/criteria/wqctable/

U. S. Environmental Protection Agency (USEPA). 2014 (May). External peer review draft aquatic life ambient water quality criterion for selenium-freshwater 2014. USEPA-820-F-14-005. U.S. Environmental Protection Agency, Office of Water, Mail code 4305T. Washington, D.C. Available at: http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/selenium/upload/External-Peer-Review-Draft-Aquatic-Life-Ambient-Water-Quality-Criterion-For-Selenium-Freshwater-2014.pdf

U.S. Environmental Protection Agency (USEPA). 1991. Technical support document for water quality-based toxics control. Report USEPA/505/2-90-001. U.S. USEPA Office of Water Enforcement and Permits, Office of Water Regulations and Standards. Washington D.C. March 1991.

U.S. Environmental Protection Agency (USEPA). 1980. Ambient Water Quality Criteria for Selenium. U. S. USEPA Report 440/5-80-070.

U.S. Environmental Protection Agency (USEPA). 1981. An exposure and risk assessment for silver. Office of Water, Regulations and Standards (WH-553). USEPA-440/4-81-017.

U.S. Environmental Protection Agency (USEPA). 1985a. Ambient Water Quality Criteria for Mercury - 1984. U. S. Environ. Protection Agency Rep. 440/5-84-026. 136 pp. Avail from Natl. Tech. Infor. Serv., 5285 Port Royal Road, Springfield, Virginia. 22161.

U.S. Environmental Protection Agency (USEPA). 1985b. Ambient Water Quality Criteria for Cyanide - 1984. U. S. USEPA Report 440/5-84-028. Office of Water, Washington , DC.

U.S. Environmental Protection Agency (USEPA). 1986. Quality Criteria for Water. U. S. USEPA Report 440/5-86-001 (Gold Book). Office of Water, Washington , DC.

U.S. Environmental Protection Agency (USEPA). 1987a. Ambient Aquatic Life Water Quality Criteria for Silver. USEPA 440/5-87-011. Office of Water, Washington , DC. 104 pp.

U.S. Environmental Protection Agency (USEPA). 1987b. Ambient Water Quality Criteria for Zinc. U.S. USEPA Report 440/5-87-003. Office of Water, Washington , DC.

U.S. Environmental Protection Agency (USEPA). 1994. Water Quality Standards Handbook: Second Edition. Report USEPA-823-B-94-005a. Office of Water. Washington D.C.

U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS). 1998. Endangered species consultation handbook. U.S. Government Printing Office, Washington D.C.

U.S. Fish and Wildlife Service (USFWS). 2002. Chapter 17, Salmon River Recovery Unit, Idaho. 194 p. *In*: U.S. Fish and Wildlife Service. Bull Trout (*Salvelinus confluentus*) Draft Recovery Plan. Portland, Oregon.

U.S. Fish and Wildlife Service (USFWS). 2010. Bull Trout Final Critical Habitat Justification. U.S. Fish and Wildlife Service, Boise, Idaho. 979 pp. plus appendix. U.S. Fish & Wildlife Service Idaho Fish and Wildlife Office, Boise, Idaho Pacific Region, Portland, Oregon.

U.S. Fish and Wildlife Service (USFWS). 2015. Letter to USEPA from FWS, Idaho Fish and Wildlife Office, 02/24/2015 for Project Name: Hecla Grouse Cr. Mine NPDES Consultation Code: 01EIFW00-2015-SLI-0309. Event Code: 01EIFW00-2015-E-00261. Idaho Fish and Wildlife Office, Boise, ID. http://www.fws.gov/idaho/

U.S. Fish and Wildlife Service(USFWS). 2003. Biological and conference opinions and concurrence for the effects of the Environmental Protection Agency's reissuance of a National Pollutant Discharge Elimination System Permit (Permit No. ID0026468) for Grouse Creek Mine into Jordan Creek, Yankee Fork, Watershed, Salmon River Drainage, Custer County, Idaho. US. Fish and Wildlife Service, Snake River Fish and Wildlife Office, Boise, Idaho. 87pp.

U.S. Forest Service (USFS). 1992. Final supplemental environmental impact statement, Volumes 1 and 2: Grouse Creek Project. Challis Nation Forest. Salmon, Idaho.

U.S. Forest Service (USFS). 1994. Biological assessment for the proposed Jordan Creek stream alteration. Yankee Fork Ranger District, Salmon-Challis Nation Forest. Salmon, Idaho.

U.S. Forest Service (USFS). 1995. Completion report for the Yankee Fork Ranger District: watershed analysis approaches for the Chinook salmon, Yankee Fork of the Salmon River, Idaho. USDA Forest Service, Rocky Mountain Research Station, Boise, Idaho.

