US ENVIRONMENTAL PROTECTION AGENCY REGION 10

Biological Evaluation for the National Pollutant Discharge Elimination System Permit for Clearwater Paper Lewiston Mill

NPDES Permit Number ID0001163

U.S. EPA Region 10 Office of Water and Watersheds
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1 Introduction

Clearwater Paper Corporation (Clearwater) operates a pulp, paper, and wood products mill (the Mill) in Lewiston, Nez Perce County, Idaho and discharges effluent from the Mill into the Snake River at its confluence with the Clearwater River. EPA is proposing to reissue the National Pollutant Discharge Elimination System (NPDES) permit for the Clearwater Mill (NPDES Permit No. ID00001163) located in Nez Perce County, Idaho, in the City of Lewiston. The draft NPDES permit will authorize discharge from one outfall, designated Outfall 001, to the Snake River and seepage from the secondary treatment pond to the Clearwater River.

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 (NOAA Fisheries) if the federal agency's actions could beneficially or adversely affect any threatened and endangered species or their critical habitat. In this case, the federal agency is the Environmental Protection Agency (EPA), and the discretionary action is the reissuance of the NPDES permit. The action evaluated in this Biological Evaluation (BE) could affect species under the jurisdiction of both the USFWS and NOAA Fisheries. This BE identifies the endangered, threatened, and proposed species and critical habitat in the project area and assesses potential effects to these species that may result from the discharge authorized in the draft Clearwater Mill NPDES permit.

The following major discussions are provided in this evaluation using the best scientific and commercial data available:

- The proposed action and the action area (including the relevance of the environmental baseline to the species' current status) are described in parts 3 and 4;
- Parts 2, 5, and 6 identify the listed species and critical habitat in the action area and define the species' biological requirements and habitat, abundance trends, and current status;
- Part 7 provides the effects analyses of the proposed action on the listed species and critical habitat;
- A summary of the effects and conclusions of the action evaluation are provided in Part 7.

In order to adhere to the recommended contents of a biological assessment for submission to USFWS (USFWS and NMFS, 1998), Table 1-1 lists the sections of this BE that correspond to the recommended content topics.

Table 1-1: Corresponding Sections of this BE to NOAA Fisheries and USFWS Recommended Contents for Biological Assessments

Recommended Content		Heading in this BE	Section(s)
Introduction		Introduction	1
List of Species	List of Species (citation) Critical Habitat (official status)	List of Species	2
Project Description	Type and scope of Project Project components pertinent to the species Management actions such as proposed monitoring of species and mitigation that may affect species	Description of Action	3
Description of Project Area	Legal description and map Define action area Current condition of habitat parameters Past and present activities related to species/habitat Analysis of cumulative effects	Description of Action Area	4
Description of Species and Habitat	General species descriptions and habitat requirements Species distribution and habitat specific to action area by life history phase Species status, distribution, and abundance trends in action area Description of Critical Habitat, if designated	Species Descriptions Habitat Characteristics of the Receiving waters	5 6
Inventories and Surveys	Describe effort to obtain information on species status Describe information used in Description of Species and Habitat in a Table	Species Descriptions	5
Analysis of Effects	Description of parameters of concern Analysis of potential direct and indirect effects Analysis of interdependent and interrelated actions Environmental baseline – track the conservation status of a species and its environment up to the present moment (starting at time of listing or earlier) Effects determinations Analysis of effects to designated critical habitat	Analysis of Effects from the Action	7
Conclusions	Summary of determinations Statements of effect of the project on the species (e.g., no affect, may affect, etc.)	Conclusions	7
References	Literature cited Copies of pertinent documents and maps List of personal communication contacts, contributors, preparers	References	1-7
Supporting Information	Supporting documents that will assist the reviewer	Appendices	A-I

1.1 References

USFWS and NMFS. 1998. Endangered Species Consultation Handbook: Procedures for Conducting Consultations and Conference Activities Under Section 7 of the Endangered Species Act. U.S. Fish and Wildlife Service and National Marine Fisheries Service.

2 List of Species

According to the USFWS species list (ID: 01EIFW00-2016-SLI-1045; WA: 01EWFW00-2016-SLI-1286) and the NOAA Fisheries species list (http://www.westcoast.fisheries.noaa.gov/protected_species/species_list/species_lists.html), the following federally-listed species are near the discharge:

Mammal Species:	Washington ground squirrel (*Urocitellus washington*) (Candidate)
Fish species:	Bull Trout (*Salvelinus confluentus*)
	Fall Chinook salmon (*Oncorhynchus tshawytscha*)
	Sockeye salmon (*Oncorhynchus nerka*)
	Spring/summer Chinook salmon (*Oncorhynchus tshawytscha*)
	Steelhead trout (*Oncorhynchus mykiss*)
Bird species:	Yellow-Billed Cuckoo (*Coccyzus amerianus*)
Plant species:	Spalding's Catchfly (*Silene spaldingii*)
	Northern wormwood (*Artemsia campestris var. wormskioldii*) (Candidate)

Additionally, NOAA Fisheries has designated Critical Habitat for Snake River Fall Chinook salmon, Spring/summer Chinook salmon, Sockeye, and Snake River Steelhead, bull trout and critical habitat for Yellow-Billed Cuckoo has been proposed by the USFWS.

3 Description of Action

This part describes the permit action proposed by EPA. The discussion includes a general overview of the proposed action, a discussion on the permit status, a description of the industrial process, a description of the outfall, and a discussion of the proposed final effluent limits in the permit.

3.1 Overview of Permit Action

Section 301(a) of the Clean Water Act (CWA) prohibits the discharge of pollutants except in compliance with CWA Section 402, among other sections. Section 402 authorizes the issuance of NPDES permits for direct dischargers (i.e., existing or new industrial facilities that discharge process wastewaters from any point source into receiving waters). The NPDES permit is developed to control the discharge using effluent limitations guidelines (ELGs) and water quality-based effluent limitations (WQBELs).

EPA establishes ELGs to require a minimum level of process control and treatment for industrial point sources. They are based on the demonstrated performance of model process and treatment technologies that are within the economic means of an industrial category. Although ELGs are based on the performance of model process and treatment technologies, EPA does not mandate the use of specific technologies; therefore, dischargers are free to use any available control technique to meet the limitations.

All receiving waters have ambient water quality standards that are established by the states or EPA to maintain and protect designated uses of the receiving water (e.g., aquatic life, public water supply, primary contact recreation). The application of the ELGs may result in pollutant discharges that exceed the water quality standards applicable to the receiving waters. In such cases, the CWA and federal guidelines require the development of more stringent WQBELs for the pollutant to ensure that the water quality standards are met. Additionally, pollutant parameters not limited in the ELGs may result in the development of WQBELs. EPA develops WQBELs in accordance with EPA guidance (EPA, 1991).

In cases where the receiving water body does not meet a water quality standard, States can use the total maximum daily load (TMDL) process as one way of quantifying the allowable pollutant loadings in receiving waters, based on the relationship between pollutant sources and in-stream water quality standards. A TMDL will provide a wasteload allocation for each point source discharge and load allocations for nonpoint discharges. A WQBEL would be developed for a point source discharge consistent with the wasteload allocation in the TMDL.

EPA is proposing to reissue an NPDES permit to the Clearwater Corporation for the Clearwater Mill in Lewiston, Idaho. The ESA regulations require the action agency to evaluate all interdependent actions (actions having no independent utility apart from the proposed action) and interrelated actions (actions that are part of a larger action and depend on the larger action for their justification). The federal regulations at 50 CFR section 402.02 define an action as all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies in the United States or upon the high seas. Because this is an existing facility that EPA is proposing to reauthorize a permitted discharge and there are no other Federal actions

associated with this facility, EPA believes that there are no interdependent or interrelated actions to this action.

EPA has no legal authority to control air emissions under its permitting authority in the Clean Water Act. As to EPA authority under the Clean Air Act, the state of Idaho implements its own State Implementation Plan (SIP)-approved Prevention of Significant Deterioration (PSD) permit program for construction/modification or major/significant projects. Idaho also has the SIP-approved minor permitting program for non-PSD air quality permits. Therefore, Idaho implements its own air permit program and EPA cannot force any changes to Clearwater Paper's air permits. Since control of air emissions is a State activity, it is required by the ESA regulations to be evaluated as a cumulative effect (effects of future State or private activities, not involving Federal activities, which are reasonably certain to occur within the action area of the Federal action subject to consultation). Cumulative effects of this action are discussed in Section IV.B of this BE. Past and ongoing effects of air emissions are included in the environmental baseline described in Section VII.E of this BE.

A copy of the draft NPDES permit is included in Appendix A of this BE. The draft NPDES permit authorizes the discharge from existing Outfall 001 to the Snake River and seeps from the secondary treatment aeration pond to the Clearwater River subject to effluent limitations, monitoring, and other conditions specified in the permit. The draft permit will be finalized following completion of this consultation.

3.2 Permit Reissuance Status

The current permit for the Clearwater Mill was issued on May 1, 2005 under the Potlach Corporation. The permit was transferred to the Clearwater Paper Corporation in 2008 and was modified on April 15, 2010 to reflect a change in the BOD_5 limit. The RBM10 model application developed by EPA was reviewed and refined in response to a request from the NPDES program for analysis of the impact of the wastewater BOD discharge from the Clearwater Paper Corporation pulp mill on the Snake River dissolved oxygen. The revised RBM10 model application was released in September 2009 and included new data and information that became available after the original EPA modeling analysis performed in 2002. Differences from the original analysis included a decreased river BOD oxidation rate, increased atmospheric reaeration rates, and the addition of a sediment oxygen demand term to the dissolved oxygen balance equation. The revised model demonstrated that Clearwater's discharge at the interim effluent limits for BOD_5 in the 2005 permit would not cause or contribute to a violation of Washington Water Quality Standards for dissolve oxygen (Gallagher and Mancilla Alarcon, 2009).

Clearwater Paper Company then requested EPA's approval to change their final water quality-based effluent limit for BOD_5 from 5,100 lb/day to 9,700 lb/day to go into effect April 1, 2010, because the original modeling was based on conservative estimates of atmospheric re-aeration rates and a limited amount of BOD decay rate data. The original water quality modeling showed that a discharge of 5,100 lb/day of BOD_5 could cause a decrease in dissolved oxygen of 0.2 mg/L, the maximum decrease allowed under Washington state water quality standards. The new information showed that discharging 9,800 lb/day of BOD_5 (100 lb/day greater than the Clearwater request) would result in a decrease in dissolved oxygen of only 0.14 mg/L, meaning that the actual impact of the requested effluent limit revision would be less than what was expected from the limits modeled in NMFS' 2004 Opinion. EPA agreed that the technical

information submitted by Clearwater was sound and that the requested permit modification was justified under NPDES regulations (40 CFR 122.62). EPA requested agreements from NMFS and USFWS to confirm that a modification to the Clearwater permit did not require re-initiation of consultation under Section 7 of the Endangered Species Act. The Services agreed that re-initiation of consultation was not required and that the requested change was within the scope and intent of the original consultation (NMFS 2010, USFWS 2010).

Currently the Clearwater Paper Corporation is operating under the 2005 permit that has been administratively continued since its expiration on April 30, 2010. The draft permit included in Appendix A will be issued upon concurrence with the Services on this Biological Evaluation, which is expected in 2019.

3.3 Facility Background

The Clearwater Corporation – Lewiston Complex (Clearwater Mill) is owned and operated by Clearwater Corporation, which has its headquarters in Spokane, Washington. Construction of a sawmill facility at the site began in 1926 by the Clearwater Timber Company. The sawmill became operational in 1927. The Clearwater Timber Company merged with two other lumber companies in 1931 to form Potlatch Forests, Inc. It later changed its name to Potlatch Corporation. In 1949, a veneer plant was completed, and construction of a pulp and paperboard mill was initiated. Operations at the pulp and paper mill began in 1950. By 1981, Potlatch had grown to include the pulp and paperboard mill, a consumer products division (tissue mill), Clearwater Lumber and panel operations, and a greenhouse. Today, the Clearwater Mill has the capability to produce 500,000 tons per day of paperboard and tissue and 160 million feet per year of lumber.

The Clearwater Mill is located approximately one mile east of the Clearwater Memorial Bridge in Lewiston, Idaho. It is situated on the south bank of the Clearwater River approximately three miles east of the Clearwater River and Snake River confluence.

Prior to 1952, the facility discharged its effluent into the Clearwater River just downstream of the city of Lewiston's Drinking Water Plant that is located on the south bank of the Clearwater River approximately 2.5 miles east of the Clearwater River and Snake River confluence. In 1952, Potlatch moved its effluent outfall to the southeast bank of the Snake River at the confluence. In preparation for the new river flow conditions after the construction of Lower Granite Dam, Potlatch relocated its outfall in 1972.

In 2008 Clearwater Paper Corporation spun off from Potlatch Corporation and controls the Lewiston Mill.

3.4 Industrial Process

This section provides an overview of the industrial process conducted at the Mill, including descriptions of the technologies used, recovery of materials used, and treatment of wastewater. Figure 3-1 shows a schematic of the industrial process and water balance information.

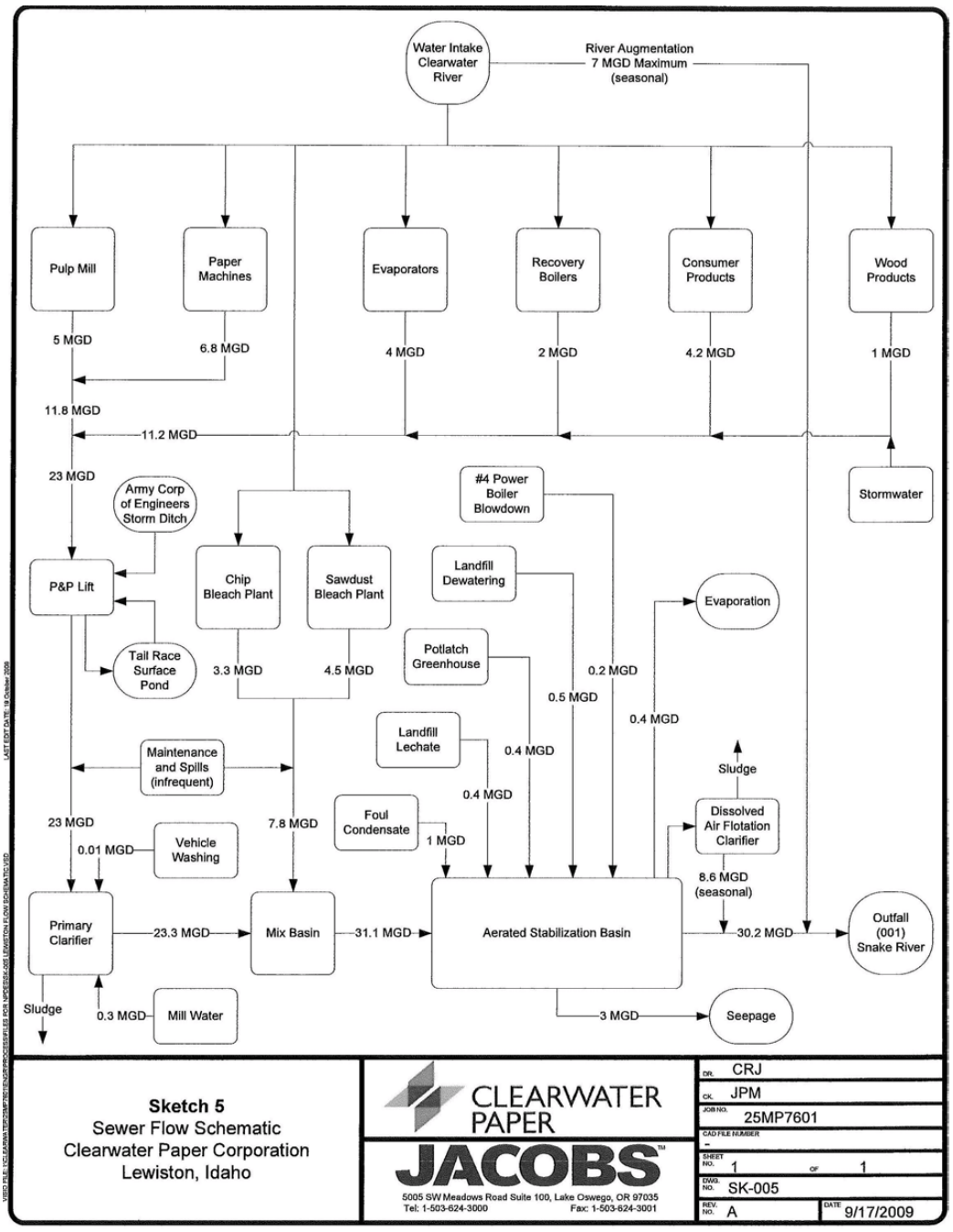

Figure 3-1: Clearwater Water Balance and Process Flow Diagram

3.4.1 Overview of Pulp, Paper and Paperboard Process

The Clearwater Mill manufactures wood products and bleached grades of paperboard, tissue and market pulp by the kraft (sulfate) process. In 1950, the mill began conversion to a kraft (sulfate) mill to produce bleached grades of paperboard, tissue and market pulp. In the 1990's, the mill completed the conversion of its bleaching process from chlorine to chlorine dioxide.

The production of pulp, paper, and paperboard involves several standard manufacturing processes including raw material preparation, pulping, bleaching, and papermaking. Raw material preparation consists of log washing, bark removal, and chipping operations. Pulping is the operation of reducing a cellulosic raw material into a form suitable for chemical conversion or for further processing into paper or paperboard. After pulping, the unbleached pulp is brown or deeply colored because of the presence of lignin and resins, or because of the inefficient washing of the spent cooking liquor from the pulp. To remove these color bodies from the pulp and to produce a light colored or white product, it is necessary to bleach to pulp. Once the pulps have been prepared from wood, further mixing, blending, and additives are necessary to prepare a suitable “furnish” for making most paper or board products.

3.4.2 Pulping Process

The Clearwater Mill operates two pulp-manufacturing processes. The chip pulp mill processes wood chips in 12 batch digesters and produces most of the pulp. The sawdust pulp mill produces pulp in continuous digesters. The wood chips and sawdust from which pulp is made are obtained from the onsite sawmill and from outside suppliers. In the early 1990s, a major rebuild of the Lewiston pulp mill’s fiber line took place. A new recovery boiler was completed in 1987, and three older units were dismantled. The chip fiber processing line was rebuilt in the 1990’s. As part of the project, most of the antiquated liquor recovery facilities were dismantled and the old chlorine-based bleaching line for chip pulp was replaced with a chlorine dioxide and oxygen based bleaching line.

The pulping process begins as the chips and sawdust are cooked in large vessels called digesters. In the digesters, the chips and sawdust are processed with cooking liquor and transformed into pulp fibers. The cooking liquor is composed of white liquor (sodium hydroxide and sodium sulfide), weak black liquor, and anthraquinone (AQ).

At the sawdust pulp mill, the pulp moves from the continuous digesters to the blow tank, then to the brownstock washers, and finally to the sawdust bleach plant. Conventional countercurrent brownstock washing is used on both the chip and sawdust pulp mills to reduce fresh water usage. After washing the brownstock is sent through the cleaning and screening system. Finally, the washed and cleaned sawdust pulp is sent to the decker, which thickens it. The brownstock is then sent to storage until needed by the sawdust bleach plant. The bleach plant removes remaining lignin and brightens the pulp. Sawdust bleaching uses chlorine dioxide, caustic, oxygen and hydrogen peroxide in a three-stage process. The first D1 stage uses chlorine dioxide, followed by an EOP stage (extraction with oxygen and peroxide) and a final D2 stage that uses chlorine dioxide. After the last bleach stage, the pulp is washed with hot water

The chip batch digesters process the chips in batches rather than continuously like the sawdust digester. Pulp moves from the digesters to the blow tank, to the brownstock screening and washing system. Prior to the chip bleach plant, the chip pulp is further processed through an oxygen delignification system, which removes lignin from the pulp. Removing lignin will

decrease chemical application within the bleach plant, resulting in cleaner effluent. At the chip bleach plant bleaching occurs in three stages, though the use of chlorine dioxide, oxygen, caustic and peroxide. The first D1 stage uses chlorine dioxide, followed by an EOP stage (extraction with oxygen and peroxide) and a final D2 stage that uses chlorine dioxide.

Most of the water used in the brownstock process is recovered and reused. The process sewer receives gland/seal water from pumps and cooling water from heater exchangers. If process water from the pulp mills enters the sewer, the Best Management Practices (BMP) Plan is implemented.

3.4.3 Paper Machines

Bleached pulp from the chip and sawdust bleach plants is stored until needed at the No. 1 and No. 2 paper machines. No. 1 machine, installed in 1950, is a one-ply machine. No. 2 machine, installed in 1955 and rebuilt in 1990, is a three-ply machine. Bleached pulp can also be used at the tissue mill or dried for market pulp in the pulp dryer. At the stock prep area of the paper machines, the pulp is combined with a large amount of water to form stock slurry. The diluted stock is passed through a series of cleaners before it can be used in the paper machines. Rejected material from the cleaning process is sent to the process sewer. Furnish is the mixture of fiber, chemicals and diluted stock. Most of the water from the furnish is removed at the wet end of the paper machines. In the first stage of the wet end, the stock is discharged onto a moving mesh wire called a Fourdrinier. This allows water drainage and sheet formation to begin. The press section of the machine removes most of the remaining water by squeezing the sheet through heavy rubber rolls. Steam-heated dryers remove the remaining moisture in the sheet. The final stages of papermaking occur at the dry end, where starch and coatings are added to the sheet. The reel then winds the sheet onto a spool to produce a parent reel.

White water is a general term for all water that has been removed from the sheet in the papermaking process. Because the sheet begins as 99.5 percent water, there is an abundance of white water for reuse. Each machine has a save-all system that recovers fiber and chemicals from the white water. Clear white water is used as dilution and shower water on the machines. White water that cannot be reused is sent to the process sewer.

Tissue production uses a process similar to paperboard production. The mill has three tissue machines. The first was installed in 1963 and rebuilt in 1994. The second and third machines were in installed in 1979 and 1993.

3.4.4 Recovery Operations

Weak black liquor recovered from the digesters and brownstock washers contains lignin and chemicals from the pulping process. The weak black liquor is concentrated in evaporators to heavy black liquor and burned in one of two recovery boilers. The organics in the liquor are combusted to produce steam. The resulting chemicals are recovered as smelt at the bottom of the furnace. The smelt is mixed with weak wash to form green liquor. Lime is added to the green liquor to form white liquor, which is sent back to the digesters completing the chemical recovery cycle. The lime mud (calcium carbonate) obtained after settling the white liquor is dewatered on rotary vacuum filters and processed in one of two lime kilns. Wastewater from the recovery operations includes weak condensates from the evaporators and wastewater from the lime mud filters.

3.4.5 Water Use and Wastewater Treatment Technologies

Operation of the processes conducted at the Mill requires the use of water. As shown in Figure 3-1, approximately 30 million gallons per day (mgd) of water is withdrawn from the Clearwater River upstream of the Mill for use in the manufacturing process. Solids are removed from the river water prior to use in the process, placed in drying beds, and disposed of on-site as clean fill.

Wastewater from the site is channeled into three main lines. One line carries higher pH wastewater from the recovery areas, tissue and paper machines and pulp mills. Because this wastewater contains fiber, it is sent to a 230-foot diameter primary clarifier for solids removal before entering the mix basin. A second line, which contains low pH water from the bleach plants runs directly to the mix basin. A third, relativity low volume line, runs from the sawmill area to the primary clarifier. The sawmill wastewater consists of non-contact cooling water, cleanup water and storm water. Wastewater enters the mix basin after primary clarification, is mixed with the lower pH bleach plant effluent and pumped to the aerated stabilization basin (ASB), or secondary treatment aeration pond, for biological treatment. Foul condensates from the evaporators and digesters are collected and routed directly to the aerated stabilization basin for treatment.

The 102-acre ASB has a residence time of approximately eight days. This secondary treatment system was constructed in the early 1970's. The objective of biological treatment is to remove organic compounds from wastewater through the growth of microorganisms, principally bacteria. Biological treatment systems maintain an inventory of biological solids within the process. It is desirable to produce floc-forming bacteria within the treatment process. Such bacteria adhere to each other due to the presence of a polysaccharide film that develops on the exterior cell wall. The floc-forming bacteria will settle out in the quiescent area of the basin prior to the discharge point. Biological solids that do not settle are discharged to the receiving water and measured as total suspended solids (TSS).

3.4.6 Stormwater Management

Most of the mill site drains into the process sewer or into the tail-race of the levy. Average rainfall is about 12" per year, so the volume of stormwater is minimal. One small area along a haul road on the eastern side of the Clearwater Mill property has the potential to drain into the adjoining wetland. Samples are taken at this location during rainfall events. Because of this drainage area, the site is required to have a SWPPP (Storm Water Pollution Prevention Plan) and is inspected regularly. Clearwater is considering modifying this to prevent roadside drainage into the wetland.

Certain peripheral areas of the site along the Clearwater River are graded and bermed such that any runoff cannot reach the river. Other peripheral areas that do have the potential for runoff into the river are kept free from industrial activity. These areas are also inspected regularly.

3.5 Outfall Description

3.5.1 Physical Description of Outfall 001

Effluent discharges through outfall 001 to the Snake River at its confluence with the Clearwater River, near the head of Lower Granite Pool. The discharge is at latitude 46° 25' 31" N, and longitude 117° 02' 15" W (approximately river mile 140). In addition to outfall 001, the facility

also discharges via a seep from the surface impoundments on the property to the Clearwater River.

The effluent is released through outfall 001 from a 400-foot diffuser pipe at a water depth of approximately 30 feet. The diffuser is in waters of the state of Idaho and upstream of the Idaho-Washington state line by 191 meters.

The diffuser consists of 79 individual ports spaced 5 feet apart rising from a common, buried 48-inch outfall pipe. Each riser pipe is angled 30 degrees from horizontal with the exit port about 1.5 feet above the river bottom. Each riser pipe is 3 inches in diameter. Only 72 of the ports are currently operating. Figure 3-2 shows the location of the diffuser at the confluence of the Snake and Clearwater Rivers. Figure 3-3 shows a more detailed view of the diffuser ports.

The discharge diffuser enhances mixing of discharged effluent in two major ways. First, by discharging effluent from numerous ports along the length of the diffuser, the effluent is spread out across a portion of the river rather than concentrated at a single point as occurs with a pipe outfall. In this way, the effluent encounters more of the river's flow and is more completely mixed into the full flow of the river. Second, there is considerable mixing created by hydrodynamic turbulence at each discharge port. The effluent discharges from each port at a velocity that is higher than the river velocity. This "jet" of effluent mixes with the ambient water, a process known as turbulent mixing, and also draws ambient water into the jet, a process known as entrainment (Jirka and Harleman, 1973). The effluent and entrained water completely mix within the jet, and thus the effluent is rapidly diluted.

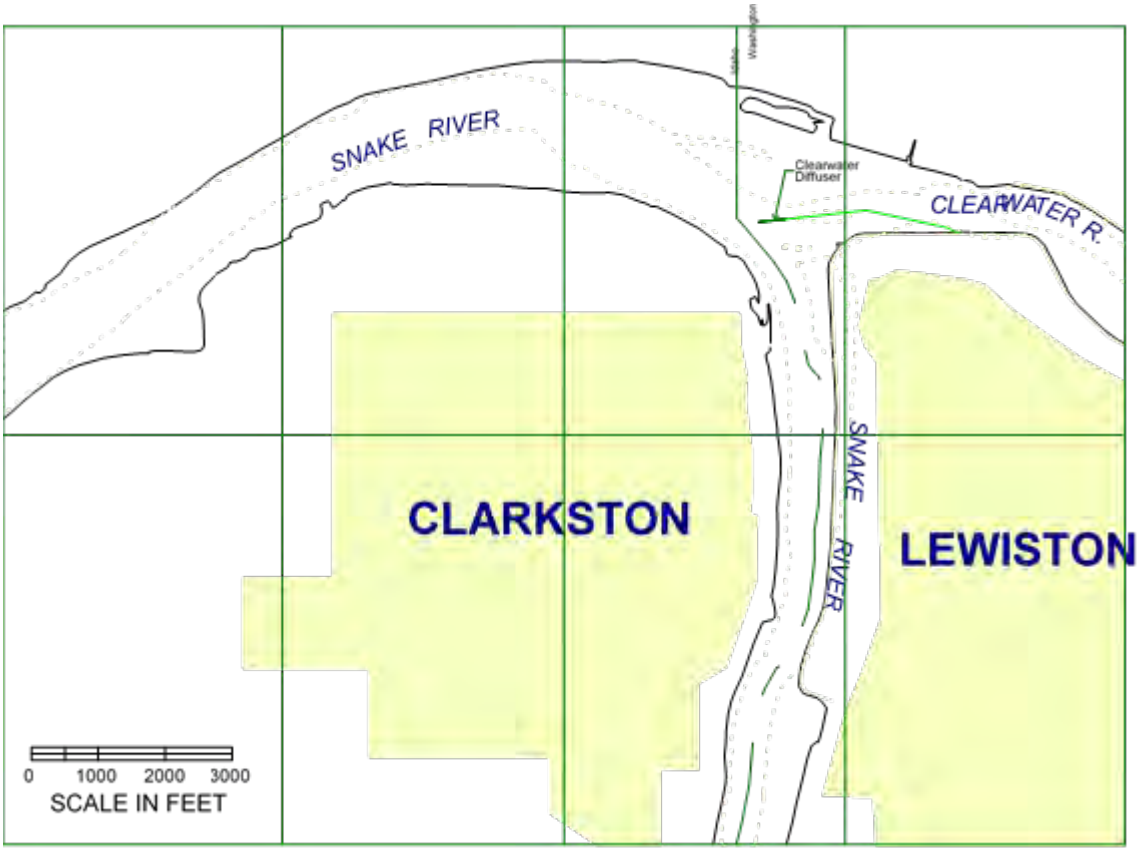

Figure 3-2: Location of the diffuser for Outfall 001 at the confluence of the Snake and Clearwater Rivers

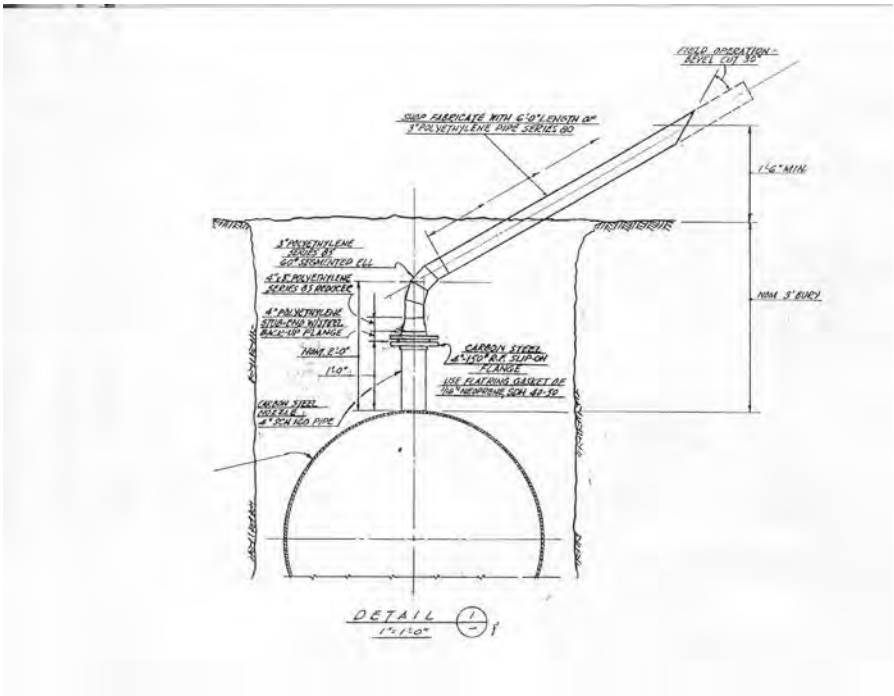

Figure 3-3: Detailed View of Outfall 001 Diffuser Port

The site of the mixing induced by the diffuser is at the discharge ports, near the bottom of the river. EPA used the CORMIX model to compute dilution factors appropriate for various types of water quality criteria (Nickel 2018).

3.5.2 Seepage to Groundwater from the Secondary Treatment Aeration Pond (ASB)

Prior to discharge through Outfall 001, the water is stored and treated in the 102-acre Aerated Stabilization Basin (ASB) located on the eastern side of the facility. Several investigations have been performed to assess seepage from the ASB. Seepage rates have been used to assess loading rates to the shallow groundwater and leakage to the Clearwater River for NPDES permit compliance. A summary of ASB construction, environmental conditions, and factors that influence seepage quantities is presented below along with a review of seepage estimates and water quality studies related to ASB seepage.

The ASB occupies the eastern portion of an old log pond that was drained just prior to ASB construction in 1973. Construction of the ASB approximately coincided with removal of a 1928 era dam and construction of the East Lewiston Levee and Lower Granite Dam by the U.S. Army Corps of Engineers (USACE). The left abutment of the 1928 dam was located in the north-central portion of the existing ASB dike. The ASB dikes were constructed from July to September 1973. Except where noted below, the dikes were not keyed into bedrock. Compacted fill was used to construct the dikes, which are composed of fine, sandy silt borrowed from the South Hill area. A construction summary (Dames & Moore, 1991) indicates that a 1-foot layer of "compacted silt and rock" was placed on the ASB bottom to meet the seepage design criteria of less than 0.25 inches/day. However, a record of how this criterion was established was not

included in the report. Based on drawings, the inside slope of the dike structure is 3 feet horizontal to 1 foot vertical. The design high water level was given by Jacob's Engineering (1973) as 775 feet elevation. Whereas the mean and nominal low water levels were given as 767.5 and 760 feet elevation, respectively and the average storage volume was 500 million gallons at the mean water level.

In 1974, a cut-off trench extending down to basalt bedrock was added along the central portion of the northern dike to control seepage. Seepage was observed along the eastern portion of the ASB and a pumping return system was installed in this area. Seepage was also observed along the Clearwater River at two locations and east of the ASB along the pre-existing inlet channel to the former log pond, which is located south of the greenhouses.

Seepage flow from the ASB is predominantly vertical downwards through the constructed bottom and the underlying finer grained native materials. Horizontal flow occurs in the coarse-grained alluvium and is generally to the north-northwest to east-northeast where groundwater discharges to the Clearwater River. Groundwater flow from the western ASB boundary flows to the northwest and ultimately discharges to the Corps Pond (groundwater sink). Seepage to the groundwater is unlikely along the southern section of the ASB where groundwater levels are higher. Because water levels in the ASB fluctuate and seasonal variations in groundwater occur, the seepage rate varies over time.

The estimated ASB seepage rate by Dames & Moore (1991) ranged between 0.3 and 9 mgd and flow from the aquifer to the river between 0.03 and 0.9 mgd. The report concluded a reasonable estimate for the rate of vertical flow out of the ASB is approximately 0.45 mgd. Based on the requirements of the 2005 NPDES permit, Clearwater Paper Corporation conducted groundwater monitoring during 2005 and 2006 (JUB Engineers 2006, 2007). The objectives of the 2005-2006 Groundwater Monitoring Program were as follows:

- Monitor the parameters specified in the NPDES permit Section I.G., Table 6 at designated sites on a quarterly basis.
- Prepare an annual report presenting the results of the monitoring program.

During the monitoring events, water quality samples were collected from eight sample sites as designated in the 2005 NPDES permit including:

- MW-1 (Southern levee of ASB Pond)
- MW-2 (Toe of North levee of ASB Pond)
- MW-2D (Toe of North levee of ASB Pond)
- MW-3 (Toe of Northwest levee of ASB Pond)
- MW-3D (Toe of Northwest levee of ASB Pond)
- MW-5 (Toe of CPD landfill)
- MW-10 (Near Toe of Northeast levee of ASB Pond)
- MW-12 (Near Building West of ASB Pond)

Results of the 2005 and 2006 Groundwater Monitoring Program are summarized in Table 3-1 (JUB Engineers, 2006; 2007). The 2nd, 3rd, and 4th quarters of 2005 and all quarters in 2006 were sampled for the required parameters. Results of the sampling indicated non-detects for all quarters in 2005 and 2006 for:

- 1,2,3,4,6,7,8-HpCDD
- 1,2,3,4,6,7,8-HpCDF
- 1,2,3,4,7,8-HxCDF

- Chloroforms (not measured in 2nd quarter 2005)
- OCDD
- Total HpCDD
- Total HpCDF
- Total HxCDD
- Total HxCDF

Therefore, the surface water of the Clearwater River is unaffected by seepage of groundwater with respect to these parameters. Other parameters monitored include field measured values of temperature, pH, conductivity, dissolved oxygen, oxygen reductive potential, and ferrous iron (Table 3-1). Conventional parameters monitored include dioxins/furans, BOD5, total suspended solids, ammonia, nitrate-nitrite, adsorbable oxygen halogens, total phosphorous, and chloroform.

Table 3-1: Results of Groundwater Samples from ASB Area

Table III-1. Results of Groundwater Samples from ASB Area

		Groundwater Monitoring Wells																										
		MW-1							MW-2							MW-2D							MW-3						
		2005 - Quarter			2006 - Quarter				2005 - Quarter			2006 - Quarter				2005 - Quarter			2006 - Quarter				2005 - Quarter			2006 - Quarter			
Parameter	Units	2nd	3rd	4th	1st	2nd	3rd	4th	2nd	3rd	4th	1st	2nd	3rd	4th	2nd	3rd	4th	1st	2nd	3rd	4th	2nd	3rd	4th	1st	2nd	3rd	4th
1,2,3,4,6,7,8-HpCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2,3,4,6,7,8-HpCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2,3,4,7,8-HxCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ammonia	mg/L	18.5	1.08	1.65	3.28	4.4	3.03	2.46	0.45	0.99	1.08	0.64	0.84	1.52	2.16	3.04	2.81	2.7	3.14	3.22	3.35	3.23	4.46	4.08	3.7	4.19	4.28	4.26	4.55
AOX	ug/L	1280	ND	ND	25	18	14	12	254	375	327	119	369	487	622	981	873	892	857	911	875	875	781	706	847	629	696	874	679
BOD	mg/L	12	ND	ND	ND	ND	ND	ND	ND	2.1	ND	1	ND	1	1	1.9	1.6	ND	2	ND	ND	2	6	3.8	5	15	ND	ND	3
Chloroforms	ug/L	NM	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Conductivity	µmhos	3586	NM	1382	1432	1293	1202	1383	1307	NM	1407	1174	1319	1614	1738	1940	NM	1862	1865	1208	1542	1853	2064	NM	2054	1889	1656	1778	2105
Dissolved Oxygen	mg/L	2.98	NM	4.47	4.79	3.45	2.14	3.66	2.78	NM	5.46	5.25	2.02	1.41	5.47	0.86	NM	4.66	6.84	2.69	0.88	5.58	2.45	NM	5.71	6.09	3.39	2.16	4.83
Iron, ferrous	mg/L	NM	NM	NM	4.4	NM	2.6	3	NM	NM	NM	0	0	3.4	4	NM	NM	NM	3	2.8	5	3	NM	NM	NM	4.2	4	4.6	3.2
Nitrate-Nitrite	mg/L	ND	0.026	0.029	ND	0.05	0.4	0.03	0.58	0.03	0.295	9.2	3.46	0.01	0.08	0.11	0.11	0.034	ND	0.11	0.06	0.07	0.22	0.23	0.12	ND	0.13	0.13	0.18
OCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ORP	mV	NM	NM	NM	-30.75	-56.75	-173.95	-102.95	NM	NM	NM	36.35	67.50	-113.65	-127.55	NM	NM	NM	-18.15	-46	-140.6	-92.15	NM	NM	NM	-32.55	-68.9	-171.65	-107.25
pH	S.U.	6.56	NM	7.13	7.21	6.62	6.85	6.97	6.71	6.93	7.48	7.77	6.84	7.09	6.93	5.95	6.58	6.8	7.01	5.94	6.67	6.66	6.28	6.45	6.67	7.04	6	6.59	6.53
Phosphorous, total	mg/L	2.94	0.2	0.09	0.22	0.25	0.22	0.24	0.15	0.68	0.19	0.21	0.21	0.53	1.39	0.84	0.82	0.61	1.2	0.71	0.76	0.6	0.91	0.9	0.79	0.88	0.93	0.85	0.93
Temperature	°C	21.84	NM	16.085	17.53	18.51	18.17	17.88	19.07	NM	13.48	9.81	17.4	21.94	19.35	17.91	NM	16.33	14.7	15.96	18.04	18.5	17.99	NM	15.25	16.12	16.86	16.8	16.66
Total HpCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total HpCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total HxCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total HxCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TSS	mg/L	236	154	ND	66	59	73	109	ND	20	ND	ND	ND	12	58	42	73	72	92	71	76	71	40	135	140	153	164	155	184

Table III-1. Continued

		Groundwater Monitoring Wells																										
		MW-3D							MW-5							MW-10							MW-12						
		2005 - Quarter			2006 - Quarter				2005 - Quarter			2006 - Quarter				2005 - Quarter			2006 - Quarter				2005 - Quarter			2006 - Quarter			
Parameter	Units	2nd	3rd	4th	1st	2nd	3rd	4th	2nd	3rd	4th	1st	2nd	3rd	4th	2nd	3rd	4th	1st	2nd	3rd	4th	2nd	3rd	4th	1st	2nd	3rd	4th
1,2,3,4,6,7,8-HpCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2,3,4,6,7,8-HpCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2,3,4,7,8-HxCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ammonia	mg/L	3.9	4.06	3.7	3.71	3.8	3.93	3.93	13.5	13.8	9.7	17.3	8.1	15.7	12.4	1.82	8.04	8	8.6	8.79	8.62	8.23	9.8	1.72	1.5	1.94	1.96	1.82	1.59
AOX	ug/L	665	641	584	487	585	800	744	612	677	721	569	526	764	882	811	779	820	807	834	391	748	112	96.5	83	64	69	50	37
BOD	mg/L	5	3.5	ND	2	ND	ND	4	256	471	326	416	157	265	1	1.4	12	ND	1	ND	ND	ND	ND	1.1	ND	3	2	1	620
Chloroforms	ug/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Conductivity	µmhos	1915	NM	1858	1833	1599	1684	2011	10130	NM	2165	2478	5195	11555	14476	1984	NM	1952	1918	1468	870	1911	495	NM	466	767	411	428	574
Dissolved Oxygen	mg/L	0.9	NM	6.09	6.52	3.91	1.08	5.94	1.32	NM	NM	1.52	4.34	0.66	2.61	2.49	NM	3.89	6.34	2.82	0.89	6.74	2.01	NM	4.53	1.29	1.11	2.14	2.09
Iron, ferrous	mg/L	NM	NM	NM	4	3.6	6.2	NM	NM	NM	NM	0	0.3	0	0	NM	NM	NM	3.6	3.1	5	3.5	NM	NM	NM	1	1.2	1	1
Nitrate-Nitrite	mg/L	0.16	0.22	0.12	ND	0.13	0.12	0.15	0.25	ND	0.53	0.05	0.08	0.1	0.01	0.1	0.06	0.042	ND	0.02	0.05	0.04	0.09	0.019	0.009	0.05	0.01	0.05	0.01
OCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ORP	mV	NM	NM	NM	-27.75	-70.5	-156.65	-110.95	NM	NM	NM	-217.35	-378.45	-436	-383.85	NM	NM	NM	-17.65	-73.05	-90.05	-100.55	NM	NM	NM	-195.4	-137.6	-107.5	-144.85
pH	S.U.	6.41	6.49	6.73	6.99	6.08	6.75	6.54	9.44	10.04	9.74	8.52	9.57	9.84	9.86	6.88	6.77	7.04	7.02	6.64	6.54	6.81	7.3	7.59	7.52	9.38	7.61	7.45	7.72
Phosphorous, total	mg/L	0.88	0.96	0.86	0.86	0.84	0.86	0.84	5.7	9.7	6.5	4.9	5.9	8.8	22.4	0.53	0.45	0.4	0.45	0.46	0.45	0.49	0.46	0.55	0.56	0.58	0.56	0.48	0.49
Temperature	°C	18.26	NM	14.745	16.11	17.29	16.9	16.78	21.54	NM	17.65	19.32	19.01	21.92	19.54	22.42	NM	20.13	19.92	21.49	23.57	21.64	15.66	NM	13.58	15.47	17.46	14.78	14.58
Total HpCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total HpCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total HxCDD	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total HxCDF	pg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
TSS	mg/L	71	82	117	147	145	149	164	7	15	67	51	9	14	18	34	40	43	40	41	43	47	ND	ND	ND	ND	ND	ND	ND

3.5.3 Wastewater Characterization

Clearwater monitors certain effluent parameters to comply with monitoring requirements specified in the permit under which the Mill currently discharges. Table 3-2 summarizes the range and average concentrations of parameters monitored in effluent from 2005 – 2016 including the 2007 High Volume sampling (Anchor, 2008) required under the 2005 permit and the 2009 permit application data.