U.S. Forest Service (USFS). 2004. Status of the amended Salmon-Challis National Forest Plans Management Indicator Species Bull Trout. U.S. Forest Service Salmon-Challis National Forest. Salmon, Idaho.

U.S. Forest Service (USFS). 2006. Biological assessment for the potential effects of managing the Salmon-Challis National Forest in the Yankee Fork Section 7 Watershed on Snake River Basin spring/Summer Chinook salmon, Snake River steelhead, and Columbia River bull trout. Prepared by J. Vacirca. USDA Forest Service, Salmon-Challis National Forest, Yankee Fork Ranger District. Salmon, Idaho.

U.S. Forest Service (USFS). 2007. Fisheries Biological assessment for the tailings impoundment closure activities at the Grouse Creek Mine. Yankee Fork Ranger District, Salmon-Challis Nation Forest. Salmon, Idaho.

U.S. Forest Service (USFS). 2008 (April). The status of fishes on the Yankee Fork Ranger

District, Salmon-Challis National Forest. Prepared by B.L. Gamett and J.A. Bartel. USDA Forest Service, South Zone Fish Program, Salmon-Challis National Forest. Challis, Idaho.

Upper Salmon River Interagency Technical Advisory Team (USRITAT). 1998. Upper Salmon River key watershed bull trout problem assessment draft (USRPA). Idaho Department of Fish and Game. Boise, ID. 171 p.

USEPA (Environmental Protection Agency). 2001. Water Quality Criterion for the Protection of Human Health : Methylmercury. USEPA-823-R-01-001. Office of Water. Washington DC. Available at: http://www.epa.gov/waterscience/criteria/methylmercury/criteria.html

USEPA. 1988. Ambient aquatic life water quality criteria for aluminum. USEPA 440/5-86-008. United States Environmental Protection Agency, Office of Water Regulation and Standards Criteria and Standards Division. Washington, DC 20460

USEPA. 2004. National Recommended Water Quality Criteria. United States Environmental Protection Agency, Office of Water, Office of Science and Technology. Available at: http://www.epa.gov/sites/production/files/2015-06/documents/nrwqc-2004.pdf

USEPA. 2015. National recommended water quality criteria: current water quality criteria. Available at: http://water.epa.gov/scitech/swguidance/standards/criteria/current/index.cfm

USFS and USEPA. 2003. Emergency fisheries biological assessment for mine, Yankee Fork Section 7 Watershed. Yankee Fork Ranger District, Salmon-Challis National Forest and U.S. Environmental Protection Agency, Region 10 Idaho Operations Office. Prepared by T. A. Montoya and E.C. Morgan.

Van Derveer, W. D. and S. P. Canton (1997). Selenium sediment toxicity thresholds and derivation of water quality criteria for freshwater biota of western streams. Environmental Toxicology and Chemistry 16: 1260-1268.

Weaver, T.M. and R.G. White. 1985. Coal Creek fisheries monitoring study No. III. Quarterly Progress Report. USDA Forest Service, Montana State Cooperative Fisheries Research Unit. Bozeman, MT. 94p.

Westin, D.T. 1974. Nitrate and nitrite toxicity to salmonid fishes. Progressive Fish Culturalist 36(2):86-89.

Woodward, D.F., J.N. Goldstein, A.M. Farag and W.G. Brumbaugh. 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.

Wren, C. D., MacCrimmon, H. R., and Loescher, B. R. 1983. Examination of bioaccumulation and biomagnification of metals in a Precambrian shield lake. Water Air Soil Pollution 19:277-291.

Zimmerman, C.E. and G.H. Reeves. 2000. Population structure of sympatric anadromous and nonanadromous *Oncorhynchus mykiss*: evidence form spawning surveys and otolith microchemistry. Canadian Journal of Fisheries and Aquatic Sciences. 57:2152-2162.

XIII. Appendices

Appendix A. Fish survey results.

Jordan Creek mean salmonid density (fish/100 m^2), 1996 – 1999 from six sites (Source: Unpublished Data, Shoshone-Bannock Tribe, contact Jeff Anderson).