Table 3-2: Summary of Outfall 001 Discharge Composition (2005 – 2016 DMR; 2007 High Volume Sampling; 2009 Permit Application)

Parameter	Units	Range of Concentrations [1]	Average Concentration [2,3]
Flow	mgd	29.8 – 37.7	29.8
pH	s.u.	6.1- 8.5	ND
Color	c.u.	750 - 750	750
Temperature (winter: Oct-Jun)	° C	23.9 – 28.7	23.9
Temperature (summer: Jul-Sep)	°C	26.8 – 29.0	26.8
Biochemical Oxygen Demand (BOD)	mg/L	25.0 – 40.0	25.0
Chemical Oxygen Demand (COD)	mg/L	428.6 - 531	428.6
Total Organic Carbon (TOC)	mg/L	Note 2	ND
Total Suspended Solids (TSS)	mg/L	38.2 - 89.4	38.2
Fecal Coliform Bacteria	MPN/100 mL	30 - 30	30
1,2,3,4,6,7,8,9-OCDD	pg/L	**0.0461 - 0.0461**	ND
1,2,3,4,6,7,8,9-OCDF	pg/L	**0.0101 - 0.0101**	ND
1,2,3,4,6,7,8-HpCDD	pg/L	**0.0153 - 0.0153**	ND
1,2,3,4,6,7,8-HpCDF	pg/L	**0.00625 - 0.00625**	ND
1,2,3,4,7,8,9-HpCDF	pg/L	**0.00242 - 0.00242**	ND
1,2,3,4,7,8-HxCDD	pg/L	**0.0039 - 0.0039**	ND
1,2,3,4,7,8-HxCDF	pg/L	**0.00129 - 0.00129**	ND
1,2,3,6,7,8-HxCDD	pg/L	**0.00625 - 0.00625**	ND
1,2,3,6,7,8-HxCDF	pg/L	**0.00127 - 0.00127**	ND
1,2,3,7,8,9-HxCDD	pg/L	**0.00625 - 0.00625**	ND
1,2,3,7,8,9-HxCDF	pg/L	**0.00158 - 0.00158**	ND
1,2,3,7,8-PeCDD	pg/L	**0.00294 - 0.00294**	ND
1,2,3,7,8-PeCDF	pg/L	**0.00625 - 0.00625**	ND
12-Chlorodehydroabietic Acid	µg/L	**20 - 20**	ND
14-Chlorodehydroabietic Acid	µg/L	**20 - 20**	ND
2,3,4,6,7,8-HxCDF	pg/L	**0.00138 - 0.00138**	ND
2,3,4,6-Tetrachlorophenol	µg/L	**0.005 - 0.005**	ND
2,3,4,7,8-PeCDF	pg/L	**0.00625 - 0.00625**	ND
2,3,7,8-TCDD	pg/L	**0.0037** - 675	2.61
2,3,7,8-TCDF	pg/L	0.0161 15.1	2.67

Parameter	Units	Range of Concentrations [1]	Average Concentration [2,3]
2,4,5-Trichlorophenol	µg/L	**0.005** - 0.32	0.32
2,4,6-Trichlorophenol	µg/L	**0.15 - 0.15**	ND
3,4,5-Trichlorocatechol	µg/L	**0.01 - 0.01**	ND
3,4,5-Trichloroguaiacol	µg/L	**0.016 - 0.016**	ND
3,4,6-Trichlorocatechol	µg/L	**0.01 - 0.01**	ND
3,4,6-Trichloroguaiacol	µg/L	**0.093 - 0.093**	ND
4,5,6-Trichloroguaiacol	µg/L	**0.093 - 0.093**	ND
9,10-Dichlorostearic Acid	µg/L	**20 - 20**	ND
Abietic Acid	µg/L	**20 - 20**	ND
Aluminum, Total	µg/L	368 - 368	368
Ammonia as Nitrogen	mg/L	2.15 - 2.15	1.12
Antimony, Total	µg/L	0.1 -0.1	0.1
Arsenic, Total	µg/L	1.6 - 1.6	1.6
Barium, Total	µg/L	263 -263	263
Bromide, Total	mg/L	1.17 - 1.17	1.17
Boron, Total	µg/L	26 -26	26
Chloroform	µg/L	1.0 - 1.0	1.0
Chromium, Total	µg/L	11.8 -11.8	11.8
Cobalt, Total	µg/L	**1**-1	1
Copper, Total	µg/L	2.5 - 2.5	2.5
Dehydroabietic Acid	µg/L	**13 - 13**	ND
Dichlorodehydroabietic Acid	µg/L	**20 - 20**	ND
Halides, Adsorbable Organic (AOX)	µg/L	3.3 - 7200	2258
Iron, Total	µg/L	342 -342	342
Isopimaric Acid	µg/L	**20 - 20**	ND
Lead, Total	µg/L	0.62 - 0.62	0.62
Linoleic Acid	µg/L	**59 - 59**	59
Magnesium	µg/L	4290 - 4290	4290
Manganese	µg/L	296 - 296	296
Mercury, Total	µg/L	**0.1 - 0.1**	ND
Molybdenum, Total	µg/L	**3.1 -3.1**	3.1
Nickel, Total	µg/L	**3.6 -3.6**	3.6
Nitrate-Nitrite	mg/L	**0.018 -0.060**	0.018
Oleic Acid/Linolenic Acid	µg/L	**23 - 23**	ND
Pentachlorophenol (PCP)	µg/L	**0.01 - 0.01**	ND
Phenols, Total	mg/L	**0.097 -0.097**	0.097
Phosphorus	mg/L	0.649 - 0.649	0.649
Pimaric Acid	µg/L	**20 - 20**	ND
Selenium, Total	µg/L	**50 - 50**	ND

Parameter	Units	Range of Concentrations [1]	Average Concentration [2,3]
Sulfate	mg/L	208 - 208	208
Surfactants	mg/L	Note 2	ND
Tetrachlorocatechol	µg/L	**0.01 - 0.01**	ND
Tetrachloroguaiacol	µg/L	**0.11 - 0.11**	ND
Thallium, Total	µg/L	**0.19 -0.19**	0.19
Titanium	µg/L	5.3 - 5.3	5.3
Total Organic Nitrogen (TON)	mg/L	4.6 - 4.6	4.6
Trichlorosyringol	µg/L	**0.005 - 0.005**	ND
Whole Effluent Toxicity (WET)	TU_c	1 - 10	-
Zinc, Total	µg/L	14.4 - 14.4	14.4

Notes:
1. Numbers in bold are equal to Detection Limits.
2. ND: Not Detected.
3. In calculating the average concentration, the detection limit is used to represent the concentration in samples in which the parameter is not detected, to account for the uncertainty that the actual concentration in that sample could range from zero to the detection limit.

3.6 Permit Limits

NPDES permits include both technology-based (ELGs) 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 limitations 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 designated uses of the water bodies 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."

Section 301(b)(2) of the Clean Water Act requires technology-based controls on effluents. This section of the Clean Water Act requires that, by March 31, 1989, all permits contain effluent limitations which: (1) control toxic pollutants and nonconventional pollutants using the "best available technology economically achievable" (BAT), and (2) represent "best conventional pollutant control technology" (BCT) for conventional pollutants (i.e., BOD_5, TSS, and pH). In no case may BCT or BAT be less stringent than "best practicable control technology currently available" (BPT), which is a minimum level of control required by section 301(b)(1)(A) the Clean Water Act.

On April 15, 1998, EPA published revised effluent guidelines for the pulp and paper industry in the Federal Register (98 FR 18503). These guidelines, known as the "Cluster Rule," replace the guidelines that were used to calculate the technology-based limitations in Potlatch's 1992 permit. They can be found in the Code of Federal Regulations (CFR) at 40 CFR Part 430. The Cluster Rule established revised subcategories for the pulp and paper industry. Due to the Cluster Rule,

Potlatch is regulated under Subpart B (Bleached Papergrade Kraft and Soda) and Subpart L (Tissue, Filter, Non-Woven, and Paperboard from Purchased Pulp).

On January 26, 1981, EPA published final effluent guidelines for the Timber Products Processing Point Source Category (46 FR 8285). These guidelines provide technology-based effluent limitations that apply to the wood products operations at the mill. The guidelines can be found at 40 CFR 129. Within these guidelines, Subpart A (Barking), Subpart K (Sawmills and Planing Mills), and Subpart L (Finishing) would apply to the sawmill if it were discharging directly to waters of the United States.

For this industrial category, the ELGs are based on the following model process and treatment technologies:

- Conventional pulping followed by complete substitution of chlorine dioxide for elemental chlorine
- Adequate chip thickness control
- Closed brown stock pulp screen room operation (i.e., screening filtrates are returned to the recovery cycle)
- Effective brown stock washing (i.e., washing that achieves a soda loss of less than or equal to 10 kg Na_2SO_4 per air dried metric ton (ADMT) of pulp (equivalent to 99% recovery of pulping chemicals from the pulp); use of TCDD- and TCDF-precursor-free defoamers (water-based defoamers or defoamers made with precursor-free oils)
- Elimination of hypochlorite (i.e., replacing hypochlorite with equivalent bleaching power, such as adding peroxide and/or oxygen to the first extraction stage and/or additional chlorine dioxide in final brightening stages)
- Use of strategies to minimize kappa factor and TCDD- and TCDF-precursors in brown stock pulp
- High-sheer mixing to ensure adequate mixing of pulp and bleaching chemicals; oxygen and peroxide enhanced extraction, which allows mills to eliminate hypochlorite and/or use a lower kappa factor in the first bleaching stage
- Efficient biological wastewater treatment, removing 90% or more of influent five-day biochemical oxygen demand (BOD_5).

The draft NPDES (2019) permit for the Clearwater Mill includes technology-based effluent limits for the following parameters:

- Biochemical oxygen demand (BOD) during December through May
- Total suspended solids (TSS)
- Upper-end pH range
- Adsorbable organic halides (AOX).

In addition to the technology-based effluent limits, the draft permit specifies technology-based limitations for the following parameters in internal fiber lines (i.e., effluent from bleaching lines) that are the only source of these chlorinated organic pollutants:

- Chloroform
- 2,3,7,8-TCDF
- 2,3,7,8-TCDD
- Trichlorosyringol
- 3,4,5-trichlorocatechol

- 3,4,6-trichlorocatechol
- 3,4,5-trichloroguaiacol
- 3,4,6-trichloroguaiacol
- 4,5,6-trichloroguaiacol
- 2,4,5-trichlorophenol
- 2,4,6-trichlorophenol
- Tetrachlorocatechol
- Tetrachloroguaiacol
- 2,3,4,6-tetrachlorophenol
- Pentachlorophenol

Section 304(h) of the Clean Water Act (CWA) requires the EPA Administrator to promulgate guidelines establishing test procedures for the analysis of pollutants. The EPA's approval of analytical methods is authorized under section 304(h) of the CWA, as well as the general rulemaking authority in section 501(a) of the Act. The EPA uses these test procedures to support the development of effluent limitations guidelines, to establish compliance with NPDES permits, for implementation of pretreatment standards, and for section 401 certifications. The section 304(h) test procedures (analytical methods) are specified in part 136 of title 40 of the Code of Federal Regulations (40 CFR Part 136). All methods specified in the permit are published in 40 CFR Part 136. Such methods have been validated by the EPA, published in the federal register for public comment, approved by the EPA and incorporated, by rulemaking, into the Code of Federal Regulations.

For many of the above listed pollutants, EPA has established ELGs that are expressed as less than the Minimum Level (<ML) of prescribed methods approved by EPA (see footnotes in Table 3-4). The Clearwater Mill is required to demonstrate compliance with those limitations and standards using EPA's Methods and ML values specified in the regulations.

The ML specified for each method is the lowest level at which laboratories calibrate their equipment. To do this, laboratories use standards (i.e., samples at several known concentrations). Calibration is necessary because laboratory equipment does not measure concentration directly; but generates signals or responses from analytical instruments that must be converted to concentration values. The calibration process establishes a relationship between the signals and the known concentration values of the standards. This relationship is then used to convert signals from the instruments for samples with unknown concentrations. In the calibration process, one of the standards will have a concentration value at the ML for the pollutant analyzed. Because the ML is the lowest level for which laboratories calibrate their equipment, measurements below the ML are to be reported as <ML.

The minimum level is defined in the glossaries to EPA methods 608.3 and 625.1 as follows:

> "The term 'minimum level' refers to either the sample concentration equivalent to the lowest calibration point in a method or a multiple of the method detection limit (MDL), whichever is higher. Minimum levels may be obtained in several ways: They may be published in a method; they may be based on the lowest acceptable calibration point used by a laboratory; or they may be calculated by multiplying the MDL in a method, or the MDL determined by a laboratory, by a factor of 3. For the purposes of NPDES compliance monitoring, EPA considers the following terms to be synonymous: 'quantitation limit,' 'reporting limit,' and 'minimum level.'"

The MDL concept origin is an article published in the peer-reviewed scientific literature in 1981 (Environmental Science and Technology 15 1426-1435). The MDL procedure has been used in the EPA's various environmental programs since it was published at 40 CFR Part 136, Appendix B in 1984. The current definition of the MDL is as follows:

> "The method detection limit (MDL) is defined as the minimum measured concentration of a substance that can be reported with 99% confidence that the measured concentration is distinguishable from method blank results."

Application of the MDL procedure to particular methods has been subject to peer review and public comment with every MDL that the EPA publishes in nearly every chemical-specific method proposed in the Federal Register since 1984. The MDL procedure is accepted and used by nearly all organizations making environmental measurements. No other detection or quantitation limit procedure or concept has achieved this level of acceptance and use.

Often, laboratories report values less than ML as "not detected" or "<ML." In some cases, however, the laboratories quantify these values. For example, even though the ML for an approved analytical method is 10 ppq for a particular pollutant, a laboratory might report a measurement of 4 ppq. These are two situations where a laboratory might report such a value. In the first situation, the laboratory could have used the method specified but referred to the measurement as "detected" although it was <ML. The second situation could occur in the future as analytical methods become more sensitive than the specified analytical method, allowing laboratories to reliably measure values less than today's MLs. Such measurements would demonstrate compliance with the <ML limitations codified in the ELGs, because these measurements are less than the ML defined in Part 430 for Subparts B and E. The Mill cannot demonstrate compliance using an analytical method with an ML above that of the designated method.

In addition to the ELGs, EPA evaluated the discharge to determine compliance with Section 301(b)(1)(C) of the Clean Water Act. To determine whether water quality based-limits are needed, EPA follows guidance in its *Technical Support Document for Water Quality-based Toxics Control* (TSD; USEPA 1991). EPA evaluated the Outfall 001 discharge to determine if "reasonable potential" exists. Effluent limits were developed for those pollutants where there was "reasonable potential" to exceed the criteria established to protect the designated uses of the receiving water. Parameters for which water quality-based effluent limitations are specified in the draft permit are:

- BOD during June through November
- Temperature
- Low-end pH range
- 2,3,7,8-TCDD

Effluent limits are not needed for those parameters that did not exhibit "reasonable potential." Monitoring was included in the draft permit for those parameters where there was not enough data to determine the need for effluent limits. A description of the reasonable potential evaluation for the draft permit is included in Appendix B. The BE evaluates the potential for chemical and physical characteristics of the effluent to affect listed species. The parameters evaluated in the BE were from the following categories:

- Parameters with effluent limitations in the 2019 draft permit

- Parameters with no effluent limitation, but monitoring is required in the 2019 draft permit

In developing WQBELs, EPA converts the criteria into limitations using the procedures in the TSD (USEPA, 1991). Factors that influence the development of effluent limits include: effluent flow, receiving water critical low flows, effluent variability, and water quality upstream of the discharge. Reasonable worst-case estimates of each of these factors were used to develop the effluent limits to ensure that they are protective of the aquatic organisms using the water quality criteria under critical conditions as a measure of the protectiveness. Each of these factors is discussed in detail in Appendix B.

The receiving water body's ability to dilute effluent is also factored into the development of effluent limitations (40 CFR 122.44(d)(1)(ii)). Available dilution increases with distance downstream of the discharge point. The availability of dilution is termed a mixing zone. Under the Idaho water quality standards, mixing zones may be authorized for discharges to meet water quality standards. Mixing zones are areas or volumes of receiving water where wastewater mixes with the receiving water and where water quality standards may be exceeded. Additional discussion of the mixing zones is provided in Section VII.A. Mixing zones were used to calculate the proposed effluent limits for the following parameters:

- Temperature
- Pentachlorophenol
- Chloroform
- Chromium VI
- Lead
- Zinc
- TSS
- Low-end pH
- Dissolved oxygen

The effluent limits and internal fiber line limits are expressed in terms of concentration (e.g., µg/l) or in terms of mass (e.g., lb/day) to ensure that the discharge to the receiving water complies with water quality standards and effluent guidelines. Mass-based limits are particularly important for control of bioconcentratable pollutants because concentration-based limits will not adequately control discharges of these pollutants if the effluent concentrations are below detection levels. However, mass-based limits alone may not assure attainment of water quality standards in waters with low dilution (i.e., less than 100-fold dilution). Therefore, some limits are expressed in both mass and concentration.

The federal regulations at 40 CFR section 122.45(d) requires effluent limitations for continuous discharges to be expressed as maximum daily and average monthly limitations for all dischargers other than publicly owned treatment works. The NPDES regulations at 40 CFR section 122.2 defines the maximum daily discharge as the highest allowable daily discharge and the average monthly discharge limitation as the highest allowable average of daily discharges over a calendar month, calculated as the sum of all daily discharges measured during a calendar month divided by the number of daily discharges measured during that month. The regulation also defines daily discharge as the discharge of a pollutant measured during a calendar day or any 24-hour period that reasonably represents the calendar day for the purposes of sampling. For pollutants with limitations expressed in units of mass (e.g., lb/day), the daily discharge is calculated as the total mass of the pollutant discharged over the day. For pollutants expressed in other units of

measurement (e.g., mg/L), the daily discharge is calculated as the average measurement of the pollutant over the day.

A comparison of the current (2005 permit with 2010 modification) and draft 2019 permit effluent concentration limits for outfall 001 are provided in Table 3-3. The internal limitations are provided in Table 3-4.

Table 3-3: Comparison of Current (2005) and Draft (2019) Effluent Limitations for Outfall 001

Parameter	Units	2005 Final Permit with 2010 Modification		2019 draft permit	
		Maximum Daily	Average Monthly [1]	Maximum Daily	Average Monthly[1]
AOX [2]	lb/day	3,950	2,590	2,979	1,951
BOD_5 [2] (December – May)	mg/L	---	---	---	---
	lb/day	55,100	28,800	50,578	26,431
BOD_5 [2] (June – November)	mg/L	---	---	---	---
	lb/day	15,000	8,400	15,000	8,400
pH	s.u.	within the range of 5.5 to 9.0 [6]		within the range of 5.7 to 8.5 [6]	
Pentachlorophenol (July – September)	µg/L	—	—	0.15	0.10
	lb/day			0.038	0.026
Pentachlorophenol (October – June)	µg/L			0.23	0.16
	lb/day			0.072	0.050
TSS	lb/day	94,400	50,600	88,030	47,081
2,3,7,8-TCDD (year-round)	mg/day	0.22 [3, 4]	0.15 [3, 4]	—	—
2,3,7,8-TCDD (July – September)	pg/L	—	—	0.94	0.65
	mg/day			0.113	0.077
2,3,7,8-TCDD (October – June)	pg/L			1.5	1.0
	mg/day			0.177	0.121
Temperature (October – June)	°C	33	—	33	—
Temperature (July)		32	—	32	—
Temperature (August – September)		31	—	31	—

Parameter	Units	2005 Final Permit with 2010 Modification		2019 draft permit	
		Maximum Daily	Average Monthly (1)	Maximum Daily	Average Monthly(1)

Notes:

1. The average monthly limit is determined as the arithmetic average of all the samples collected within the month. For the purpose of calculating monthly average, the permittee must use all values greater than the method detection level; however, zeros may be used for values less than the method detection level.
2. To calculate the maximum daily loading in lb/day, multiply the concentration (mg/L) by a conversion factor of 8.34 lb·L/mg·gal and the daily average effluent flow rate (mgd). For BOD_5 and AOX, 3 mgd must be added to the daily average effluent flow to account for pond seepage.
3. This effluent limit is not quantifiable using EPA approved analytical methods. The permittee will be in compliance with the effluent limit provided the measured concentration is at or below the compliance level of 10 pg/L and the calculated quantity is < 0.72 mg/day using EPA Method 1613.
4. To calculate the maximum daily loading in mg/day, multiply the measured concentration (pg/L) by a conversion factor of 0.003786 mg·L/pg·gal·10^6 and the daily effluent flow rate (in mgd or 10^6 gallons per day) plus 3 mgd for pond seepage. If the measured concentration is not detectable, then use one half the detection level as the concentration in the calculation and report as "< {calculated value}" on the DMR.
5. Monitoring is required only during the first, second, and fourth year of the permit.
6. Per 40 CFR 401.17, the permittee must maintain the pH of the effluent within the range specified, except excursions from the range are permitted subject to the following limitations: The total time during which the pH values are outside the required range of pH values shall not exceed 7 hours and 26 minutes in any calendar month; and no individual excursion from the range of pH values shall exceed 60 minutes.

Table 3-4: Internal Fiber Line Limitations

Parameter	Units	2005 Permit		2019 draft permit	
		Maximum Daily	Average Monthly	Maximum Daily	Average Monthly
2,3,7,8-TCDD	pg/L	<10 [1]	---	<10 [1]	---
2,3,7,8-TCDF	pg/L	31.9	---	31.9	---
Chloroform (total)	lb/day	28.8	17.2	N/A	N/A
Chloroform: chip fiber line	lb/day	N/A	N/A	15.0	9.0
Chloroform: sawdust fiber line	lb/day	N/A	N/A	6.7	4.0
Trichlorosyringol	µg/L	<2.5 [2]	---	<2.5 [2]	---
3,4,5-trichlorocatechol	µg/L	<5.0 [2]	---	<5.0 [2]	---
3,4,6-trichlorocatechol	µg/L	<5.0 [2]	---	<5.0 [2]	---
3,4,5-trichloroguaiacol	µg/L	<2.5 [2]	---	<2.5 [2]	---
3,4,6-trichloroguaiacol	µg/L	<2.5 [2]	---	<2.5 [2]	---
4,5,6-trichloroguaiacol	µg/L	<2.5 [2]	---	<2.5 [2]	---
2,4,5-trichlorophenol	µg/L	<2.5 [2]	---	<2.5 [2]	---

Parameter	Units	2005 Permit		2019 draft permit	
		Maximum Daily	Average Monthly	Maximum Daily	Average Monthly
2,4,6-trichlorophenol	µg/L	<2.5 [2]	---	<2.5 [2]	---
Tetrachlorocatechol	µg/L	<5.0 [2]	---	<5.0 [2]	---
Tetrachloroguaiacol	µg/L	<5.0 [2]	---	<5.0 [2]	---
2,3,4,6-tetrachlorophenol	µg/L	<5.0 [2]	---	<5.0 [2]	---
Pentachlorophenol	µg/L	<5.0 [2]	---	<5.0 [2]	---
Flow	mgd	---	---	---	---

Notes:

1. The permittee must use EPA Method 1613 for the analysis of this parameter. The permittee must achieve a minimum level equal to or less than this concentration. For purposes of reporting on the DMR, if a value is less than the minimum level but greater than the method detection level, the permittee must report the actual value. If a value is less than the method detection level, the permittee must report "less than {numerical method detection limit}" on the DMR.
2. The permittee must use EPA Method 1653 for the analysis of this parameter. The permittee must achieve a minimum level equal to or less than this concentration. For purposes of reporting on the DMR, if a value is less than the minimum level but greater than the method detection level, the permittee must report the actual value. If a value is less than the method detection level, the permittee must report "less than {numerical method detection limit}" on the DMR.

3.7 References

Anchor Environmental, LLC. 2008. High Volume Quarterly Study Sampling and Analysis Report – Potlatch NPDES Permit Renewal Compliance Monitoring. January 2008.

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4 Description of Action Area

This part describes the action area for the permit action proposed by EPA. The discussion includes a definition of the action area, a description of the terrain and climate in the action area, and a description of the receiving water condition in the action area.

4.1 Definition of Action Area

The ESA implementing regulations define action area as all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR section 402.02). Indirect effects are defined as those effects that are caused by or will result from the proposed action and are later in time, but are still reasonably certain to occur (50 CFR section 402.02). Neither the ESA regulations nor guidance (USFWS and NMFS, 1998) provides a definition of direct effects; however, correspondence from USFWS (2000) defines "direct effects" under the ESA consultation process as direct or immediate effects of the proposed action on the species or its habitat.

Since the proposed action is the re-issuance of the NPDES permit, the direct effects are those that would cause toxicity to a listed species from individual and combined pollutant concentrations within the hydrodynamic mixing zone. The presence of parameters regulated by the draft permit could potentially be present at a concentration that could cause toxicity to a listed species at different distances downstream from the discharge, depending upon the effluent limit, available dilution from the river, and the physical and chemical characteristics of the parameter. Section VII.E and Appendix D provide the analysis of the potential direct effects from the action that define the action area (the area within which each individual parameter may have an effect) and Section VII.G discusses the potential combined direct effects from the action that define the action area due to combined effects of parameters within the whole effluent.

The area where direct effects may occur commences at the point of discharge. Therefore, on the Snake River the action area is bounded on the upper end at Outfall 001 (i.e., Snake River Mile 139). However, pond seepage from the ASB occurs at the Clearwater Mill, therefore, the action area commences at the upper end of the ASB on the Clearwater River (i.e., Clearwater River Mile 3). The action area downstream for a specific parameter depends on the physical and chemical properties that cause it to degrade or dilute as it travels downstream. A parameter that is highly volatile or readily biodegradable in a river may be present over a relatively small downstream area at a concentration that could potentially cause toxicity, because several mechanisms effectively remove the parameter from the river. On the other hand, a parameter that is persistent in the environment and is not readily biodegraded in a river system might be present over a longer downstream distance at a concentration that could potentially cause toxicity, because removal mechanisms are less effective in eliminating this parameter from the river.

Indirect effects for the proposed action are those that would cause an effect to a listed species or habitat from individual and/or combined pollutant concentrations within the waterbody at a later time. These effects would result from delayed exposure (e.g., uptake of deposited effluent constituents from sediment resuspension, consumption of prey species, and habitat modification (e.g., deposited effluent constituents on the riverbed, decrease in photosynthesis). Any of these indirect effects could occur as long as there is influence on the Snake River water column and

sediment quality from the Clearwater Mill discharges. Therefore, the indirect action area extends to the point downstream where an indirect adverse effect could occur (e.g., where the concentration of a parameter in the sediment resulting from the effluent discharge is high enough to cause an adverse effect to threatened and endangered fish species).

From the analysis conducted in this BE, the action area occurs from River Mile 3 of the Clearwater River to the mouth of the Snake River at its confluence with the Columbia River. A map showing the action area is provided in Figure 4-1.

4.2 Terrain and Climate

The Snake River flows through terrain that is warmer and drier on an annual basis than the upper Columbia Basin or other drainages to the north. Geologically, the land forms are subject to high amounts of erosion. Collectively, the environmental factors of the Snake River Basin result in a river that is high in alkalinity, pH, and turbidity. The Upper Snake Basin is characterized by mountainous terrain and flat to gently sloping plains, changing to semidesert in the plateau lands. The numerous mountain ranges drain to the Snake River and the Snake River Plain Aquifer, one of the largest aquifers in the United States. This is one of the most productive agricultural areas in the country, producing sugar beets, corn, potatoes, and dry beans. Three large water-supply reservoirs dominate the edge of the Snake River Plain: the American Falls Reservoir on the upper Snake River; the Palisades Reservoir on the Snake River at the Idaho/Wyoming border; and the Blackfoot Reservoir in the upper Blackfoot River. Terrain along the lower Snake River is steep, with 2,000 feet breaklands of basalt cliffs and talus slopes near Lewiston, gradually diminishing in height downstream to 100-200 feet at the confluence with the Columbia River. The surrounding rolling plateau country to the north and west is predominantly devoted to dryland wheat production. Many of the soils of the region are naturally light and highly susceptible to erosion. Soil erosion is a problem in the region; consequently, the Snake River has a noticeable sediment load during the spring runoff season.

The climate is semi-arid with precipitation mostly in the winter and spring. Annual precipitation along the Snake River averages 13 to 18 inches. In the river canyons, strong winds are common, generally blowing in a westerly direction. Yearly average wind speeds range from four to six miles per hour. The summers are hot, with temperatures often in the 90s and occasionally over 100°F (32-38°C). It is not uncommon to have periods of a month or more in the summer without precipitation. Climate change is also forecasted to bring significantly less rainfall to the region, and moderate and extreme droughts could occur in the future.

4.3 Receiving Water

4.3.1 Cumulative Effects

Cumulative effects are defined at 50 CFR section 402.02 as those effects of future State or private activities, not involving federal activities, which are reasonably certain to occur within the action area of the Federal action subject to consultation. Since the action area is within the confines of the waterbody of the lower Snake River and lower Clearwater River, cumulative effects would be those that affect the waterbody.

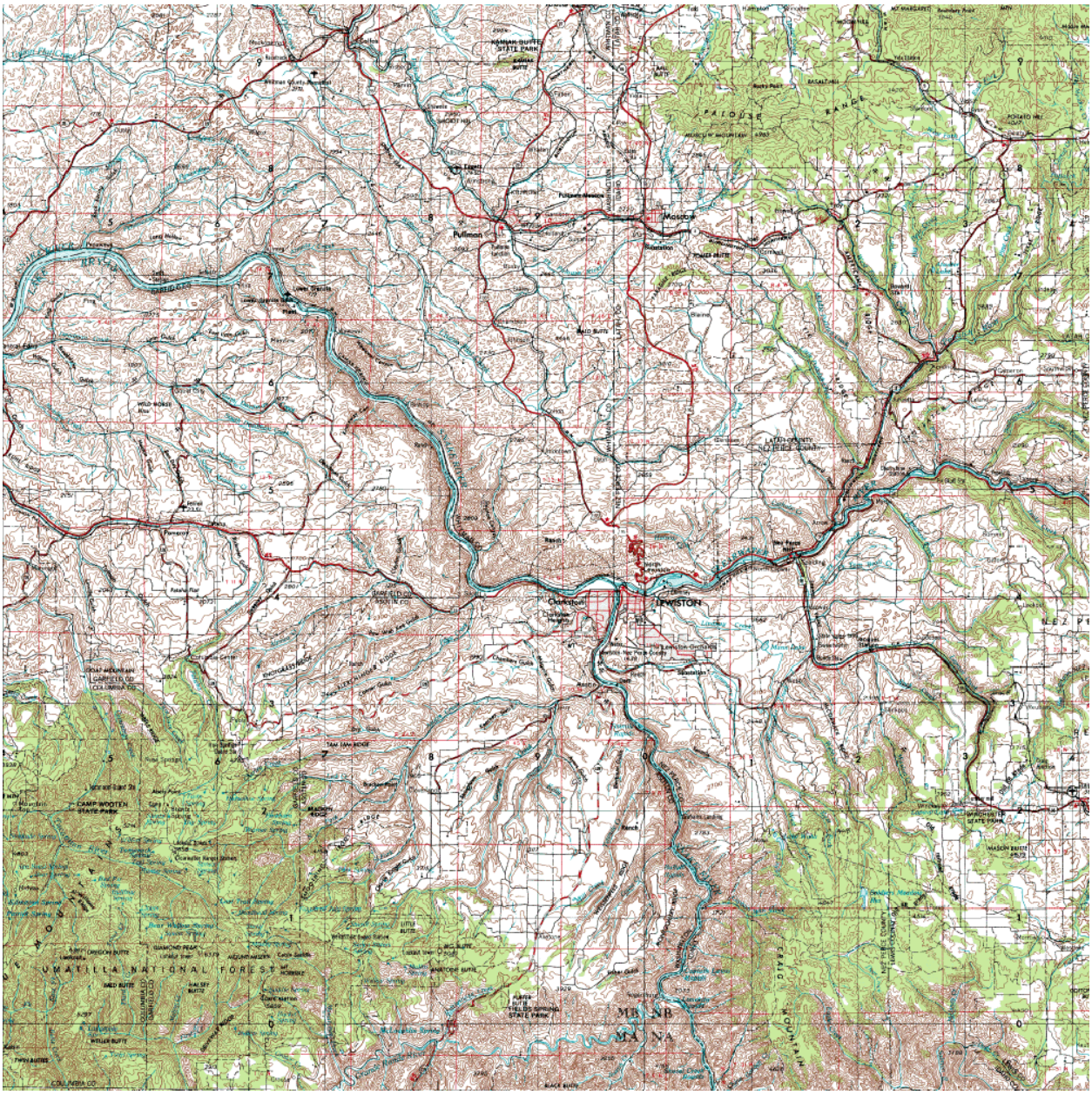

Figure 4-1: Map indicating action area of Lower Snake River below confluence with Clearwater River.

Future anticipated non-Federal actions likely to continue having adverse effects on the endangered and threatened species that may occur in or near surface waters in the action area include:

- Air deposition from the Clearwater Mill stacks
- Air deposition (global)
- Urban stormwater runoff (pesticides, herbicides, hydrocarbons, metals, temperature)
- Recreational boating (hydrocarbons)
- Recreational fishing
- Recreational swimming (bacteria)

- Agricultural practices – irrigation (flow diversion) and irrigation returns (pesticides, herbicides, nutrients, sedimentation and temperature)
- Timber harvest (sedimentation and temperature)
- Grazing (nutrients, sedimentation, bacteria, and temperature)
- Dam operations (temperature, flow augmentation, dissolved gas)
- Clearwater Mill water rights (flow diversion) – two rights for removing water from the Clearwater River. Total of 75 cfs managed through Idaho Water Resources.
- Urban development (sedimentation, hydrocarbons, copper, and temperature)
- Road building (sedimentation, hydrocarbons, and temperature)
- Sand and gravel operations (sedimentation and temperature)
- Fish hatcheries (introduction of nonnative fishes and nutrients)
- Discharges from publicly-owned treatment works (POTWs)

There are also non-Federal actions likely to occur in or near surface waters in the State of Idaho, which are likely to have beneficial effects on the endangered and threatened species. These include implementation of riparian improvement measures; best management practices associated with timber harvest; animal grazing; agricultural activities; urban development; road building and abandonment and recreational activities; and other nonpoint source pollution controls. EPA is unaware of any other currently planned or reasonably foreseeable future activities in the lower Snake River drainage that could affect listed species.

4.3.1.1 Publicly Owned Treatment Works

POTWs discharging within the action area are listed in Table 4-1, below.

Table 4-1: POTWs Discharging within the Action Area

Facility Name	Permit Number	Permit Effective Date	Permit Expiration Date	Design Flow (mgd)	Parameters with Limits	Receiving Water
Lewiston WWTP	ID0022055	February 1, 2016	January 31, 2021	5.71	BOD5, TSS, E. coli, total residual chlorine, pH.	Clearwater River
Clarkston WWTP	WA0021113	May 1, 2016	April 30, 2021	2.2	Carbonaceous biochemical oxygen demand ($CBOD_5$), TSS, pH, fecal coliform bacteria, ammonia	Snake River

The EPA determined that its issuance of an NPDES permit to the City of Lewiston WWTP would have no effect on threatened or endangered species.

4.3.1.2 Stormwater

Facilities in the action area in Idaho with industrial stormwater permits are listed in Table 4-2, below.

Table 4-2: Industrial Stormwater Permits in the Action Area (Idaho)

Facility Name	Permit Number	SIC Code and Description	Latitude	Longitude	Receiving Water
Pacific Steel and Recycling	IDR053088	5093 = Scrap and Waste Materials	46.42668	-117.015248	Clearwater River

Facility Name	Permit Number	SIC Code and Description	Latitude	Longitude	Receiving Water
Herco, Inc. Asphalt Paving Plant	IDR053215	2951 = Asphalt Paving Mixtures and Blocks	46.427577	-117.011834	Clearwater River
Clearwater Paper Corp.	IDR053113	2631 = Paperboard Mills	46.426403	-116.956768	Lost Creek Wetland
Port of Lewiston	IDR053166	4013 = Switching and Terminal Services	46.4267	-117.015	USACE Ponds, Clearwater River
Port of Lewiston	IDR053167	4013 = Switching and Terminal Services	46.4255	-117.0119	USACE Ponds, Clearwater River
Port of Lewiston	IDR053168	4449 = Water Transportation of Freight	46.426581	-117.009566	USACE Ponds, City of Lewiston MS4 into Clearwater River, Clearwater River

Facilities in the action area in Washington with stormwater permits are listed in Table 4-3, below.

Table 4-3: Washington Stormwater Permits in the Action Area

Facility Name	Permit Number	Description	Latitude	Longitude	Receiving Water
City of Clarkston	WAR046502	Municipal SW Phase II Eastern WA GP	—	—	Snake River
Equipment Yard on Sacajawea Park Road	WAR304242	Construction SW GP	46.21006616	-119.029782	Snake River
Tidewater Terminal Co Clarkston	WAR000716	Industrial SW GP	—	—	Snake River
Sunnyslope Townhomes	WAR305022	Construction SW GP	46.41611	-117.08278	Snake River
SSI Burbank Yard	WAR010603	Industrial SW GP	46.206667	-119.025	Snake River
Koncrete Industries Burbank Batch Plant	WAR305176	Construction SW GP	46.20336	-119.021867	Snake River
Tidewater Terminal Co Pasco	WAR000023	Industrial SW GP	46.223632	-119.013851	Snake River

The EPA issued a draft permit for the City of Lewiston and Lewis-Clark State College MS4 on December 18, 2018. The EPA issued a draft permit for the Idaho Transportation Department

District 2 MS4 on February 5, 2019. More information is available on the EPA's website at: https://www.epa.gov/npdes-permits/stormwater-discharges-municipal-sources-idaho-and-washington.

4.3.1.3 Other Permitted Point Sources

The City of Lewiston's water treatment plant is covered under the EPA's general NPDES permit for drinking water treatment plants. The water treatment plant discharges to the Clearwater River. The permit number is IDG380003. The EPA determined that its issuance of the general NPDES permit for water treatment plants in Idaho would have no effect on threatened or endangered species.

The State of Washington has issued a general permit coverage (#WAG137006) to the Lyons Ferry Hatchery. This facility discharges to the Snake River near Starbuck. Outfall locations are listed in Table 4-4, below.

Table 4-4: Lyons Ferry Hatchery Outfall Locations

Outfall Number	Latitude	Longitude
001	46.59462413	-118.2327883
002	46.5977705502734	-118.233031054569

General information about NPDES permits issued by the EPA (i.e., Idaho permits and permits for federal facilities and facilities on Tribal land in Washington) can be found at: https://www.epa.gov/npdes-permits/about-region-10s-npdes-permit-program. General information about NPDES permits issued by the Washington Department of Ecology (non-federal and non-tribal facilities in Washington) can be found at: https://ecology.wa.gov/Water-Shorelines/Water-quality/Water-quality-permits.

4.3.1.4 Clearwater Air Emissions

EPA has long known that pulp and paper mills emit chlorine and chloroform to the air. In addition, pulp mills are known to be a source of odor due to total reduced sulfur (TRS).

It is possible that dioxins and furans will be emitted from the facility to ambient air. Stack emissions of dioxins and furans are more likely to be adsorbed onto emitted particles than in vapor form. If dioxins and furans are emitted from the facility adsorbed to particles from stack emissions, such particles may deposit within the watershed of the Snake and Clearwater Rivers.

Figure 4-2 shows the watershed of the Snake and Clearwater Rivers near the confluence. Erosion and runoff may cause particles that have been deposited in the watershed to drain into the Snake or Clearwater Rivers. The portion of the area around the Mill most likely to receive aerial deposition is in the predominant wind direction.

Because the Mill is in an east-west valley, the wind generally blows either to the east or to the west. Clearwater collects meteorological data such as wind speed and direction from an on-site meteorological tower. Using data collected from this tower, Idaho DEQ prepared a wind rose, depicting the frequency of wind speed and direction, on a quarterly basis. The wind rose for January 2007 through December 2011 is shown in Figure 4-3. It is clear from these on-site meteorological data that the predominant wind direction is east-to-west. Winds blow west-to-east to a lesser extent, and from the north and south rarely.

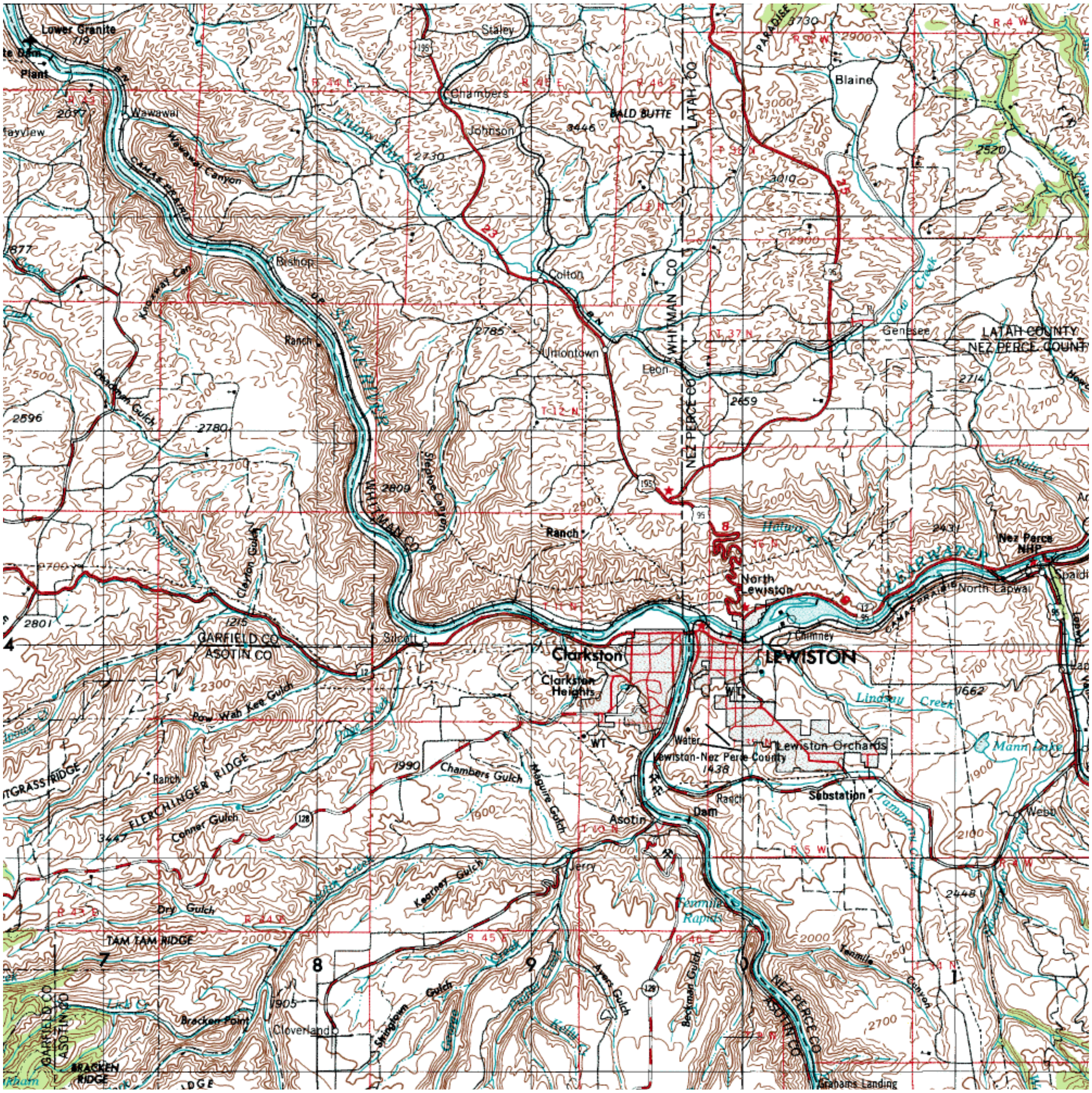

Figure 4-2. Map showing watershed around the Clearwater Mill in Lewiston, Idaho.

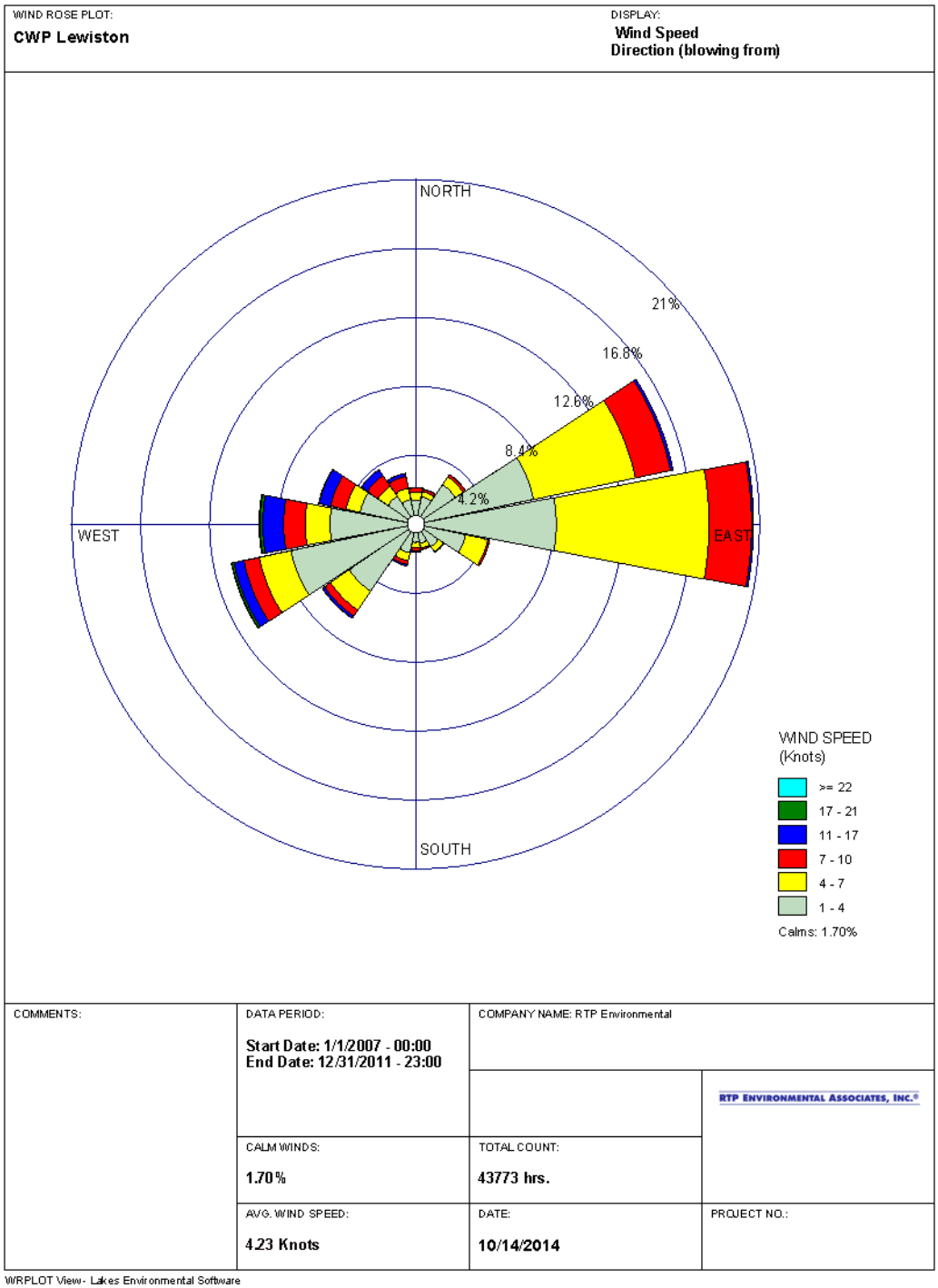

Figure 4-3: Wind roses depicting the frequency of wind speed and direction from Clearwater on-site meteorological data (2007-2011).