	Mean density (Standard error)						
Year	**Age 0+ Chinook salmon**	**Age 0+ steelhead**	**Age 1+/2+ steelhead**	**Mountain whitefish**	**Bull trout**	**Westslope cutthroat trout**	**Rainbow trout**
1996	0.00 (0.00)	0.00 (0.00)	1.85 (0.77)	0.24 (0.26)	0.00 (0.00)	0.30 (0.33)	0.00 (0.00)
1997	0.00 (0.00)	0.00 (0.00)	0.42 (0.32)	0.19 (0.12)	0.00 (0.00)	0.42 (0.32)	0.09 (0.09)
1998	1.62 (1.62)	0.00 (0.00)	0.15 (0.15)	0.00 (0.00)	0.00 (0.00)	0.42 (0.42)	0.37 (0.24)
1999	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.10 (0.10)	0.14 (0.14)	1.37 (0.84)	0.00 (0.00)

Jordan Creek fish counts from snorkel surveys. (Source: Unpub. Data, Shoshone-Bannock Tribe, contact Jeff Anderson).

Species	Year	Count	Location
Cutthroat trout (75-170mm)	2000	11	all locations
Steelhead (110mm)	2000	1	below 002
Bull trout (120mm)	2000	1	below 002
Chinook	2000	0	below 002
bull trout (90mm)	1999	1	below 002
Chinook (50-80mm)	1998	4	(200m above Yankee Fork confluence)

Snorkel survey results from Jordan Creek and the Yankee Fork in July-Sept. 2006. (Source: USFS 2008).

Species	Upper Jordan headwaters	Mid Jordan near outfall 002	Lower Jordan above Yankee fork confluence	Yankee Fork above Salmon R confluence	Yankee Fork below the Jordan Creek confluence
Cutthroat trout	44	10	2	1	0
Steelhead/rainbow	0	4	17	32	5
Bull trout	0	0	0	0	2
whitefish	0	0	4	60	19
Chinook juveniles	0	0	0	5	1
Unidentified salmonid	0	0	0	0	1

Appendix B. Summary of water quality monitoring data collected at six monitoring sites by Hecla Limited (Hecla Limited 2013): outfall 002 (Jordan Creek), outfall 003 (Yankee Fork), Site S-6 (Jordan Creek upstream of Outfall 002), Site S-4 (Jordan Creek downstream of Outfall 002), **Site S-9 (Yankee Fork upstream of Outfall 003), and Site S-10 (Yankee Fork downstream of Outfall 003).** The sampling frequency is as follows: Outfalls 002 and 003 are sampled weekly for all parameters when discharging. The other four monitoring sites are sampled quarterly in April, June, August, and October. These samples were not collected if streams are frozen, dry, or if access is unsafe. Temperature data are collected at a different frequency for Site S-4 and the two outfalls. At Site S-4 temperature data are collected daily (Jordan Creek downstream of Outfall 002) and temperature is collected daily at outfalls 002 and 003 when discharging.

Appendix B. Summary statistics of water quality monitoring data collected at both outfalls and 4 monitoring sites 2003-2013 (excluding 2011). (source: Hecla Limited).

Outfall 002 Jordan Creek							
Variable	**Units**	**N**	**Mean**	**Median**	**Min.**	**Max.**	**Std.Dev.**
Ammonia	mg/L	141	0.098	0.060	0.050	0.740	0.0944
Cadmium, TR	mg/L	376	0.000	0.000	0.000	0.001	0.0001
Calcium, TR	mg/L	95	125.089	123.000	28.200	284.000	47.4777
Conductivity	mmho	990	643.198	656.500	211.000	1463.000	153.9121
Copper, TR	mg/L	376	0.004	0.004	0.001	0.021	0.0032
Cyanide (WAD)	mg/L	595	0.003	0.002	0.002	0.014	0.0017
Hardness	mg/L	95	341.168	340.000	87.000	734.000	120.5048
Lead, TR	mg/L	376	0.000	0.000	0.000	0.001	0.0001
Magnesium, TR	mg/L	95	6.909	7.100	3.800	10.100	1.7132
Mercury, T	mg/L	379	0.000	0.000	0.000	0.000	0.0001
Nitrate + Nitrite, T	mg/L	92	0.909	0.730	0.020	3.260	0.6275
pH	SU	990	7.174	7.100	6.600	8.800	0.3228
Selenium, TR	mg/L	147	0.002	0.002	0.000	0.009	0.0014
Silver, TR	mg/L	376	0.000	0.000	0.000	0.000	0.0000
Temperature	C	990	6.400	4.600	0.010	18.700	4.7440
TSS	mg/L	383	5.173	4.000	1.000	80.000	7.0467
Turbidity	NTU	989	2.520	2.100	0.200	21.900	1.8969
WET (acute)	TUA	10	1.100	1.000	1.000	1.500	0.2108
WET (chronic)	TUC	27	2.163	1.000	1.000	16.000	3.1854
Zinc, TR	mg/L	376	0.014	0.010	0.002	0.210	0.0141

Appendix B continued. Summary statistics of water quality monitoring data collected at both outfalls and 4 monitoring sites 2003-2013 (excluding 2011) (source: Hecla Limited).