Deposition occurring in the easterly direction may land on the Clearwater River or Snake River watersheds upstream of the Mill, and deposition in the westerly direction may land on the Snake River watershed downstream of the Mill. The wind direction, however, likely has little to no effect on the extent to which dioxins and furans emitted from the Mill may eventually be transported to the Clearwater and/or Snake Rivers. This is because the watershed of the Snake

and Clearwater Rivers (the land area that drains into the rivers) covers a large area surrounding the rivers in all directions, not just in the easterly or westerly directions. Particles depositing in all areas of the watershed may be eroded or may runoff into the Snake and Clearwater River, potentially contributing adhered compounds that may be moved into the Rivers.

To remain in compliance with Clean Air Act requirements, the Mill has taken steps to reduce air emissions in the early 2000's. Due to improvements in air pollution control technologies and bleaching process changes instituted by the Mill, emissions of dioxins and furans under current operating conditions (if any) are lower than historical emissions and may even be zero. Clearwater does not monitor concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF in air emissions from the Mill. Although no measured concentrations in air emissions are available, Clearwater has estimated concentrations in air emissions using default emission factors developed by EPA for industrial sources, assuming the use of certain air pollution control equipment. Emission factors are numeric estimates of the quantity or concentration of a parameter in air emissions from certain types of facilities, based upon statistical analysis of measurements from numerous facilities of a given type. While they provide a general estimate of potential emissions from a certain type of source, they are not specific to any given facility. The values provided in Table 4-5 are EPA-generated emission factors for 2,3,7,8-TCDD and 2,3,7,8-TCDF for facilities similar to Clearwater's Mill. These values should not be assumed to represent actual emissions of 2,3,7,8-TCDD and 2,3,7,8-TCDF in air emissions from Clearwater's Mill. Rather, they represent EPA's estimate of emissions from facilities similar to the Mill.

Dioxins and furans have been measured in sediment upstream as well as downstream of the diffuser, as part of the sediment studies conducted from 2005 to 2006. A discussion of baseline conditions of dioxins and furans is provided in paragraph VII.E.1.e of this BE. Dioxins and furans contributed to the Snake and/or Clearwater River because of air deposition from the Clearwater facility would have been captured during this sediment sampling. There is no indication that effects of air deposition will increase above current baseline levels. In addition to the Clearwater Mill's stack emissions, other potential sources of dioxin deposition in the action area include forest fires and backyard burn barrels.

4.3.1.5 Global Air Deposition

The National Academy of Sciences (NAS, 1978) provided data on the occurrence of six halomethanes in the air. The general background tropospheric concentration of chloroform ranged from 9.8×10^{-5} to 19.6×10^{-5} mg/m^3, with higher concentrations in marine air, lower levels were normally found in continental air samples. Over urban areas, there can be higher concentrations of carbon tetrachloride, chloroform, and methylene chloride. Historically, automobile exhausts have been implicated in high urban area chloroform concentrations. However, in 1988, the California Air Resources Board studied chloroform emissions in southern California and concluded automobile emissions were a negligible source of chloroform, due in part to legislation reducing lead content in gasoline (State of California Air Resources Board, 1988).

Table 4-5: Estimated Emissions of Dioxin and Furans Based on EPA Emission Factors

	No. 4 Power Boiler Firing Wood/WTP Sludge			No. 4 Power Boiler Firing Oil			No. 4 Power Boiler Firing Paper			No. 2 Power Boiler Firing Oil			No. 4 Recovery			No. 5 Recovery			No. 3 Lime Kiln			No. 4 Lime Kiln		
PCDD/F Compound	Emission Factor (µg/ton)	Production Value (tons wood and sludge)	Emissions (g/yr)	Emission Factor (ng/gallon)	Production Value (gallons)	Emissions (g/yr)	Emission Factor (µg/ton)	Production Value (tons paper)	Emissions (g/yr)	Emission Factor (ng/gallon)	Production Value (gallons)	Emissions (g/yr)	Emission Factor (ng/lb BLS)	Production Value (tons)	Emissions (g/yr)	Emission Factor (ng/lb BLS)	Production Value (tons)	Emissions (g/yr)	Emission Factor (ng/lb CaO)	Production Value (tons)	Emissions (g/yr)	Emission Factor (ng/lb CaO)	Production Value (tons)	Emissions (g/yr)
Threshold Calculations	0.062	337938	0.0210	12.033	0	0	0.99	575	0.0006	12.033	6642	7.99E-05	0.453	179000	0.1622	0.453	649000	0.587994	0.378	12228.2301	0.0092	0.378	17297.04	0.013077
2,3,7,8-TCDD	0		0.0000	0		0	0		0.0000	0		0	0		0.0000	0		0	0		0.0000	0		0
1,2,3,7,8-PeCDD	0		0.0000	0.094		0	0		0.0000	0.094		6.24E-07	0		0.0000	0		0	0		0.0000	0		0
1,2,3,4,7,8-HxCDD	0		0.0000	0.24		0	0		0.0000	0.24		1.59E-06	0		0.0000	0		0	0		0.0000	0		0
1,2,3,6,7,8-HxCdd	0		0.0000	0.249		0	0.004		0.0000	0.249		1.65E-06	0.002		0.0007	0.002		0.002596	0.001		0.0000	0.001		3.46E-05
1,2,3,7,8,9-HxCdd	0		0.0000	0.302		0	0.005		0.0000	0.302		2.01E-06	0.005		0.0018	0.005		0.00649	0		0.0000	0		0
1,2,3,4,6,7,8-HpCDD	0		0.0000	1.806		0	0.105		0.0001	1.806		1.2E-05	0.049		0.0175	0.049		0.063602	0.028		0.0007	0.028		0.000969
1,2,3,4,6,7,8,9-OCDD	0.848		0.0178	7.779		0	0.569		0.0003	7.779		5.17E-05	0.142		0.0508	0.142		0.184316	0.256		0.0063	0.256		0.008856
2,3,7,8-TCDF	0.152		0.0032	0		0	0.044		0.0000	0		0	0.005		0.0018	0.005		0.00649	0.008		0.0002	0.008		0.000277
1,2,3,7,8-PeCDF	0		0.0000	0.243		0	0.015		0.0000	0.243		1.61E-06	0.002		0.0007	0.002		0.002596	0.002		0.0000	0.002		6.92E-05
2,3,4,7,8-PeCDF	0		0.0000	0.187		0	0.01		0.0000	0.187		1.24E-06	0.003		0.0011	0.003		0.003894	0		0.0000	0		0
1,2,3,4,7,8-HxCDF	0		0.0000	0.29		0	0.009		0.0000	0.29		1.93E-06	0.004		0.0014	0.004		0.005192	0.009		0.0002	0.009		0.000311
1,2,3,6,7,8-HxCDF	0		0.0000	0.134		0	0.007		0.0000	0.134		8.9E-07	0.002		0.0007	0.002		0.002596	0.002		0.0000	0.002		6.92E-05
1,2,3,7,8,9-HxCDF	0		0.0000	0		0	0.021		0.0000	0		0	0		0.0000	0		0	0		0.0000	0		0
2,3,4,6,7,8-HxCDF	0		0.0000	0.09		0	0.009		0.0000	0.09		5.98E-07	0.004		0.0014	0.004		0.005192	0		0.0000	0		0
1,2,3,4,6,7,8-HpCDF	0		0.0000	0.621		0	0.028		0.0000	0.621		4.12E-06	0.006		0.0021	0.006		0.007788	0		0.0000	0		0
1,2,3,4,7,8,9-HpCDF	0		0.0000	0		0	0.011		0.0000	0		0	0		0.0000	0		0	0		0.0000	0		0
1,2,3,4,6,7,8,9-OCDF	0		0.0000	0		0	0.021		0.0000	0		0	0.026		0.0093	0.026		0.033748	0		0.0000	0		0
Total CDD/Fs			0.0210			0			0.0005			7.99E-05			0.0895			0.3245			0.0075			0.010586

4.3.1.6 Historical DDT use

Dichloro-diphenyl-trichloroethane (DDT) is a chlorinated hydrocarbon that is used as a pesticide. DDT is effective against many organisms, but it's most known for its success in control of the *Anopheles* mosquito, which transmits malaria. Despite the value DDT has in combating diseases, such as malaria, the use of DDT has been abused. It is a "hard" insecticide, in that its residues accumulate in the environment. Although it is not especially toxic to mammals (the fatal human dose is 500 mg/kg of body weight, about 35 g for a 150-lb person), it is concentrated by lower organisms such as plankton and accumulates in the fatty tissues of fish and birds. In 1949, the Fish and Wildlife Service first noted the toxicity of DDT, but indiscriminate use as an agricultural pesticide for the control of crop-destroying pests continued to grow. In the State of Washington, the Snake River is impaired by 4,4' DDE, a DDT breakdown product (Washington State Department of Ecology, 2011).

4.3.2 Physical Description of Receiving Water

The Clearwater Mill is in Lewiston, Idaho, at Township 36 North, Range 5 West, within the Lower Snake-Asotin Subbasin, HUC 17060103. The Clearwater River is a tributary to the Snake River, and the Snake River is a tributary to the Columbia River, which are all part of the Columbia River Basin. The Columbia River Basin is highly regulated by dams. Figure 4-4 shows the location of the dams regulating the Columbia River Basin.

Upstream of the discharges from the Clearwater Mill, both the North Fork of the Clearwater and Snake Rivers are regulated by dams. Dworshak Dam (1972)[1] is located on the North Fork of the Clearwater River and greatly influences the flow and temperature of the Clearwater River. In the Snake River, there are several Idaho Power dams upstream of the outfall in Hells Canyon known as the Hells Canyon Complex. The Brownlee Dam (1958) is the furthest Hells Canyon Complex dam to the outfall.

Four dams impound the lower Snake River downstream of the discharge: Ice Harbor (1961), Lower Monumental (1969), Little Goose (1970), and Lower Granite (1975). Lower Granite Dam is located 39 miles downstream of the outfall and is the closest downstream dam to the outfall. The reservoir behind Lower Granite Dam is Lower Granite Reservoir (LGR). Impoundment of LGR is considered to end near Asotin, Washington, in the Snake River arm and near the Clearwater Mill in the Clearwater River arm.

The Lower Granite Reservoir (LGR) includes the confluence of the Snake and Clearwater Rivers. The uppermost portion of LGR is riverine in nature, while the lower portions more resemble a reservoir. The retention time of LGR is 7 to 10 days.

[1] http://www.nww.usace.army.mil/Locations/District-Locks-and-Dams/Dworshak-Dam-and-Reservoir/

Figure 4-4: Map of Dams Regulating the Columbia River Basin (USACE, 2003)

4.3.2.1 Snake River and Clearwater River Confluence Mixing

At the confluence of the Snake and Clearwater Rivers, the circulation dynamics are determined by the discharge and density (primarily a function of temperature) of both rivers. These processes have been modeled numerically by Cook et al. (2003) and described by Cook et al. (2006) and can be approximated by examining only the momentum balance between the two rivers. Cook et al. (2003) reports four "modes" of

mixing dynamics, while Cook et al. 2006 describes three general circulation patterns at the confluence of the Snake and Clearwater Rivers.

The dynamics of mixing between the Snake River and the inflowing Clearwater River near the Clearwater discharge diffuser may affect the fate of the effluent in the receiving-water system. Aerial photographs and in-stream measurements have recorded varying interaction between the rivers as a function of relative flow and temperature.

There are two main modes of the rivers mixing as identified by Cook et al. (2003). In the stratified mode (identified by Cook et al. as mode 4), the Clearwater River is significantly cooler than the Snake River. This scenario occurs during the summer when cold water is released from the Dworshak Reservoir to the Clearwater River. The colder Clearwater River water is sufficiently denser than the Snake River water under these circumstances to create vertical temperature stratification.

The occurrence of stratified temperature conditions is shown by field data collected by Clearwater as part of their routine monitoring program and as reported by Cook et al. (2003). Under conditions of low flow, the cold Clearwater River water has been observed to form a submerged stagnant wedge in the Snake River upstream of the confluence (Cook et al., 2003; Olivares, 2002).

During vertical thermal stratification, large differences in temperature have been observed throughout the reservoir downstream of the confluence (Cook et al., 2003). The temperature differences observed by Cook between epilimnetic (upper water column, above the thermocline) and hypolimnetic (lower water column, below the thermocline) waters occurred from June through September and peaked in July. The strength of stratification varied from site to site, however, differences greater than 10°C were observed between the epilimnetic and hypolimnetic layers.

Figure 4-5 illustrates an example of vertical and horizontal thermal stratification when the Clearwater River was approximately 10°C cooler causing it to abruptly plunge beneath the Snake River (Cook and Richmond, 2004). Downstream of the confluence and through the bend downstream, surface water temperatures remain constant in the satellite image, indicating vertical stratification of the river. This can be confirmed by examining temperature logger data in the Cook report (Cook et al., 2003).

A second mode (as defined by Cook et al. as mode 1) is an unstratified scenario in which the rivers do not mix but flow side by side. This scenario is illustrated in Figure 4-6, an infrared satellite image that records an occasion in which the temperature of the two rivers differed by approximately 1ºC. Under these conditions, the temperature and density difference between the rivers is not sufficiently great to cause stratification. Rather, the inertial force of the flowing Snake River overcomes the relatively weak force associated with the difference in the density of the two rivers. If the rivers have comparable discharge flows, the rivers do not intermix, but instead flow side by side.

Two other modes (modes 2 and 3) observed by Cook et al. (2003) occurred when one river flows with a much greater discharge than the other does. Under these conditions, the river of greater discharge may dominate the flow dynamics in the confluence and cause the two flows to intermix.

Cook et al. (2006) further generalizes the circulation patterns at the confluence of the Snake and Clearwater Rivers into three categories dependent on temperature and discharge rate. When the temperatures as well as the discharge rates of the two rivers are similar, the two rivers flow parallel to each other, with little mixing occurring between the two rivers for several miles downstream from the confluence.

When there is a small difference in temperature but a large difference in discharge rates between the two rivers, the two rivers will mix together within a short distance downstream of the confluence.

When there is a large difference in temperature between the two rivers, the colder Clearwater River plunges beneath the warmer Snake River at the confluence, creating a vertically stratified temperature profile. During July and August, the Clearwater River is significantly cooler (10 degrees or more) than the Snake River, and the resulting density difference is sufficient to stratify Lower Granite Reservoir. This vertical stratification due to large temperature differences occurs over a wide range of discharge rates.

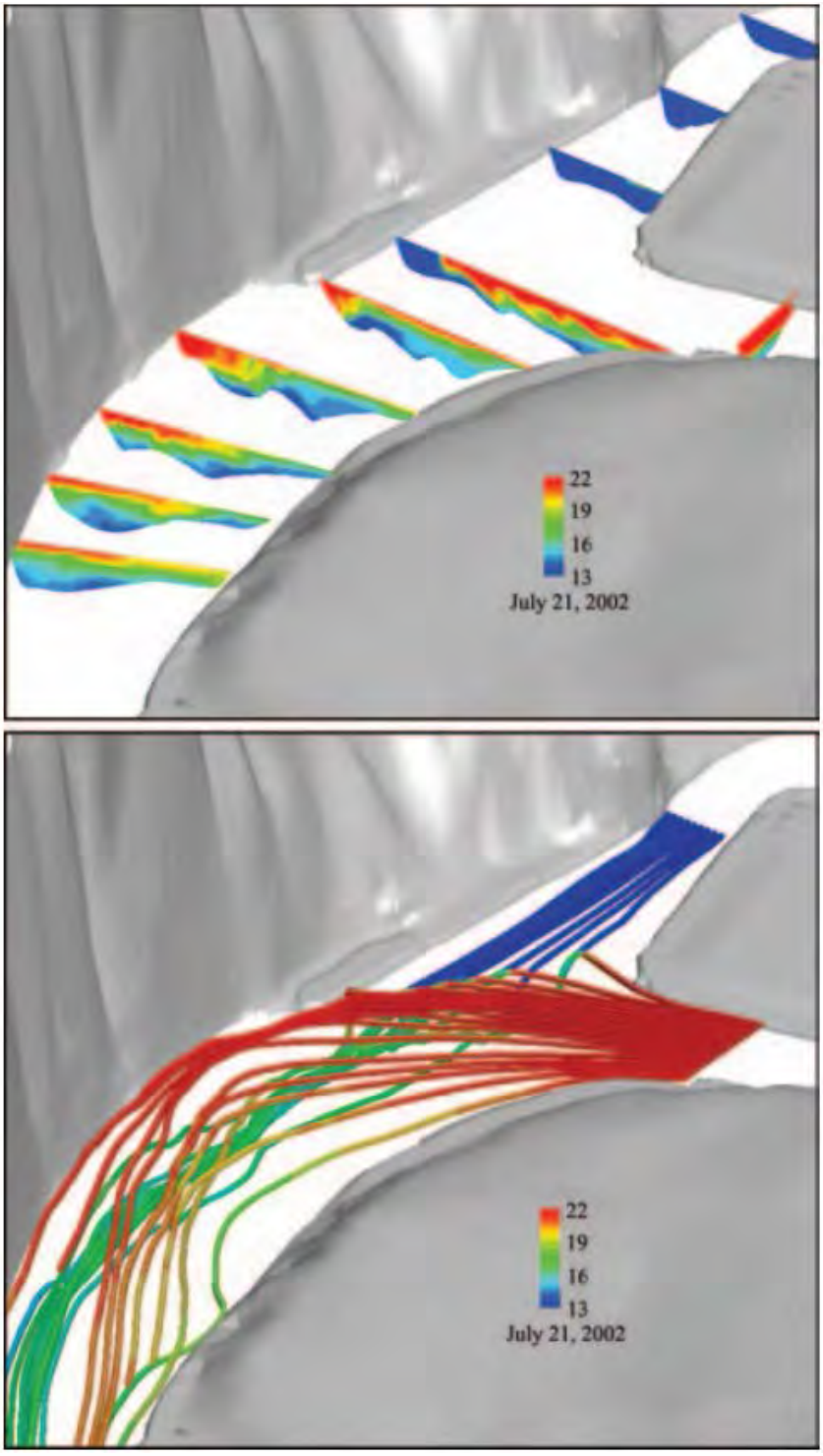

Figure 4-5: Computational fluid dynamics modeling of water temperature on July 21, 2002 at 11 a.m. (Legend is water temperature in degrees Celsius. (Cook and Richmond, 2004)

Figure 4-6: Infrared satellite image of horizontal stratification at the Snake and Clearwater confluence at midnight, 4/4/2002. The red and blue arrows indicate the directions of the Clearwater and Snake River flows respectively. (Cook et al., 2003)

4.3.2.2 Lower Granite Reservoir Velocity

A Receiving Water Monitoring Study was conducted by Potlatch Corporation as part of Endangered Species Act Tier 1 studies in 2005 and 2006 (AMEC, 2006; 2007). Water velocity was among the parameters selected for weekly monitoring during the studies. A mean daily value was calculated from all measurements collected from several depths on a sample day. Sampling locations are depicted in Figure 4-7. In both the 2005 and 2006 monitoring studies, mean water velocity at the Clearwater River reference location was typically greater or more variable than velocity measured at all locations in the Snake River. Velocity in the Clearwater River decreased from a maximum of 1.8 ft/s to a minimum of 0.4 ft/s over the entire 2005 monitoring period, while velocity in the Snake River remained fairly uniform and generally remained within 0.4 and 0.01 ft/s. In 2006, velocity in the Clearwater River decreased over the monitoring period and ranged from 0.32 ft/s to 3.67 ft/s, while velocity in the Snake River showed little variability with averages between 0.01 and 0.25 ft/s. Figure 4-8, Table 4-6 and Table 4-7 summarize the water velocity measurements collected during the studies.

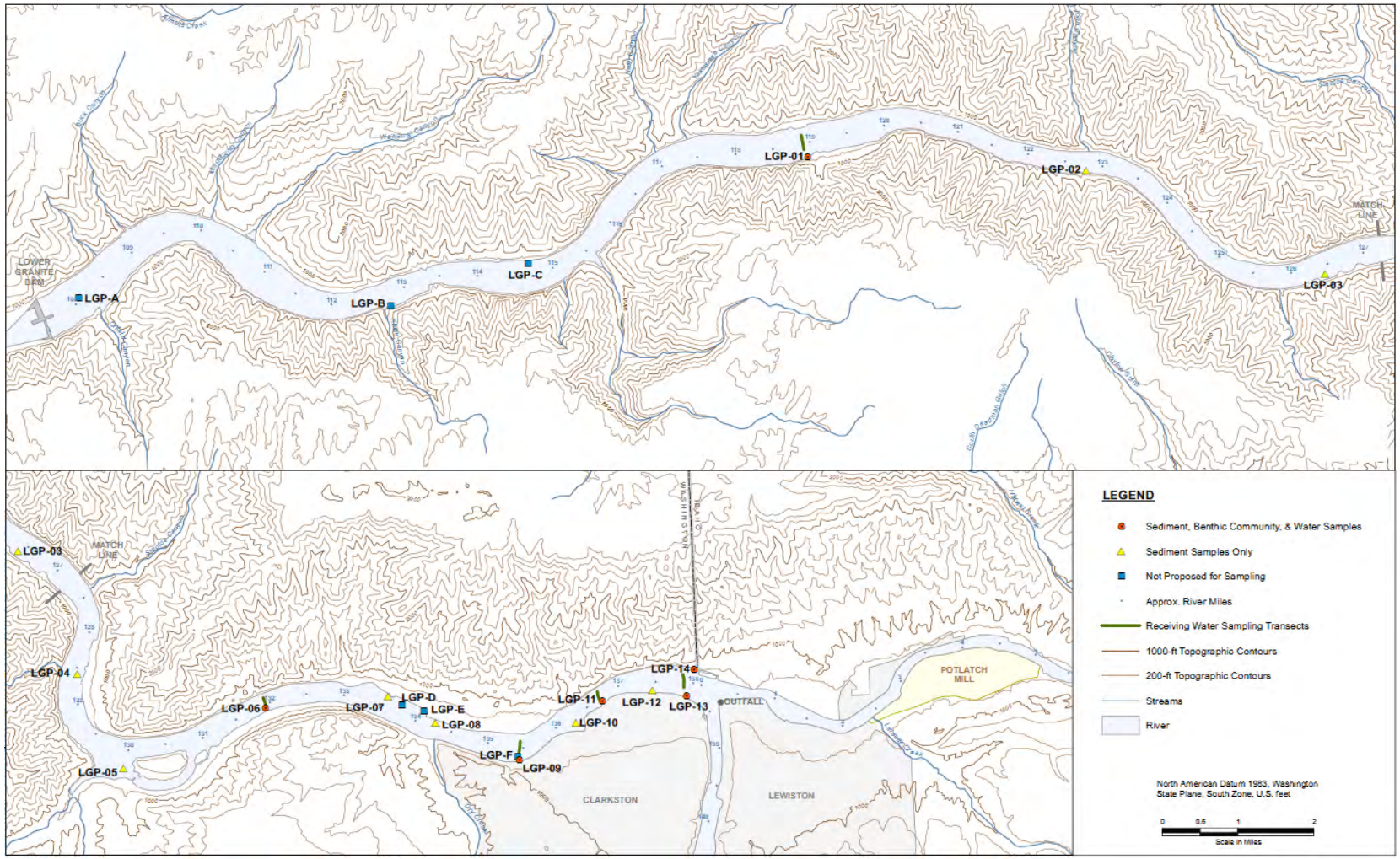

Figure 4-7: Location of sediment, benthic community, and receiving water samples in 2005 and 2006. (AMEC 2006, 2007)

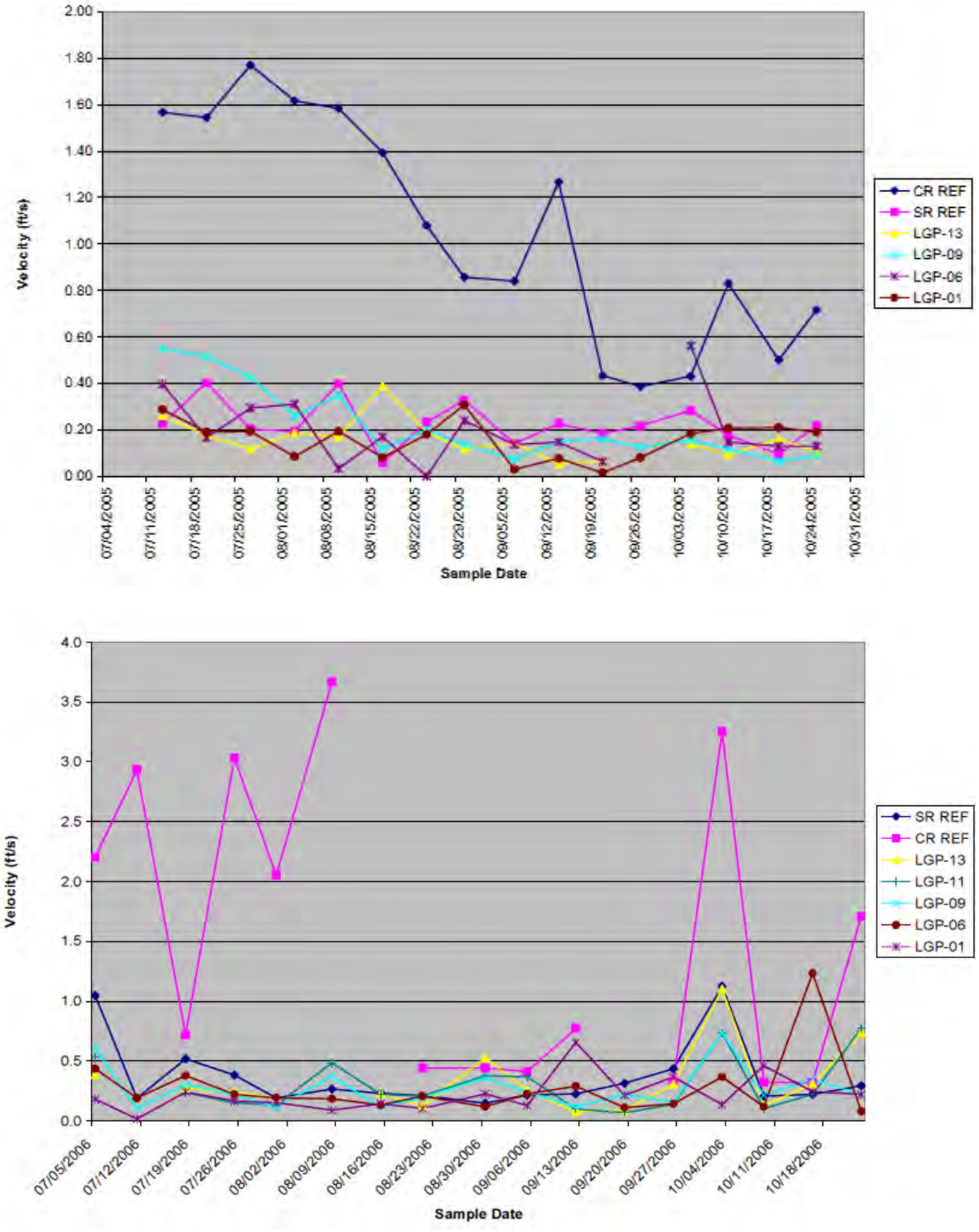

Figure 4-8: Velocity measured in Snake and Clearwater Rivers upstream and downstream of the Clearwater diffuser in 2005 and 2006 (AMEC 2006, 2007).

Table 4-6: Summary of velocity measures made in the Snake and Clearwater Rivers during 2005 receiving water sampling (AMEC, 2006)

Mean Velocity (ft/sec)

Sampling Date	SR REF	CR REF	LGP-13	LGP-09	LGP-06	LGP-01
07/07/2005						
07/13/2005	0.23	1.57	0.26	0.55	0.40	0.29
07/20/2005	0.40	1.54	0.18	0.52	0.17	0.19
07/27/2005	0.20	1.77	0.12	0.43	0.29	0.19
08/03/2005	0.19	1.62	0.19	0.26	0.31	0.09
08/10/2005	0.40	1.58	0.17	0.35	0.03	0.19
08/17/2005	0.06	1.39	0.39	0.12	0.17	0.08
08/24/2005	0.23	1.08	0.19	0.20		0.18
08/30/2005	0.33	0.86	0.12	0.14	0.24	0.31
09/07/2005	0.14	0.84	0.14	0.07	0.14	0.03
09/14/2005	0.23	1.27	0.05	0.15	0.15	0.08
09/21/2005	0.18	0.43	0.07	0.16	0.06	0.02
09/27/2005	0.22	0.39		0.13		0.08
10/05/2005	0.28	0.43	0.14	0.16	0.56	0.18
10/11/2005	0.18	0.83	0.09	0.12	0.15	0.21
10/19/2005	0.10	0.50	0.16	0.06	0.13	0.21
10/25/2005	0.22	0.72	0.11	0.09	0.13	0.19
Average	**0.22**	**1.05**	**0.16**	**0.22**	**0.21**	**0.16**
Median	**0.22**	**0.97**	**0.14**	**0.15**	**0.16**	**0.19**
Min	**0.06**	**0.39**	**0.05**	**0.06**	**0.03**	**0.02**
Max	**0.40**	**1.77**	**0.39**	**0.55**	**0.56**	**0.31**

Table 4-7: Summary of velocity measures made in the Snake and Clearwater Rivers during 2006 receiving water sampling (AMEC, 2007)

Sampling Date	SR REF	CR REF	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
07/05/2006	1.05	2.20	0.39	0.53	0.61	0.44	0.18
07/11/2006	0.19	2.93	*	*	0.10	0.19	0.01
07/18/2006	0.52	0.72	0.27	0.23	0.30	0.38	0.24
07/25/2006	0.38	3.03	0.24	0.15	0.22	0.22	0.17
07/31/2006	0.19	2.05	0.19	0.13	0.11	0.19	0.15
08/08/2006	0.26	3.67	*	0.48	0.37	0.18	0.09
08/15/2006	0.23	*	0.23	0.22	0.11	0.13	0.14
08/21/2006	0.21	0.44	0.12	0.21	0.20	0.21	0.10
08/30/2006	0.15	0.44	0.53	0.38	0.36	0.12	0.22
09/05/2006	0.22	0.41	0.25	0.37	0.24	0.22	0.13
09/12/2006	0.22	0.77	0.07	0.10	0.12	0.29	0.66
09/19/2006	0.31	*	0.09	0.07	0.22	0.11	0.21
09/26/2006	0.44	0.33	0.31	0.14	0.15	0.14	0.37
10/03/2006	1.12	3.25	1.10	0.73	0.73	0.37	0.13
10/09/2006	0.21	0.32	0.13	0.10	0.22	0.12	0.46
10/16/2006	0.22	0.32	0.31	0.22	0.35	1.23	0.24
10/23/2006	0.29	1.71	0.73	0.77	0.22	0.08	0.22
Average	**0.37**	**1.51**	**0.33**	**0.30**	**0.27**	**0.27**	**0.22**
Median	**0.23**	**0.77**	**0.25**	**0.22**	**0.22**	**0.19**	**0.18**
Min	**0.15**	**0.32**	**0.07**	**0.07**	**0.10**	**0.08**	**0.01**
Max	**1.12**	**3.67**	**1.10**	**0.77**	**0.73**	**1.23**	**0.66**

Notes:
*: Measurement not available for specific date/location

4.3.3 Water Quality Standards

Section 303(c) of the Clean Water Act requires every State to develop water quality standards applicable to all water bodies or segments of water bodies that lie within the State. A water quality standard defines the water quality goals of a water body, or a portion thereof, by designating the use or uses to be made of the water, by setting criteria necessary to protect the uses, and by establishing antidegradation policies and implementation procedures that serve to maintain and protect water quality. States adopt water quality standards to protect public health or welfare, enhance the quality of water, and serve the purposes of the Clean Water Act. A water quality standard should (1) include provisions for restoring and maintaining chemical, physical,

and biological integrity of State waters; (2) provide, wherever attainable, water quality for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water; and (3) consider the use and value of State waters for public water supplies, propagation of fish and wildlife, recreation, agriculture and industrial purposes, and navigation.

EPA has established water quality standards regulations at 40 CFR Part 131. Under section 510 of the Clean Water Act, States may develop water quality standards more stringent than required by this regulation. Water quality standards are composed of three parts: use classifications, numeric and/or narrative water quality criteria, and an antidegradation policy. The use designations required under the Clean Water Act include public water supply, recreation, and propagation of fish and wildlife. The States are free to designate more specific uses (e.g., cold water aquatic life, agricultural), or to designate uses not mentioned in the CWA, except for waste transport and assimilation which is not an acceptable designated use. Section 303(a-c) of the Clean Water Act requires States to adopt criteria sufficient to protect designated uses for State waters. These criteria may be numeric or narrative.

Water quality criteria set ambient levels of individual pollutants or parameters or describe conditions of a waterbody that, if met, will generally protect the designated use of the water. Water quality criteria are developed to protect aquatic life and human health, and, in some cases, wildlife from the deleterious effects of pollutants. Section 304(a) of the Clean Water Act directs EPA to publish water quality criteria guidance to assist States in developing water quality standards. EPA criteria consist of three components: magnitude (the level of pollutant that is allowable, generally expressed as a concentration), duration (the period of time over which the instream concentration is averaged for comparison with criteria concentrations), and frequency (how often criteria can be exceeded). Currently, EPA has developed criteria for over 150 pollutants including priority toxic pollutants, non-priority pollutants, and organoleptic effects criteria[2]. EPA criteria for the protection of aquatic life address both short-term (acute) and long-term (chronic) effects on freshwater species while human health criteria are designed to protect people from exposure resulting from consumption of water and fish or other aquatic live.

Narrative criteria are statements that describe the desired water quality goal and supplement the numeric criteria. Narrative criteria can be the basis for limiting specific pollutants where the State has no numeric criteria for those pollutants or they can be used to limit toxicity where the toxicity cannot be traced to a specific pollutant (e.g., whole effluent toxicity).

The federal regulations at 40 CFR section131.12 require States to adopt an antidegradation policy and implementation methods that provide three tiers of protection from degradation of water quality. Tier 1 protects existing uses and provides the absolute floor of water quality for all waters of the United States. Tier 2 protects the level of water quality necessary to support propagation of fish, shellfish, and wildlife and recreation in and on the water in waters that are currently of higher quality than required to support these uses. Tier 3 protects the quality of outstanding national resources, such as waters of national and State parks and wildlife refuges, and waters of exceptional recreational or ecological significance. As defined by the State of Idaho, the Snake River and Clearwater River are protected as Tier 1 from degradation of water quality.

[2] https://www.epa.gov/wqc/national-recommended-water-quality-criteria

As described above, Outfall 001 discharges to the Snake River at its confluence with the Clearwater River and seeps from the treatment pond discharge to the Clearwater River upstream of the confluence. Both discharges are near the head of Lower Granite Reservoir. The *Idaho Water Quality Standards* (IDEQ, 2018) designate this section of the Clearwater and Snake Arms of Lower Granite Pool as protected for the following uses: cold water biota, primary contact recreation, domestic water supply, wildlife habitats, and aesthetics. Table 4-8 provides the numeric water quality criteria that apply to the Snake River and the Clearwater River for these uses. The narrative water quality criteria are as follows:

4.3.3.1 Idaho Narrative Water Quality Criteria

- Surface waters of the state shall be free from hazardous materials in concentrations found to be of public health significance, or hazardous materials, toxic substances, and deleterious materials in concentrations that impair designated beneficial uses.
- Surface waters of the state shall be free from toxic substances in concentrations that impair designated beneficial uses. These substances do not include suspended sediment produced as a result of nonpoint source activities.
- Surface waters of the stat shall be free from deleterious material sin concentrations that impair beneficial uses. These materials do not include suspended sediment produced as a result of nonpoint source activities.
- Surface waters of the state shall be free from floating, suspended, or submerged matter of any kind in concentrations causing nuisance or objectionable conditions or that may impair designated beneficial uses.
- Surface waters of the state shall be free from excess nutrients that can cause visible slime growths or other nuisance aquatic growths impairing designated beneficial uses.
- Surface waters of the state shall be free from oxygen-demanding materials in concentrations that would result in an anaerobic water condition.

Because Clearwater's discharge is immediately upstream from the State of Washington, their standards were also considered to ensure that Washington's waters quality standards were not violated by the discharge. Washington's water quality standards are found in the Washington Administrative Code at WAC 173-201A. In the State of Washington, the Snake River, from its mouth to the Washington-Idaho-Oregon border (River Mile 176.1) is designated for salmonid spawning; rearing and migration; primary contact recreation; domestic, industrial, and agricultural water supply; stock watering; wildlife habitat; harvesting; commerce and navigation; boating; and aesthetics (WAC 173-201A-602). Table 4-8 provides the numeric water quality criteria that apply to the Snake River for Washington. The narrative water quality criteria are as follows:

4.3.3.2 Washington Narrative Water Quality Criteria

- Toxic, radioactive, or deleterious material concentrations shall be below those that have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic conditions to the most sensitive biota dependent upon those waters, or adversely affect public health.
- Aesthetic values shall not be impaired by the presence of materials or their effects, excluding those of natural origin, which offend the senses of sight, smell, touch, or taste.

Once standards are developed and adopted by States, EPA must review and approve or disapprove them. EPA's review is to ensure that the State water quality standards meet the requirements of the Clean Water Act and the water quality standards regulation. EPA may promulgate a new or revised standard for a State where necessary to meet the requirements of the Clean Water Act. Currently, States are required to review their water quality standards at least once every three years and revise them as necessary. The most current State water quality standards are used for the development of permit limitations.

The Idaho Department of Environmental Quality (IDEQ) began a Triennial Review of several of their water quality standards (IWQS) in 2014, intending to be completed in 2016. A report entitled, "2014 Triennial Review: Report of Findings to EPA" that included public input and IDEQ findings was submitted to EPA in November of 2014. Included were details of three workshops that resulted in the identification of findings divided into three categories: high priority, medium priority, and low priority. The findings were prioritized on a 3-4-year timeline with the high priority issues scheduled for 2015 and 2016. The high priority findings were as follows:

- Update Idaho's toxics criteria for the protection of human health to take into account newer Idaho-specific information of exposure from fish consumption.
- Undertake rulemaking to provide guidance for the designation of uses and development of use attainability analyses.
- Update aquatic life criteria for copper.
- Use work done on identification of salmonid spawning timing and location to complete designation of waters in Idaho which provide for or could provide a habitat for active self-propagating populations of salmonid fishes to support adoption of EPA's regionally recommended temperature criterion.
- Adopt new §304(a) recommendation for ammonia criteria.

IDEQ began a subsequent triennial review of their WQS in 2017 (IDEQ 2017). The high priority items identified in the 2017 triennial review were:

- Update recreational use and criteria and consider EPA's 2012 recommended criteria.
- Update aquatic life criteria for the following toxics with new or revised EPA recommended criteria: acrolein, carbaryl, and diazinon.
- Update aquatic life criteria for ammonia.
- Develop a performance-based approach for deriving site-specific temperature criteria.
- Designate appropriate aquatic life uses for Jacks Creek in the Bruneau subbasin.

Table 4-8: Numeric Water Quality Criteria for the Snake River and the Clearwater River

Parameter	Units	Most Stringent Criteria in Idaho		Most Stringent Criteria in Washington
		Snake River	Clearwater River	Snake River
Ammonia, total (as N)[8] acute chronic (4-day) chronic (30-day)	mg/L	 0.885 - 32.6 0.753 - 12.4 0.301 - 4.98	 0.885 - 32.6 0.753 - 12.4 0.301 - 4.98	 0.885 - 32.6 0.753 - 12.4 0.301 - 4.98
Antimony	µg/L	5.2	5.2	6

Parameter	Units	Most Stringent Criteria in Idaho		Most Stringent Criteria in Washington
		Snake River	Clearwater River	Snake River
Arsenic [2]	µg/L	10	10	0.018 [1]
Chloroform	µg/L	5.7	5.7	100
Chromium VI [2] acute chronic	µg/L	 15.7 10.6	 15.7 10.6	 16 11
Copper [2, 9] acute chronic	µg/L	 9.7 6.8	 4.6 3.5	 9.7 6.8
Dissolved Oxygen [7] water column intergravel one day 7-day average	mg/L	 6.0 [6] NA NA	 6.0 5.0 6.0	 8.0
	% of saturation	NA	90%	
Lead [2,9] acute chronic	µg/L	 33 1.3	 14 0.54	 33 1.3
Nickel [2]	µg/L	31.3	16.1	80
pH	s.u.	6.5 - 9.0	6.5 - 9.0	6.5 - 8.5 [10]
Pentachlorophenol	µg/L	0.11	0.11	0.002
2,3,7,8-TCDD	pg/L	0.013	0.013	0.013
Temperature daily maximum daily average	°C	 22 19	 22 19	 20.0 [3]
2,4,5-Trichlorophenol	µg/L	140	140	—
2,4,6-Trichlorophenol	µg/L	1.5	1.5	0.25
Turbidity instantaneous 10-day average	NTU	 50 25	 increase 5 NTU or 10% [5]	 increase 5 NTU or 10% [5]
Whole Effluent Toxicity (WET) acute chronic	TU	 0.3 1.0	 0.3 1.0	 0.3 1.0
Zinc [2, 9] acute chronic	µg/L	 68 62	 35 32	 68 62

Parameter	Units	Most Stringent Criteria in Idaho		Most Stringent Criteria in Washington
		Snake River	Clearwater River	Snake River

Footnotes:

1 Washington's human health criterion for arsenic is 0.14 µg/l, measured as the inorganic form only. However, because there is no EPA-approved test method to measure inorganic arsenic, the State does not apply this criterion in NPDES permits.
2 Metals criteria (except arsenic) are expressed as dissolved metal.
3 When natural conditions exceed 20°C, no temperature increase will be allowed which will raise the receiving water temperature by greater than 0.3°C, nor shall such temperature increases, at any time, exceed t=34/(T+9), where "t" represents the maximum permissible temperature increase measured at the mixing zone boundary; and "T" represents the background temperature as measured at a point or points unaffected by the discharge and representative of the highest ambient water temperature in the vicinity of the discharge.
4 Based on the geometric mean of a minimum of five (5) samples taken every three (3) to seven (7) days over a thirty (30) day period.
5 When background is 50 NTU or less, increase is restricted to 5 NTU. When background is greater than 50 NTU, increase is restricted to 10% or 25 NTU, whichever is less.
6 This standard does not apply to the bottom 20% of the reservoir or the hypolimnion strata.
7 The dissolved oxygen criteria are minimum values.
8 Since the criteria for ammonia is temperature and pH dependent, the ammonia criteria were developed based on the criteria for temperature and pH. Acute criteria are based on the daily maximum temperature and the chronic criteria are based on the daily average criteria.
9 These criteria are hardness dependent. The criteria were based on the 5th percentile of the data. The Snake River was hardness was determined to be 54.9 mg/L. Since the hardness of the Clearwater River is below 25 mg/L, the criteria must be based on 25 mg/L (see discussion in Appendix B).
10 With a human-caused variation within the above range of less than 0.5 units.
11 Those samples obtained for calculating the geometric mean value.

4.3.4 Status of Receiving Water Quality

USEPA's re-issuance of the Clearwater Mill NPDES permit in 2005 constituted a discretionary action that could beneficially or adversely affect threatened and endangered species or their critical habitat near the discharge. USEPA's BE was evaluated by U.S. Fish and Wildlife and National Marine Fisheries and as part of the biological opinion concluded that the permit re-issuance would not jeopardize the continued existence of listed species. The BO did specify non-discretionary terms and conditions which needed to be met by USEPA and Clearwater to minimize potential "take" of listed species because of permit reissuance. The implementation of a monitoring and assessment plan to characterize conditions in effluent, receiving water, sediment, and biological media near the Facility was one of the non-discretionary items. The Tier 1 monitoring was performed during the first two years of the current permit (2005 and 2006) (Appendix C).

The Surface Water and Effluent Study principally addressed the measurement of trace organic compounds and dioxins/furans (some of which were required for compliance with the NPDES permit). To do this, a specialized sampling technique known as High Volume Sampling was employed. The Receiving Water Monitoring Study primarily evaluated conventional water quality parameters that are routinely measured in the field (such as BOD, temperature, pH, and TSS). Both studies used an upstream/downstream study design. Field parameters including water velocity, pH, dissolved oxygen (DO), and temperature; conventional parameters analyzed included total suspended solids (TSS) and biochemical oxygen demand (BOD); nutrients including nitrogen-containing nutrients (i.e., ammonia-nitrogen, nitrate and nitrite nitrogen, and

total Kjeldahl nitrogen) and phosphorous-containing nutrients (i.e., total phosphorous and orthophosphate); dioxins/furans (i.e., 2,3,7,8-TCDD and 2,3,7,8-TCDF); resin acids; phytosterols; chlorophenolics; dissolved and total organic carbon; individual dioxin congeners; and furans were measured at seven sampling locations including:

- Clearwater River reference location (CR REF)
- Snake River reference location (SR REF)
- Locations downstream of the effluent from nearest to farthest:
    - LGP-13
    - LGP-09
    - LGP-06
    - LGP-01

The 2005 and 2006 Sampling results are summarized in Appendix C. As noted by AMEC (2006a and 2007), the results of the 2005 and 2006 weekly receiving water monitoring study and the quarterly surface water and effluent study revealed no indications that the Clearwater Facility's effluent has any influence on downstream parameter measurements. No meaningful difference between reference location conditions and downstream conditions were observed. In conclusion, the results of sampling and analysis upstream and downstream of the Facility support the finding in EPA's Biological Evaluation and the Services' Biological Opinions that the EPA's re-issuance of Potlatch's NPDES permit is not likely to jeopardize the continued existence of Snake River steelhead, Snake River spring/summer and fall Chinook salmon, and Snake River sockeye salmon, nor result in the destruction or adverse modification of designated critical habitat for Snake River spring/summer and fall chinook salmon and Snake River sockeye salmon. Tier 2 studies were not completed due to the results of the Tier 1 studies.

4.3.5 Mixing Zone

When an effluent discharge is released to an ambient waterbody in concentrations that vary from the waterbody (either greater than or less than), the effluent discharge will mix with the receiving waterbody until equilibrium is reached. This area of mixing is termed a mixing zone. The outfall for any effluent discharge should be designed to maximize mixing with the receiving water and decrease the size of the mixing zone. This BE refers to two types of mixing zones: the hydrodynamic mixing zone (HMZ) and the regulatory mixing zone (RMZ). Each type of mixing zone is described in the paragraphs below.

Mixing zones are areas where an effluent discharge undergoes initial dilution and are extended to cover the secondary mixing in the ambient waterbody. When effluent is discharged into a waterbody, its transport may be divided into two stages with distinctive mixing characteristics. The extent of mixing and dilution in the first stage are determined by the initial momentum and buoyancy of the discharge. This initial area of effluent contact with the receiving water is where the concentration of the effluent will be its greatest in the water column. The design of the discharge outfall should provide ample momentum to dilute the concentrations in the immediate contact area as quickly as possible.

The second stage of mixing covers a more extensive area in which the effect of initial momentum and buoyancy is diminished, and the effluent is mixed with the surrounding water primarily by ambient turbulence. In a large river or estuary, this second-stage mixing area may extend for miles before uniformly mixed conditions are attained. The general definition for a completely mixed condition is when no measurable difference in the concentration of the

pollutant (e.g., does not vary by more than 5 percent) exists across any transect of the waterbody. The HMZ is the area in which an effluent discharge is diluted until it becomes indistinguishable from the surrounding water, which generally encompasses the area in both the first and second stages of mixing.

In October and November of 1997, Potlatch conducted two field programs to study the HMZ of the effluent plume in the Snake River (Potlatch, 1997). Due to difficulties encountered during the October study, divers were used to assist in locating and measuring the plume during the November study. Temperature and conductivity values were used to measure the plume. Measurements were taken downstream of six diffuser ports at 1, 5, 10, 20, and 40 feet from the port opening. Values were also measured at 15 feet from one of the ports, 60 feet from two of the ports and 75 feet and 80 feet from 1 port each. The divers followed the centerline of the plume based on visual observations of the plume. Conductivity and temperature values were measured one foot above and below the centerline to ensure that the actual plume centerline had been identified. At 60 feet, the divers noted that the plume was difficult to distinguish from background. Width measurements of the plume were not measured.

During the study, effluent flow, temperature, and conductivity were collected at one-hour intervals from 7:00 am to 3:00 pm. The effluent measurements were relatively constant during the study (flow varied from 25210 to 25340 gpm; temperature varied from 24.8 to 25.1˚C; and conductivity varied from 1.687 to 1.781 µS/cm). Flow, temperature and conductivity were also measured at upstream stations of the Snake River and Clearwater River. The Clearwater River flow was 6340 cfs and the Snake River flow was 16400 cfs. Temperature and conductivity values were collected at the surface, mid-depth, and near the bottom. The Snake River temperature varied in depth by 0.2˚C during the day (9.7 to 9.9˚C) and the conductivity varied by 0.006 µS/cm (0.312 to 0.316 µS/cm) while the Clearwater River temperature varied in depth by 0.6˚C (8.8 to 9.4˚ C) and the conductivity varied by 0.18 µS/cm (0.104 to 0.284 µS/cm). The Clearwater River was slightly stratified.

From the study conducted by Potlatch (1997), the HMZ extends 60 feet downstream of the diffuser under the observed conditions. However, the width of the HMZ was not measured. Verification models using PLUMES were used to estimate the mixing zone width and dilution. The width was estimated to be 425 feet wide and the average dilution of the diffuser was estimated to be between 87 and 97.

A regulatory mixing zone is an allocated impact zone where water quality criteria (see discussion in IV.C.3) can be exceeded as long as a number of protections are maintained, including freedom from the following: materials in concentrations that settle to form objectionable deposits; floating debris, oil, scum, and other matter in concentrations that form nuisances; substances in concentrations that produce objectionable color, odor, taste, or turbidity; and substances in concentrations that produce undesirable aquatic life or result in a dominance of nuisance species. Since these areas of impact, if disproportionately large, could potentially adversely impact the productivity of the waterbody, and have unanticipated ecological consequences, they are carefully evaluated and appropriately limited in size. Therefore, a regulatory mixing zone is smaller than the hydrodynamic mixing zone. Appendix D provides the evaluation conducted for the regulatory mixing zones authorized for this NPDES permitted discharge.

In 2018, EPA used the CORMIX model to evaluate the mixing properties of the discharge (Nickel, 2018). CORMIX is a comprehensive software system for the analysis, prediction, and

design of outfall mixing zones resulting from discharge of aqueous pollutants into diverse water bodies.

A screening analysis was performed to evaluate the effect upon mixing of the variability in ambient temperatures and, in turn, densities (including ambient temperature stratification) throughout the year. At least one model simulation was set up for each month. Multiple simulations were set up for July through October, to reflect different ambient temperature stratification conditions that have been observed during July and September and to investigate the effect of changes in effluent temperature (and therefore density) upon plume behavior in a stratified ambient density field during these months. The simulation producing the poorest mixing in the screening analysis was then adapted for use sizing the mixing zones. The CORMIX model predicted that the poorest mixing will occur in early September and when the ambient temperature is strongly vertically stratified, and the effluent plume will "trap" below the thermocline. After identifying these conditions as the critical condition for mixing, additional modeling scenarios were run by EPA to evaluate mixing properties for acute and chronic life water quality criteria and for human health criteria for carcinogens and non-carcinogens.

The definition EPA is using to define the action area for the proposed permit is where the furthest effect is expected from the proposed action. The furthest effect is expected to occur near the mouth of the Snake River at its confluence with the Columbia River.

4.3.6 Status of Receiving Water Sediment Quality

As part of the Services concurrence with the biological evaluation that was submitted for the 2005 NPDES permit re-issuance to Clearwater (formerly Potlatch), the permittee was required to conduct monitoring studies in the vicinity and downstream of Outfall 001 including collecting and analyzing sediment samples from discrete locations in the receiving water system between the confluence of the Clearwater and Snake Rivers and Lower Granite Dam. The sediment quality data was combined with data from the other related investigations (i.e., receiving water, effluent, and biotic tissue) to evaluate potential impacts of discharges from the Clearwater Mill on juvenile salmon. The goal of the monitoring program was to support the effort to characterize the potential effects of discharges from Clearwater's Mill to the Clearwater and Snake Rivers on endangered and listed species and the environment.

In July 2005, sediment samples from 14 locations in the Snake River downstream of the confluence and at two reference locations in the Snake and Clearwater Rivers upstream of the outfall and the mill's settling pond were collected. Results of chemical and conventional analysis are reported in Table 4-9. The Snake and Clearwater Reference locations were upstream of the Clearwater discharge and the downstream sampling location names decrease as the distance downstream increase, with LPG-14 being the closest downstream sampling location and LPG-01 being the furthest downstream. As per the Anchor (2006) report, none of the concentrations of chemicals exceeded their respective benchmark criteria, either in any single replicate or in the arithmetic average of the respective four replicates for a given station. Most of the analytes that were detected at sample stations downstream of the Clearwater diffuser were also detected at the reference stations on both the Clearwater and Snake Rivers.

4.3.7 Benthic Macroinvertebrate Sampling

In the re-issued permit as part of the concurrence by the Services, Clearwater Paper Corporation (formerly Potlatch) was required to have a comparative study of the benthic macroinvertebrates

in the Snake River downstream of the discharge point and the upstream reference locations in the Snake and Clearwater Rivers. The purpose of the benthic community study was to evaluate the benthic macroinvertebrate community composition to determine whether any potential shifts in the benthic community composition that could affect the prey base for listed fish species and, if so, whether such shifts may be related to the Mill's effluent discharge.

Table 4-9: Sediment quality in Clearwater and Snake Rivers up and downstream of the Clearwater discharge (Anchor 2006)

		CR-REF		SR-REF		LPG-01		LPG-02		LPG-03		LPG-04		LPG-05		LPG-06	
Location ID	**CR-REF**	**Notes/**	**SR-REF**	**Notes/**	**LGP-01**	**Notes/**	**LGP-02**	**Notes/**	**LGP-03**	**Notes/**	**LGP-04**	**Notes/**	**LGP-05**	**Notes/**	**LGP-06**	**Notes/**
Conventionals																
Oxidation Reduction Potential (mv)	**59**		**-54.7**		**180**		**-82.3**		**-104.6**		**-97**		**-88.4**		**-38.8**	
Total organic carbon (%)	**0.9**		**2.8**		**1.5**		**5.1**		**5**		**10**		**2.2**		**2.7**	
Total solids (%)	**63.2**		**51**		**66.9**		**33**		**34.4**		**29.6**		**49.6**		**49.4**	
Grain size (%)																
Gravel	**0.1**		**0.4**		**11.8**		**0**		**0**		**2.1**		**0**		**0**	
Sand, Very Coarse	**0.3**		**0.6**		**10.3**		**1**		**1.2**		**3.5**		**0.8**		**0.4**	
Sand, Coarse	**1.3**		**1.7**		**12.3**		**0.9**		**0.8**		**5.8**		**0.6**		**1**	
Sand, Medium	**13.1**		**6.6**		**12.7**		**1**		**0.9**		**5.9**		**0.8**		**3.3**	
Sand, Fine	**46.6**		**33.2**		**19.9**		**2.7**		**2.4**		**4.4**		**4.3**		**9.2**	
Sand, Very Fine	**23.7**		**30.8**		**12.5**		**10.2**		**10.4**		**17**		**26.9**		**41.9**	
Silt	**11.9**		**22.4**		**16.4**		**76.7**		**79.9**		**54.7**		**65.2**		**39**	
Clay	**1.3**		**3.6**		**2.8**		**9.1**		**7.9**		**5.1**		**4.6**		**2.6**	
Dioxins and Furans (ng/kg)																
1,2,3,4,6,7,8-HpCDD	**7.51**		**7**		**3.54**	**J**	**21.9**		**21.55**		**24.23**		**10.99**		**11.95**	
1,2,3,4,6,7,8-HpCDF	**1.29**	**J**	**1.69**	**J**	**0.51**	**J**	**4.45**	**J**	**4.57**	**J**	**5.5**	**J**	**2.44**	**J**	**2.48**	**J**
1,2,3,4,7,8,9-HpCDF	**0.08**	**J**	**0.15**	**J**	0.07	U	**0.28**	**J**	**0.29**	**J**	**0.36**	**J**	**0.15**	**J**	0.07	U
1,2,3,4,7,8-HxCDD	**0.09**	**J**	**0.1**	**J**	0.16	U	**0.33**	**J**	**0.29**	**J**	**0.34**	**J**	**0.17**	**J**	0.14	U
1,2,3,4,7,8-HxCDF	0.15	U	**0.13**	**J**	0.13	U	**0.32**	**J**	**0.34**	**J**	**0.34**	**J**	**0.17**	**J**	0.12	U
1,2,3,6,7,8-HxCDD	**0.3**	**J**	**0.29**	**J**	0.11	U	**1.08**	**J**	**1.03**	**J**	**1.13**	**J**	**0.56**	**J**	**0.37**	**J**
1,2,3,6,7,8-HxCDF	0.15	U	**0.11**	**J**	0.14	U	**0.25**	**J**	**0.25**	**J**	**0.25**	**J**	**0.14**	**J**	0.13	U
1,2,3,7,8,9-HxCDD	**0.34**	**J**	**0.22**	**J**	0.16	U	**0.88**	**J**	**0.82**	**J**	**0.83**	**J**	**0.47**	**J**	0.14	U
1,2,3,7,8,9-HxCDF	0.15	U	**0.07**	**J**	0.13	U	0.14	U	0.14	U	0.13	U	0.13	U	0.12	U
1,2,3,7,8-PeCDD	0.15	U	**0.07**	**J**	0.14	U	**0.19**	**J**	**0.18**	**J**	**0.22**	**J**	**0.11**	**J**	0.13	U
1,2,3,7,8-PeCDF	0.13	U	**0.07**	**J**	0.12	U	**0.13**	**J**	**0.13**	**J**	**0.13**	**J**	**0.09**	**J**	0.11	U
2,3,4,6,7,8-HxCDF	0.09	U	**0.1**	**J**	0.08	U	**0.2**	**J**	**0.2**	**J**	**0.22**	**J**	**0.12**	**J**	0.07	U
2,3,4,7,8-PeCDF	0.14	U	**0.09**	**J**	0.13	U	**0.17**	**J**	**0.17**	**J**	**0.18**	**J**	**0.12**	**J**	0.12	U
2,3,7,8-TCDD	0.06	U	0.05	U	0.05	U	0.05	U	0.05	U	0.05	U	0.05	U	0.05	U
2,3,7,8-TCDF	0.07	U	**0.14**	**J**	**0.25**	**J**	**0.41**	**J**	**0.38**	**J**	**0.36**	**J**	**0.32**	**J**	**0.16**	**J**
OCDD	**56.48**		**51.73**		**24.73**		**160.5**		**162.5**		**174.25**	**J**	**87.25**	**J**	**85.15**	**J**
OCDF	**4.56**	**J**	**4.18**	**J**	**1.25**	**J**	**12.14**		**12.48**		**19.7**		**8.45**	**J**	**6.28**	**J**
Total HpCDD	**20.45**		**12.83**		**8.31**		**45.1**		**42.55**		**47.75**		**22.5**		**24.2**	
Total HpCDF	**3.93**		**3.87**		**1.27**		**11.51**		**11.5**		**15.9**		**6.48**		**6.49**	
Total HxCDD	**2.92**		**1.8**		**1.12**		**8.62**		**7.62**		**7.88**		**4.14**		**3.37**	
Total HxCDF	**1.21**		**1.49**		0.14	U	**5.54**		**5.61**		**6.23**		**2.79**		**3.22**	
Total PeCDD	**0.08**		**0.14**		0.14	U	**1.3**		**1.13**		**1.27**		**0.59**		0.13	U
Total PeCDF	**0.41**		**0.54**		0.13	U	**2.67**		**2.63**		**2.77**		**1.5**		**0.8**	
Total TCDD	**0.12**		**0.23**		**0.28**		**1.37**		**1.34**		**1.58**		**0.7**		**0.46**	
Total TCDF	**0.21**		**0.91**		**1.25**		**3.88**		**3.69**		**3.84**		**1.93**		**1.25**	
Guaiacols (µg/kg)																
3,4,5-Trichloroguaiacol	0.06	U	**0.21**	**J**	**0.33**	**J**	**0.6**	**J**	**3.45**	**J**	**2.08**	**J**	0.08	U	0.08	U
3,4,6-Trichoroguaiacol	0.16	U	0.21	U	0.18	U	0.32	U	0.31	U	0.32	U	0.19	U	0.21	U
4,5,6-Trichloroguaiacol	0.07	U	0.09	U	**2.11**	**J**	**2.95**	**J**	**9.77**	**J**	**9.23**	**J**	**0.77**	**J**	**0.39**	**J**
Tetrachloroguaiacol	0.07	U	0.09	U	0.08	U	0.14	U	0.14	U	0.14	U	0.08	U	0.09	U
Phytosterols (µg/kg)																
3,4,5-Trichlorocatechol	**0.28**	**J**	**3.71**	**J**	**1.9**	**J**	**5.2**	**J**		--	**17.8**	**J**	**2.41**	**J**	**7.11**	**J**
3,4,6-Trichlorocatechol	0.1	U	0.13	U	0.31	J	0.2	U		--	**1.78**	**J**	**0.11**	**U**	**0.45**	**J**
beta-Sitosterol	**10185**		**34325**		**7055**		**46650**		**46650**		**218500**		**16900**		**31550**	
Campesterol	**289**		**1190**		**300.5**	**J**	**1835**		**1685**		**4537.5**		**620**		**1069.5**	
Stigmastanol	**409**	**J**	**1735**	**J**	**737.25**	**J**	**4007.5**		**3927.5**		**8250**		**1327.5**	**J**	**1770**	
Stigmasterol	**273.5**		**746.5**	**J**	**281.5**	**J**	**1345**		**1247.5**		**2527.5**		**560.75**	**J**	**847.5**	
Tetrachlorocatechol	3.14	U		NR		NR		NR		NR		NR		NR		NR
Trichlorosyringol	0.07	U	0.1	U	0.08	U	0.15	U	0.14	U	0.15	U	0.08	U	0.09	U
Resin Acids (µg/kg)																
1,2-Chlorodehydroabietic acid	7.5	U	9	U	8.7	U	16	U	15	U	18	U	8.7	U	9.3	U
1,4-Chlorodehydroabietic acid	6.7	U	9.1	U	7.8	U	15	U	14	U	16	U	9	U	9.2	U
9,10-Dichlorostearic Acid	25	U	34	U	29	U	51	U	48	U	56	U	33	U	37	U
Abietic acid	**109**	**J**	**557.5**	**J**	**64.125**		**265**	**J**	**242.5**	**J**	**13950**		**59.25**	**J**	**600**	
Dehydroabietic Acid	**294**	**J**	**1975**		**120.25**		**977.5**	**J**	**900**	**J**	**11700**		**252.5**	**J**	**1557.5**	**J**
Dichlorodehydroabietic Acid	14	U	19	U	16	U	29	U	27	U	31	U	18	U	20	U
Isopimaric Acid	**69.75**		**275**		**39.375**		**185**	**J**	**197.5**	**J**	**5775**		**38.5**	**J**	**375**	
Linoleic Acid	**232.5**	**J**	**265**	**J**	**78.25**	**J**	**185**	**J**	**143**	**J**	**665**	**J**	**59.5**	**J**	**237.5**	**J**
Oleic Acid	**962.5**	**J**	**932.5**	**J**	**357.5**	**J**	**1067.5**	**J**	**835**	**J**	**5900**	**J**	**410**	**J**	**867.5**	**J**
Pimaric Acid	**21.125**		**159.25**		**76.75**		**68.25**	**J**	**64.5**	**J**	**850**		**17.125**	**J**	**159.5**	
Retenes (µg/kg)																
Retene	**21.28**		**106.43**	**J**	**17.68**		**190.5**		**215.25**		**4322.5**		**49.33**		**227.5**	
Phenols (µg/kg)																
2,3,4,6-Tetrachlorophenol	0.08	U	0.11	U	0.09	U	0.16	U	0.16	U	0.16	U	0.09	U	0.1	U
2,4,5-Trichlorophenol	0.07	U	**0.22**	**J**	0.08	U	0.15	U	0.15	U	0.15	U	0.09	U	0.1	U
2,4,6-Trichlorophenol	0.07	U	**0.26**	**J**	0.08	U	0.14	U	0.14	U	**0.45**	**J**	0.08	U	0.09	U
Pentachlorophenol	0.07	U	0.1	U	0.08	U	**0.43**	**J**	0.14	U	0.15	U	**0.26**	**J**	**0.27**	**J**

Notes:
Reference samples are from the Snake and Clearwater Rivers.
Refer to the data aggregation section of the data summary report for asumptions used in calculating the result average value.
Refer to the data summary report for an explanation of missing data values.
Bold The analyte was detected.
U The analyte was not detected above the sample reporting limit.
J The analyte was positively identified, and the estimated concentration is between the sample detection limit and the sample reporting limit.
NR The analytical laboratory did not report any data for this compound.

Table 4-9. Continued.

		LPG-07		LPG-08		LPG-09		LPG-10		LPG-11		LPG-12		LPG-13		LPG-14
Location ID	**LGP-07**	**Notes/**	**LGP-08**	**Notes/**	**LGP-09**	**Notes/**	**LGP-10**	**Notes/**	**LGP-11**	**Notes/**	**LGP-12**	**Notes/**	**LGP-13**	**Notes/**	**LGP-14**	**Notes/**
Conventionals																
Oxidation Reduction Potential (mv)	**-31.9**		**-3.9**		**-97.2**		**-71.5**		**-67.5**		**-75.9**		**-65.3**		**-100**	
Total organic carbon (%)	**1.6**		**1.9**		**4.3**		**4.1**		**3.9**		**2.9**		**2.2**		**2.2**	
Total solids (%)	**55**		**52.7**		**39.3**		**37.2**		**41.9**		**46.1**		**50.5**		**52.5**	
Grain size (%)																
Gravel	**0.3**		**0**		**0**		**0**		**0.2**		**0.1**		**0.1**		**0**	
Sand, Very Coarse	**0.3**		**0.7**		**0.8**		**1.4**		**1.4**		**1**		**1.3**		**0.5**	
Sand, Coarse	**0.3**		**1.4**		**1**		**2.6**		**2.7**		**2.6**		**3.8**		**1.3**	
Sand, Medium	**1.5**		**4.9**		**2.1**		**4.5**		**4.3**		**6.8**		**13.8**		**3.9**	
Sand, Fine	**16.6**		**17.5**		**8.5**		**11.7**		**31.7**		**38.1**		**47.5**		**15.6**	
Sand, Very Fine	**32.3**		**34.4**		**32.1**		**28.9**		**23.9**		**23.3**		**17.2**		**35.8**	
Silt	**46**		**38.6**		**47**		**45.9**		**28.1**		**22.7**		**12.4**		**37.7**	
Clay	**3**		**3.1**		**7.3**		**5.5**		**5.2**		**5.2**		**3.4**		**4.6**	
Dioxins and Furans (ng/kg)																
1,2,3,4,6,7,8-HpCDD	**8.09**		**10.33**		**17.28**		**16.43**		**10.78**		**8.06**		**5.71**	**J**	**25.75**	
1,2,3,4,6,7,8-HpCDF	**1.46**	**J**	**2.05**	**J**	**5.34**	**J**	**5.56**	**J**	**2.87**	**J**	**1.68**	**J**	**0.89**	**J**	**4.15**	**J**
1,2,3,4,7,8,9-HpCDF	**0.13**	**J**	**0.12**	**J**	**0.34**	**J**	**0.41**	**J**	0.07	U	**0.11**	**J**	**0.08**	**J**	**0.29**	**J**
1,2,3,4,7,8-HxCDD	**0.13**	**J**	**0.13**	**J**	0.16	U	**0.28**	**J**	0.16	U	**0.13**	**J**	**0.06**	**J**	**0.31**	**J**
1,2,3,4,7,8-HxCDF	**0.1**	**J**	**0.17**	**J**	0.14	U	**0.71**	**J**	0.13	U	**0.14**	**J**	**0.1**	**J**	**0.32**	**J**
1,2,3,6,7,8-HxCDD	**0.39**	**J**	**0.46**	**J**	**0.64**	**J**	**0.89**	**J**	**0.4**	**J**	**0.41**	**J**	**0.26**	**J**	**1.43**	**J**
1,2,3,6,7,8-HxCDF	**0.08**	**J**	**0.1**	**J**	0.14	U	**0.7**	**J**	0.14	U	**0.11**	**J**	**0.06**	**J**	**0.35**	**J**
1,2,3,7,8,9-HxCDD	**0.34**	**J**	**0.39**	**J**	0.16	U	**0.78**	**J**	0.16	U	**0.3**	**J**	**0.21**	**J**	**0.87**	**J**
1,2,3,7,8,9-HxCDF	0.13	U	0.13	U	0.14	U	0.13	U	0.13	U	0.13	U	0.14	U	0.14	U
1,2,3,7,8-PeCDD	**0.06**	**J**	**0.08**	**J**	0.14	U	**0.19**	**J**	0.14	U	0.14	U	0.15	U	**0.17**	**J**
1,2,3,7,8-PeCDF	0.12	U	**0.06**	**J**	0.12	U	**0.28**	**J**	0.12	U	0.12	U	0.13	U	**0.12**	**J**
2,3,4,6,7,8-HxCDF	**0.06**	**J**	**0.08**	**J**	0.08	U	**0.61**	**J**	0.08	U	**0.07**	**J**	0.09	U	**0.3**	**J**
2,3,4,7,8-PeCDF	**0.06**	**J**	**0.07**	**J**	0.13	U	**0.59**	**J**	0.13	U	**0.06**	**J**	0.14	U	**0.21**	**J**
2,3,7,8-TCDD	0.05	U	0.05	U	0.05	U	**0.05**	**J**	0.05	U	0.05	U	0.05	U	0.05	U
2,3,7,8-TCDF	**0.2**	**J**	**0.2**	**J**	**0.2**	**J**	**0.61**	**J**	**0.2**	**J**	**0.15**	**J**	**0.13**	**J**	**0.33**	**J**
OCDD	**62.25**		**76.8**		**129.1**		**112.18**		**75.38**		**58.25**		**42.15**		**158.5**	
OCDF	**4.04**	**J**	**4.95**	**J**	**15.24**	**J**	**15.42**	**J**	**7.4**	**J**	**4.07**	**J**	**3.74**	**J**	**8.79**	**J**
Total HpCDD	**17.58**		**20.8**		**31.5**		**31.73**		**21.23**		**15.55**		**10.92**		**56.5**	
Total HpCDF	**3.69**		**4.87**		**14.48**		**14.89**		**7.15**		**3.81**		**3.05**		**10.52**	
Total HxCDD	**2.95**		**3.33**		**4.45**		**6.69**		**3.37**		**2.56**		**1.57**		**8.83**	
Total HxCDF	**1.63**		**2.19**		**5.52**		**10.87**		**3.02**		**1.95**		**1.25**		**6.66**	
Total PeCDD	**0.13**		**0.33**		**0.34**		**1.58**		0.14	U	**0.13**		**0.07**		**0.75**	
Total PeCDF	**0.65**		**0.9**		**1.57**		**7.5**		**1.09**		**0.74**		**0.53**		**3.29**	
Total TCDD	**0.46**		**0.62**		**0.57**		**0.7**		**0.44**		**0.28**		**0.09**		**0.8**	
Total TCDF	**0.9**		**1.19**		**1.84**		**3.71**		**1.61**		**1.03**		**0.66**		**2.18**	
Guaiacols (µg/kg)																
3,4,5-Trichloroguaiacol	**0.26**	**J**	**1.24**	**J**	**1.36**	**J**	0.1	U	**1.25**	**J**	**0.32**	**J**	**0.4**	**J**	0.08	U
3,4,6-Trichoroguaiacol	0.19	U	0.19	U	0.31	U	0.24	U	0.24	U	0.23	U	0.19	U	0.19	U
4,5,6-Trichloroguaiacol	**0.82**	**J**	**6.17**	**J**	**0.28**	**J**	0.11	U	**0.48**	**J**	0.1	U	**7.72**	**J**	0.08	U
Tetrachloroguaiacol	0.08	U	0.08	U	**0.3**	**J**	0.11	U	0.1	U	0.1	U	0.08	U	0.08	U
Phytosterols (µg/kg)																
3,4,5-Trichlorocatechol	**1.69**	**J**	**4.62**	**J**	**8.27**	**J**	**3.01**	**J**	**9.01**	**J**	**2.32**	**J**	**6.42**	**J**	**3.43**	**J**
3,4,6-Trichlorocatechol	0.12	U	**0.28**	**J**	0.19	U	**1.46**	**J**	**1.1**	**J**	**0.84**	**J**	**0.61**	**J**	0.11	U
beta-Sitosterol	**12350**		**18200**		**52350**		**53400**		**58700**		**32950**		**30700**		**26100**	
Campesterol	**481.75**	**J**	**641**	**J**	**1877.5**		**1882.5**		**2377.5**		**1347.5**		**1242**		**885.75**	**J**
Stigmastanol	**1107.5**	**J**	**1397.75**	**J**	**2962.5**	**J**	**3105**	**J**	**3170**		**1485**	**J**	**1024.25**	**J**	**1905**	**J**
Stigmasterol	**429.5**	**J**	**511.25**	**J**	**1320**	**J**	**1297.5**		**1450**		**869.25**	**J**	**715**	**J**	**599**	**J**
Tetrachlorocatechol		NR		NR		NR		NR		NR		NR	4.94	U		NR
Trichlorosyringol	0.09	U	0.09	U	0.14	U	0.11	U	0.11	U	0.11	U	0.09	U	**0.27**	**J**
Resin Acids (µg/kg)																
1,2-Chlorodehydroabietic acid	9	U	9.4	U	15	U	14	U	14	U	9.8	U	9.9	U	9.2	U
1,4-Chlorodehydroabietic acid	8.1	U	8.5	U	13	U	13	U	9	U	9.8	U	8.9	U	8.2	U
9,10-Dichlorostearic Acid	30	U	31	U	47	U	46	U	**31.375**	**J**	36	U	33	U	30	U
Abietic acid	**64.75**		**90.5**	**J**	**642.5**		**2570**	**J**	**4005**	**J**	**1947.5**		**732.5**		**545**	**J**
Dehydroabietic Acid	**307.5**		**465**	**J**	**1825**		**5675**	**J**	**4175**		**4775**		**2950**	**J**	**987.5**	**J**
Dichlorodehydroabietic Acid	17	U	17	U	26	U	26	U	25	U	20	U	18	U	17	U
Isopimaric Acid	**49.5**		**85.75**	**J**	**360**		**477.5**	**J**	**857.5**	**J**	**350**		**530**		**212.5**	**J**
Linoleic Acid	**90.75**		**99**	**J**	**270**	**J**	**180**	**J**	**425**	**J**	**215**		**360**	**J**	**129.5**	**J**
Oleic Acid	**750**	**J**	**692.5**	**J**	**1300**	**J**	**4550**	**J**	**1700**	**J**	**2025**	**J**	**2075**	**J**	**900**	**J**
Pimaric Acid	**28.875**		**29.5**		**195**		**167.5**	**J**	**277.5**	**J**	**127.25**		**415.25**		**57.25**	**J**
Retenes (µg/kg)																
Retene	**49.38**		**69.65**		**385.25**	**J**	**1482.75**		**243**		**96.58**	**J**	**117.28**		**250**	**J**
Phenols (µg/kg)																
2,3,4,6-Tetrachlorophenol	0.1	U	0.09	U	0.11	U	0.12	U	0.12	U	0.12	U	**0.38**	**J**	**0.52**	**J**
2,4,5-Trichlorophenol	0.09	U	0.09	U	**0.32**	**J**	0.12	U	0.11	U	0.11	U	0.09	U	0.09	U
2,4,6-Trichlorophenol	0.08	U	0.08	U	**0.4**	**J**	0.11	U	**0.32**	**J**	0.1	U	0.08	U	0.08	U
Pentachlorophenol	0.09	U	0.08	U	0.1	U	**0.44**	**J**	0.11	U	0.1	U	**0.29**	**J**	0.08	U

Notes:

Reference samples are from the Snake and Clearwater Rivers.

Refer to the data aggregation section of the data summary report for asumptions used in calculating the result average value.

Refer to the data summary report for an explanation of missing data values.

Bold The analyte was detected.

U The analyte was not detected above the sample reporting limit.

J The analyte was positively identified, and the estimated concentration is between the sample detection limit and the sample reporting limit.

NR The analytical laboratory did not report any data for this compound.

AMEC (2006b) indicates that the evaluation of the benthic community data indicates the following:

- Taxa Richness: AMEC (2006) indicates that taxa richness at downstream sampling locations is not different from taxa richness at the Snake River reference location but does differ from taxa richness at the Clearwater River reference location. AMEC (2006) notes that the observed difference is likely attributable to differences in water temperature and/or habitat characteristics and not to influence from the Mill's effluent.
- Abundance: Species abundance ate downstream locations is not different from species abundance at the Clearwater River reference location but does differ from Snake River reference abundance. In evaluating the difference in abundance, abundance was correlated to temperature and not to concentrations of chemicals measured in sediment samples.
- Percent dominant taxa: No difference exists between downstream and reference locations with respect to percent dominant taxa except for the furthest downstream location LGP-01 (the only location where an amphipod was observed).
- Tolerance Index: There were no differences between tolerance indices for downstream sampling locations and those for the Snake River reference location. Although there was some difference in tolerance indices between downstream locations and the Clearwater River reference location, these were correlated to differences in percent fine sand and water temperature and not concentrations of chemicals measured in sediment samples.

The overall results of the macroinvertebrate sampling reveal no clear indications that the Facility's effluent has any significant influence on the downstream macroinvertebrate community (AMEC, 2006b).

4.3.8 Resident Fish Tissue Sampling and Analysis

In accordance with the renewal of the NPDES permit for Clearwater Paper Corporation (formerly Potlatch), the permittee was required to conduct resident fish tissue monitoring studies in the vicinity and downstream of Outfall 001 associated with Clearwater's Mill in Lewiston, Idaho. A total of 24 field sample replicate composites, field duplicates for six replicate composites and eight reference station composites were analyzed in this study. Analytes including dioxins/furans, resin acids, retene, beta-sitosterol, and chlorophenol were measured. Toxicity equivalence factors (TEFs) were calculated for all dioxin/furan congeners. The total concentration for each dioxin/furan congener was multiplied by its TEF and the results for each congener were expressed in terms of 2,3,7,8-TCDD equivalents (TEQ). The benchmark dioxin/furan toxicity equivalency factor (TEF) for 2,3,7,8-TCDD and total dioxin/furan TEQ is 9 TEQ. All TEQs calculated from resident fish were less than 0.1 thus no bioaccumulation effects are expected from the concentration of dioxins/furans in resident fish (Anchor, 2008a).

The concentration of most of the compounds analyzed were non-detect in the resident fish tissue. Seven different analytical methods were used and a total of 4590 individual analytes tested. Of the 4590, only 617 were detected in fish tissue. None of the analytes exceeded their respective benchmark criteria and of the analytes detected, 90 of the 617 detected results were attributed to linoleic acid and oleic acid/linolenic acid values. Most of the analytes that were detected in resident fish from sample stations downstream of the Clearwater diffuser were also detected at the reference stations on both the Clearwater and Snake Rivers, although tissue concentrations

tended to be lower at the reference stations as compared to the downstream sample stations (Anchor, 2008a).

4.3.9 Caged Bivalve Bioaccumulation Study

Clearwater Paper Corporation (formerly Potlatch) was required in the 2005 permit to conduct bioaccumulation studies downstream of Outfall 001 to evaluate potential impacts of the discharges from Clearwater Mill on listed species. A caged bivalve study was performed in accordance with the ESA Tier 1 Monitoring Plan approved by NOAA Fisheries and U.S. Fish and Wildlife Services. Locations for the caged bivalve study coincided with the sample stations for the receiving water and sediment studies previously discussed in this BE. Bivalve were placed at two upstream reference locations and five downstream locations between the Clearwater Mill outfall and the Lower Granite Dam.

Caged bivalve tissue collection resulted in 10 composite samples from below the Mill's outfall, one field duplicate composite, two baseline composites collected prior to deployment, and three reference site composites from upstream locations. Analytes including dioxins/furans, resin acids, retene, beta-sitosterol, and chlorophenol were measured. Toxicity equivalence factors (TEFs) were calculated for all dioxin/furan congeners. The total concentration for each dioxin/furan congener was multiplied by its TEF and the results for each congener were expressed in terms of 2,3,7,8-TCDD equivalents (TEQ). The benchmark dioxin/furan toxicity equivalency factor (TEF) for 2,3,7,8-TCDD and total dioxin/furan TEQ is 9 TEQ. All TEQs calculated from resident fish were less than 0.1 thus no bioaccumulation effects are expected from the concentration of dioxins/furans in resident fish (Anchor, 2008b).

The concentration of most of the compounds analyzed were non-detect in the resident fish tissue. Seven different analytical methods were used and a total of 688 individual analytes tested. Of the 688, only 131 were detected in fish tissue. Of the 131 detected results, 32 were attributed to linoleic acid and oleic acid/linolenic acid values. Most of the analytes that were detected in caged bivalves from sample stations downstream of the Clearwater diffuser were also detected at the reference stations on both the Clearwater and Snake Rivers, and the downstream sample stations showed that all tended to be very similar in the types and concentrations of analytes detected (Anchor, 2008b).

4.3.10 Washington State Toxics Monitoring Program

The Washington State Department of Ecology collected fish tissue samples within the action area in 2004, 2005, and 2009 (Seiders et al. 2007, Seiders et al. 2011). Results are summarized in Appendix I.

4.4 References

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5 Species Description

This section describes the threatened and endangered species that may occur in the Action Area as indicated by the USFWS[3] and NOAA[4] . The discussion includes the life history, habitat use, and habitat concerns as well as specific information on the abundance and timing of occurrence of each species within the Action Area. Additional species including Spalding's Catchfly, Yellow-billed Cuckoo, Northern Wormwood, and the Washington Ground Squirrel may be present in the action area but are not considered in this BE because they are not aquatic or aquatic dependent. These species are presented in this section but are not considered any further in the evaluation of effects of the action. The species addressed in this BE and their status is listed in Table 5-1. The presence of threatened and endangered fish species by month is summarized in Figure 5-1.

Table 5-1. List of the threatened and endangered species addressed by this BE.

Common Name	Scientific Name	Listing Status	Critical Habitat
Bull Trout	*Salvelinus confluentus*	Threatened	Yes
Snake River Fall Chinook salmon	*Oncorhynchus tshawytscha*	Threatened	Yes
Snake River Sockeye salmon	*Oncorhynchus nerka*	Endangered	Yes
Spring/Snake River Summer Chinook salmon	*Oncorhynchus tshawytscha*	Threatened	Yes
Steelhead	*Oncorhynchus mykiss*	Threatened	Yes

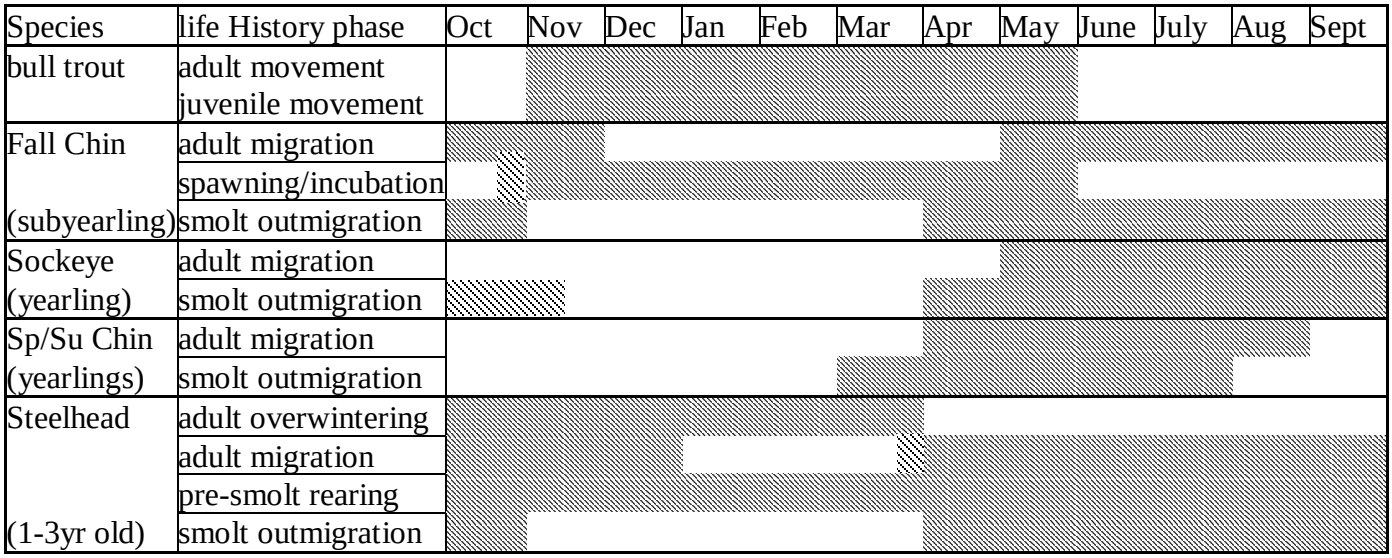

Figure 5-1: Timing of presence of salmon/steelhead species in the Action Area by life history phase based on passage data collected at the Lower Granite Dam (DART). Bull trout presence data are very limited, and timing is therefore estimated (USACE, 1999).

[3] ID: 01EIFW00-2016-SLI-1045; WA: 01EWFW00-2016-SLI-1286

[4] http://www.westcoast.fisheries.noaa.gov/protected_species/species_list/species_lists.html)

5.1 Inventories and Surveys

Descriptions of each species were synthesized from numerous documents and reports relevant to both the broad aspects of each species as well as information pertinent to the Action Area. Data presented on salmon, steelhead, and bull trout presence and abundance are from the following sources:

- Adults migrating upstream through the fish ladders are counted at the Lower Granite Dam March 1st through December 15th at the Lower Granite Dam (Larry Basham, Field Coordinator, Fish Passage Center, Pers. Comm., L. Herger, USEPA, August 20, 2003). These data were accessed from the Columbia River Data Access in Real Time database (DART)[5], which stores data from numerous sources and projects.
- Smolt data are available from the Lower Granite Dam smolt trap that is operated March through June. There are some year-to- year differences in these collection periods due to factors such as equipment and flow levels. These data were accessed from the DART database.
- The Columbia Basin Pit Tag Information System (PTAGIS)[6] database was queried for pit tag data collected at various sites in the basin. These data are useful for describing abundance and period of presence of smolts in the action area. The Columbia Basin Pacific States Marine Fisheries Commission maintains this database.

5.2 Bull Trout (Salvelinus confluentus) – Threatened

5.2.1 Status and Description

Bull trout are within the char subgroup of the family Salmonidae. The species is native to the Pacific Northwest and western Canada and is widespread throughout the tributaries of the Columbia River Basin (USFWS, 1998a). The USFWS listed the Columbia River population segment of the bull trout as threatened on June 10, 1998 (63FR 31647). Currently, Critical Habitat for bull trout has been designated throughout their U.S. range, and was set in place on September 30, 2010 (50 CFR, RIN1018-AW88). It includes all streams, lakes, and reservoirs in which bull trout are found, in Idaho, Washington, Oregon, Montana, and Nevada. Within Washington, 1213 km of marine shoreline have also been designated as bull trout critical habitat.

The USFWS recognizes 386 bull trout stocks within the Columbia River population segment in Montana, Idaho, Oregon, and Washington (USFWS, 1997). The area covered by the Columbia River population segment includes the entire Columbia River and eleven of its tributaries, excluding isolated populations in the Jarbridge River (a Snake River tributary) in Nevada.

Bull trout are present in the Snake River, (USFWS, 2000), and occupy large areas of contiguous habitat in the Snake River basin downstream of Hells Canyon Dam (USFWS, 1998a). Major Snake River tributaries below Hells Canyon Dam with bull trout subpopulations include the Tucannon, Grande Ronde, Imnaha, Clearwater and Salmon rivers and Asotin Creek (USFWS, 2000). Subpopulations occurring upstream of Hells Canyon Dam are generally small, isolated, and fragmented (USACE, 2002).

[5] http://www.cbr.washington.edu/dart
[6] http://www.ptagis.org

Historically, bull trout occurred throughout the Columbia River (IDFG, 1999). Bull trout were likely dispersed widely throughout the Snake River drainage (except in eastern Idaho) limited only by natural passage and thermal barriers (USACE, 1999). They were not known to occur above Shoshone Falls on the Snake River or in the Wood River basin. Today, bull trout are primarily found in upper tributary streams and some lake and reservoir systems as they have been eliminated from the main-stems of most large rivers (USFWS, 1998b). Generally, known bull trout populations in the entire Columbia River population segment are declining and occupy about 45 percent of their estimated historic range (Quigley and Arbelbide, 1997). In the Snake River and its tributaries, some bull trout populations appear stable, such as those in the Grande Ronde, Tucannon, and Malheur rivers, while others have a moderate to high risk of extinction (USFWS, 1997). Many bull trout have been observed in the Imnaha, Clearwater, and Salmon rivers (USFWS, 1998a), but these fish occur as isolated subpopulations in headwater tributaries. They exhibit lost or restricted life history forms and have reduced spawning areas and low abundances (USFWS, 1998a).

5.2.2 Life History

Bull trout populations exhibit four distinct life history forms: resident, fluvial, adfluvial, and anadromous. Fluvial, adfluvial, and anadromous fish are migratory, spawning in tributary streams where juveniles rear for one to four years. Fluvial bull trout juveniles then migrate to rivers where they grow to maturity. Adfluvial bull trout, after rearing, migrate to lakes where they remain until they reach maturity (Fraley and Shepard, 1989). Anadromous juvenile bull trout migrate to saltwater/coastal areas. This form does not occur in the Snake River basin and will not be discussed. The resident bull trout form inhabits their natal streams or nearby tributaries for their entire life cycle. More than one life history form, such as resident and adfluvial or fluvial, may occur in the same stock or population. Offspring of these fish may exhibit any one of these life history type behaviors (USFWS, 1998a). In the Snake River basin, bull trout exhibit both migrant and resident life history forms (USACE, 1999).

Adult bull trout begin to migrate from feeding to spawning grounds during the spring and summer, usually ending by mid to late July (USFWS, 1999a). Spawning occurs from August to November, with a peak during September and October (IDFG, 1999). A decrease in water temperature to below 10°C typically induces spawning (IDFG, 1999; USFWS, 1999a). Bull trout spawning occurs in the coldest stream reaches within river basins that are clean and free of sediment (USACE, 1999). Spawning sites are typically found in runs, tails, and pools with water depth ranging from 0.2 to 0.8 m. Eggs are buried 10 to 20 cm in the gravel with a water velocity ranging from 0.2 to 0.6 m/s (IDFG, 1999). Adult bull trout migrate back to wintering areas, lakes and large rivers, once spawning is complete in the fall.