Outfall 003 Yankee Fork							
Variable	**Units**	**N**	**Mean**	**Median**	**Min.**	**Max.**	**Std.Dev.**
Ammonia	mg/L	468	2.477	0.190	-0.050	16.400	4.2562
Arsenic, TR	mg/L	38	0.013	0.009	0.001	0.045	0.0108
Cadmium, TR	mg/L	468	0.000	0.000	0.000	0.005	0.0003
Calcium, TR	mg/L	468	131.709	131.500	27.100	250.000	28.1348
Chromium, TR	mg/L	38	0.008	0.010	-0.010	0.010	0.0054
Conductivity	mmho	2977	1010.559	931.000	7.500	1760.000	369.8651
Copper, TR	mg/L	468	0.004	0.004	0.001	0.105	0.0059
Cyanide (WAD)	mg/L	2527	0.005	0.002	-0.002	0.138	0.0080
Flow	(gpm)	2978	367.120	282.100	13.300	1156.300	287.7734
Hardness	(mg/L)	468	359.092	360.000	85.000	657.000	73.4717
Lead, TR	(mg/L)	471	0.000	0.000	0.000	0.017	0.0010
Magnesium, TR	(mg/L)	468	7.249	7.100	4.300	13.300	1.4668
Mercury, T	(mg/L)	473	0.000	0.000	0.000	0.000	0.0001
Nickel, TR	(mg/L)	38	0.009	0.010	-0.010	0.020	0.0051
pH	SU	2977	7.410	7.400	6.300	8.900	0.3957
Selenium, TR	(mg/L)	475	0.012	0.006	0.000	0.044	0.0110
Silver, TR	(mg/L)	468	0.000	0.000	0.000	0.001	0.0001
Temperature	C	2978	8.119	6.600	0.300	20.400	5.4012
TSS	(mg/L)	471	4.118	4.000	1.000	180.000	9.2808
Turbidity	(NTU)	2960	3.549	3.000	0.700	301.000	6.1830
WET (chronic)	TUC	15	7.213	1.000	-1.000	52.100	15.5787
Zinc, TR	(mg/L)	468	0.016	0.010	-0.002	0.348	0.0243

Appendix B continued. Summary statistics of water quality monitoring data collected at both outfalls and 4 monitoring sites 2003-2013 (excluding 2011)(Source: Hecla Limited 2010, 2011, 2013, 2014).

Site S-6: Jordan Creek upstream of Outfall 002.							
Variable	**Units**	**N**	**Mean**	**Median**	**Min.**	**Max.**	**Std.Dev.**
Ammonia	mg/L	69	0.057	0.050	0.050	0.120	0.01777
Calcium, TR	mg/L	39	10.497	11.800	5.000	15.100	3.13499
Conductivity	mmho	69	72.980	77.900	18.000	149.000	23.74350
Copper, D	mg/L	39	0.001	0.001	0.001	0.002	0.00029
Cyanide (WAD)	mg/L	69	0.002	0.002	0.002	0.002	0.00000
Hardness	mg/L	39	29.205	32.000	15.000	42.000	8.55071
Lead, D	mg/L	39	0.000	0.000	0.000	0.000	0.00005
Magnesium, TR	(mg/L)	39	0.685	0.700	0.300	1.100	0.20201
Mercury, T	(mg/L)	22	0.000	0.000	0.000	0.000	0.00009
Nitrate + Nitrite, T	(mg/L)	69	0.027	0.020	0.020	0.110	0.01548
pH	SU	69	7.197	7.300	5.800	7.800	0.38652
Selenium, TR	(mg/L)	39	0.000	0.000	0.000	0.001	0.00041
Temperature	C	69	6.788	5.900	0.500	16.300	3.71715
TSS	(mg/L)	23	1.078	1.000	1.000	2.000	0.26104
Turbidity	(NTU)	69	1.616	1.000	0.220	8.810	1.73970
Zinc, D	(mg/L)	39	0.009	0.004	0.002	0.040	0.00901