Bull trout embryos incubate over the 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).

Juvenile migratory bull trout typically remain in tributary streams for one to three years before migrating to river main-stems and lakes when they are six to eight inches long. Rearing juvenile bull trout spend most of their time near stream substrate (USFWS, 1998b). They require high levels of in-channel woody debris, undercut banks, and rock/cobble substrate for use as cover (Rieman and McIntyre, 1993; IDFG, 1999; USFWS, 1999a). Juvenile bull trout are more sensitive to temperature changes than other life stages. Hillman and Essig (1998) found that the

optimal temperature for juvenile growth and rearing is likely 12 to 14°C. Juvenile bull trout prey on terrestrial and aquatic insects, and become more piscivorous as they mature (USFWS, 1998b).

Juveniles of the fluvial and adfluvial forms migrate during the spring, summer, and fall. Once reaching the river main-stem or lake, they will remain there until reaching sexual maturity at age 4 to 7 years (USFWS, 1998b). Migratory bull trout are typically larger than the resident form reaching approximate lengths of 24 inches as compared to 6 to 12 inches for residents (USFWS, 1998a).

Adfluvial mature bull trout associated with reclamation projects in the upper region of the Snake River basin appear to reside in reservoirs for approximately six months between November and June (USACE, 1999). During this period, with water temperatures of 7 to 12 °C, adult adfluvial bull trout live in shallow areas, depending on prey availability (Flatter, 1997). Most bull trout, even those not sexually mature, appear to migrate upstream beginning in May and June and return in November to December (USACE, 1999).

5.2.3 Habitat Concerns

All life history stages of bull trout have complex habitat requirements compared to many other salmonids (Fraley and Shepard, 1989; Rieman and McIntyre, 1993). The five parameters necessary for bull trout success, as outlined by Rieman and McIntyre (1993), are adequate channel and hydrologic stability, substrate, cover, temperature, and the presence of migration corridors. Also, stream flow, bed load movement, and channel instability influence the survival of juvenile bull trout (Weaver, 1985; Goetz, 1989).

Preferred spawning habitat includes low gradient streams with loose, clean gravel and cobble substrate and high water quality (Fraley and Shepard, 1989; USFWS, 1998b). The relatively long incubation period makes bull trout eggs and embryos vulnerable to fine sediment accumulation and water quality degradation (Fraley and Shepard, 1989). Cover, such as large woody debris, undercut banks, boulders, pools, side margins, and beaver ponds, is heavily utilized by all life stages of bull trout for foraging and resting habitat (USFWS, 1998a). Bull trout prefer cold waters, and temperatures greater than 15^{o}C are considered to limit their distribution (Rieman and McIntyre, 1993). Finally, migration corridors are important for sustaining bull trout populations, allowing for gene flow and connecting wintering areas to summer/foraging habitat (Rieman and McIntyre, 1993).

Bull trout distribution, abundance, and habitat quality have declined range-wide (Bond, 1992; Schill, 1992; Thomas, 1992; Ziller, 1992; Rieman and McIntyre, 1993; Newton and Pribyl, 1994; McPhail and Baxter, 1996). Threats to bull trout in the coterminous United States fall into several categories including: habitat degradation (e.g., land management activities with negative impacts on water quality or spawning habitat); passage restrictions; mortality or entrapment at dams; and competition from non-native lake and brook trout (USFWS, 1998b). Specific land and water management activities that depress bull trout populations and degrade habitat include dams and other diversion structures, forest management practices, livestock grazing, agriculture, agricultural diversions, road construction and maintenance, mining, and urban and rural development (Beschta et al., 1987; Chamberlin et al., 1991; Meehan, 1991; Frissell, 1993; Wissmar et al., 1994).

Bull trout populations associated with lower Snake River hydropower dams and reservoirs are likely affected by dam operation and/or flow augmentation (i.e., spill) used to mitigate effects on

salmon migration by increasing fish passage efficiency (USACE, 1999). Spill could result in the entrainment (and associated stranding, isolation, and/or delayed upstream migration) of individual bull trout that migrate to lower Snake River reservoirs seasonally to feed (USACE, 1999). Dams themselves can potentially harm bull trout through turbine mortality or gas super-saturation (USFWS, 1999b). Reservoir operations could negatively impact bull trout habitat quality and quantity, availability of Chinook smolts (the bull trout's most abundant prey in the lower Snake River main-stem), and access to tributary streams below the dams (USACE, 1999). Dam operations also cause water temperature shifts that could prolong warm water periods, delaying bull trout migration to cooler tributaries. Finally, fish passage ladders at dams have been designed for salmon and are not fully compatible with bull trout swimming style, resulting in blockage or delay for bull trout entrained in dam tail waters, though some bull trout have been observed to pass at fish ladders (USACE, 1999). Critical habitat has been noted by the USFWS as depicted in Figure 5-2.

5.2.4 Presence in the Action Area

Information to describe bull trout use of the Action Area is very limited but it is likely that fluvial and adfluvial form of both adults and juveniles may be present in the action area during their migration periods. One bull trout was spotted at Lower Granite Dam in 1998 (USFWS, 2000), possibly indicating that fluvial bull trout are migrating to some of the several bull trout subpopulations upstream of Lower Granite Dam (USACE, 2002).

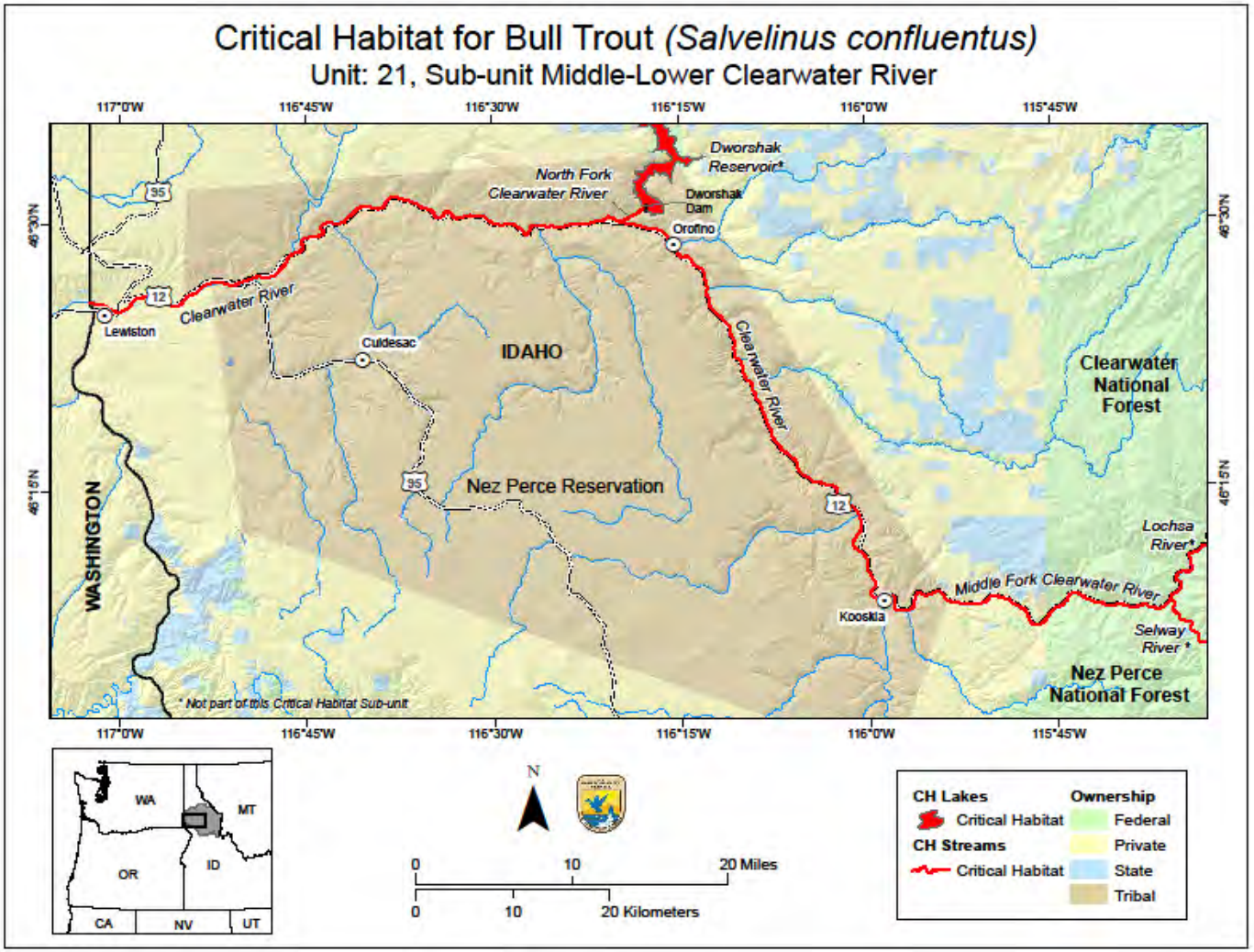

Figure 5-2: Bull trout critical habitat

Conversely, downstream movement of migrants from upstream of the Lower Granite reservoir (i.e., from Asotin Creek, and the Grande Ronde, Imnaha, Salmon, and Clearwater rivers) can also potentially move freely to and from Lower Granite Reservoir. However, the USFWS (2000) has found little evidence to suggest that these subpopulations use habitat associated with the lower Snake River main-stem dams. Seasonal use of the Snake River by bull trout is likely in upriver tributaries such as the Grande Ronde and Imnaha rivers, but these areas are substantially upstream of the Action Area.

In the Clearwater basin, there are known subpopulations of bull trout in the Selway, Lochsa, and North Fork and South Fork Clearwater rivers (IDFG, 1999). While little is known of the status or trends of these subpopulations, the migratory form is known to exist. Their use of the main-stem Clearwater River is seasonal, as summer water temperatures exceed those used by bull trout.

Besides the restrictions of movement caused by the dams and the overall low population status of bull trout in the basin, use of the Action Area is limited by physical habitat conditions. Spawning and rearing habitat between the Clearwater/Snake confluence and Lower Granite Dam is limited due to high water temperatures, lack of in-stream woody debris (cover), and poor gravel substrate. The combination of these factors likely results in a low abundance of bull trout in the Action Area. Estimated periods of presence of adult adfluvial bull trout in reservoirs like the Lower Granite Reservoir are November through May (USACE, 1999).

5.2.5 Abundance and Timing Data

At the Lower Granite Dam, one bull trout was caught in the Snake River every year or two, indicating that bull trout are present in the upper end of Lower Granite Reservoir during the spring of at least some years (Bueftner, 2000). Basham (2000) indicated that the Idaho Department of Fish and Game smolt trap at Lewiston, Idaho captures an occasional bull trout, at catch rates of no more than one bull trout annually. Data from the Fish Passage Center shows no bull trout captured in the Lower Granite Dam smolt trap during the years 2006 through 2015. Because the trap is only operated during the spring, the catch information cannot be used to confirm that bull trout are absent any time of the year. Likewise, it is possible that bull trout may be passing through the Lower Granite Dam during periods when the smolt trap is not operational or the counts are not being made at the ladders (July through February and January through February, respectively).

Recent telemetry studies indicate that adult bull trout typically occur in or near the action area for about 7-8 months annually. Baxter (2002) used telemetry to observe bull trout seasonal migrations from the Wenaha River, a tributary to the Grande Ronde River, to the Snake River downstream of Hells Canyon. Of the bull trout that migrated from the Wenaha to the Snake River, most reached their furthest distance from the Wenaha River from late October to mid-December. Among those bull trout, return migration occurred between May and early July.

5.3 Fall Chinook salmon (Oncorhynchus tshawytscha) - Threatened

5.3.1 Status and Description

The Chinook salmon is the largest of the Pacific salmon (NMFS, 1997a). In the Snake River Basin, there are spring, summer, and fall Chinook runs. Due to differences in genetics, as well as spawning location and timing, fall Chinook are considered a separate Evolutionarily Significant Unit (ESU) from the other Snake River Chinook runs (Matthews and Waples, 1991). Spring and summer Chinook are discussed in a separate section. The Snake River fall Chinook was listed as threatened on April 22, 1992 (NMFS, 1992) and Critical Habitat was designated for this run on December 28, 1993. The designated Critical Habitat includes all river reaches accessible to Chinook salmon in the Columbia River from the Dalles Dam upstream to the confluence with the Snake River in Washington (NMFS, 1998). Critical Habitat in the Snake River basin includes its tributaries in Idaho, Oregon, and Washington.

Historically, fall Chinook were found throughout the Snake River main-stem and the lower sections of its major tributaries, occurring all the way upstream to Shoshone Falls (607 RM upstream). Snake River fall Chinook are now restricted to the portions of the Snake River below Hells Canyon Dam. Currently, Snake River fall Chinook spawn in the main-stem Snake River from the head of Lower Granite Dam Reservoir at RM 148.3 to Hells Canyon Dam at RM 184.3, the lower reaches of the Clearwater and Salmon rivers in Idaho, and the Grande Ronde, Imnaha, and Tucannon rivers of Oregon and Washington.

5.3.2 Life History

Columbia Basin fall Chinook salmon are ocean-type outmigrants, typically migrating to the ocean at a much younger age than stream-type spring/summer Chinook (Waples et al. 1991). Although they may remain in fresh water for up to a year after hatching, most juveniles begin outmigrating within the first three months of life. Ocean-type Chinook are more likely to remain in coastal waters in the ocean, rather than undergoing extensive offshore migrations, and to utilize estuaries and coastal areas for juvenile rearing to a larger degree than stream-type Chinook.

Adult fall Chinook begin entering the Columbia River in July and migrate to the Snake River primarily from August to November. By October, most fall Chinook have passed Lower Granite Dam (USACE, 1999). Preferred water temperature ranges for adult Chinook salmon have been variously described as 12.2 to 13.9 °C (Brett, 1952), 10 to 15.6 °C (Burrows, 1963), and 13 to 18 °C (Theurer et al., 1985). From 1993 to 1995, Groves and Chandler (2001) observed that the peak spawning by fall Chinook salmon in the Snake River occurred from about November 1 to November 20 with the earliest spawning observed on October 21; the latest spawning was December 13. During their study, spawning generally began as water temperatures dropped below 16.0°C, and concluded as temperatures approached 5.0°C.

In the Snake River Basin, fall Chinook salmon spawn from October to December in the main channel of the Snake River and in the lower reaches of its major tributaries. Female Chinook deposit eggs in redds constructed in gravel beds. Fry emerge from the gravel from late April to late May. Fry rear near the shoreline of the main-stem river and in

river reservoirs for a few months before migrating to the ocean as subyearlings in the summer and fall (Garland and Tiffan, 1999). Analysis of Passive Integrated Transponder (PIT) tagged fall Chinook indicates significant numbers of subyearling migrants continue moving through the lower Snake River corridor of reservoirs during July and August. Out-migration is initiated when juvenile fall Chinook reach a threshold size believed to be about 85 mm; this often occurs as water temperature increases and river flow drops in June and July (Williams et al., 1996). Fall Chinook continue to rear and grow as they migrate and actively feed in the juvenile migration corridor/rearing area. Fall Chinook juveniles use the shoreline habitat during outmigration. They preferred low water velocity and sandy substrate to rip rap, due to better foraging opportunities (Williams et al. 1996). Bottom gradients, also influenced the habitat selection of subyearling fall Chinook salmon with most being found in areas with slope less than 20 percent (Garland and Tiffan, 1999). Subyearling fall Chinook salmon in the lower Snake River reservoirs are either pelagic-oriented or found over sandy, mostly unvegetated substrate. Young fall Chinook become more pelagic as shore temperatures exceed 20 °C. High water temperatures may limit juvenile fall Chinook salmon rearing in reservoirs along the main-stem of the Snake River after July (USACE, 1995). Migration spikes documented at Lower Granite Dam are typical before and after August. These are likely caused by the formation of elevated water temperatures that cause a thermal barrier preventing fish passage down river.

Once fall Chinook young enter the Columbia River estuary, they forage and grow before moving to the ocean (Van Hynig 1968). Fall Chinook remain in the ocean for one to four years before migrating back to their natal rivers and streams. Most Snake River fall Chinook return after three years (Chapman et al. 1991).

5.3.3 Habitat Concerns

Factors influencing the decline of fall Chinook include the destruction, modification, or curtailment of its habitat or range. Human activities that contribute to habitat loss and modification are water diversions, timber harvest, agriculture, mining, and urbanization (NMFS, 1998). Over-fishing of the species for commercial, recreational, scientific or educational purposes is also a contributing factor. Finally, factors such as predation, introduction of non-native species, and habitat loss or impairment increase stress on any surviving individuals and thus increases potential susceptibility to diseases. The continued straying by non-native hatchery fish into natural production areas is an additional source of risk to the Snake River fall Chinook salmon (NMFS, 1998).

Hydroelectric development on the main-stem Columbia and Snake rivers continues to affect juvenile and adult migration (NMFS, 1998). Almost all historical Snake River fall-run Chinook salmon spawning habitat in the Snake River Basin was blocked by the Hells Canyon Dam complex (NMFS 1998). Inundation of the main-stem Snake and Columbia rivers have reduced the remaining habitat. Critical habitat has been noted by the USFWS as depicted in Figure 5-3.

Figure 5-3: Chinook salmon critical habitat.

5.3.4 Presence in Action Area

The Action Area is within the Snake River fall Chinook salmon migration corridor used by adults migrating to upstream spawning habitat and by smolts outmigrating to the ocean. Returning adult fall Chinook salmon migrate upstream through this section of the Snake River from May through September and smolts migrate downstream through the area primarily from April through October.

Orientation within the water column is not specifically known for adult fall Chinook. However, hydroacoustic surveys (USACE 1991) found larger fish are typically oriented near the bottom in the Lower Granite Reservoir. Outmigrating juveniles were located throughout the water column with the greatest concentration in the upper 15 meters. Subyearling Chinook use shoreline areas of islands and other shallow areas within the Lower Granite Reservoir during migration (Bennett et al. 1993).

5.3.5 Abundance and Timing Data

Fall Chinook passage data has been collected at the Lower Granite Dam beginning in 1975 and are available from the DART database[7]. These data are collected at the dam starting on August 18 and ending on December 15th, as the Corps considers this time frame the counting window (USACE as cited at http://www.cbr.washington.edu/dart/adultruns.html). This window of data collection effort may not capture the earliest dates of passage. Data for 2006 through 2015 are presented to describe abundance and passage near the Action Area **(**Appendix E**)**.

Upstream passage of adult fall Chinook into the Lower Granite Reservoir occurred from late August to early November (Table 5-2 and Figure 5-4). The date of early passage for the Lower Granite Dam is August 17, 2008 and 2012 (earliest date of data collection) and the date of late passage is assumed to be December 15, 2010 (latest date of data collection). Thus, data presented in Table 5-2 collected from 2006 through 2015 reflects the start of monitoring rather than the date of first passage. For these years, the data end between December 2 (in 2009) and December 15 (in 2010).

Table 5-2: Dates of Adult Fall Chinook Passage at Lower Granite Dam 2006 - 2016.

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2006	Average	8/18	8/30	9/18	10/16	12/3
2007	Average	8/18	8/31	9/21	10/20	12/3
2008	Average	8/17	8/31	9/12	10/09	12/4
2009	Average	8/18	8/29	9/14	10/16	12/2
2010	Average	8/18	9/3	9/19	10/15	12/15
2011	High	8/18	8/28	9/19	10/21	12/7
2012	Average	8/17	9/2	9/18	10/11	12/12
2013	Average	8/18	9/5	9/20	10/13	12/4
2014	Average	8/18	9/3	9/20	10/13	12/4
2015	Low	8/18	9/1	9/19	10/15	12/14

[7] http://www.cbr.washington.edu/dart/adultruns.html

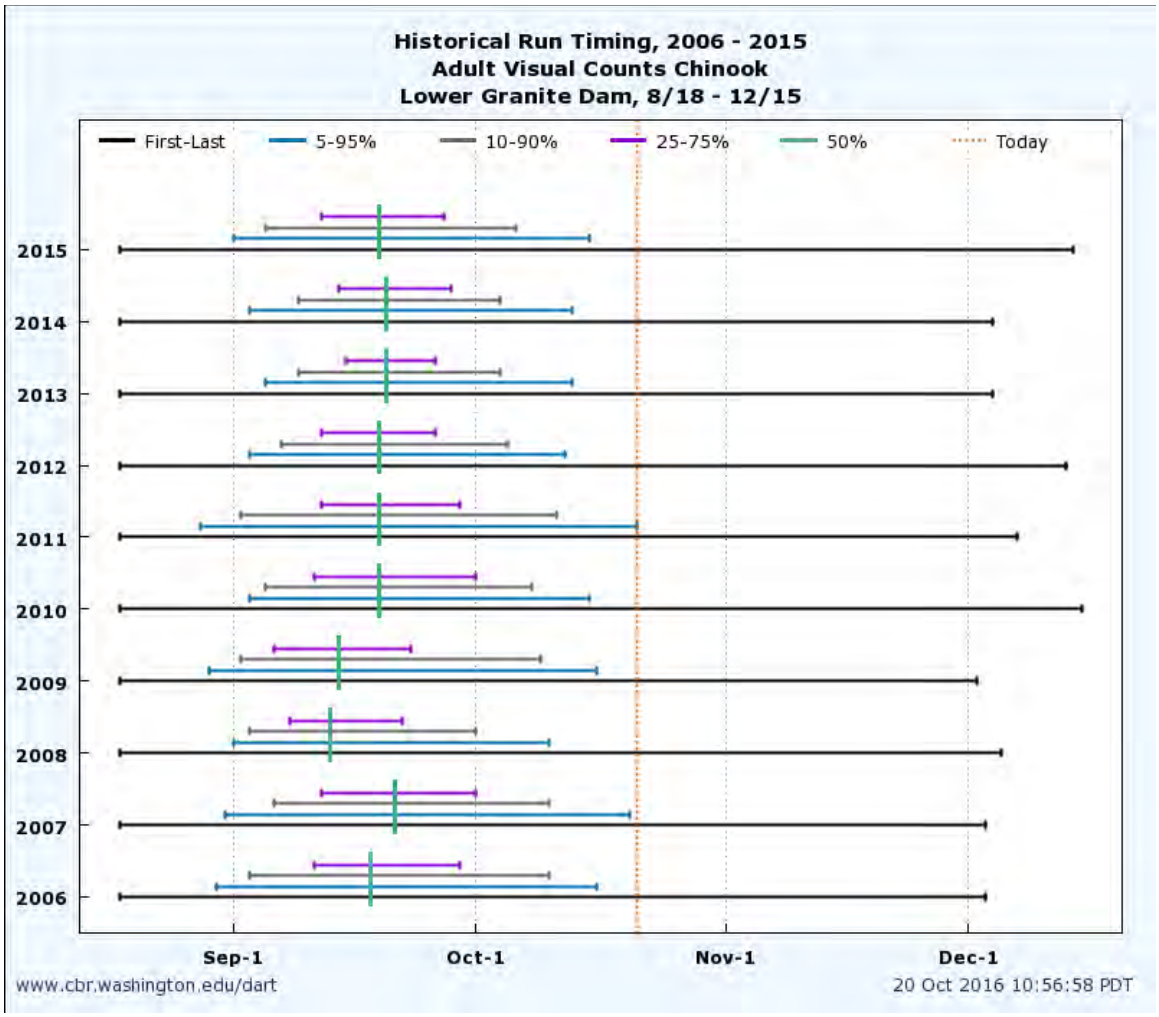

Figure 5-4: Average adult fall Chinook passage at Lower Granite Dam, 2006 - 2015 (source: DART).

Dam passage data, obtained through the University of Washington's Columbia Basin Research DART website, show that the subyearling Chinook are passing through Lower Granite Reservoir from before March 26 (in 2011 - 2015) through November 1 (in 2006, 2008, 2011), respectively (Table 5-3 and Figure 5-5). Most out-migrating subyearling wild fall Chinook passed over the Lower Granite Dam in June and July in sampled years 2006 to 2015. During this time period, total numbers of fish for each year ranged from approximately 338,000 (2007) to 1,177,374 (2011). The timeframe for the majority of wild fall Chinook out-migration is relatively narrow (during June, July, and August over the monitored years) and the fraction of the total population outmigrating during a given week is relatively constant from one year to another.

Table 5-3. Dates of sub-yearling wild fall Chinook passage at Lower Granite Dam 2006 - 2015 (source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2006	Average	3/31	5/20	6/5	7/8	11/1
2007	Average	3/30	6/1	6/10	7/28	10/31

Table 5-3. Dates of sub-yearling wild fall Chinook passage at Lower Granite Dam 2006 - 2015 (source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2008	Average	4/3	5/23	6/16	8/9	11/1
2009	Average	4/2	5/26	6/9	7/11	10/31
2010	Average	3/27	5/31	6/8	7/26	10/31
2011	High	3/26	5/20	6/10	7/24	11/1
2012	Average	3/26	5/23	6/14	7/22	10/31
2013	Average	3/26	5/24	6/9	9/3	10/31
2014	Average	3/26	5/26	6/11	8/6	10/31
2015	Low	3/26	5/25	6/7	8/3	10/31

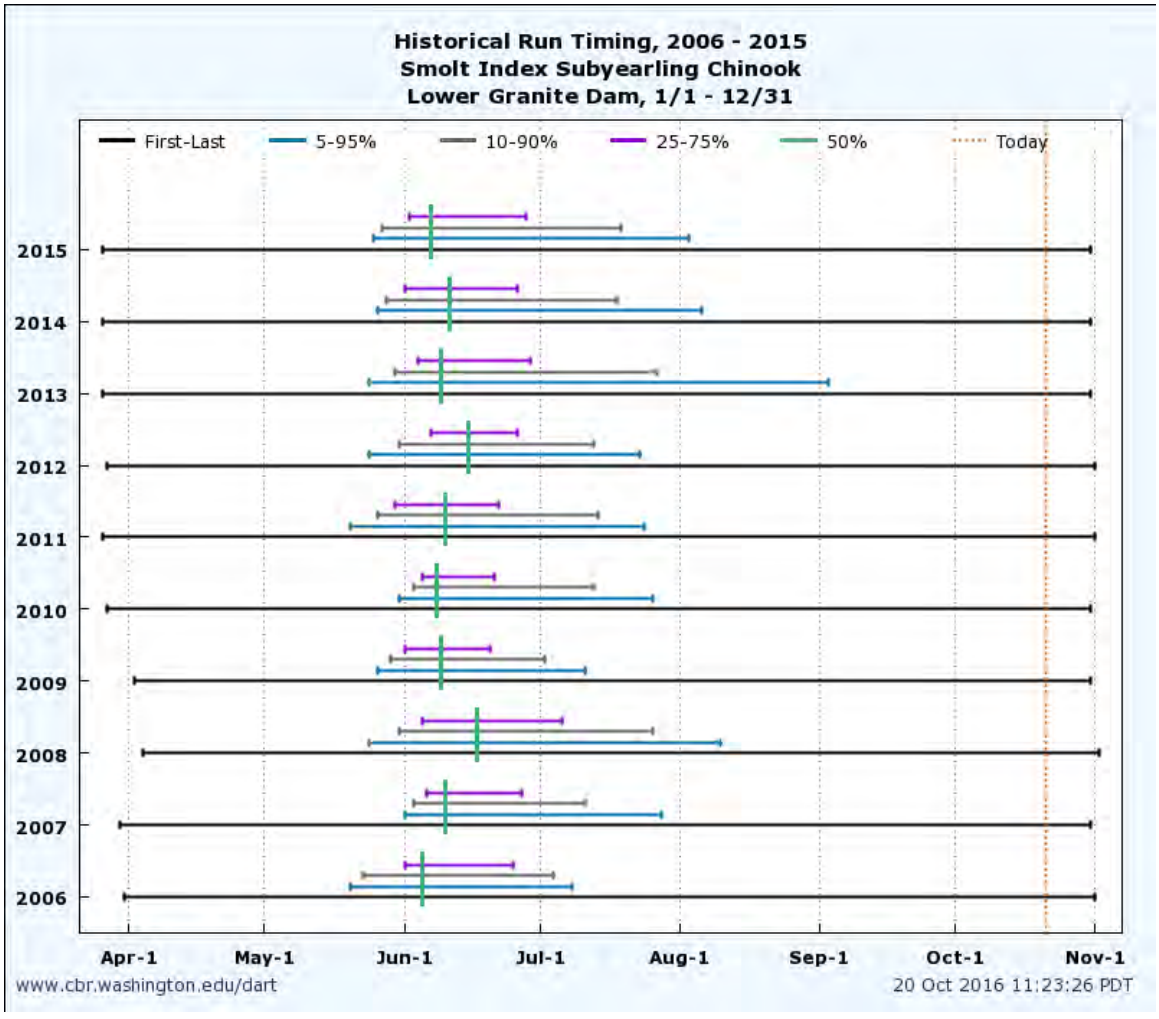

Figure 5-5: Average sub-yearling wild fall Chinook passage at Lower Granite Dam 2006 - 2015 (source: DART)

Currently, hatchery-reared fall Chinook make up most of the juvenile fall Chinook population in the Snake River. Downstream migrating subyearling hatchery fall Chinook

passed over the Lower Granite Dam primarily between 2 May and 22 November in sampled years 2013 and 2010 (Table 5-4). The periods of migration through the Snake River for hatchery-reared fall Chinook were not always consistent with those of the wild population. This may be attributable to the timing of their release from the hatcheries, or other factors such as hatchling survival, predation, or passage mortality. Total numbers of hatchery fall Chinook out migrating in each year ranged from approximately 3000 (2013) to 58,000 (2012).

Table 5-4. Subyearling hatchery fall Chinook passed over the Lower Granite Dam, 2006 - 2015 (source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2006	Average	3/25	4/16	6/4	11/11	12/16
2007	Average	3/27	4/20	5/13	6/16	10/28
2008	Average	3/27	4/20	6/2	10/20	12/13
2009	Average	1/1	4/17	5/28	11/11	12/5
2010	Average	3/25	4/23	6/5	11/22	12/16
2011	High	3/23	4/8	5/30	8/11	12/15
2012	Average	3/22	4/4	6/3	11/7	12/20
2013	Average	3/18	4/7	5/2	6/10	8/8
2014	Average	4/5	4/15	6/2	7/8	11/11
2015	Low	4/3	4/25	5/29	6/13	8/31

5.3.6 Travel Time

Keefer et al. (2002) investigated adult passage efficiency and travel time of fall Chinook at eight main-stem dams and reservoirs in the lower Columbia and Snake rivers, all major tributaries between Bonneville and Priest Rapids Dams on the Columbia River, and the Snake River and its tributaries upstream to Hells Canyon Dam during the fall (August-October) over a period of three years. Median values reported for the three-year duration ranged from 19 km/day to 31 km/day, with a mean of 27.2 km/day. Keefer et al. (unpublished manuscript) also studied fall Chinook migration speed in Columbia and Snake River reservoirs (Bonneville, Dalles, John Day, McNary to Ice harbor, McNary to Hanford receiver, Ice Harbor, Lower Monumental, Little Goose, Lower Granite to Snake River receiver, and Lower Granite to Columbia River receiver) over the same three-year duration. Median values reported for the three-year duration ranged from 8 km/day to 71 km/day, with a mean of 49.6 km/day.

Skalski et al. (1996) measured juvenile fall Chinook migration speed during both moderate and low river flows in the Columbia River, downstream of its confluence with the Snake River. At free flowing and impounded stretches, where flow rates were approximately 8500 m^3/s, migration speeds were 40 km/day to 55 km/day. At lower flows, approximately 4250 m^3/s, migration speeds were 24 km/day to 27 km/day.

For both juvenile and adult fall Chinook, a range of mean migrations speeds of approximately 25 to 50 km/day has been observed. This distance from the confluence of

the Snake and Clearwater rivers and the Lower Granite Dam is approximately 31 miles, or 50 km. Given the mean migration speeds observed in the literature, fall Chinook may require one to two days to travel between the confluence of the Snake and Clearwater rivers and Lower Granite Dam.

Travel times from the free-flowing section of the Snake River through the Lower Granite Dam were calculated for subyearling fall Chinook using pit-tagged hatchery fish (Smith et al. 2003). In this study, juveniles reared at Lyons Ferry Hatchery were released upstream at two sites on the Snake: Pittsburg Landing (173km above dam) and Bill Creek (92 km above dam). A total of 52,813 fish were tracked 1995-2000. Subyearlings were detected at the Lower Granite Dam from mid to late May through the end of October when the detection system was turned off. The average travel time from the flowing portion of the river to the dam was 43.5 days. According to Connor et al. (2003), wild fall Chinook juveniles spend a significant portion of this time period rearing (feeding and growing) or dispersing passively downstream rather than actively migrating downstream.

Travel times of wild Chinook juveniles were estimated from the PTAGIS database by NOAA (2003). Juveniles trapped and tagged at a Snake River and a Clearwater River trap from 1990-2003 were detected at the Lower Granite Dam allowing for estimates of travel time. This analysis has three caveats: 1) length data were available but no distinction between spring/summer and fall run fish was possible, 2) fish samples were collected from the surface, where juveniles that are actively migrating are most likely to be oriented in the water column. This may bias the sample away from the portions of the cohort that may be feeding/rearing as they progress downstream, 3) data were collected during the peak migration, again focusing the study on one portion of the entire cohort.

The following travel times were estimated in days for subyearling Chinook juveniles (<91mm):

Table 5-5: Travel Times for Subyearling Chinook Juveniles

Snake River Trap (n=287)		Clearwater River Trap (n=260)	
Mean	23.9	Mean	25.3
Median	19.0	Median	21.4
99.5 percentile	99.6	99.5 percentile	77.5

5.4 Snake River Sockeye salmon (Oncorhynchus nerka) – Endangered

5.4.1 Status and Description

Snake River sockeye salmon were listed as an endangered species in December 1991 (NMFS, 1991). This species was once found in the many lakes of the Payette, Salmon, and Wallowa River systems of Idaho and Oregon (USACE, 2002). Numbers have declined precipitously over the past century and the species was reduced to a remnant population close to extinction by the late 1980s and early 1990s. Current Snake River sockeye production is limited to Redfish Lake in the Salmon River Basin in Idaho. In 1990, no adult sockeye salmon returned to Redfish Lake (USACE, 1995). Since then, only small numbers of fish have successfully migrated back to Redfish Lake.

Critical Habitat for Snake River sockeye salmon was designated by the NMFS on December 28, 1993 (NMFS, 1993). In the Snake River, this includes spawning and juvenile rearing areas, juvenile downstream migration corridors, areas for maturation and development to adulthood, and adult upstream migration corridors (NMFS, 1993). The designated habitat for Snake River sockeye salmon consists of river reaches of the Columbia, Snake, and Salmon rivers, Alturas Lake Creek, Valley Creek and Stanley, Redfish, Yellow Belly, Pettit, and Alturas lakes. The Snake River upstream to its confluence with the Salmon River, which encompasses the Action Area, is within the areas designated as Critical Habitat. Critical Habitat includes the substrate and water of the waterways, and the adjacent riparian zone (defined as the area within 300 ft of the normal high-water line of a stream channel or 300 ft from the shoreline of a standing body of water) (NMFS, 1993).

Since 1991, a captive broodstock program has been part of the Snake River sockeye recovery strategy (USACE, 2002). The short-term objective of the program is to prevent the extinction of the species, with the longer-term goal of accelerating the re-establishment of sockeye runs to waters of the Stanley Basin. This program cultures progeny that supplement the wild population (USACE, 2002). In 1998, approximately 160,000 sub-yearling parr (presmolts) and smolts were released to lakes in the Stanley Basin.

5.4.2 Life History

Most adult Snake River sockeye enter the Columbia River in June and July and pass over Lower Granite Dam from June 25 to August 30 (USACE, 2002). A spring water temperature of 15.5°C has been used as an index for correlation with sockeye runs in the Columbia River (Quinn and Adams, 1996). Adults, which do not feed during their upstream migration (NMFS, 1991), return to Redfish Lake via the lower Snake River and Salmon River from mid-July through August, with the return peaking in August. They remain in the Lake to spawn in September and October.

Sockeye eggs are deposited in redds of fine gravel constructed along the lake's gravel beaches. Flowing water with dissolved oxygen at or near saturation (5.0 mg/L, depending on temperature) and cool temperatures ranging from 4 to 14 °C (Foerster, 1968; Ricker, 1976; Reiser and Bjornn, 1979) are needed for good egg survival. Eggs incubate for a period of 80 to 140 days and hatch in the spring. Upon hatching, fry remain in the gravel for three to five weeks, emerging in April and May. They immediately move to the deeper portion of the lake where, as visual predators, they feed on plankton and insect larvae for a year or more before migrating toward the ocean (NOAA, 1997). In Redfish Lake, most juveniles outmigrate as yearlings, rarely remaining for more than two years after emergence (NMFS, 1991).

Juveniles migrate from Redfish Lake primarily in late April and May, concurrent with the start of peak river flows and warmer water temperatures (38 to 50 °F) (Bjornn et al., 1968 as cited in NMFS, 1991). Juvenile migration corridors must have adequate substrate, water quality and quantity, temperature, velocity, cover, food, riparian vegetation, space, and safe passage conditions. In recent years, most outmigrants have passed Lower Granite Dam (first dam on the Snake River downstream from the Salmon River) from mid-May through mid-July (USACE, 2002). Most of the wild juvenile sockeye pass

Lower Granite Dam from March through early September, with outmigration lasting into November (USACE, 2002).

In the ocean, sockeye salmon feed on copepods, amphipods, larvae of crustaceans and fish, and squid (NOAA, 1997). Snake River sockeye generally spend two years in the ocean before returning to the Snake River and Redfish Lake in their fourth or fifth year of life. The survival rate for wild Snake River sockeye, from the time they migrate from Redfish Lake as smolts until they return as adults, averaged 0.07% from 1991 through 1996 (USACE, 2002).

5.4.3 Habitat Concerns

Factors influencing the decline of sockeye salmon include the destruction, modification, or curtailment of its habitat or range. According to NMFS (1991), the eight hydroelectric projects between upriver rearing areas and the ocean adversely affect sockeye salmon including, barriers to movement and habitat change where reservoirs have replaced free flowing river making fish susceptible to predation, disease, and elevated temperatures. Snake River sockeye smolts are relatively small with comparatively limited energy reserves. Delay or stress during migration can result in increased mortality in Snake River sockeye smolts. Other contributors to habitat loss and modification are water diversions, timber harvest, agriculture, mining, and urbanization. Over-fishing of the species for commercial, recreational, scientific or educational purposes is also a contributing factor. Finally, factors such as predation, introduction of non-native species, and habitat loss or impairment increase stress on any surviving individuals and thus increases potential susceptibility to diseases. Critical habitat has been noted by the USFWS as depicted in Figure 5-6.

5.4.4 Presence in Action Area

The Action Area is within the migration corridor used by Snake River Sockeye salmon adult and smolt life history forms. Returning adult sockeye salmon migrate upstream through this section of the Snake River from May to September and smolts migrate downstream through the area from March through mid-November. The Action Area encompasses the confluence of the Snake and Clearwater rivers where the cold waters of the Clearwater flow into the warmer waters of the Snake River. This area may be an attractive thermal refuge to migrating fish as smolts and adult salmon often “dip-in” to non-natal rivers to rest or seek cold-water refuge (NMFS, 2003). Some of these migrating fish may remain a few hours or days en route.

Orientation within the water column is not specifically known for adult sockeye. However, hydroacoustic surveys (USACE 1991) found larger fish are typically oriented near the bottom in the Lower Granite Reservoir. Hydroacoustic surveys conducted in May and June found outmigrating juveniles were located throughout the water column with the greatest concentration in the upper 15 meters.

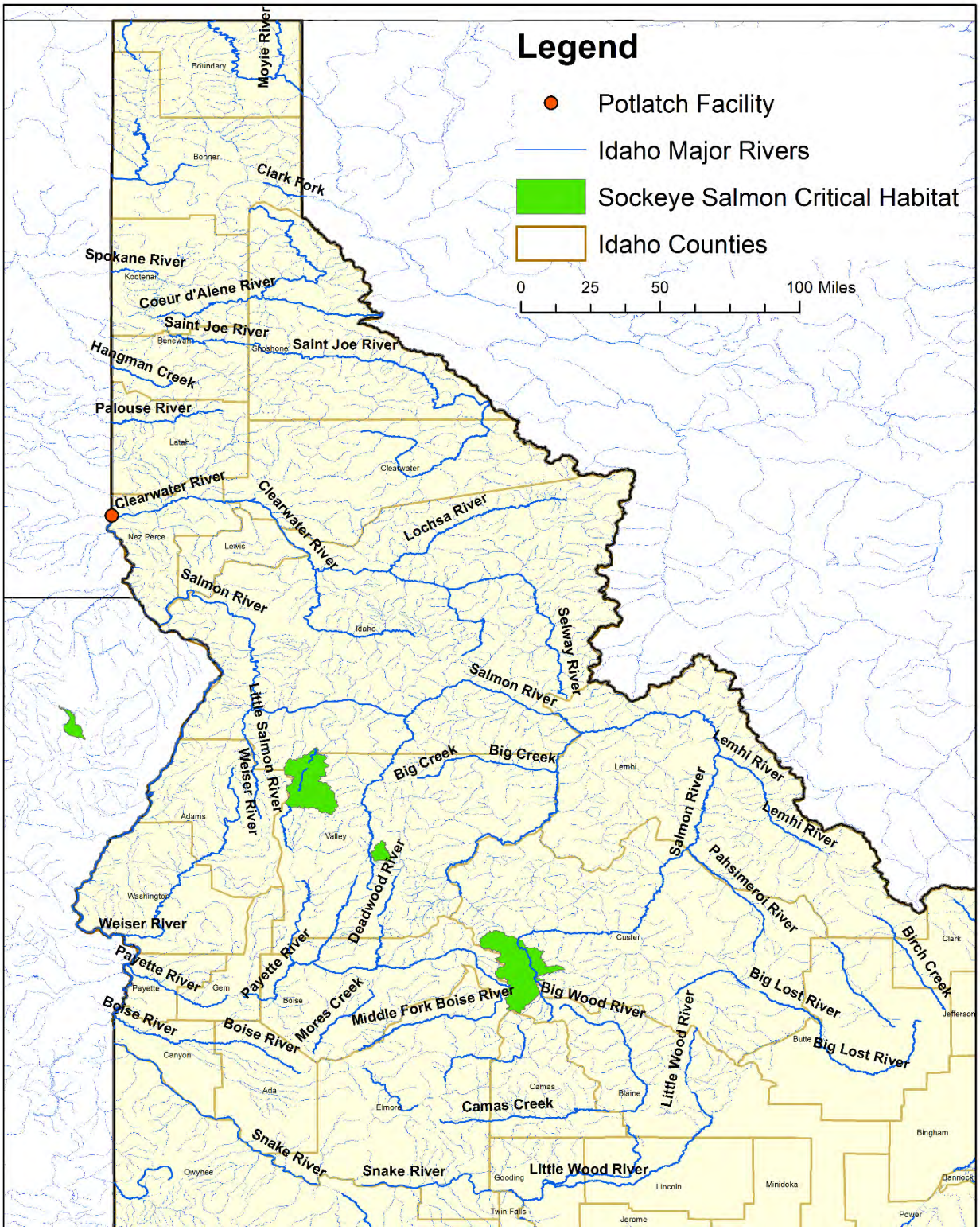

Figure 5-6: Sockeye salmon critical habitat.

5.4.5 Abundance and Timing Data

Based on travel time estimates between Columbia River dams developed by Quinn et al. (1997), it is estimated that sockeye would take less than two days to travel from lower Granite Dam to the confluence of the Snake and Clearwater rivers (based on the slowest calculated travel rate of 16.9 mi/day). Therefore, the fish data collected at the dam can be used to estimate run timing as well as fish abundance in the Action Area.

Dam passage data were obtained through University of Washington's Columbia Basin Research DART website. Sockeye passage data at the Lower Granite Dam are summarized for 2006 to 2015 based on daily count data. Summarized DART data are in Appendix E. This time period includes a low flow year (2015), an average flow year (2014), and a high flow year (2011).

Very few adult sockeye were observed passing Lower Granite Dam during the 2006 through 2015 time period with the run size ranging from 1 to 339 during the migration period (Figure 5-7). First passage of adult sockeye across the dam into the Lower Granite Reservoir downstream of the mouth of the Clearwater River ranged from June 22, 2009, to June 30, 2008. As shown in Table 5-6, the date of last adult passage for the Lower Granite Dam ranged from July 31, 2010, to October 21, 2014.

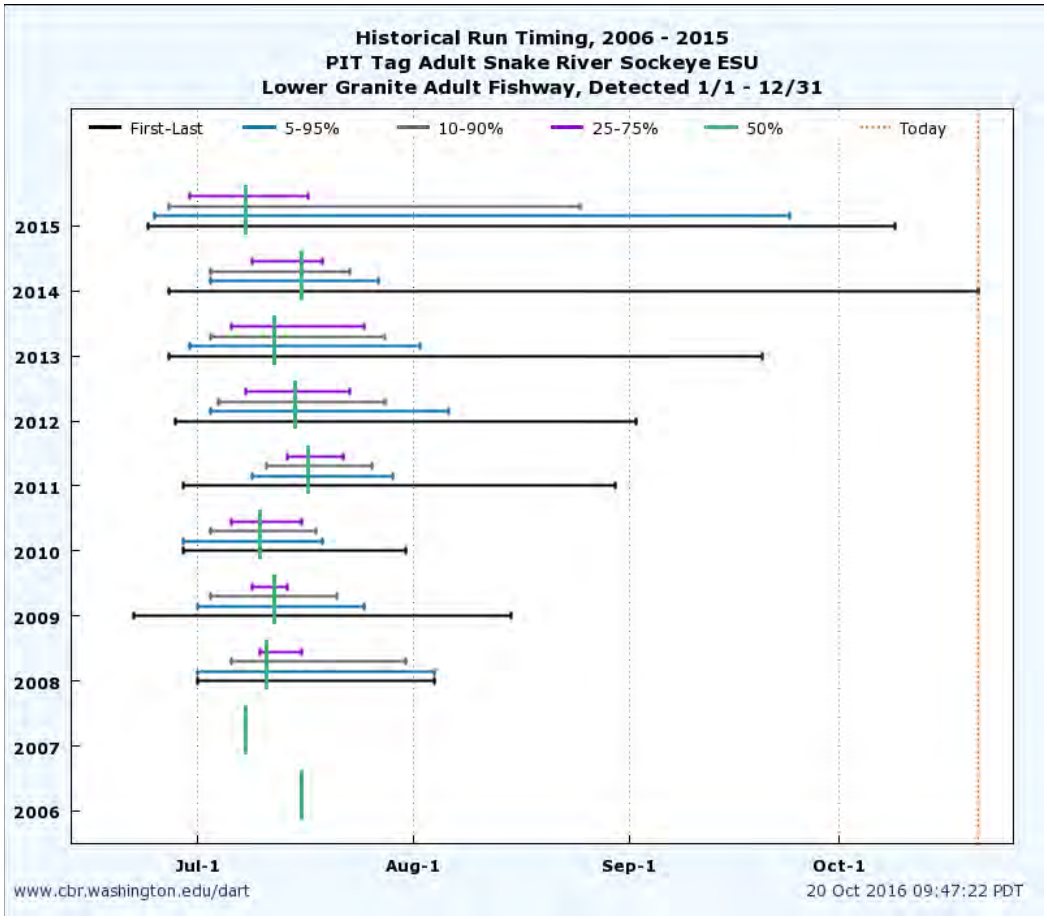

Figure 5-7: Average adult sockeye passage at Lower Granite Adult Fishway, 2006 - 2015 (Source: DART).

Table 5-6. Dates of adult sockeye passage at Lower Granite Adult Fishway, 2008 - 2016 (Source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2008	Average	6/30	6/30	7/10	8/3	8/3
2009	Average	6/22	7/1	7/12	7/25	8/15
2010	Average	6/29	6/29	7/10	7/19	7/31
2011	High	6/29	7/9	7/17	7/29	8/30
2012	Average	6/27	7/2	7/14	8/5	9/1
2013	Average	6/27	6/30	7/12	8/2	9/20
2014	Average	6/27	7/3	7/16	7/27	10/21
2015	Low	6/24	6/25	7/8	9/24	10/9

Downstream migrating juvenile wild sockeye passed over the Lower Granite Dam primarily between April and early June from 2006 to 2015, with a small proportion of late outmigrants passing the dam into November (Table 5-7). The dates of first juvenile sockeye passage through Lower Granite Dam ranged from April 25, 2012, through May 16, 2010. The dates of last juvenile sockeye passage through Lower Granite Dam ranged from July 3, 2014, through November 30, 2009. Total annual numbers of juvenile sockeye passing the Lower Granite Dam during the time period ranged from approximately 300 (1998) to 11,700 (2011) (Figure 5-8, Appendix E).

Table 5-7. Dates of juvenile sockeye passage at Lower Granite Dam, 2006 - 2016 (Source: DART).

Year	Flow	First	5th percentile	50th percentile	95th percentile	Last
2006	Average	5/6	5/10	5/17	6//1	11/15
2007	Average	4/30	5/5	5/16	5/27	10/20
2008	Average	5/13	5/17	5/24	6/14	7/4
2009	Average	5/8	5/15	5/19	5/28	11/30
2010	Average	5/16	5/17	5/22	6/2	6/22
2011	High	5/13	5/19	5/23	6/2	7/28
2012	Average	4/25	5/17	5/20	5/31	8/1
2013	Average	5/4	5/14	5/16	5/19	7/23
2014	Average	5/8	5/16	5/17	5/23	7/3
2015	Low	4/28	5/9	5/16	5/19	5/31

Juvenile hatchery sockeye, from the captive broodstock program have been counted at Lower Granite Dam and Lower Monumental Dam since 1995. Dates of migration through the Snake River for hatchery-reared sockeye were comparable to those for wild sockeye.

5.4.6 Travel Time

Although only two studies of sockeye migration speed were identified, the findings of the two studies are similar. Bjornn et al. (2000) studied adult sockeye migration speed, with speeds ranged from 29 km/day to 43.8 km/day, with a mean of 25.6 km/day. Discover the Outdoors8 reported a mean migration speed of 20.9 km/day for adult sockeye migrating upstream in the Columbia, Snake, Fraser, Nass, Stuart, and Skeena rivers.

A range of mean migrations speeds for adult sockeye of approximately 21 to 26 km/day has been observed. This distance from the confluence of the Snake and Clearwater rivers and the Lower Granite Dam is approximately 31 miles, or 50 km. Given the mean migration speeds observed in the literature, sockeye may require two to two and one-half days to travel between the confluence of the Snake and Clearwater Rivers and the Lower Granite Dam. Data on travel times for juvenile sockeye were not found.

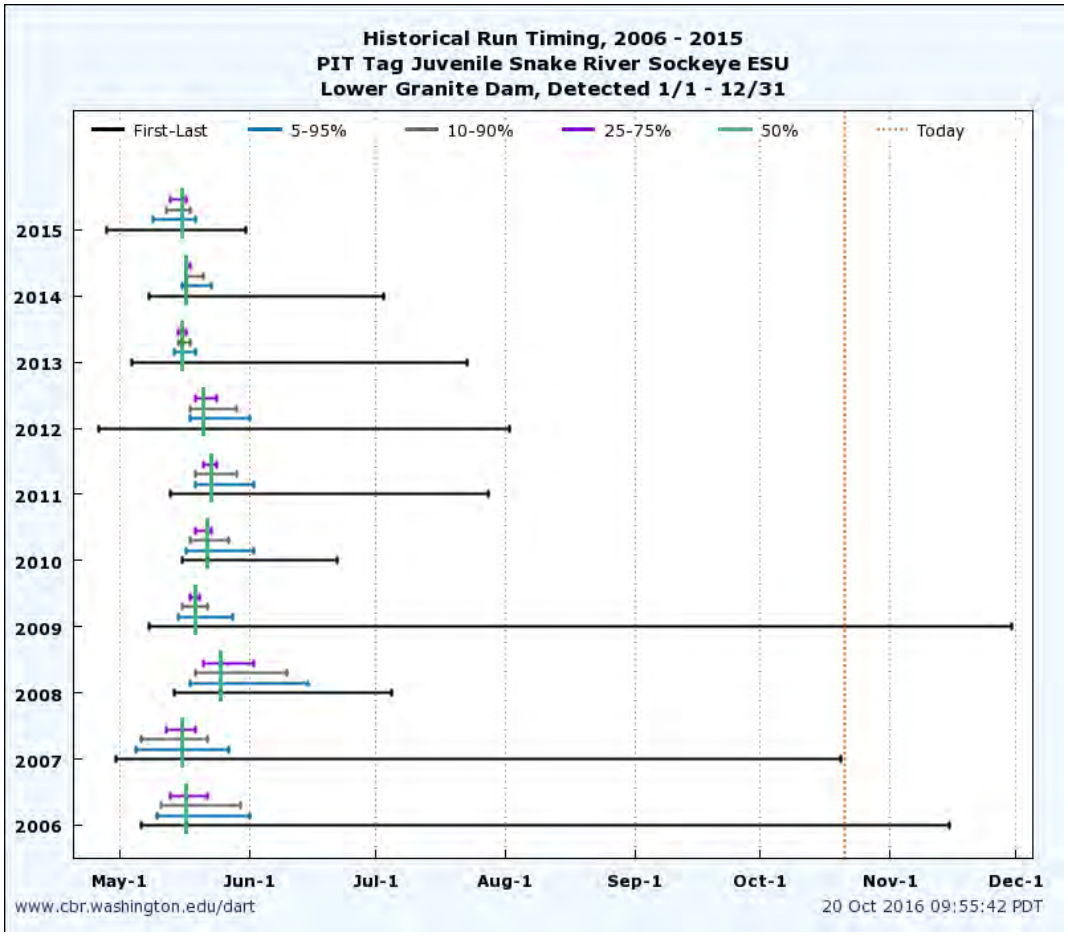

Figure 5-8: Average juvenile sockeye passage at Lower Granite Dam, 1995-1999 (Source: DART).

[8] http://www.dto.com/fwfishing/speciesProfile/356

5.5 Snake River spring/summer Chinook salmon (Oncorhynchus tshawytscha) – Threatened

5.5.1 Status and Description

The Chinook salmon is the largest of the Pacific salmon species (NMFS, 1997a). In the Snake River Basin, there are spring, summer, and fall Chinook runs. As previously discussed, spring and summer Chinook are grouped into one ESU and are addressed separately from fall Chinook. The spring/summer Chinook in the Snake River were listed as threatened on April 22, 1992 (NMFS, 2000a). Spring Chinook salmon of the Clearwater River were exempt from the listing because of uncertainty associated with the genetic integrity of this stock. Allegedly construction of the Lewiston Dam in the early 1900s eliminated all runs of native spring Chinook salmon into the Clearwater basin and those currently found in the basin are exclusively of hatchery origin. Critical Habitat was designated on December 28, 1993 (NMFS, 1998) and updated on February 16, 2000 (NMFS, 2000b). Critical Habitat consists of river reaches of the Columbia River in Oregon and Washington, the Snake and Salmon rivers, and all tributaries of the Snake and Salmon rivers (except the Clearwater River) presently or historically accessible to Snake River spring/summer Chinook salmon. This excludes reaches above impassable natural falls and Hells Canyon Dam on the Snake River. Spring/summer Chinook in the Snake River Basin are stream-type outmigrants, which means they spend more time in freshwater (primarily headwater streams) than ocean-type fish and are therefore more dependent on freshwater stream ecosystems.

Historically, the Snake River was estimated to produce approximately 39 percent of the total spring Chinook and 45 percent of the total summer Chinook salmon in the Columbia River Basin (Mallett, 1974). Spring/summer Chinook spawned in practically all the accessible and suitable habitat in the Snake River upstream from its confluence with the Columbia River (Matthews and Waples, 1991). Since the 1960s, spring/summer Chinook counts at Snake River dams have declined considerably. However, in the last three years (2013-2015), the combined return of hatchery and wild spring/summer Chinook was 263,969. In 2015, the passage of 122,658 adult spring/summer Chinook past Lower Granite Dam was more than 60% greater than the 10-year average of 76,661. The present range and rearing habitat for naturally spawned Snake River spring/summer Chinook salmon is primarily limited to the Salmon, Grande Ronde, Imnaha, and Tucannon River sub-basins with limited spawning in the lower Clearwater River subbasin.

5.5.2 Life History

Adult spring and summer Chinook enter the Columbia River during the months of March through May, and May through July, respectively (USACE, 1995). Migration past the Lower Granite Dam on the Snake River generally occurs over the period from April through August. Migrating adults may pass through water with dissolved oxygen levels as low as 3.5 to 4.0 mg/L (Fujioka, 1970; Alabaster, 1988 and 1989). Spawning for each stock generally occurs from August through October, peaking in the Snake River system in September. Spring Chinook occupying higher elevation reaches of the river basin tend to spawn earlier than summer Chinook positioned further downstream.

Eggs are deposited in redds built in gravel beds. The eggs remain in the gravel for 90 to 150 days until they hatch, usually by December. Stream flow, gravel quality, and silt load all significantly influence the survival of developing eggs. Temperatures for optimal Chinook salmon egg incubation have been reported to be between 5.0 and 14.4 °C (Bell, 1984). An upper lethal limit of 25.1 °C has been reported by Brett (1952) but may be lower depending on other water quality factors (Ebel et al., 1971). Alevins remain in the gravels for two to three weeks while the yolk is absorbed (Scott and Crossman, 1973) before emerging from the gravels. Timing of emergence varies by basin. Typically, spring Chinook fry emerge from November to April in the Salmon River Basin (USACE, 1999). Emergence occurs from February to April in the Clearwater Basin. Summer Chinook fry emergence occurs from late March through mid-June (USACE, 1999). Optimal survival of fry occurs with water temperatures ranging from 12 to 14°C, and Scott and Crossman (1973) reported an upper lethal temperature for fry of 25.1°C. The fry begin migrating downstream into larger rivers and streams where they will grow and forage for approximately one year. The optimal growth temperature for Chinook salmon is approximately 15°C (Beer, 1999).

The following spring, yearling spring/summer Chinook salmon begin their outmigration toward the ocean between March and July, with spring run fish outmigrating a few weeks earlier than the summer run fish. Because they spend nearly a year in fresh water, these smolts are quite large (approximately 10 to 15 inches in length) when they migrate to the ocean. This enables them to move offshore fairly quickly and undertake extensive offshore migrations (Healey, 1983 and 1991; Myers et al., 1984). They will spend from one to six years (typically two to four) growing and feeding. Upon reaching sexual maturity, they migrate back to their natal streams. Exceptions to this are some yearling males (jack salmon), which mature almost completely in freshwater and only spend a few months to a year in the ocean before returning to spawn (Myers et al., 1998).

5.5.3 Habitat Concerns

Factors influencing the decline of spring/summer Chinook are similar to those affecting fall Chinook include the destruction, modification, or curtailment of its habitat or range. Contributors to habitat loss and modification are water diversions, timber harvest, agriculture, mining, and urbanization (NMFS, 1998). Excessive silt loads have been reported to halt Chinook salmon movements or migrations (Reiser and Bjorn, 1979). Over-fishing of the species for commercial, recreational, scientific or educational purposes is also a contributing factor. Hydroelectric development on the main-stem Columbia and Snake rivers continues to affect juvenile and adult migration (NMFS, 1998) including blocking of access to spawning habitat, modification of flowing habitat by inundation, increased predation from warm-water fishes, delayed migration, and mortality during passage past dams and through turbines.

Factors such as predation, introduction of non-native species, and habitat loss or impairment increase stress on any surviving individuals and thus increases potential susceptibility to diseases. Spring/summer Chinook use tributaries for spawning thus their spawning habitat is vulnerable to degradation due to sedimentation from logging activities (NMFS, 1997a). Introduced and/or artificially propagated fish can affect the indigenous stocks through competition between hatchery and native stocks, interbreeding

between hatchery and native Chinook salmon stocks, and disease introductions by artificially propagated fish (NMFS, 1997a). Critical habitat has been noted by the USFWS as depicted in Figure 5-6.

5.5.4 Presence in Action Area

The Action Area is within the spring/summer Chinook salmon migration corridor used by adult and smolt life history forms. Returning adult Chinook salmon migrate upstream through this section of the Snake River from March through August and smolts migrate downstream through the area primarily from April through June. The Action Area has a rather unique habitat feature in the occurrence of the confluence of the Snake and Clearwater rivers. This area is an attractive thermal refuge to migrating fish. Smolts and adult salmon often "dip-in" to non-natal rivers to rest or seek cold water refuge. Some of these fish may remain a few hours or days in-route, while others may attempt to hold for extended periods of time, such as weeks or months (NMFS, 2003). In the Action Area, the colder waters of the Clearwater River provide a refuge from the warmer waters of the Snake River migration route.

Orientation within the water column is not specifically known for adult spring/summer Chinook. However, hydroacoustic surveys (USACE 1991) found larger fish are typically oriented near the bottom in the Lower Granite Reservoir. Hydroacoustic surveys conducted in May and June found outmigrating juvenile salmonids were located throughout the water column with the greatest concentration in the upper 15 meters.

5.5.5 Abundance and Timing Data

Spring/summer Chinook passage data collected at the Lower Granite Dam for the years 2006 through 2015 are presented to describe abundance and passage near the Action Area (Appendix E). Most adult spring/summer Chinook migrate upstream across the dam into the Lower Granite Reservoir from mid-April to mid-July (Table 5-8). For the years 2006 through 2015, the date of early passage for the Lower Granite Dam was March 31, 2014 through May 8, 2006. The date of late passage was not determined from the Columbia River DART9 database, which obtains data from USACE (2002), because they counted Chinook salmon without distinguishing between spring, summer, or fall Chinook. However, the convention for separating spring-, summer- and fall-run fish is based on date of passage. At the Lower Granite Dam, the spring run is considered to occur from March 1 to June 17; the summer run is considered to occur from June 18 to August 17; and the fall run is considered to occur from August 18 to December 15. This convention has been used to allocate fish counts from the DART database into spring/summer- and fall-run Chinook. The migration timing data for adult spring/summer Chinook is summarized in (Figure 5 9). Migrating adult run sizes ranged from approximately 37,000 fish (2006) to 179,000 fish (2015) (Appendix E).

Downstream migrating yearling wild spring/summer Chinook passed over the Lower Granite Dam primarily between mid-April and end of May from 2006 to 2016 (Figure 5-10). Juveniles first passed through Lower Granite Reservoir from March 20, 2014, through March 31, 2010 (Table 5-9). The dates of last juvenile passage ranged from July 6, 2007, through December 12, 2012. The timing of this outmigration is relatively narrow.

[9] http://www.cbr.washington.edu/dart/

Table 5-8. Dates of adult spring/summer Chinook passage at Lower Granite Adult Fishway, 2006 - 2015 (Source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2006	Average	5/8	5/15	6/9	7/29	10/5
2007	Average	4/25	5/5	6/8	7/25	10/20
2008	Average	4/22	5/10	6/13	7/7	9/28
2009	Average	4/28	5/13	6/11	7/11	10/11
2010	Average	4/20	5/5	6/5	7/6	11/18
2011	High	4/15	5/11	6/24	7/27	10/25
2012	Average	5/4	5/18	6/1	7/16	9/26
2013	Average	4/23	5/8	6/5	7/5	8/10
2014	Average	3/31	5/7	6/2	7/4	11/6
2015	Low	4/3	4/29	5/21	7/28	9/29

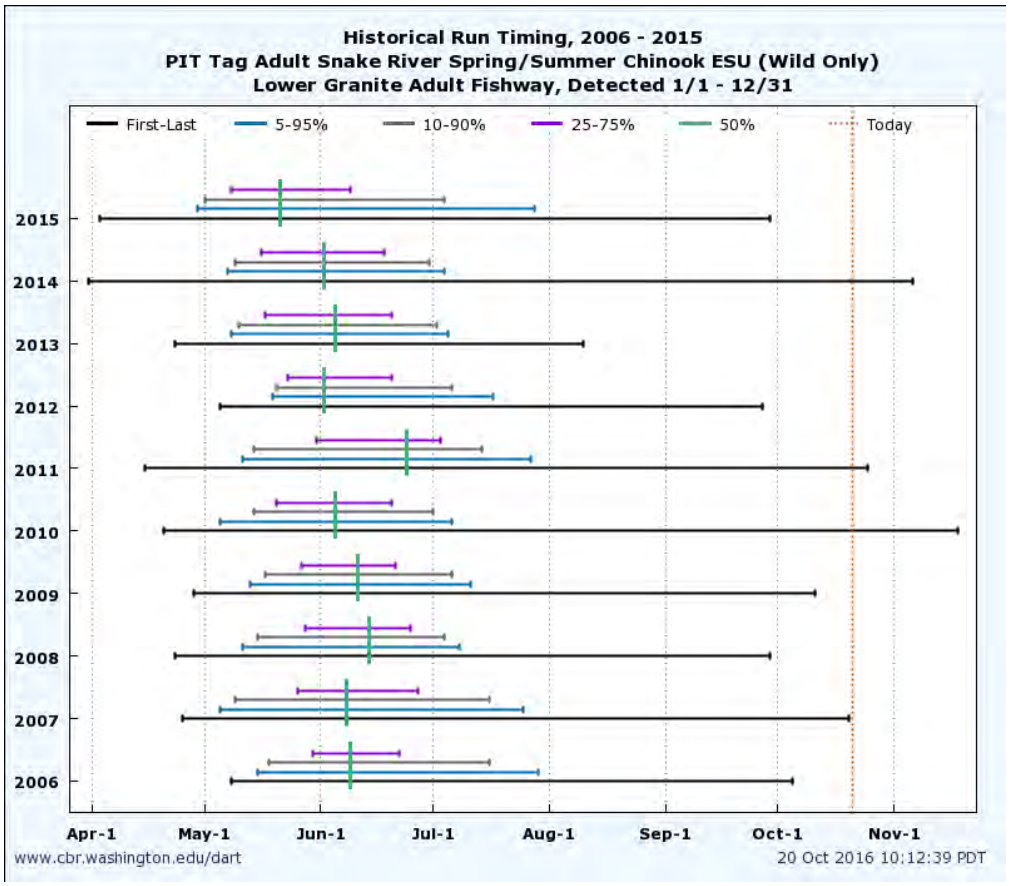

Figure 5-9: Average of adult spring/summer Chinook passage at Lower Granite Dam, 2006 - 2015 (Source: DART).

Although wild spring/summer Chinook are the focus of this analysis, hatchery-reared fish make up most of the juvenile spring/summer Chinook population in the Snake River.

Downstream migrating yearling hatchery spring/summer Chinook passed over the Lower Granite Dam primarily between mid-April and end of May from 2006 to 2015 (Figure 5-11). The dates of first passage ranged from March 28, 2012, to April 18, 2010 (Table 5-10). The dates of last passage ranged from June 9, 2015, to July 17, 2011. The periods of migration through the Lower Snake River for hatchery-reared Chinook were comparable to those for wild spring/summer Chinook.

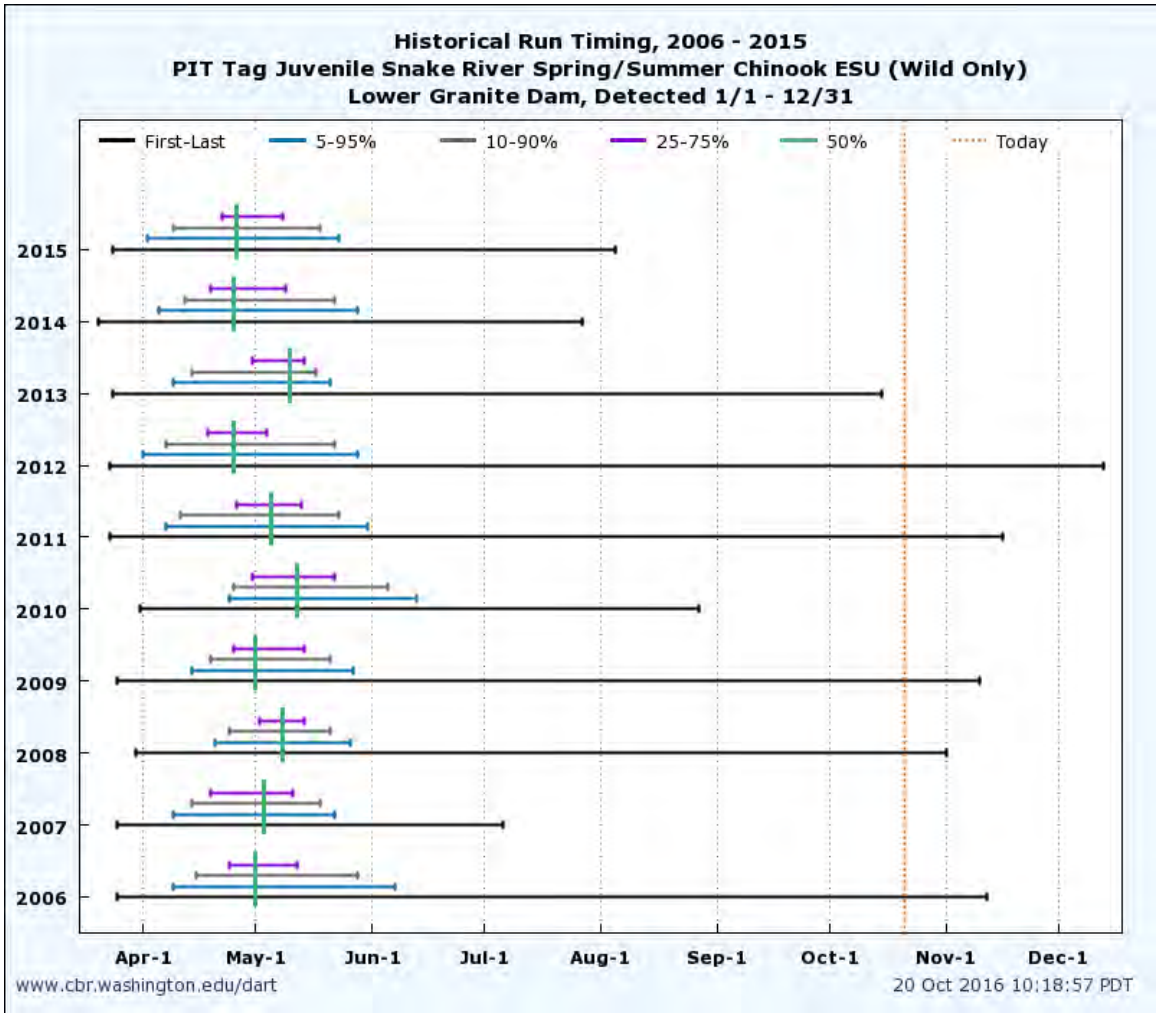

Figure 5-10: Average wild yearling spring/summer Chinook Passage at Lower Granite Dam, 2006 - 2015 (source: DART).

Table 5-9. Dates of wild yearling spring/summer Chinook passage at the Lower Granite Dam, 2006 - 2015 (Source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2006	Average	3/25	4/9	5/1	6/7	11/12
2007	Average	3/25	4/9	5/3	5/22	7/6
2008	Average	3/29	4/19	5/7	5/25	10/31
2009	Average	3/25	4/14	5/1	5/27	11/10
2010	Average	3/31	4/24	5/12	6/13	8/27
2011	High	3/23	4/7	5/5	5/31	11/16
2012	Average	3/22	3/31	4/24	5/27	12/12
2013	Average	3/24	4/9	5/10	5/27	11/10
2014	Average	3/20	4/5	4/25	5/28	7/27
2015	Low	3/24	4/2	4/26	5/23	8/5

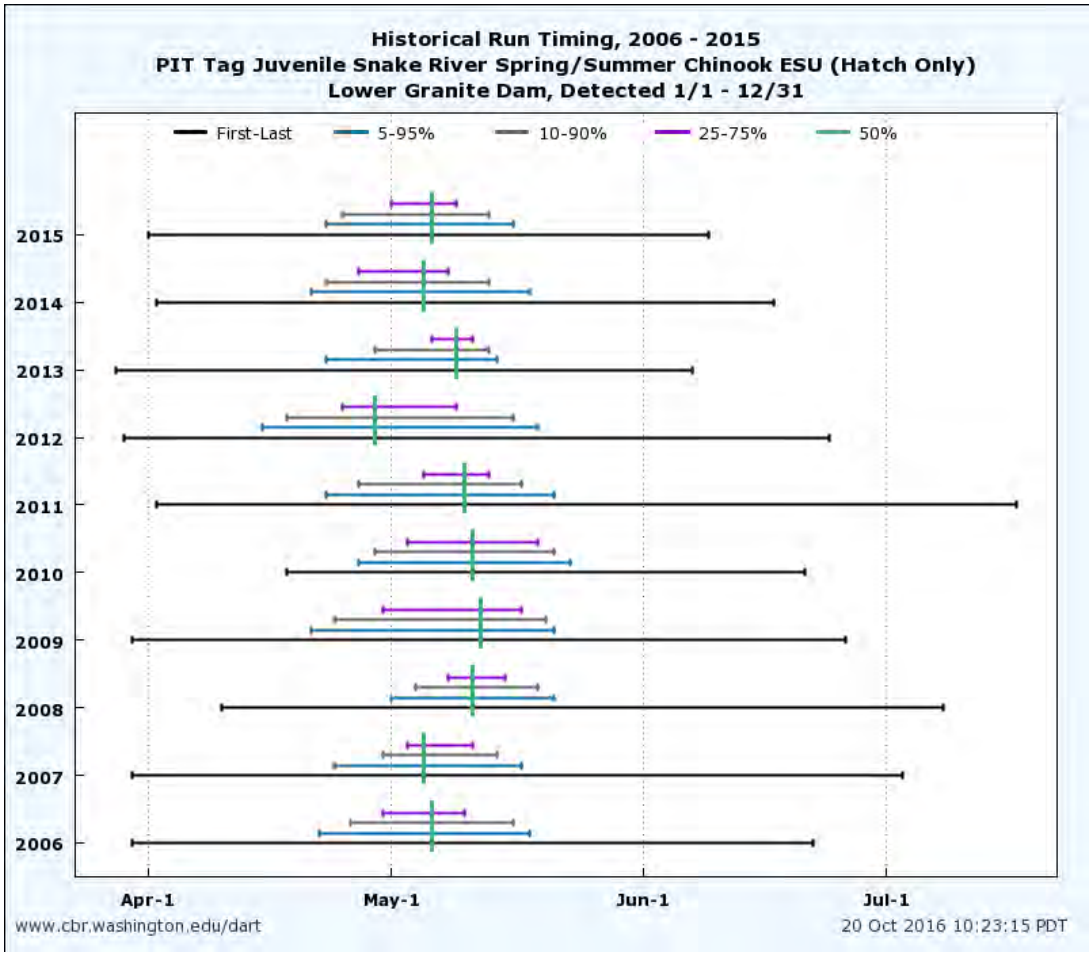

Figure 5-11: Average hatchery yearling spring/summer Chinook passage at Lower Granite Dam, 2006 - 2015 (source: DART).

Table 5-10. Dates of hatchery yearling spring/summer Chinook passage at Lower Granite Dam, 2006 - 2015 (source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2006	Average	3/30	4/22	5/6	5/18	6/22
2007	Average	3/30	4/24	5/5	5/17	7/3
2008	Average	4/9	4/30	5/10	5/20	7/7
2009	Average	3/30	4/21	5/12	5/21	6/26
2010	Average	4/18	4/27	5/11	5/23	6/21
2011	High	4/2	4/23	5/10	5/21	7/17
2012	Average	3/28	4/14	4/28	5/18	6/23
2013	Average	3/28	4/14	4/28	5/18	6/23
2014	Average	4/2	4/21	5/5	5/18	6/17
2015	Low	4/1	4/23	5/6	5/16	6/9

5.5.6 Travel Time

The DART database was used to estimate travel time for juvenile Chinook. Most recent annual data showed travel time for spring/summer Chinook during March through October. Migration speeds ranged from 2.1 km/day to 21.2 km/day, with a mean migration speed of 11.8 km/day. Wild and hatchery adult spring/summer Chinook salmon are monitored by DART as well, with 2003 mean velocities of 6.35 and 5.88 km/day, respectively. Keefer et al. (2003) measured migration speed of adult spring/summer Chinook at eight main-stem dams and reservoirs in the lower Columbia and Snake rivers, all major tributaries between Bonneville and Priest Rapids dams on the Columbia River and the Snake River and its tributaries upstream to Hells Canyon Dam during the spring (April-May) over a period of five years. Median values reported for the five-year duration ranged from 12 km/day to 38 km/day, with a mean of 25.7 km/day. Keefer et al. (2003) also studied adult spring/summer Chinook migration speed in Columbia and Snake River reservoirs (Bonneville, Dalles, John Day, McNary to Ice harbor, McNary to Hanford receiver, Ice Harbor, Lower Monumental, Little Goose, Lower Granite to Snake River receiver, and Lower Granite to Columbia River receiver) over the same five-year period. Median values reported for the five-year duration ranged from 16 km/day to 83 km/day, with a mean of 61.3 km/day. Bjornn et al. (2000) also studied adult spring/summer Chinook migration speed through pools. Migration speeds ranged from 43.2 km/day to 61.5 km/day, with a mean of 51.4 km/day.

A range of mean migrations for adult spring/summer Chinook of approximately 12 to 50 km/day has been observed. The distance from the Lower Granite Dam to the confluence of the Snake and Clearwater rivers is approximately 31 miles, or 50 km. Given the mean migration speeds observed in the literature, adult spring/summer Chinook may require one to four days to travel between the Lower Granite Dam the Clearwater and Snake river confluence.

As presented in the section discussion the spring/summer Chinook, travel times of wild Chinook juveniles were estimated from the PTAGIS database by NOAA (2003). Juveniles trapped and tagged at a Snake River and a Clearwater River trap from 1990-2003 were detected at the Lower Granite Dam allowing for estimates of travel time. This analysis has three caveats: 1) length data were available but no distinction between spring/summer and fall run fish was possible, 2) fish samples were collected from the surface, where juveniles that are actively migrating are most likely to be oriented in the water column. This may bias the sample away from the portions of the cohort that may be feeding/rearing as they progress downstream, 3) data were collected during the peak migration, again focusing the study on one portion of the entire cohort.

The following travel times were estimated in days for yearling Chinook juveniles (>90 mm):

Table 5-11: Travel Times for Yearling Chinook Juveniles

Snake River Trap (n=4770)		Clearwater River Trap (n=1045)	
Mean	7.3	Mean	13.8
Median	5.5	Median	11.9
99.5 percentile	29.8	99.5 percentile	52.8

5.6 Steelhead (Oncorhynchus mykiss) – Threatened

5.6.1 Status and Description

Steelhead exhibit a complex life cycle and may be either anadromous or a freshwater resident. The anadromous form, called steelhead, is unlike Pacific salmon species in that individuals can spawn multiple times before dying. The species supports an important recreational fishery throughout its range and is one of the top five sport fish in North America. The Snake River steelhead was listed as a threatened species on August 18, 1997 (NMFS, 1997b) for the spawning range upstream of the confluence of the Snake River with the Columbia River. Only the anadromous life forms of *O. mykiss* (steelhead) were listed. Critical Habitat was designated for the Snake River steelhead ESU on February 16, 2002 (50 CFR 226.212). This designation encompasses historically accessible reaches of all rivers and tributaries with this ESU's range (excludes areas above Hells Canyon Dam, Dworshak Dam, and Napias Falls on Napias Creek.

Steelhead inhabit the Snake River Basin streams of southeast Washington, northeast Oregon, and Idaho (USACE, 1995). This ESU comprise two groups, A-run and B-run, distinguished based on migration timing, ocean residence duration, and adult size (NMFS, 1997b). A-run steelhead predominately have a one-year ocean residence time and B-run fish have a mostly two-year ocean residency. A-run steelhead were historically present in all Snake River drainages while B-run fish were found only in the Clearwater River and Salmon River drainages (USACE, 1995). Currently, most steelhead in the Snake River are B-run fish. Because A-run and B-run steelhead are not clearly distinguishable above the Bonneville Dam, they are considered one Evolutionary Significant Unit (ESU) in the listing. The Snake River ESU includes wild fish as well as three hatchery populations: summer runs from the Dworshak National Fish Hatchery stock, Imnaha River stock, and Oxbow Hatchery stock (NMFS, 1997b).

In the Snake River Basin ESU, major tributaries include the Clearwater, Grande Ronde, Salmon, Selway, and Tucannon rivers. Of these, only the Tucannon River is located downstream of the Clearwater Paper Mill (RM 139) between RM 62 and RM 63 of the Snake River. The primary rivers within the Snake River Basin supporting Snake River steelhead include the Clearwater, Salmon, Tucannon, Imnaha, Grande Ronde, and Asotin rivers.

The Salmon River drainage contains primarily A-run steelhead, except for the South Fork Salmon and Middle Fork Salmon rivers, which contain primarily naturally producing B-run steelhead. The Clearwater River drainage also contains A-run fish, except for the Selway River drainage, which contains primarily naturally producing B-run fish (Rich and Petrosky, 1994; Busby et al., 1996).

Snake River Basin steelhead formerly inhabited most of the major tributaries and streams of the Snake River, limited only by natural barriers. Today, no naturally occurring steelhead are found above Hells Canyon Dam on the Snake River due to the lack of fish passage provisions at the dam. Similarly, the Dworshak Dam (located at RM 1.9) blocks steelhead passage on the North Fork Clearwater River. The basin supported large numbers of steelhead. NMFS and USFWS (1972) estimated that 114,000 steelhead were produced annually in the Snake River Basin from 1954 to 1967. Snake River Basin steelhead recently suffered severe declines in abundance relative to historical levels. The natural component for steelhead escapement above Lower Granite Dam was about 9,400 from 1990-1994. Low run sizes over the last 10 years are most pronounced for naturally produced (wild) steelhead. Based on surveys in the mid 1990's approximately, 86% of adult steelhead at the Lower Granite Dam are of hatchery origin (Busby et al., 1996).

5.6.2 Life History

Steelhead in the Snake River Basin (both A-run and B-run) exhibit summer run timing characterized by entering rivers sexually immature and spending several months maturing in fresh water before they spawn. Steelhead enter the Columbia River throughout the year. Upriver summer A-run steelhead enter the Columbia River from June to early August, while B-run steelhead migrate later, from August to October (USACE, 2002). However, spawning does not occur until late winter or the following spring, meaning that many adult steelhead may spend close to a year in fresh water. Thus, all adult steelhead that survive to spawn must over winter somewhere in the Columbia River system. Keefer et al. (2002) report that most (93.5%) steelhead were last recorded in their final tributaries or upstream from Lower Granite and Priest Rapids dams prior to the onset of winter and apparently over winter in those areas prior to spawning. During this time, they may be vulnerable to predation and disturbance. Cover such as logs, rocks, undercut banks, and vegetation is required to reduce disturbance and predation of pre-spawning and spawning steelhead. Steelhead over winter in cool, deep holding pools (Nickelson et al., 1992).

Spawning in the Snake River Basin occurs from March through May, with A-run steelhead spawning a few weeks earlier and at lower elevations than B-run steelhead. Snake River steelhead spawn at higher elevations (up to 2,000 m) and migrate farther from the ocean (up to 1,500 km) than any other steelhead in the world (Busby et al., 1996). Although steelhead are iteroparous, they rarely spawn more than twice (Nickelson et al., 1992). Before most of the lower Columbia River and Snake River dams were constructed, the proportion of repeat-spawning steelhead in the Snake and

Columbia rivers was less than five percent (USACE, 2002). The current proportion of repeat spawners is unknown but assumed to be near zero.

Snake River steelhead spawn in cool, clear tributaries of the river, with water temperatures ranging from 10 to 15.5 °C (Scott and Crossman, 1973). Preferred spawning habitat includes small and medium-sized gravel in riffles located upstream of pools. Depending on the water temperature, steelhead eggs may incubate in redds for up to 4 months before hatching. Snake River fry emerge from the gravel in July through September (BPA et al., 1994).

Juveniles prefer water temperatures of 12 to 15 °C and occupy shallow riffles for the first year of life before moving to pools and runs. Juvenile steelhead rear primarily upstream of the four Lower Snake River dams (USACE, 2002). Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Winter rearing habitat is characterized by complexity, primarily in the form of large and small wood. Some older juveniles move downstream to rear in larger tributaries and river main-stems (Nickelson et al., 1992). Young steelhead remain in freshwater for one to four years, before migrating toward the ocean. Steelhead smolts, 15 to 20 cm in total length (Meehan and Bjornn, 1991), pass the Lower Granite Dam on their way to the ocean from mid-April through early July (BPA et al., 1994; USACE, 1999). A-run steelhead, as mentioned above, typically stay in the ocean for one year, while B-run steelhead stay for two years before returning to the river for their spawning migration (BPA et al., 1994).

5.6.3 Habitat Concerns

Factors similar to those affecting other salmonids, such as habitat destruction and modification, over utilization, and natural and human-made factors, have contributed to the decline of Snake River Basin steelhead. NMFS (1997b) identified several threats to steelhead including timber harvest, agriculture, mining, and urbanization that have degraded, simplified, and fragmented habitat. NMFS (1997b) also identified water diversions for agriculture, flood control, domestic, and hydropower purposes (especially in the Columbia River) that have greatly reduced or eliminated historically accessible habitat. Other potential threats to steelhead include (NMFS, 1997b) over-harvest by recreational fisheries, predation by pinnipeds and piscivorous fish species, effects of artificial propagation, and the deterioration or loss of freshwater and marine habitats. Critical habitat has been noted by the USFWS as depicted in Figure 5-12.

5.6.4 Presence in Action Area

The Action Area is within the migration corridor used by Snake River steelhead adult and juveniles. Adult steelhead may feed and rest in the confluence of the Snake and Clearwater rivers prior to moving farther upstream in each river for spawning in the spring. In a tagging study conducted from 1969 to 1971 in the confluence area, adult steelhead were found to migrate and rest in near shore areas, traveling 20 to 30 m out into the channel and migrating in mid-channel only when crossing to the other shore (USEPA, 1974).

Recent telemetry studies indicate that adult steelhead typically occur near the Snake/Clearwater confluence for about 4-5 months annually. Bjornn et al. (2003) used telemetry studies from 1991-1995 to study the migration of adult steelhead past dams and

through reservoirs in the lower Snake River and into the tributaries. They observed many adult steelhead entered the Clearwater River in the fall, but large numbers of steelhead destined for the Clearwater River wintered over in Lower Granite Reservoir, near the confluence of the Snake and Clearwater rivers and in the Snake River between Lewiston, ID, and Asotin. For the four-year period, they determined over wintering reaches for 327 of 491 steelhead whose last telemetry record was in the Clearwater River. Of the 327, 70.3% wintered over upstream of Lower Granite Dam and the remainder wintered over downstream from the dam. The wintering- over locations were subdivided for 245 of the 327 steelhead: 48.6% wintered over in the Lower Granite reservoir and Clearwater/Snake River confluence area upstream to the Snake River receiver site located near Asotin WA (RM 145.3), and the lower Clearwater receiver site located upstream of the Clearwater Paper Mill (RM 144).

Redds are built typically in smaller tributaries and in main river reaches above the confluence area. However, a small number of A-run steelhead spawn in Snake River tributaries that enter Lower Granite Reservoir and downstream of the Lower Granite Dam (USACE, 1995). Juvenile steelhead may use the confluence area of the Snake and Clearwater rivers and Lower Granite Reservoir for rearing habitat, although most smolts migrate rapidly through the area.

Orientation within the water column of the Action Area is not specifically known for adult steelhead. However, hydroacoustic surveys (USACE 1991) found larger fish are typically oriented near the bottom in the Lower Granite Reservoir. Yearling steelhead have been collected at mid (6-12 m) and shallow (< 6m) depth (Bennett et al. 1993) as well as depths >18m (USACE 1991). Hydroacoustic surveys (USACE 1991) conducted in May and June found outmigrating juvenile salmonids were located throughout the water column with the greatest concentration in the upper 15 meters.

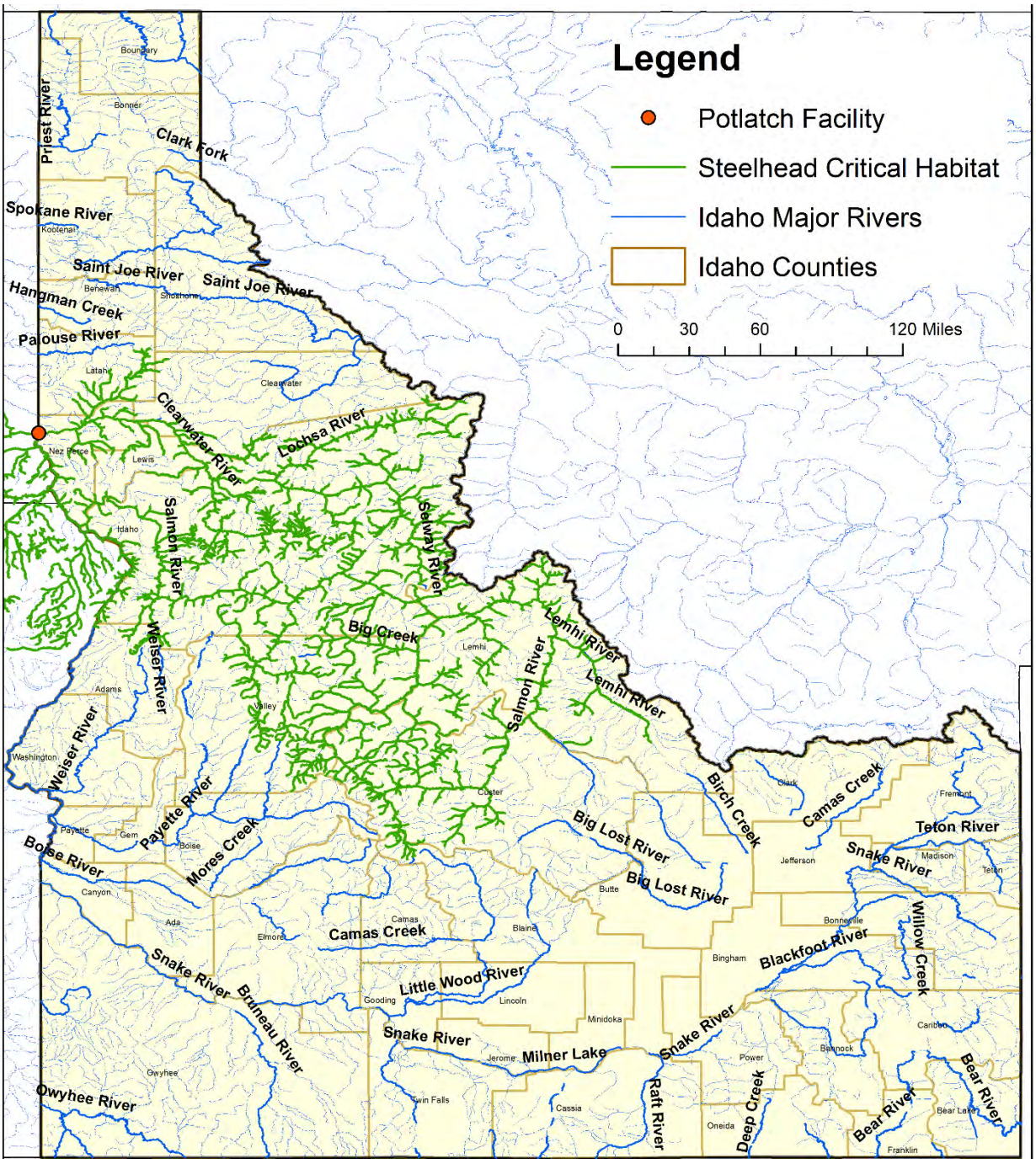

Figure 5-12: Steelhead salmon critical habitat.