Site S-4: Jordan Creek downstream of Outfall 002							
Variable	**Units**	**N**	**Mean**	**Median**	**Min.**	**Max.**	**Std.Dev.**
Ammonia	mg/L	216	0.061	0.050	0.050	0.160	0.02220
Cadmium, D	mg/L	4	0.000	0.000	0.000	0.000	0.00000
Calcium, TR	mg/L	72	23.393	24.300	6.800	44.000	8.36307
Conductivity	mmho	218	155.398	159.650	8.500	318.000	47.56517
Copper, D	mg/L	38	0.001	0.001	0.001	0.010	0.00169
Cyanide (WAD)	mg/L	220	0.002	0.002	0.002	0.003	0.00007
Hardness	mg/L	72	66.139	68.500	20.000	120.000	23.05126
Lead, D	mg/L	38	0.000	0.000	0.000	0.000	0.00002
Magnesium, TR	(mg/L)	72	1.844	2.000	0.600	2.900	0.62323
Mercury, T	(mg/L)	39	0.000	0.000	0.000	0.000	0.00000
Nitrate + Nitrite, T	(mg/L)	217	0.134	0.120	0.020	0.640	0.08873
pH	SU	218	7.420	7.400	5.800	8.500	0.32531
Selenium, TR	(mg/L)	39	0.001	0.000	0.000	0.001	0.00035
Silver, D	(mg/L)	4	0.000	0.000	0.000	0.000	0.00000
Temperature	C	423	5.984	6.300	0.010	16.700	3.44960
TSS	(mg/L)	45	5.704	2.000	1.000	140.000	20.65157
Turbidity	(NTU)	216	2.562	1.335	0.300	94.300	6.77407
Zinc, D	(mg/L)	38	0.008	0.005	0.002	0.040	0.00787

Appendix B continued. Summary statistics of water quality monitoring data collected at both outfalls and 4 monitoring sites 2003-2013 (excluding 2011) (Source: Hecla Limited).

Site S-9: Yankee Fork upstream of Outfall 003.							
Variable	**Units**	**N**	**Mean**	**Median**	**Min.**	**Max.**	**Std.Dev.**
Ammonia	mg/L	41	0.058	0.050	0.050	0.120	0.01883
Calcium, TR	mg/L	41	6.178	6.100	4.800	7.700	0.60808
Conductivity	mmho	196	67.481	66.000	35.000	326.000	25.54469
Copper, D	mg/L	40	0.001	0.001	0.001	0.002	0.00039
Cyanide (WAD)	mg/L	192	0.002	0.002	0.002	0.002	0.00000
Hardness	mg/L	41	19.220	19.000	15.000	24.000	1.91719
Lead, D	mg/L	44	0.000	0.000	0.000	0.001	0.00011
Magnesium, TR	(mg/L)	41	0.922	0.900	0.700	1.300	0.12750
Mercury, T	(mg/L)	25	0.000	0.000	0.000	0.000	0.00010
Nitrate + Nitrite, T	(mg/L)	37	0.026	0.020	0.020	0.100	0.01819
pH	SU	196	7.074	7.165	5.800	8.300	0.45473
Selenium, TR	(mg/L)	42	0.000	0.000	0.000	0.001	0.00043
Temperature	C	196	5.807	5.150	0.300	16.900	3.73647
TSS	(mg/L)	40	3.430	2.000	1.000	20.000	4.22564
Turbidity	(NTU)	185	3.691	2.490	0.960	43.000	4.81617
Zinc, D	(mg/L)	40	0.008	0.004	0.002	0.030	0.00725

Site S-10:Yankee Fork downstream of Outfall 003.							
Variable	**Units**	**N**	**Mean**	**Median**	**Min.**	**Max.**	**Std.Dev.**
Ammonia	mg/L	41	0.065	0.050	0.050	0.200	0.0307
Calcium, TR	mg/L	40	10.283	10.250	5.000	14.400	2.2549
Conductivity	mmho	177	123.014	116.000	49.000	1472.000	106.1510
Copper, D	mg/L	40	0.001	0.001	0.001	0.003	0.0005
Cyanide (WAD)	mg/L	217	0.002	0.002	0.002	0.002	0.0000
Hardness	mg/L	40	30.700	30.500	16.000	43.000	6.4855
Lead, D	mg/L	40	0.000	0.000	0.000	0.000	0.0000
Magnesium, TR	(mg/L)	41	1.215	1.200	0.800	1.700	0.2455
Mercury, T	(mg/L)	24	0.000	0.000	0.000	0.000	0.0001
Nitrate + Nitrite, T	(mg/L)	37	0.072	0.050	0.020	0.360	0.0708
pH	SU	177	6.988	7.000	6.000	8.000	0.3970
Selenium, TR	(mg/L)	42	0.000	0.000	0.000	0.001	0.0004
Temperature	C	176	6.209	5.650	0.400	16.300	3.7211
TSS	(mg/L)	40	4.180	2.000	1.000	37.400	6.5310
Turbidity	(NTU)	169	3.067	2.000	0.700	40.300	4.0531
Zinc, D	(mg/L)	40	0.007	0.004	0.002	0.030	0.0063