5.6.5 Abundance and Timing Data

Steelhead passage data collected at the Lower Granite Dam for the years 2006 through 2015 are presented to describe abundance and passage near the Action Area. These data are summarized in Appendix E. Most upstream migration of wild adult steelhead across the dam into the Lower Granite Reservoir downstream of the mouth of the Clearwater River occurred from April to December (Figure 5-13). As shown in Table 5-12, the date of early passage adult steelhead at the Lower Granite Dam, 1993 through 1999, ranges from

January 2, 2006 to March 4, 2007. The date of late passage ranges from December 15 in 2006 and 2012 to December 31, 2010. A-run migration in this portion of the Snake River occurs between March and May, followed by B-run migration between August and November. Annual run size of wild steelhead ranged from approximately 29,000 to 76,000 fish.

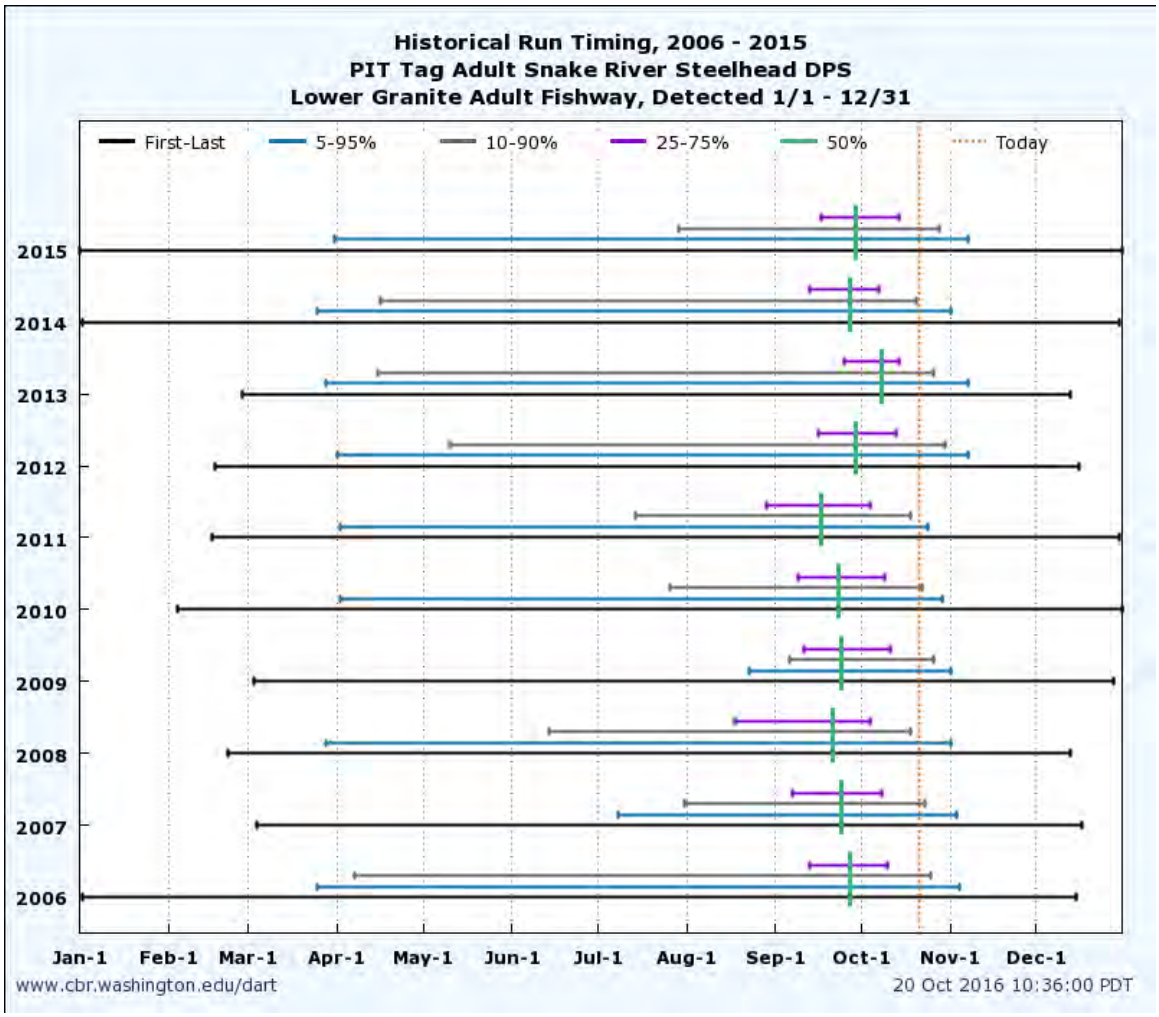

Figure 5-13: Average adult steelhead passage at Lower Granite Adult Fishway, 2006 - 2015 (source: DART).

Data from 2006 - 2015 shows downstream migrating juvenile wild steelhead passed over the Lower Granite Dam primarily between April and mid-May, with a small proportion of late outmigrants passing the dam in late July and early August. First passage dates ranging from March 18, 2006, through April 2, 2010 (Table 5-13 and Figure 5-14). Dates of last passage over Lower Granite Dam ranged from June 23, 2007, through December 16, 2012. These dates probably represent the dates of sampling rather than the actual dates of passage.

Table 5-12. Dates of adult steelhead passage at Lower Granite Adult Fishway, 2006 - 2015 (source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last

2006	Average	1/2	3/25	9/27	11/4	12/15
2007	Average	3/4	7/8	9/24	11/3	12/17
2008	Average	2/22	3/27	9/20	10/31	12/12
2009	Average	3/3	8/23	9/24	11/1	12/28
2010	Average	2/4	4/2	9/23	10/29	12/31
2011	High	2/16	4/2	9/17	10/24	12/30
2012	Average	2/17	3/31	9/28	11/6	12/15
2013	Average	2/27	3/28	10/8	11/7	12/13
2014	Average	1/2	3/25	9/27	11/1	12/30
2015	Low	1/1	3/31	9/29	11/7	12/31

Table 5-13. Dates of juvenile steelhead passage at Lower Granite Dam, 2006 - 2015 (source: DART).

Year	Flow	First	5th %tile	50th %tile	95th %tile	Last
2006	Average	3/28	4/12	5/2	5/22	6/25
2007	Average	3/25	4/22	5/8	5/24	6/23
2008	Average	4/5	4/21	5/9	5/26	11/21
2009	Average	3/25	4/19	4/26	5/21	7/19
2010	Average	4/2	4/24	5/7	6/4	7/8
2011	High	3/23	4/2	5/10	5/31	10/6
2012	Average	3/22	4/14	4/29	5/24	12/16
2013	Average	3/20	4/14	5/11	5/22	10/19
2014	Average	3/20	4/19	5/7	5/27	10/31
2015	Low	3/18	4/12	5/5	5/29	8/20

Hatchery adult and juvenile steelhead make up most of the steelhead population in the Snake River. Dates of migration through the Snake River for hatchery-reared steelhead were comparable to those for wild steelhead.

5.6.6 Travel Time

Keefer et al. (2002) investigated adult steelhead travel time and passage efficiency in the lower Columbia and Snake rivers using radio telemetry. Migration speeds ranged from 24 km/day to 42.5 km/day, with a mean migration speed of 32.8 km/day. Keefer et al. (2003**)** measured migration speed of adult steelhead at eight main-stem dams and reservoirs in the lower Columbia and Snake rivers, all major tributaries between

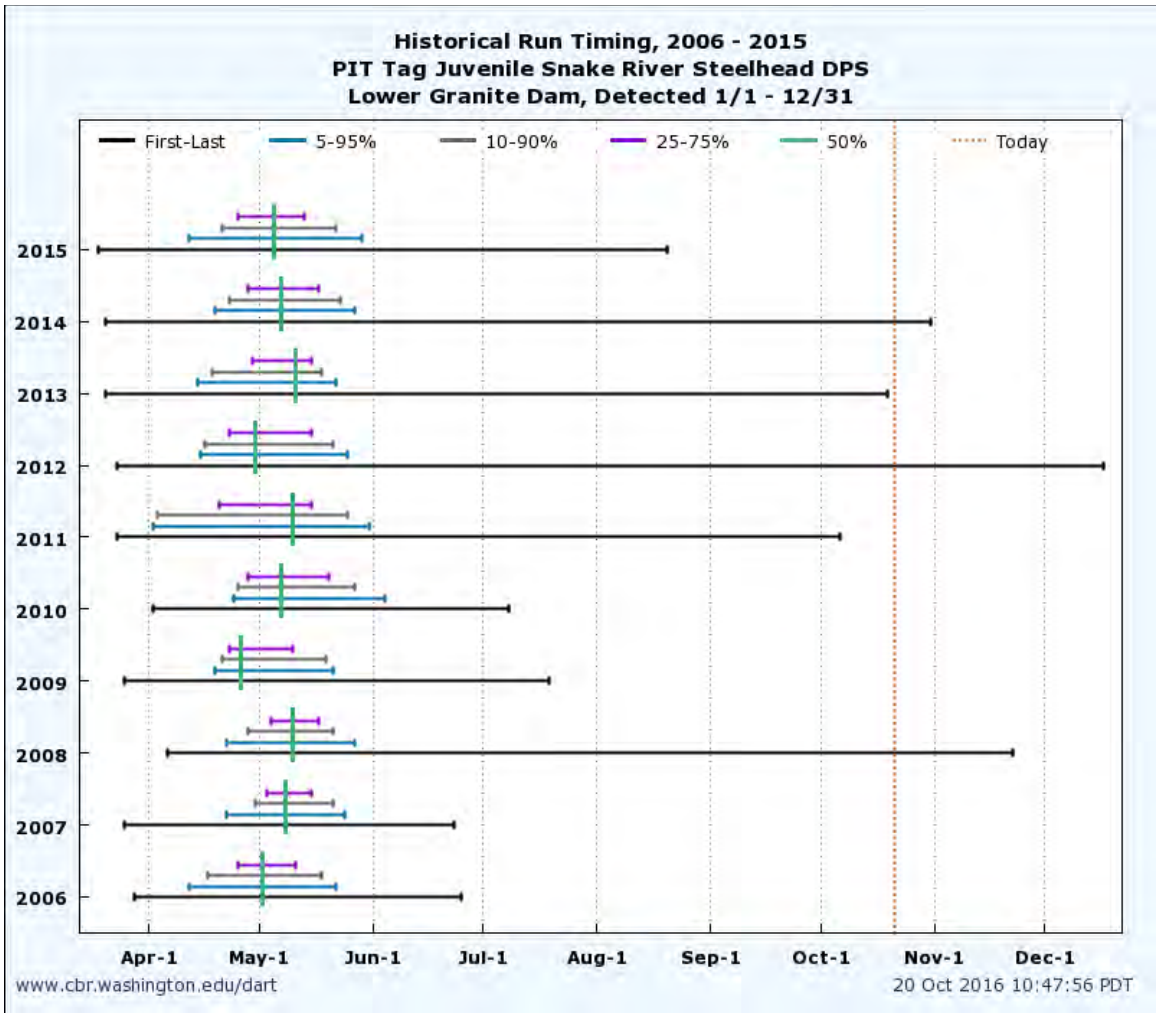

Figure 5-14: Average juvenile steelhead passage at Lower Granite Dam, 2006-2015 (source: DART).

Bonneville and Priest Rapids dams on the Columbia River and the Snake River and its tributaries upstream to Hells Canyon Dam during warmer months (June-October) over a period of four years. Median values for adult steelhead reported for the four-year duration ranged from 7 km/day to 21 km/day, with a mean of 13.1 km/day. Keefer et al. (2003) also studied adult steelhead migration speed in Columbia and Snake River reservoirs (Bonneville, Dalles, John Day, McNary to Ice harbor, McNary to Hanford receiver, Ice Harbor, Lower Monumental, Little Goose, Lower Granite to Snake River receiver, and Lower Granite to Columbia River receiver) over the same four-year duration. Median values reported for the four-year duration ranged from 10 km/day to 49 km/day, with a mean of 29.5 km/day. In a study conducted by Bjornn et al. (2000), adult steelhead migration speeds ranged from essentially zero to 150.6 km/day, with a mean of 15.6 km/day.

A range of mean migrations speeds for adult steelhead of approximately 13 to 33 km/day has been observed. This distance from the confluence of the Snake and Clearwater rivers and the Lower Granite Dam is approximately 31 miles, or 50 km. Given the mean

migration speeds observed in the literature, adult steelhead may require one and one half to four days to travel between the Lower Granite Dam and the confluence.

Travel times of wild steelhead juveniles were estimated from the PTAGIS database by NOAA (2003). Juveniles trapped and tagged at a Snake River and a Clearwater River trap from 1990-2003 were detected at the Lower Granite Dam allowing for estimates of travel time. The size range for sampled juvenile steelhead was 80-340 mm at the Snake River trap and 120-270 mm at the Clearwater trap. This analysis has two caveats: 1) fish samples were collected from the surface, where juveniles that are actively migrating are most likely to be oriented in the water column. This may bias the sample away from the portions of the cohort that may be feeding/rearing as they progress downstream, 2) data were collected during the peak migration, again focusing the study on one portion of the entire cohort.

The following travel times were estimated in days for steelhead juveniles:

Table 5-14: Travel Times for Steelhead Juveniles

Snake River Trap (n=13887)		Clearwater River Trap (n=4447)	
Mean	3.7	Mean	5.6
Median	3.0	Median	4.9
99.5 percentile	20.2	99.5 percentile	20.1

5.7 Spalding's Catchfly (Silene spaldingii) - Threatened

5.7.1 Status and Description

Spalding's catchfly (*Silene spaldingii*) is a long-lived perennial herb in the family Caryophyllacea. It has four to seven pairs of lance-shaped leaves and small greenish-white flowers. The plant is distinguished by its very sticky foliage and petals that are shallowly lobed (USDA, n.d.). The species is presently listed as threatened, since 2001. Natural Heritage Programs in Idaho and Montana consider the plant to be rare and imperiled. Both the Bureau of Land Management and the U.S. Forest Service consider the plant a sensitive species.

5.7.2 Life History

Spalding's catchfly produce up to several vegetative or flowering stems, all of which originate from a simple or branched underground stem, called a caudex. This stem runs along a long and narrow taproot (USFWS, n.d.) Typically the plants grow to 20-40 cm in height and each stem bears 4-7 pairs of leaves of 5-8 cm in length and 2-4cm in width, arranged opposite one another. When individuals reproduce, they produce 3-20 pink, light green, or cream-colored flowers borne in a branched, terminal inflorescence (USFWS, n.d.) The plant reproduces solely by seed and lacks the ability to reproduce vegetatively (USFWS, n.d.).

5.7.3 Habitat Concerns

The catchfly's grassland habitat once was widespread in the region but has been reduced by more than 95 percent over the past century, primarily because of conversion to agricultural and urban uses. Fire suppression also has allowed an unnatural increase in

woody plants, which overtake catchfly habitat, decreasing its numbers. Threats to this species may also include livestock grazing, herbicide spraying, noxious weed infestation, and recreation. All populations are potentially vulnerable to naturally occurring events or human activities. No critical habitat has been designated by the USFWS, but the known range is depicted in Figure 5-15.

5.7.4 Presence in the Action Area

Spalding's catchfly has a known range which extends along the entirety of Lewis, Nez Perce, Idaho, and Latah counties. However, since it is not an aquatic species, its proximity is not close enough to warrant consideration under this BE.

5.7.5 Abundance and Timing Data

The majority of remaining Spalding's catchfly populations are extremely small and isolated, often bordering agricultural fields or rangelands. The plant prefers open grasslands with rough fescue or bluebunch wheatgrass with some occasional conifers, and the deep-soiled valley/foothill zones, typically associated with the Palouse region of southeastern Washington, northwestern Montana, and portions of Idaho, Oregon and British Columbia, Canada. It is native to grassland prairie habitats that range from 1,500 to 5,000 feet in elevation (USFWS, n.d.). This rare plant may be found at 52 sites in west-central Idaho, northwestern Montana, northeastern Oregon, eastern Washington and British Columbia. Only 18 population sites contain more than 50 individuals, and of those, only six contain more than 500 plants. More than half the known sites are located on private land. The total number of Spalding's catchfly for all populations is about 16,500. Plants emerge in mid to late May and flowering occurs in mid-July until August, and sometimes into October. The vegetative portion lying above ground dies back at the end of the growing season, and will remain dormant until the following spring, or even for several years (USFWS, n.d.).

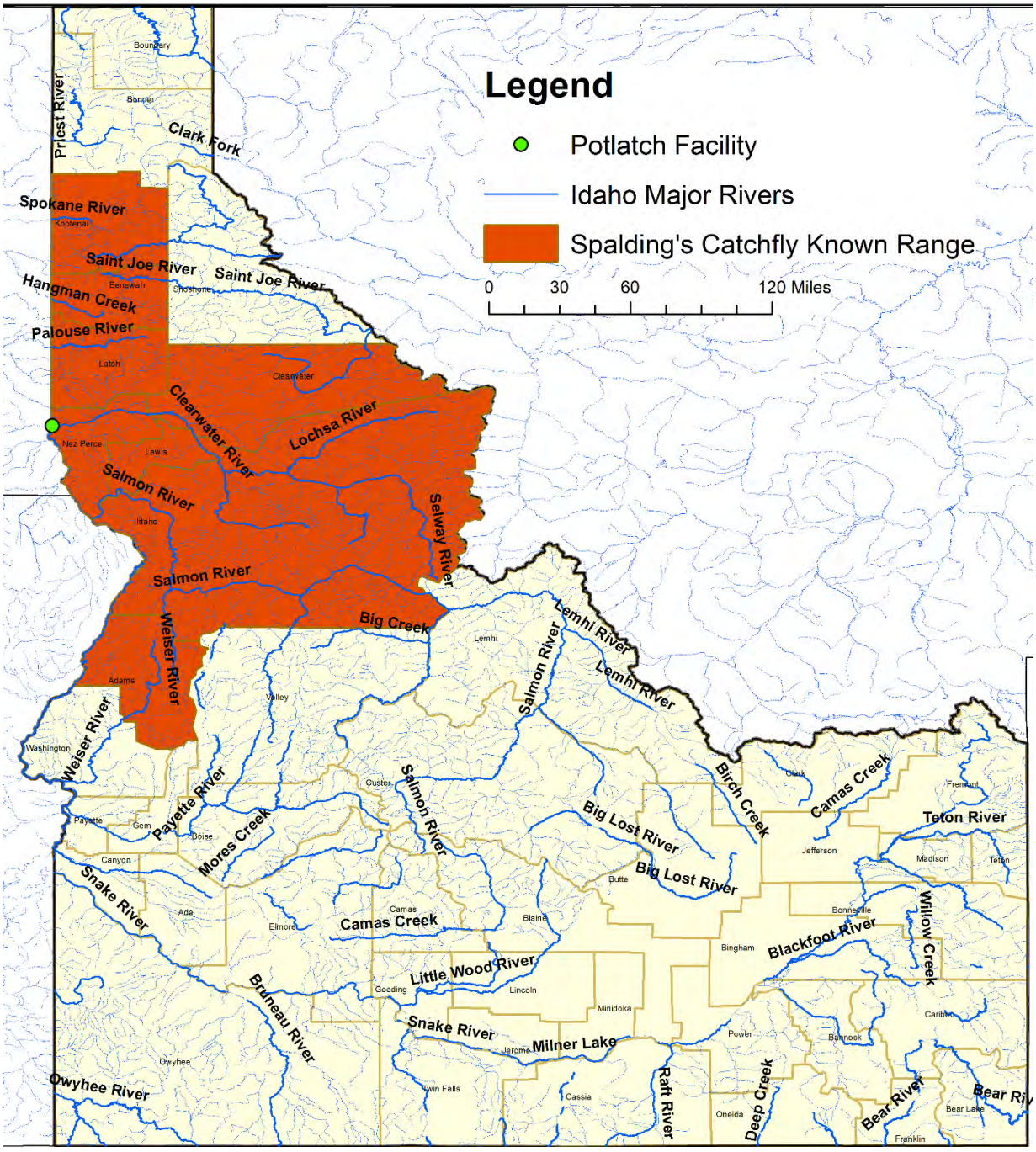

Figure 5-15: Spalding's Catchfly known range.

5.8 Yellow-billed Cuckoo (Coccyzus americanus) - Threatened

5.8.1 Status and Description

The yellow-billed cuckoo (*Coccyzus americanus*) is a long and slender songbird with a yellow bill nearly as long as its head, which is relatively flat. They possess a long tail and are typically brown above and white below in color, with a black face and yellowish coloring around the eyes. Cuckoos typically live in the western United States in willow

and cottonwood forests along rivers and streams. The birds are generally absent from heavily forested areas and large urban areas. Yellow-billed cuckoos primarily eat large insects such as caterpillars and cicadas, as well as an occasional small frog or lizard (USFWS, n.d.).

Populations have declined rapidly throughout the western U.S. in the twentieth century, and are extirpated from British Columbia, Washington, and possibly Nevada. The yellow-billed cuckoo is listed as endangered under the California Endangered Species Act. At one time, this cuckoo numbered more than 15,000 pairs, but has been reduced to about 30 pairs in less than 100 years. In Arizona, where the largest cuckoo population west of the Rocky Mountains continues to be found, the Arizona Department of Fish and Game considers the bird to be a species of concern. The bird is designated as threatened in Utah. In Idaho, the species is considered a rare visitor and breeder in the Snake River Valley. These state listings do not confer the same regulatory protection as the federal Endangered Species Act (USFWS 2001).

5.8.2 Life History

The yellow-billed cuckoo once ranged throughout most of the United States, southern Canada, and Mexico, but has experienced severe population declines, particularly west of the Rocky Mountains. By the 1920s, the yellow-billed cuckoo had disappeared from its former range in British Columbia, and by the 1950s the species no longer bred in the northwestern United States, including northern California. Today, only small remnant populations persist in the West (CLO 2001).

5.8.3 Habitat Concerns

Because the birds are primarily found in riparian areas, potential threats include conversion of this habitat to agriculture, dams and river flow management, bank protection, livestock overgrazing, agricultural water use, pesticide use, and competition from exotic plants. USFWS has proposed critical habitat for the yellow-billed cuckoo but it is not near the Clearwater facility (Figure 5-16).

5.8.4 Presence in the Action Area

Yellow-billed cuckoos breed from southern Canada south to the Greater Antilles and Mexico. While the yellow-billed cuckoo is common east of the Continental Divide, biologists estimate that more than 90 percent of the bird's riparian habitat in the West has been lost or degraded due to conversion to agriculture, dams and river flow management, bank protection, overgrazing, pesticide use, and competition from exotic plants such as tamarisk.

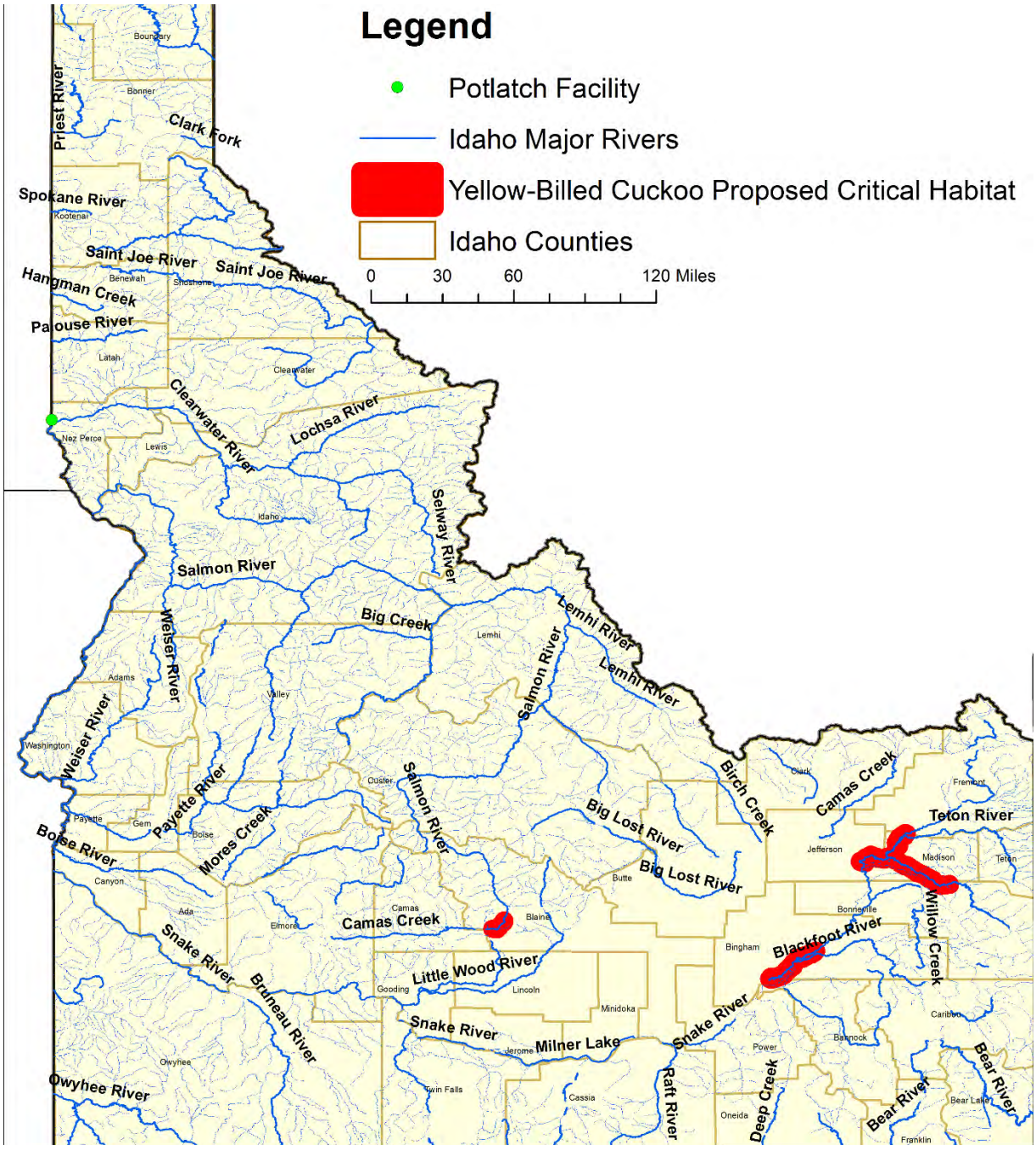

Figure 5-16: Yellow-Billed Cuckoo proposed critical habitat.

The yellow-billed cuckoo is listed as Threatened on the federal Endangered Species Act, as well as having proposed critical habitat. However, the proposed critical habitat is not near the Clearwater facility, therefore this species is not considered in the biological evaluation.

5.8.5 Abundance and Timing Data

Populations have declined rapidly throughout the western U.S. in the twentieth century, and are extirpated from British Columbia, Washington, and possibly Nevada. The

yellow-billed cuckoo is listed as endangered under the California Endangered Species Act. At one time, this cuckoo numbered more than 15,000 pairs, but has been reduced to about 30 pairs in less than 100 years. In Arizona, where the largest cuckoo population west of the Rocky Mountains continues to be found, the Arizona Department of Fish and Game considers the bird to be a species of concern. The bird is designated as threatened in Utah. In Idaho, the species is considered a rare visitor and breeder in the Snake River Valley. These state listings do not confer the same regulatory protection as the federal Endangered Species Act.

5.9 Northern Wormwood (Artemisia campestris. Var. wormskioldii) – Proposed Threatened

5.9.1 Status and Description

Northern Wormwood is a perennial in the aster family Asteraceae and is also commonly known as Pacific Sagebrush (USFWS 2013). It is a low-growing plant, typically reaching 6-12 inches, but can grow to be 16 inches. It has a taproot, and rosette arranged basal leaves which are 1-4 inches long, and usually covered in silky hairs, as is the stem of the plant (USFWS 2013). The Northern Wormwood is federally listed as a candidate species.

5.9.2 Life History

Northern Wormwood is found in exposed basalt, sand, and cobbly-sandy habitats, and flowers in April and May. Outer female flowers are fertile, the sterile disk flowers have undeveloped ovaries (USFWS 2013).

5.9.3 Habitat Concerns

The construction of dams along the Columbia River, resulted in habitat loss as well as individuals and populations of Northern Wormwood. Erosion by wind and water on sandy substrate has caused mortality of adult plants, and decreased survival of seedlings (USFWS 2013). Both trampling, during recreational usage of Wormwood habitat, and above average rainfall, can stress many populations (USFWS 2013).

5.9.4 Presence in the Action Area

The Northern Wormwood is not presently found near the Clearwater facility, and thus will not be considered in this BE.

5.10 Washington Ground Squirrel (Urocitellus washington) – Proposed Threatened

5.10.1 Status and Description

This squirrel ranges from 7.3-9.6 inches in length and is distinguished from other Washington and Oregon ground squirrels due to its smaller size, shorter tail, and white speckled dorsum (USFWS 2012). Females are quite social, and often will form groups with up to 3 other females within their communities, while males are more mobile. The species is federally listed as a candidate species, and as endangered under the Oregon Endangered Species Act (USFWS 2012).

5.10.2 Life History

This squirrel is diurnal, and spends much of the year underground, with adults emerging from hibernation between January and early March (USFWS 2012). Typically, this squirrel's lifespan is less than 5 years, with high annual mortality, up to 66% for males, and 76% for females; starvation, freezing, disease, and human interference are all causes (USFWS 2012). One litter is produced annually, with females becoming briefly sexually receptive within a few days of emergence from hibernation. Average litter size is between 5-8, with pups emerging above ground in the early spring, between March and April (USFWS 2012). Washington ground squirrels tend to live in shrub-steppe and grassland habitat in the Columbia Basin, and require sufficient forage and suitable soils, such as sandy or silt-loam.

5.10.3 Habitat Concerns

Often habitats with deeply disturbed soils are preferred by the Washington ground squirrel, with agriculture (plowing, discing, crops, livestock) causing most of the disturbance. There has also been a 51-85% degradation or loss of historic Washington ground squirrel habitat throughout the Columbia Basin, attributed mostly to agriculture (USFWS 2012). Intensive livestock grazing also encourages the spread of invasive weeds like cheatgrass, outcompeting native forbs and grasses that make up most of the squirrel's food sources. Residential, military, and commercial development also reduces squirrel habitat (USFWS 2012).

5.10.4 Presence in the Action Area

The Washington ground squirrel is not presently found near the Clearwater facility, and thus will not be considered in this BE. The designated Habitat Conservation Areas (HCAs) and their lines of connectivity are shown in Figure 5-17; these are defined geographical areas where state, local, or federal agencies concentrate their efforts on maintaining habitat for an endangered or candidate species.

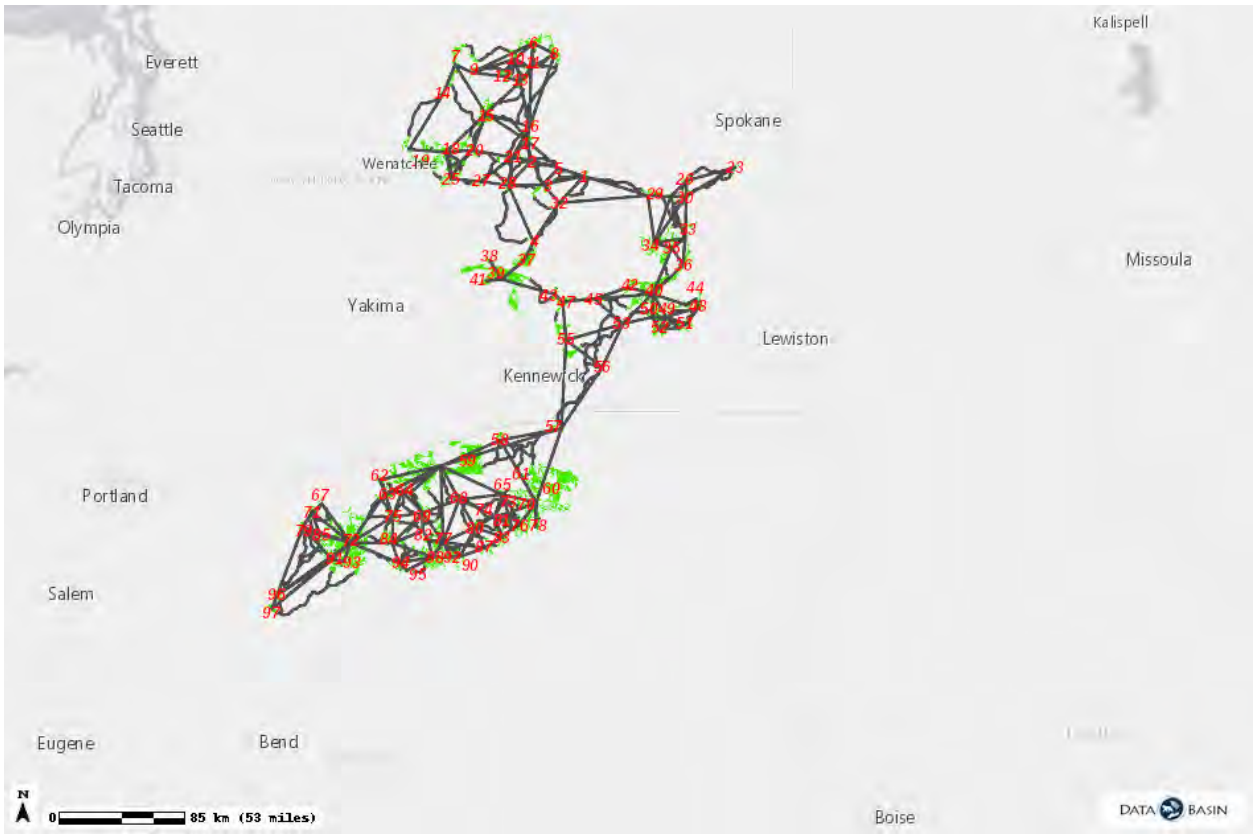

Figure 5-17: Habitat conservation areas (HCAs), and connectivity lines, for the Washington Ground squirrel.

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6 Physical Habitat Characteristics of the Receiving Water

This section describes physical habitat availability and condition for the listed species in the Action Area. Presence of critical habitat, and habitat values are discussed. More recent investigation has verified the original numbers. Habitat conditions associated with water quality are presented in Section IV.

6.1 Area Description

The climate of the lower Snake River basin is dry (11 inches precipitation at lower elevations to 23 inches at higher elevations) with average maximum winter air temperatures of 40°F and typical summer highs ranging from 80-90°F. The past three years (2013-2016) have been the hottest on record, and the Corps has made permanent improvements to the Lower Granite Reservoir to assist federally endangered salmon that were suffering thermal restrictions along the fish ladders (USACE website 2016). The topography ranges from areas of broad valleys with gentle slopes to areas of deep confined canyons with steep walls. Elevation of the lower Snake River basin ranges from 340 to 3,000 feet. The area is within the Columbia Basin physiographic province and includes two major vegetation zones: steppe communities dominated by bunchgrasses and shrub-steppe dominated by sagebrush species. The steeper slopes have grasslands habitat type dominated by cheatgrass and remnant bunchgrasses. The shrub-steppe habitat is characterized by big sagebrush, rabbit brush, and cheatgrass.

There are 9,220 acres of project lands surrounding Lower Granite Reservoir. These include fee lands that are federally owned and managed by the Army Corps of Engineers (the Corps). Port districts own land adjacent to the project for industrial development. Most of the project lands are used for public recreation, wildlife habitat, wildlife mitigation, and water-connected industrial development. The area is developed for recreation including: boat ramps, marinas, day-use facilities, and campgrounds.

Habitat Management Units (HMUs) were established along the lower Snake River to compensate for wildlife habitat lost resulting from inundation following dam installation. There are 17 HMUs totaling 5,002 acres, along lower Granite Reservoir. Six municipal and industrial pump stations withdraw water from the Lower Granite Reservoir. Also, there are port facilities at Lewiston, Clarkston, and Wilma that are used to transport grain, wood products, and other commodities.

The main urban areas are Lewiston and Clarkston. The lower Snake River receives discharge of urban runoff as well as treated effluent from multiple municipal wastewater treatment plants at these towns, as well as the secondary-treated wastewater effluent from the Clearwater Mill.

The middle Snake River basin has large amounts of agriculture and agricultural return flows are considered the largest nonpoint source of pollution entering the lower Snake River.

The Lower Snake River has four locks/dams in the State of Washington: Ice Harbor, Lower Monumental, Little Goose, and Lower Granite dams. All are run-of-the river

facilities. They have limited storage capacity and flow rate through the dam is approximately the same as that entering the reservoir. These dams were built for navigation, hydropower generation, irrigation, and recreation. The dams have had a major influence on the quantity and quality of salmonid habitat available in the Action Area. All populations and life histories of Snake River salmonids are affected by this major habitat alteration (USACE, 2002, Appendix H).

6.2 Habitat Characteristics

Prior to construction of the four dams (1961-1975), the lower Snake River had an alluvial morphology consisting of a longitudinal profile of pool-riffle-run sequences. Water levels were less controlled and fluctuated by as much as 20-30 feet. The impoundment of the river converted the lower Snake River to a continuous reservoir system. The only areas that retain riverine characteristics are the relatively short and discontinuous tailrace areas just downstream of each dam.

6.2.1 River morphology

The Snake River has mean annual discharge over 54,000 cfs and is the largest tributary to the Columbia River. The Clearwater River is the largest tributary to the Snake River and historically contributes about 39% of the flow to the Snake River. During summer low flow periods the Clearwater contributes about 50% of the flow with Dworshak releases. Lower Granite Dam creates the pool that is the dominant habitat feature of the Action Area. The dam is located on the Snake River at river mile 107 near Almota, Washington. The dam creates a pool that extends 39.3 miles upstream in the Snake and a further 4.6 miles into the lower Clearwater River. Impoundment of the Lower Granite Reservoir is considered to end near Asotin in the Snake River arm and near the Clearwater Corporation in the Clearwater River arm. The Dam is 3,200 feet wide and has a hydraulic head of 100 feet.

The lower Granite Reservoir created behind the dam has a capacity of 49,000 acre-feet (normal operating range) and normal pool operation range is 733-738 ft elevation. Other physical features of the reservoir are in the following table:

Table 6-1: Physical Features of Lower Granite Reservoir

Normal pool fluctuation-weekly	1.5 m
Reservoir length	62.8 km (39.0mi)
Surface area	3,602.0 h (8,900ac)
Proportion of impounded reach	25.6%
Maximum depth, flat pool	42.3 m (138ft)
Mean depth, flat pool	16.6 m (54ft)
Maximum width	1128.0 m (3,700ft)
Mean width	6473.0 m (2,110ft)
Major tributaries	Clearwater River
(From Bennett et al., 1983 as cited in USACE, 2002, Appendix B)	

The reservoir area exhibits a typical longitudinal impoundment gradient composed of three reach types. The uppermost portion of the Lower Granite Reservoir is almost a riverine environment (approximately 5-15% of the impoundment gradient). This reach includes the confluence of the Clearwater and Snake Rivers, which is an important fish

habitat area in the Lower Granite Reservoir due to greater water velocity and cooler water inflow from the Clearwater. A mid-reservoir reach represents the largest section of each impoundment and is a transition area from the lotic character to the more lentic conditions nearer the dam (67 – 72%). The reach immediately above the dam is the forebay (13-18%) that has entirely lentic characteristics (Zimmerman and Parker, 1995 as cited USACE, 2002, Appendix B).

Approximately 10% of the Lower Granite Reservoir is shallow water habitat (Bennett et al., 1993). Many of these areas are created from in-water disposal of dredged sediment. Shallow areas are located at the shoreline of in-channel islands and some mid-channel shelf areas. Shallow water areas in the reservoir are maintained due to the relatively small fluctuations in water level (<5ft). Consistent water levels maintain benthic habitat thereby maintaining production of benthic invertebrates (fish food source). Backwaters areas, with very low water velocity, slightly warmer water, and fine substrate, are very limited in the Lower Granite Reservoir. These areas are favored by resident warm water species (e.g., centrarchids) for spawning and rearing. Aquatic macrophyte production in the Lower Granite Reservoir is very minor due to lack of shallow areas and backwaters.

6.2.2 Riparian characteristics

Prior to inundation, the riparian habitat was composed of riparian forest palustrine scrub-shrub, and mesic shrubland. Cottonwood, white alder, and black locust dominated forested areas. Currently, riparian vegetation communities cannot develop due to the steep shorelines along the reservoirs and because these shorelines are typically covered in riprap. Riparian vegetation is limited to a narrow corridor and backwater areas. The extent of woody plant communities that once characterized the riparian zone are very limited.

6.2.3 Sediment

The lower Snake River reservoir system accumulates approximately 3 to 4 million cubic yards of sediment per year, primarily within the Lower Granite Reservoir and the Palouse and Tucannon river deltas (USACE, 2002). Approximately half of these sediments are fine-grain and the rest are coarser sand. Sediment in the lower Snake River is characterized as aerobic brown or gray silt (Falter, 2001 as cited in NMFS, 2003). The large inputs of sediment have necessitated dredging which began in 1986. Sediments in the Action Area have accumulated chemical contaminants, including dioxin, metals, pesticides, herbicides, ammonia, and nitrogen. Contaminants in the sediments can are bioavailable to benthic organisms and can bioaccumulate in higher trophic levels via the food chain.

6.2.4 Substrate

Substrate size in Lower Granite Reservoir differs significantly between shallow and deep-water areas although silt is the predominant substrate class based on a study from six sample sites (Bennett and Shrier, 1986 as cited in USACE, 2002, Appendix B). Clay content generally increased with distance downstream and organic content is <5%. A substrate study of five shallow areas in the Lower Granite Reservoir found a high degree of embeddedness for substrates <150mm diameter (Bennett et al., 1988 as cited in USACE, 2002). Organic content ranged from 5.2-8.8%. In a study by the Corps (2000)

emphasizing depositional areas, average particle size was distributed among small-size classes (silt and clay 17% and sand 74% and gravel 8%). Generally, samples collected near the Snake/Clearwater confluence that were more than 75m from the shoreline contained <1% fines.

6.2.5 Fish assemblage

The Lower Snake Reservoirs contain 18 native species and 17 introduced species. Seasonal sampling conducted between 1979 and 1980 found bridgelip sucker, redside shiner, largescale sucker, small mouth bass and northern pikeminnow had the highest relative abundance in the Lower Snake reservoirs (Bennett et al., 1983 as cited in USACE, 2002), accounting for approximately 80% of all fish sampled. Of these five species, all except smallmouth bass are native to the lower Snake River. Less abundant fish were a combination of native and introduced species. Introduced crappies, yellow perch, and sunfish were highly abundant in off-channel habitats. Other introduced fish such as catfish and bullheads were present, but in lower abundance.

The most significant salmonid predators in lower Snake River reservoirs are smallmouth bass because of their high abundance, habitat overlap rearing salmonids, and reduced abundance of their other prey source (crayfish) (Bennett et al., 1993). Smallmouth bass consume mostly subyearlings and wild steelhead because of their limited mouth gape. The small proportion of larger-sized smallmouth bass has the potential to consume limited numbers of yearling Chinook salmon, but the magnitude of this predation is probably low. Also, yearling Chinook migrate earlier in the year (March-May) when water temperatures are lower and predators less active. Other than fall Chinook, fish predation appears to be relatively low in yearling Chinook and steelhead. Crappie and yellow perch are relatively minor predators on juvenile salmonids in the reservoir and their small body size restricts consumption to sub-yearling Chinook and smaller yearling Chinook and wild steelhead.

6.2.6 Fish passage at the dam

Upstream migrating adult salmonids use a fish ladder to pass over the lower Granite Dam. The ladder has two south shore entrances as well as devices to attract adults to the ladder. The ladder is operated year-round except for a two-week annual maintenance period (January – March). Downstream migrating juveniles have three possible passage routes through the dam: the turbines, the spillway, or the juvenile bypass system. Depending on operational choices, the latter route can result in either diversion to the transport system (truck/barge system) or release in the tailrace. Spillway passage is generally considered the safest route, with juvenile bypass systems a close second, and turbine passage the least safe. Entrainment into the turbines can result in physical damage to fish. Although spillway passage is the safest route of passage, the physical process of spilling water can result in elevated concentrations of dissolved gas in downstream waters, which can in turn cause death or injury of juvenile migrants, irrespective of which route they passed the dam.

6.3 Habitat Values

Habitat characteristics that are important to survival and conservation of Snake River salmon as describe by their Critical Habitat designation (Federal Register Vol.58, No.

247, Dec. 28, 1993) are listed below. These features are also relevant to the bull trout that also has designated critical habitat. This section describes physical habitat values of the Action Area for these habitat components.

Table 6-2: Essential Features of Habitat Components in the Action Area

Habitat Component	Essential features
Juvenile migration corridors	substrate, water quality and quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, safe passage conditions
Spawning and juvenile rearing areas	spawning gravel, water quality and quantity, water temperature, food, riparian vegetation, cover/shelter, and space
Adult migration corridors[1]	substrate, water quality and quantity, water temperature, water velocity, cover/shelter, riparian vegetation, space, safe passage conditions
[1]Adult steelhead spend a protracted period of time in the Action Area as they move to spawning grounds; therefore, food is an essential feature of their adult migration corridor.	

6.3.1 Juvenile Migration

The conversion from a lotic to a lentic system affects habitat availability and the duration of movement of juveniles in the system. The slow, deep-water habitat that dominates the Action Area provides little cover in the form of riparian features, large woody debris, substrate, and off channel areas. The reservoir has low water velocity, thus, the natural transport provided to juveniles migrating downstream is reduced which may increase stress and energy expenditure. Juveniles are more susceptible to predation from both piscivorous fishes and birds in this modified habitat (less cover and longer duration of exposure). Finally, passage at the dam facility can result in injury or mortality.

Dam operation has been modified to reduce some of the negative impacts to migrating juveniles including the following:

Flow augmentation. Dworshak Dam releases water to increase flow to reduce travel time of juvenile migrants through the system. The decreased travel time reduces exposure of juveniles to predators and to reservoir conditions /potential hazards. Approximately 1.9 MAF of the Snake River Basin storage is made available for augmentation.

Reservoir drawdown. Lower Granite Dam is operated within one foot of the Minimum Operating Pool (MOP) from April 3 through November 15 annually to increase water velocity, decreasing juvenile travel time.

Temperature Control. Summer releases of cold water from the Dworshak Dam reduce temperatures in the Lower Granite Reservoir to improve water conditions for migrating adults (fall Chinook and sockeye) and juvenile fall Chinook salmon. Noteworthy, however is the fact that the reduced water temperature in the lower Clearwater River tends to reduce the growth of fall Chinook rearing in this area and retards the onset of smoltification and downstream migration.

Surface bypass collector. Collects downstream migrants and routes them through a low volume spillway or to collection area for downstream transport. This system reduces stress to juveniles because they do not experience the pressure changes associated with screen bypass systems. Also, fish enter the bypass near the surface which is where they are normally located in the water column.

Behavioral guidance structures. Attracts surface-oriented fish in the dam forebay and directs fish away for the powerhouse and towards the surface bypass collector.

Spillway flow deflectors. Decrease water turbulence as the water plunges over the dam. This reduces levels of total dissolved gas that are harmful to migrating juveniles. The mainstem Snake River from its confluence with the Clearwater River to its mouth at the Columbia River is under a TMDL that addresses total dissolved gases (TDG) (WA Ecology, 2003). TDG are elevated to levels that exceed state standards due to spill events at four hydroelectric dams on the Lower Snake River: Lower Granite, Little Goose, Lower Monumental, and Ice Harbor dams.

6.3.2 Spawning habitat

Of the ESA fish species addressed in this BE, only fall Chinook salmon would possibly use the Action Area for spawning. Installation of the Lower Granite Dam effectively eliminated Chinook salmon spawning habitat in most of the Action Area. Chinook require lotic habitat for spawning, with gravel/small cobble substrate with adequate water movement or upwelling to oxygenate eggs and to remove built up of nitrogenous waste. Groves and Chandler (1999) describe the range of fall chinook spawning habitat in the Snake River as having substrate-level water velocities of 0.1-2.1 m/s and substrate size of 2.5-15.0cm. Some incidental spawning by fall Chinook salmon has been found to occur in the tailrace of the Lower Granite Dam (Dauble et al., 1999). However, physical characteristic required for adequate spawning habitat are not found in the Action area above the dam.

6.3.3 Rearing and maturation habitat

Juvenile Chinook salmon rear in a wide-variety of environments ranging from small infertile streams to large rivers and impoundments. Rearing juvenile fall chinook have been documented to use the limited island shorelines and other shallow areas available in the Action Area. These areas are important habitat for rearing subyearlings and for short- term foraging for outmigrating yearling chinook and steelhead smolts. These areas have low gradient shoreline and fine sediment substrate.

6.3.4 Adult migration

Adult salmon/steelhead have an open deepwater migration corridor through the Reservoir that primarily provides migration space. Besides deep water cover, the reservoir habitat offers little habitat diversity in terms of substrate, velocity, cover, or riparian features. The confluence of the Snake/Clearwater does provide greater habitat value to migrants due to presence of pool habitat, greater flow velocity, and cold-water inflow from the Clearwater.

6.4 Conclusion

The major habitat use of this portion of the Snake River and the lower Clearwater River is as a salmon/steelhead migration corridor for juveniles and adults, holding area for steelhead adults, and rearing area for juvenile fall Chinook. As an adult migration corridor, the action area is not of high quality due to lack of essential features including cover/shelter. Habitat for outmigrating juveniles is also not of high quality due to reduced water velocity, lack of cover, abundance of predators, and the difficulty of passing through the dam. Spawning habitat that could be used by main stem spawning Chinook salmon is limited to tailrace areas as these are the only areas with adequate water velocity, depth and substrate. All other spawning habitat was inundated by the reservoir. Rearing habitat for juveniles is limited due to limited shallow areas with adequate cover/shelter.

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7 Analysis of Effects from the Action

7.1 Technical Approach for Analysis of Effects

This analysis consists of evaluating the incremental impact to threatened and endangered species due to the continued or new effluent limitations in the EPA draft NPDES permit for the Clearwater facility. The elements of the effects analysis are:

- Identifying and describing the parameters of concern in the effluent discharge, including discussions of the environmental baseline (receiving water concentrations), water quality standards, effluent limits, and exposure volume computations;
- Reviewing available toxicity data and identifying the lowest concentration at which toxicity was observed in toxicity tests for each species (or appropriate surrogate species) to establish levels which are considered safe (i.e., toxicity benchmarks);
- Distribution and mobility of the listed species; and
- Determining the likelihood that threatened and endangered species will be exposed to concentrations above those considered safe by evaluating the magnitude, frequency, and duration to determine the direct and indirect effects of this permit.

50 CFR 402 requires that:

> *"The Federal agency requesting formal consultation shall provide the Service with the best scientific and commercial data available or which can be obtained during the consultation for an adequate review of the effects that an action may have upon listed species or critical habitat."*

To comply with this requirement, the BE uses the following recently collected data:

- Parameter measurements in effluent from 2005 – 2016;
- Parameter measurements in the Clearwater River and Snake River upstream and downstream of the diffuser collected by Clearwater in 2005 and 2006 to characterize the environmental baseline; and
- Temperature (2005) and river velocity measurements (Cook et al., 2006).

Additionally, the BE includes information from a thorough review of the scientific literature regarding:

- Biology of the species;
- Characteristics of critical habitat; and
- Potential toxicity of parameters of concern.

All effluent values used in this analysis are final effluent concentrations that are calculated from the permit limitations required by this permit. These values represent the maximum effluent concentrations that are permissible under the 2019 draft permit.

Since the discharge is continuous, the effect to the species (listed species and their prey) would be from direct exposure to the discharge plume, including exposures to concentrations in the mixing plume of the discharge.

USFWS and NMFS provide guidance (USFWS and NMFS, 1998) for evaluating potential effects to listed species. According to USFWS and NMFS (1998), several different types of effects must be evaluated for each listed species: direct effects, indirect effects, cumulative effects, interdependent effects, and interrelated effects. In this BE, direct effects were evaluated for exposure of the listed species to the parameters of concern in the water, sediments, or their food. This will account for accumulation from both direct (uptake through the gills and direct ingestion of water) and indirect (sediment and food prey) pathways. Indirect effects include potential toxicity to prey or potential habitat degradation due to migration barriers or benthic smothering. Cumulative effects are addressed through the WET testing studies. Interdependent and interrelated effects are discussed in the uncertainty analysis.

The following methodology was used to determine effects:

1. The water column environmental baseline data was compared to the water column toxicity benchmark. The EPA's toxicity benchmarks in this BE are based on the no effect concentration for listed species or suitably sensitive surrogate species and the prey of listed species. For bioaccumulative parameters, sediment and fish tissue, environmental baseline data were also compared to the respective toxicity benchmarks, if available. When the environmental baseline was unknown or exceeds the toxicity benchmark, the analysis looked at the incremental impact of the river due to the discharge by assuming a small or immeasurable impact at the edge of the exposure volume.

2. The maximum effluent concentration, at the diffuser, was compared to the toxicity benchmark(s).

a. When the maximum effluent concentration, at the diffuser, was less than the toxicity benchmark(s), a determination of "not likely to adversely affect" was made.

b. When the maximum effluent concentration at the diffuser was greater than a toxicity benchmark, a dilution factor was applied. A computer simulation model (CORMIX v11.0 GTD) was used to determine that the available dilution factor at 25% of the stream width under 7-day, 10-year low river flow (7Q10) conditions for the month of September. September river flow conditions were used because the temperature stratification that occurs in early September results in relatively poor mixing, even though stream flow is slightly lower in October and December. Having identified early September conditions as the critical condition for mixing, the EPA ran additional modeling scenarios with the critical stream flow rates specified in IDAPA 58.01.02.210.03.b to evaluate mixing properties for acute and chronic aquatic life water quality criteria and for human health criteria for carcinogens and non-carcinogens. Modeling results evaluating conditions at 25% of the stream width, as well as mass balance calculations for 25% of the critical stream flow volumes, are summarized in Table 7-1, below.

Table 7-1. CORMIX Results (Nickel 2018).

Criteria Type	River Flow Statistic	River Flow Value (CFS)	Calculated Ambient Velocity (ft/s)	Dilution Factor at 25% of Stream Flow Volume (mass balance)	Dilution Factor at 25% of Stream Width
Acute Aquatic Life[2]	1Q10	14,061	0.234	72.9	35.9
Chronic Aquatic Life	7Q10	16,285	0.271	84.3	36.5
Chronic Aquatic Life (ammonia)	30Q10	18,457	0.307	95.4	37.5
Human Health Non-Carcinogen	30Q5	18,829	0.387	102	39.3
Human Health Carcinogen (July – September)[1]	Harmonic Mean	29,154	0.486	150	49.7
Human Health Carcinogen (October – June)[1]	Harmonic Mean	33,951	0.566	175	78.1

Notes:
1. In the harmonic mean scenario, a mixing zone encompassing 25% of the stream width would extend downstream past the Washington border. The State of Idaho cannot authorize a mixing zone that extends into another State. Thus, the conditions at the Washington border (191 meters downstream) are reported.
2. See discussion in Section 4.2.2.2 in USEPA 2018.

The mixing zone for chronic aquatic life criteria was used in the effects analysis in this BE. The chronic mixing zone has the following dimensions:

- Width: 152.5m (25% of the stream width of 610m)
- Downstream distance: 99.55m
- Plume thickness at mixing zone boundary: 4.00m
- Plume cross-sectional area at mixing zone boundary: 609 m^2 (10.9% of the river's cross-sectional area)
- Travel time to mixing zone boundary: 952 seconds (15.9 minutes)

If the concentration of a given parameter of concern is above the toxicity benchmark at the point of discharge, but below the toxicity benchmark at the edge of the chronic mixing zone, organisms will experience no adverse effects from the parameters of concern at points beyond the chronic mixing zone but may experience some adverse effects within the mixing zone. However, the EPA believes any such adverse effects will be insignificant due to the small size of the chronic mixing zone and the short time (15.9 minutes) necessary for the plume to reach the edge of the chronic mixing zone. Therefore, a determination of "not likely to adversely affect" was made for effects when the concentration at the edge of the chronic mixing zone (i.e., after applying a dilution factor of 36.5) was less than the toxicity benchmark. In cases where the background concentration is zero, the concentration at the edge of the chronic mixing zone is simply the maximum effluent concentration divided by the chronic dilution factor.

A determination of "likely to adversely affect" was made for direct effects when the concentration at the edge of the chronic mixing zone was greater than the toxicity benchmark.

For bioaccumulative parameters, the effluent concentration was converted to both tissue and sediment concentrations using procedures outlined in the Water Quality Guidance for the Great Lakes Watershed (USEPA 1995a), commonly known as the Great Lakes Initiative (GLI), and measured tissue concentration in both resident fish and caged

invertebrates and measured sediment concentrations were also used. A determination of "likely to adversely affect" was made for indirect effects when the effluent converted tissue and sediment concentrations or the measured values were above the tissue and sediment toxicity benchmarks.

Synergistic effects were determined from the whole effluent toxicity (WET).

The effects analysis in this BE assumes that the effect is to the species when they are present in the Action Area. Section V discusses when the species are present within the Action Area. The following summarizes the information from Section V:

- Bull trout are estimated to be in the Action Area from November through May. They are not known to reside or spawn in the Action Area but may use the Action Area as a migration corridor.
- Fall Chinook salmon are estimated to be in the Action Area during various life history stages throughout the year; adult migration occurs from May through November, spawning and incubation occurs from mid to late October through May, and smolt outmigration occurs from April through October. Spawning areas are limited to the tailrace areas below dams. Migrating adult fall Chinook require one to two days travel time between the Lower Granite Dam and confluence of the Snake and Clearwater rivers and may spend hours or days in the confluence of the Snake and Clearwater rivers due to cold-water refuges from the cooler Clearwater River. Average travel time for outmigrating juveniles, from the flowing portion of the Snake River to the Lower Granite Dam, is 43.5 days. Juveniles use shallows and shoreline areas for feeding and rearing during their outmigration.
- Sockeye salmon are estimated to be in the Action Area from April through mid-November. Adults use the Action Area only as a migration corridor, migrating from May through September. Smolts outmigrate through the Action Area from April through mid-November. Adults do not feed during their upstream migration; therefore, indirect effects to adults are not considered in this BE. Adult migrating sockeye may require two to two and one-half days to travel between the Lower Granite Dam and the confluence of the Snake and Clearwater rivers and may spend hours or days in the confluence due to cold-water refuges from the cooler Clearwater River.
- Spring/summer Chinook salmon are estimated to be in the Action Area from March through August. Adults use the Action Area only as a migration corridor, migrating from April through August. Smolts outmigrate from March through July. Migrating spring/summer adult Chinook may require up to four days travel between the Lower Granite Dam and the confluence of the Snake and Clearwater rivers and may spend hours or days in the confluence due to cold-water refuges from the cooler Clearwater River.
- Steelhead are estimated to be in the Action Area in various life stages throughout the year. Adults overwinter in the Action Area from October through March and migrate through the Action Area from mid-March through December to reach upstream spawning areas. Rearing pre-smolt juveniles may be present in the Action Area throughout the year. Smolts outmigration occurs from April through October. Migrating adult steelhead may require up to four days travel between

the Lower Granite Dam and the confluence of the Snake and Clearwater Rivers and may spend hours or days in the confluence due to cold-water refuges from the cooler Clearwater River.

- Orientation within the water column of ESA salmonid species in the Action Area can be generalized as follows: out-migrating juveniles are located throughout the water column with the greatest concentration in the upper 15 meters. Juveniles are often associated with shallow areas. Yearling juvenile steelhead have been captured at >18 meters throughout the Lower Granite Reservoir. Orientation of adults of each salmonid species within the water column is not known. However, hydroacoustic surveys of the Lower Granite Reservoir found large fish are typically near the bottom and that limnetic densities were low.

Additionally, the Action Area is designated critical habitat for Snake River Fall Chinook salmon, Spring/summer Chinook salmon, Sockeye, Snake River Steelhead, and bull trout. The effects analysis in this BE considered direct effects to prey species and fish passage, and indirect effects to the benthic community. Section VI provides a more in-depth discussion of the critical habitat in the Action Area.

7.2 Selection of Parameters of Concern

This section presents general information for the parameters of concern in the discharge and discusses the related environmental baseline, water quality standard, effluent limits, and toxicity benchmarks. The parameters of concern in this BE were selected because they have been measured in the effluent or they are controlled by effluent limitations guidelines (ELGs) for this industry. Not all the parameters of concern had "reasonable potential" for release into the ambient environment at concentrations above water quality standards in the development of effluent limits for the draft permit. Other pollutants may be present in the effluent discharge, but at concentrations that are well below the applicable water quality standards.

EPA conducted a study of the pulp, paper, paperboard industry to establish ELGs and standards reflecting the best practicable control technology currently available (BPT), best conventional pollutant control technology (BCT), and best available technology economically achievable (BPT) that would apply to this action (USEPA, 1993). The study included a review of existing regulations, a review of available literature, and an evaluation of existing data, data obtained from an industry-wide questionnaire, data from foreign mills, as well as data obtained from short- and long-term sampling at 19 separate facilities. EPA identified 24 pollutants or pollutant parameters as present in pulp, paper, and paperboard wastewaters and determined them as parameters to consider for limitations under the BPT, BCT, and BAT ELGs. In addition, temperature, fecal coliform bacteria, turbidity, sulfate, surfactants, ammonia, dissolved oxygen, nutrients (nitrogen and phosphorus), and whole effluent toxicity are included as non-conventional pollutants of concern specific to this discharge. The pollutants of concern are as follows:

7.2.1 Conventional Pollutants

- Biochemical oxygen demand (BOD),
- Total suspended solids (TSS), and
- pH
- Fecal coliform bacteria

7.2.2 Nonconventional Pollutants

- Adsorbable organic halides (AOX)
- Ammonia
- Chemical oxygen demand (COD)
- Color
- Dissolved oxygen (DO)
- Nutrients
- Tetrachlorocatechol (TeCC)
- Tetrachloroguaiacol (TeCG)
- 2,4,5-trichlorophenol (2,4,5-TCP)
- 3,4,5-trichlorocatechol (3,4,5-TCC)
- 3,4,5-trichloroguaiacol (3,4,5-TCG)
- 3,4,6-trichlorocatechol (3,4,6-TCC)
- 3,4,6-trichloroguaiacol (3,4,6-TCG)
- 4,5,6-trichloroguaiacol (4,5,6-TCG)
- 2,3,4,6-tetrachlorophenol (2,3,5,6-TeCP)
- 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8-TCDF)
- Trichlorosyringol
- Temperature
- Whole effluent toxicity (WET)

7.2.3 Priority Pollutants

- Antimony (Sb)
- Arsenic (As)
- Chloroform
- Copper (Cu)
- Hexavalent Chromium (Cr VI)
- Lead (Pb)
- Nickel (Ni)
- Pentachlorophenol (PCP)
- 2,4,6-trichlorophenol (2,4,6-TCP)
- 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD)
- Thallium (Th)
- Zinc (Zn)

Conventional pollutants are those defined in 40 CFR 401.16, namely TSS, BOD, oil and grease, fecal coliform, and pH. Analytical measures of TSS, BOD, and oil and grease are not chemical-specific determinations but aggregate measures of suspended particulates, oxygen-demanding substances, and Freon-extractable substances in water, respectively. Specific compounds contributing to these measures may or may not exhibit toxic effects and may or may not be among the 126 priority pollutants defined by the CWA. The priority pollutants are specifically designated elements or compounds that exhibit toxic effects in aquatic systems and, if determined to be present at significant levels, must be regulated by categorical technology-based effluent limitations guidelines and standards pursuant to Section 301(b)(2)(A) of the CWA.

Non-conventional pollutants are all other pollutants that are neither the five listed conventional pollutants nor the designated 126 priority pollutants. Non-conventional pollutants may be aggregate measures such as COD or AOX or specific elements or compounds such as chlorine, ammonia, and 2,3,7,8-TCDF. The agency has the authority and discretion to limit non-conventional pollutants in categorical effluent limitations guidelines and standards as appropriate based upon the presence of these pollutants and findings that the removal or treatment of the pollutants is technically and economically achievable.

Additionally, EPA must establish water quality-based effluent limitations to control all pollutants or pollutant parameters (either conventional, non-conventional, or toxic pollutants) which EPA determines 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.

A total of 443 specific pollutants were the subject of extensive study for the ELGs (USEPA, 1993). These 443 pollutants included 124 of the 126 priority pollutants and 319 non-conventional pollutants. Asbestos and cyanide were the two priority pollutants not included because they are not expected to be present at concentrations of concern in pulp and paper mill effluents.

Of the 443 pollutants that were analyzed as part of the development of the ELGs, 363 were not detected in the final effluent with the use of analytical methods promulgated pursuant to Section 304(h) of the CWA or with other state-of-the-art methods and 28 were detected at levels below concentrations of concern. EPA eliminated pollutants as pollutants of concern for the development of the ELGs that were not detected in the final effluent or that were detected at levels below concentrations of concern. Appendix F provides these pollutants and the maximum concentration or the detection limits. Although the analytical detection levels of many of the pollutants that were not detected are greater than the water quality criteria, EPA did not include water quality-based effluent limits for these parameters because they are effectively controlled through the limitation of similar pollutants (40 CFR 122.44(d)(1)(vi)(c)).

EPA eliminated the following pollutants that were detected in the effluents for the following reasons:

- Titanium was eliminated because this pollutant was detected at concentrations below those considered treatable. There is currently no water quality standard for this parameter and there is no reason to believe that this parameter would be present in the effluent above concentrations in the intake water to the Clearwater Mill from the Clearwater River.
- Even though other dioxin and furan congeners may be present in the effluent, studies EPA conducted during the development of the ELGs (USEPA, 1993) showed that 2,3,7,8-TCDD and 2,3,7,8-TCDF were the predominant chlorinated dibenzo-*p*-dioxins (CDDs) and chlorinated dibenzo-*p*-furans (CDFs) found in pulp and paper matrices. The EPA is proposing to regulate 2,3,7,8-TCDD and 2,3,7,8-TCDF and, in so doing, will effectively minimize generation of the most toxic CDDs and CDFs.

- Acetone, methylene chloride, and methyl ethyl ketone (MEK) are volatile organic compounds that are not expected to be present in pulp and paper mill effluents. EPA has reviewed data from both hardwood and softwood mills employing a variety of bleaching processes in an effort to identify factors that contribute to the formation of acetone, methylene chloride, and MEK in the bleach plant. Acetone, methylene chloride, and MEK are used in analytical chemistry laboratories during sample preparation procedures. Sometimes, concentrations of these compounds are reported in environmental samples as a result of the sample preparation steps using these compounds and not because the compounds were actually present in the environmental samples. In the EPA and Industry long-term study (USEPA, 1993), methylene chloride was found to be a sample- and laboratory-contaminant. EPA believes that this is the case with acetone and MEK as well. Consequently, because these compounds are most likely associated with laboratory contamination and are not likely present in effluent, these compounds are not evaluated in the BE.
- Other pollutants (magnesium, manganese, potassium, sodium, and sulfur) were eliminated because they were detected at concentrations not considered treatable with end-of-pipe treatment technologies suitable for large effluent flows. There are currently no water quality standards for these parameters, except for manganese, which has only been measured in the Clearwater Mill effluent at concentrations below the applicable water quality criteria.

The draft 2019 permit specifies effluent limitations or monitoring requirements for 22 parameters, including 19 chemicals (12 chlorinated organic compounds, chloroform, thallium, 2,3,7,8-TCDD and 2,3,7,8-TCDF), AOX, BOD, TSS, pH, and temperature. Other parameters in addition to the 20 limited parameters, which include COD, color, ammonia, nutrients (including ammonia as Total N, nitrate+nitrite, and phosphorous) and metals (including antimony, arsenic, chromium, copper, lead, nickel, thallium and zinc), were also identified as parameters of concern in the effluent and discussed in this BE, although they are not limited by the 2019 draft permit. Based on the available data, EPA has established effluent limitations for the following parameters in this discharge:

7.2.4 Conventional Pollutants

- Adsorbable Organic Compounds (AOX)
- Biochemical oxygen demand (BOD)
- Total suspended solids (TSS)
- pH

7.2.5 Nonconventional Pollutants

- Tetrachlorocatechol (TeCC)
- Tetrachloroguaiacol (TeCG)
- 2,4,5-trichlorophenol (2,4,5-TCP)
- 3,4,5-trichlorocatechol (3,4,5-TCC)
- 3,4,5-trichloroguaiacol (3,4,5-TCG)
- 3,4,6-trichlorocatechlol (3,4,6-TCC)
- 3,4,6-trichloroguaiacol (3,4,6-TCG)
- 4,5,6-trichloroguaiacol (4,5,6-TCG)

- 2,3,4,6-tetrachlorophenol (2,3,4,6-TeCP)
- 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8-TCDF)
- Trichlorosyringol
- Temperature

7.2.6 Priority Pollutants

- Chloroform
- Pentachlorophenol (PCP)
- 2,4,6-trichlorophenol (2,4,6-TCP)
- 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD)

7.3 Selection of Environmental Baseline

Regulations implementing the ESA (50 CFR 402.02) define the environmental baseline as the past and ongoing impacts of all Federal, State, or private actions and other human activities leading to the current status of a species, its habitat, and ecosystem within the Action Area. Also included in the environmental baseline are the anticipated impacts of all proposed Federal projects in the Action Area that have undergone section 7 consultation, and the impacts of state and private actions which are contemporaneous with this consultation. Therefore, the environmental baseline would include the current and past discharges from this facility and this BE also compares the proposed action to the current permit issued to this facility. The environmental baseline may not be known for all parameters of concern because they either have not been measured in the Action Area or they were not detected in the Action Area.

7.4 Selection of Toxicity Benchmarks

Toxicity benchmarks are derived from studies that are conducted using the species of interest, exposed the test species to the chemical of interest for a relatively long period of time, monitored sensitive endpoints, and representative of how exposures occur in a natural setting. The toxicity benchmarks were based on the no effect concentration for listed species or suitably sensitive surrogate species and the prey of listed species.

A thorough review of the scientific literature was conducted to identify as many sources of toxicity data for these parameters as possible. In some cases, toxicity data were obtained from a previously compiled collection of toxicity information. In other cases, individual papers published in the scientific literature were reviewed. In still other cases, the toxicity data used by EPA to derive water quality criteria were reviewed.

The quantity and quality of toxicity data available for permitted parameters varies widely. For some parameters, dozens of toxicity studies (examining several different types of toxicity) have been published, while other parameters may only one or a few published toxicity studies. In some studies, potential toxicity to listed species was examined, while other studies looked for toxicity in non-listed salmonids or non-salmonid aquatic species. Still other studies obtained toxicity data from experiments using a single chemical in a controlled exposure setting (such as an aquarium).

Accordingly, toxicity studies with certain characteristics were considered unsuitable for use in the BE and were eliminated from the collection of toxicity data for permitted parameters. These studies were considered unacceptable because they were incomplete.

USEPA's ECOTOX database reports the level of study "completeness." A "complete" study thoroughly described the methods and results of the experiment. A "moderate" study was generally considered satisfactory, but one or more pieces of information about the methods or results were missing. A study is rated "incomplete" when important information about the study's methods or results is not reported. Data from "incomplete" studies were not used in the BE.

Ideally, each permitted parameter would have been studied in toxicity tests using each life stage of each listed species and in the prey and predators of each listed species. Furthermore, each toxicity study would have:

- Evaluated the parameter at several different concentrations or amounts and reported a dose-dependent increase in toxicity with increasing concentration or amount;
- Reported a no observed effect level (NOEL);
- Used a chronic duration; and
- Observed toxicity to individual organisms, such as reduced growth or reproductive impairments that compromised the survival or reproductive capacity of the organism.

Actual direct testing of potential toxicity has not been conducted for all chemicals and listed species. While some toxicity data have been collected for nearly all the parameters of concern, toxicity data are generally not available for every life stage of a listed species. In cases where little or no toxicity data are available for a parameter of concern to each life stage of a listed species, toxicity data from a similar parameter, species, or life stage was used as a surrogate.

In some cases, the study reporting the lowest concentration for a parameter did not report a 'no observed effect concentration' (NOEC) endpoint. For direct effects, the lowest endpoint reported was used and then NOEC endpoints were extrapolated using safety factors (e.g., if only a LC_{50} was reported, then a safety factor would be applied to obtain an estimated LOEC and another safety factor would be applied to obtain an estimated NOEC). For indirect effects (affects to prey species) only one safety factor was used to extrapolate from an effect concentration to a no effect concentration.

Although using surrogate toxicity data from a similar species, life stage, or parameter increases the uncertainty associated with the BE, this approach is preferable to omitting the evaluation of a species or parameter with no toxicity data. The following subsections describe the BE's approach to assign surrogate toxicity data to a parameter or species when ideal toxicity data are unavailable.

7.4.1 Surrogate species

In general, few toxicity studies have been conducted using listed species. However, toxicity studies using similar or other highly sensitive (i.e., indicator) species have often been conducted and can be used as a surrogate for the non-tested species of interest. In judging whether other (tested) species can be used as a surrogate for listed species, it is important to know whether the tested species is more sensitive than, less sensitive than, or about equally sensitive as the listed species. In this case, "sensitivity" refers to the relative severity of the observed toxicity in one species as compared to the other. A

highly sensitive species exposed to a certain concentration of a parameter would experience more severe toxicity than a less sensitive species exposed to the same concentration.

When a tested species is more sensitive or about equally sensitive to a non-tested species, the tested species can be considered a suitable surrogate for the non-tested species. The comparative sensitivity of listed species and surrogate species can be ascertained by comparing the toxicity observed in surrogate species to the toxicity observed in other species exposed to certain well-studied chemicals. Dwyer et al. (1995) used this type of comparative sensitivity approach to estimate the potential toxicity of several chemicals to endangered and threatened fish species for which no toxicity data were available. Generally, the rainbow trout (*Oncorhynchus mykiss*) has been considered a suitable surrogate for coldwater fishes, and the Fathead minnow (*Pimephales promelas*) has been considered a suitable surrogate for warm water fishes (Dwyer et al., 1995).

7.4.2 Chemical surrogate

For some parameters with effluent limits in the draft permit, little or no toxicity data using aquatic species are available. Therefore, parameter-specific toxicity data cannot be used to assess potential effects to listed species. Toxicity data for a similar parameter were used as surrogates for the following parameters:

- No aquatic toxicity studies were found for certain chlorinated phenolic compounds (i.e., 3,4,5-trichlorocatechol, 3,4,6-trichlorocatechol, 3,4,6-trichloroguaiacol, 4,5,6-trichloroguaiacol, and trichlorosyringol). Toxicity data are available, however, for structurally similar compounds, and will be used in the BE as a surrogate. This BE uses the benchmarks established for 2,4,5-trichlorophenol as surrogates for direct and indirect toxicity for threatened and endangered salmonids because it had the lowest direct effect of compounds with similar chemical structure and properties.
- Whole effluent toxicity is facility-specific so no data from literature can help to evaluate the level of protection provided by the permit limits. Therefore, this BE uses EPA's recommended magnitude of 1 TU_c for WET as a surrogate for direct and indirect toxicity for threatened and endangered salmonids.

7.4.3 Life stage surrogate

A review of the scientific literature found that younger life stages of fish are generally more sensitive to chemical toxicants than older fish (e.g., Buhl, 1997; Hutchinson et al., 1998), though this was not always found to be the case. Mayes et al. (1983) did not find fathead minnow fry, juveniles, or adults to vary significantly in sensitivity to nine organic compounds tested. Additionally, Ingersoll et al. (1990) found that the sensitivity of brook trout to aluminum toxicity increased with age. Relative sensitivity likely varies depending on the substance used in the toxicity test, the toxicological effect observed (e.g., survival, growth) (Pickering et al., 1996), and the endpoint measured (e.g., NOEC, LOEC). This seems to be the case with aquatic invertebrates. Hutchinson et al. (1998) analyzed EC_{50} and NOEC data from the European Centre for Ecotoxicology and Toxicology of Substances (ECETOC) Aquatic Toxicity database and found that based on EC_{50} data, juvenile invertebrates exhibited equal or greater sensitivity than adults to 54%

of substances, while they exhibited equal or greater sensitivity than adults to 91% of substances based on NOEC data. While some investigators have found that younger invertebrates are more sensitive to some contaminants than older life stages (Nebeker et al., 1984), others found that older and younger invertebrates exhibited similar sensitivities to acute toxicity (Nebeker et al., 1986).

7.5 Discussion of Parameters of Concern

7.5.1 Parameters Limited and Monitored in Permit

7.5.1.1 Chloroform

7.5.1.1.1 Introduction

Chloroform ($CHCl_3$), also known as trichloromethane or methyl trichloride, at ordinary temperatures and pressures, is a clear, colorless, volatile liquid with a pleasant, etheric, nonirritating odor and sweet taste (Hardie, 1964; Windholz, 1976). It has a boiling point range of 61-62ºC, a melting point of –63.5ºC, and is nonflammable. There is no flash point (Hardie, 1964; Windholz, 1976). The n-octanol/water partition coefficient (K_{ow}) for chloroform is 83 (log K_{ow}=1.9). Chloroform is slightly soluble in water (7.42×10^6 µg/L of water at 25ºC). It is miscible with alcohol, benzene, ether, petroleum ether, carbon tetrachloride, carbon disulfide, and oils (Windholz, 1976). Chloroform is highly refractive and has a vapor pressure of 200 mm Hg at 25ºC (Irish, 1962; Windholz, 1976). Because of its volatile nature, chloroform has the potential for evaporation to the air from pollution sources or from the water column.

Most of the chloroform found in the environment comes from industry. Chloroform was one of the first inhaled anesthetics to be used during surgery, but it is not used for anesthesia today. Nearly all the chloroform made in the United States today is used to make other chemicals, but some is sold or traded to other countries. Chloroform enters the environment from chemical companies and paper mills. It is also found in wastewater from sewage treatment plants and drinking water to which chlorine has been added. Chlorine is added to most drinking water and many wastewaters to destroy bacteria. Small amounts of chloroform are formed as an unwanted product during the process of adding chlorine to water. Chloroform can enter the air directly from factories that make or use it and by evaporating from water and soil that contain it. It can enter water and soil when wastewater that contains chlorine is released into water or soil. It may enter water and soil from spills and by leaks from storage and waste sites. There are many ways for chloroform to enter the environment, so small amounts of it are likely to be found almost everywhere.

Chloroform evaporates very quickly when exposed to air. Chloroform also dissolves easily in water but does not adhere to the soil very well. This means that it can travel down through soil to groundwater where it can enter a water supply. Chloroform lasts for a long time in both the air and in groundwater. Most chloroform in the air eventually breaks down, but this process is slow. Some chloroform may break down in soil. Chloroform does not appear to build up in great amounts in plants and animals, but some small amounts of chloroform may be found in foods (ATSDR, 1997). McConnell et al. (1975) reviewed the incidence, significance, and movement of chlorinated hydrocarbons

in the food chain. They concluded that chloroform is widely distributed in the environment and is present in fish, water birds, marine mammals, and various foods.

Chloroform is an extremely volatile compound that is generated during the bleaching of pulp with hypochlorite, chlorine, or chlorine dioxide. Hypochlorite bleaching results in the greatest amount of chloroform generation while chlorine dioxide bleaching results in the least amount of chloroform generation. Because the Clearwater Mill uses 100 percent chlorine dioxide for bleaching, this results in the formation of low levels of chloroform. As chloroform is generated, it partitions to the air, and to the bleach plant effluent (though, some of the chloroform remains with the pulp).

At ambient environmental temperatures, chloroform is thermostable and resists decomposition (Hardie, 1964). However, slow decomposition occurs following prolonged exposure to sunlight and in darkness when air is present (Hardie, 1964). There is no appreciable decomposition of chloroform at ambient temperatures in water, even in the presence of sunlight (Hardie, 1964). Aqueous degradation of chloroform is accelerated in the presence of aerated waters and metals, such as iron, with hydrogen peroxide representing a reaction product (Hardie, 1964).

Environmental persistence data indicate that several removal mechanisms may be responsible for reducing concentrations of chloroform in river systems. Information in the Hazardous Substances Data Base (HSDB, 2000) indicates that volatilization, photolysis, and biodegradation have been identified as removal mechanism. With a half-life of 36-hours in a stream system (HSDB, 2000), volatilization is the primary removal mechanisms. Half-lives for photolysis (a few months) and biodegradation (one week to a few months) were substantially longer than the half-life for volatilization (HSDB, 2000; Howard et al., 1991).

7.5.1.1.2 ii. Environmental Baseline

Chloroform sampling of surface water and groundwater performed in 2005 and 2006 as part of NPDES compliance monitoring have indicated only non-detects. The only chloroform found at any appreciable level was in the direct effluent from Outfall 001. The range found from sampling Outfall 001 in 2002 was 1.4 – 2.5 µg/L, with an average concentration of 1.95 µg/L, well below the water column benchmark of 12.4 µg/L. Even if the environmental baseline were assumed to be at the criterion of 5.7 µg/L, the environmental baseline would still be below levels that are considered safe for threatened and endangered salmonids.

7.5.1.1.3 Water Quality Standard

The most stringent water quality standard in Idaho and Washington for chloroform is Idaho's Clean Water Act effective criterion of 5.7 µg/l as a long-term average for the protection of human health.

7.5.1.1.4 Effluent Limitation

Any chloroform found in bleach plant effluent that is not emitted to the air prior to reaching the wastewater treatment plant is volatilized and degraded during secondary treatment. Any residual chloroform that remains in the pulp may be found in the fraction

of untreated pulp that may comprise a fraction of the total suspended solids in the effluent.

The 2019 draft permit establishes separate effluent limits for the chip and sawdust fiber lines. The limits for the two fiber lines sum to a maximum limit of 21.7 lb/day and an average monthly fiber line limit of 13.0 lb/day. The equivalent chloroform maximum daily and average monthly concentrations in the final effluent due to the fiber line limitations would be **82.2 µg/L** and **49.2 µg/L**, respectively.

The 2005 permit consisted of a maximum daily fiber line limit of chloroform of 28.8 lb/day and an average monthly fiberline limit of 17.2 lb/day. The equivalent chloroform maximum daily and average monthly concentrations in the final effluent due to the fiberline limitations would be 86.3 µg/L and 51.5 µg/L, respectively. The proposed limitations are more stringent than the 2005 permit.

7.5.1.1.5 Toxicity Benchmarks

7.5.1.1.5.1 Direct Effects

Chloroform has been most commonly tested under static conditions with no measurement of the concentrations of chloroform to which the organisms are exposed. Consequently, the acute toxicity database will probably underestimate the toxicity because concentrations in static tests are likely to diminish during the progress of the exposure due to loss from water to air.

Several studies have measured the concentration of chloroform required to cause mortality in aquatic organisms. The range of LC50 values (the concentration that is expected to be lethal to 50% of the organisms tested) for the salmonid, *Oncorhynchus mykiss*, was 1,240 µg/L to 67,500 µg/L (Bentley et al., 1979; Birge et al., 1979; Black and Birge, 1980; Qureshi et al., 1982). The NOECs of 24,000 µg/L and 42,000 µg/L have also been reported for *O. mykiss* (Bentley et al., 1979). This seemingly high level of variability in test endpoint values can be attributed to variations in testing including exposure duration and magnitude and possible differences in test organism characteristics (e.g., age), as well as varying test conditions such as test temperature and type of water used.

Birge et al. (1993) studied potential avoidance brought on by chloroform in rainbow trout. In an acute (20 minute) laboratory test under flow-through conditions, the NEC, maximum acceptable toxicant concentration (MATC), and LOEC for trout exposed to chloroform were 4,180 µg/L, 7,050 µg/L, and 11,900 µg/L.

Two studies by Slooff (1978, 1979) examined physiological toxicity of chloroform on rainbow trout (*O. mykiss*). Both found changes in trout physiology at chloroform concentrations of 20,000 µg/L in acute (24 hour) flow through laboratory tests.

The direct toxicity concentration (the lowest concentration observed to cause direct toxicity to salmonid species) for chloroform is 1240 µg/L, which is based on the LC50 value reported by Birge and Black (1979). The behavioral endpoint reported by Birge et al. (1993) for rainbow trout was not selected because it was a very short-term bioassay (20 minutes). Application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC) to the direct toxicity concentration would generate a benchmark of 12.4 µg/L.

This BE uses 12.4 µg/L as the direct water column toxicity benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.1.5.2 Indirect Effects

Non-salmonid fish have been found to have chloroform LC50s ranging from 2,030 µg/L to 660,000 µg/L. The NOECs ranged from 75,000 µg/L in bluegill sunfish to 122,000 µg/L in medaka (Slooff, 1978 and 1979; Hazdra et al., 1979; Bentley et al., 1979; Black and Birge, 1980; Mattice et al., 1981; Mayes et al., 1983; Schell, 1987).

Birge et al. (1980) reported an EC50 of 270 µg/L for developmental toxicity in spring peeper.

Other behavior studies have been conducted using bluegill and green sunfish (*Lepomis macrochirus* and *L. cyanellus*). Avoidance behavior was observed at concentrations ranging from 20,000 µg/L to 33,200 µg/L (Summerfelt and Lewis, 1967; Black and Birge, 1980; Birge et al., 1993). Behavioral tests using the invertebrate *Cypris subglobosa* (Khangarot, B.S., and S. Das, 2009) have indicated immobilization at concentrations of 2,803 and 4325 µg/L (EC50).

Histological toxicity (which could include necrosis, edema, lesions, and hemorrhaging) has been observed in fish (i.e., medaka, *Oryzias latipes*) exposed to chloroform. One or more of these types of toxicity were found to occur in *O. latipes* at chloroform concentrations of 100,000 µg/L over a period of 10 days (Schell 1987).

Schell (1987) exposed the medaka (*O. latipes*) to 215,000 µg/L of chloroform over a 10-day period. Over that time period, changes in medaka physiology were observed.

Mortality tests done using the invertebrates *Brachionus calyciflorus*, *Ceriodaphnia dubia*, *Daphnia magna*, *Lumbriculus variegatus*, and *Penaeus duorarum* have resulted in LC_{50}s ranging from 2,000 µg/L to 758,000 µg/L and NOECs of 3,400 µg/L (Ceriodaphnia), 120,000 µg/L (Daphnia), and 32,000 µg/L (Penaeus) (Bentley et al., 1979; LeBlanc, 1980; Qureshi et al., 1982; Gersich et al., 1986; Snell et al., 1991; Cowgill and Milazzo, 1991; Rogge and Drewes, 1993).

The reproductive toxicity of chloroform has been studied in two species of invertebrates, *D. magna* and *C. dubia*. Chloroform was found to change the reproductive capabilities of 50% of the *D. magna* examined (EC50) at concentrations ranging from 288,000 µg/L to 336,000 µg/L in a 9 to 11-day laboratory study (Cowgill and Milazzo, 1991). The EC50 for reproductive toxicity on Ceriodaphnia dubia was 311,000 µg/L to 368,000 µg/L (Kuhn et al., 1989; Cowgill and Milazzo, 1991). The NOECs for reproductive toxicity were reported for Daphnia (6,300 µg/L to 200,000 µg/L) and Ceriodaphnia (200,000 µg/L) (Kuhn et al., 1989; Cowgill and Milazzo, 1991).

For prey species, the lowest observed effect was related to developmental toxicity in the spring peeper (270 µg/L). Using a factor of 10 to convert from an effect concentration to a no effect level, the toxicity benchmark is 27 µg/L for non-salmonid prey. **This BE uses 27 µg/L as an indirect toxicity benchmark for prey species.**

7.5.1.1.6 Effects Analysis

Since the maximum effluent chloroform concentration allowed under this permit (56.3 µg/L) is greater than the direct water column toxicity benchmark (12.4 µg/L), this analysis looks at the direct effects within the exposure volume of the effluent (i.e., the area where the concentration of the plume exceeds the toxicity benchmark) and the effects at and beyond the exposure volume boundary.

7.5.1.1.6.1 Direct Effects

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the chronic mixing zone would be 36.5, therefore the maximum exposure concentration at the edge of the mixing zone would be the maximum daily limit of 82.2 µg/L divided by the available dilution of 36.5, or 2.25µg/L. The calculated maximum exposure concentration of 2.25 µg/L is less than the toxicity benchmark of 12.4 µg/L and the water quality standard of 5.7 µg/L.

Therefore, EPA concludes that the discharge of chloroform at the maximum effluent concentration **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.1.6.2 Indirect Effects

As predicted by the model, the available dilution (36.5) is enough to reduce the maximum exposure concentration below indirect effect toxicity benchmark of 27 µg/L.

At and beyond the exposure volume, the permit limits are designed to protect the water quality standard for chloroform (5.7 µg/L). Since the water quality standard is almost five times lower than the indirect toxicity benchmark (27 µg/L), it is not likely that prey species would be exposed to unsafe levels of chloroform.

Therefore, the discharge of chloroform **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.1.6.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA concludes that the discharge of chloroform at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.2 Hydrogen Ion (pH)

7.5.1.2.1 Introduction

The definition of pH is the negative logarithm of the hydrogen "activity" (APHA, 1998). It is mathematically related to hydrogen ion activity according to the expression:

pH = -log [H^+], where [H^+] is the hydrogen ion activity. In dilute solutions, hydrogen ion activity is approximately equivalent to the molar concentration of hydrogen ions (APHA, 1998). According to APHA (1998), pure water has a pH of 7.0 standard units (su), but in equilibrium with atmospheric carbon dioxide, the pH of distilled water is approximately 5.6 su. Solutions with a pH above 7 indicate that the solution is alkaline, while a pH below 7 indicates that the solution is acid.

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. The principal system regulating pH in natural waters is the carbonate system with is composed of carbon dioxide (CO_2), carbonic acid (H_2CO_3), bicarbonate ion (HCO_3), and carbonate ions (CO_3). Stumm and Morgan (1970) have described the interactions and kinetics of this system. Because of the nature of the chemicals causing alkalinity, and the buffering capacity of carbon dioxide in water, very high pH values are seldom found in natural waters.

pH is an important factor in the chemical and biological systems of natural waters. The degree of dissociation of weak acids or bases is affected by changes in pH. This effect is important because the toxicity of many compounds is affected by the degree of dissociation.

The pH of a water body does not indicate ability to neutralize additions of acids or bases without appreciable change. This characteristic, termed "buffering capacity," is controlled by the amounts of alkalinity and acidity present.

7.5.1.2.2 Environmental Baseline

From 2005-2006, a groundwater monitoring program collected samples from 8 different sites adjacent to an aerated stabilization basin for the facility. In 2005, a minimum pH of 6.0, and a maximum pH of 10.04, were recorded in 2005; in 2006, the minimum and maximum pH recordings were 5.94 and 9.86, respectively (JUB Engineers 2006 a & b and 2007). Average readings are shown in Figure 7-1 and Figure 7-2 and Table 7-2 and Table 7-3.

A surface water study was conducted in 2005, in which water quality measurements from both the facility's effluent, and waters above and below the facility were measured. Mean pH was largely stable at all locations, and ranged from 7.31 to 8.72, as seen in Figure 7-3 and Figure 7-4(AMEC Earth and Environmental 2006). In 2006, as part of the NPDES annual monitoring report, pH measurements were collected during the weekly receiving water monitoring study; the mean pH of these readings fluctuated between 7.62 to 9.05, generally within the range of the IDEQ water quality standard of 6.5 to 9.0 (AMEC Earth and Environmental 2007). These readings are shown in Figure 7-4 and Table 7-5.

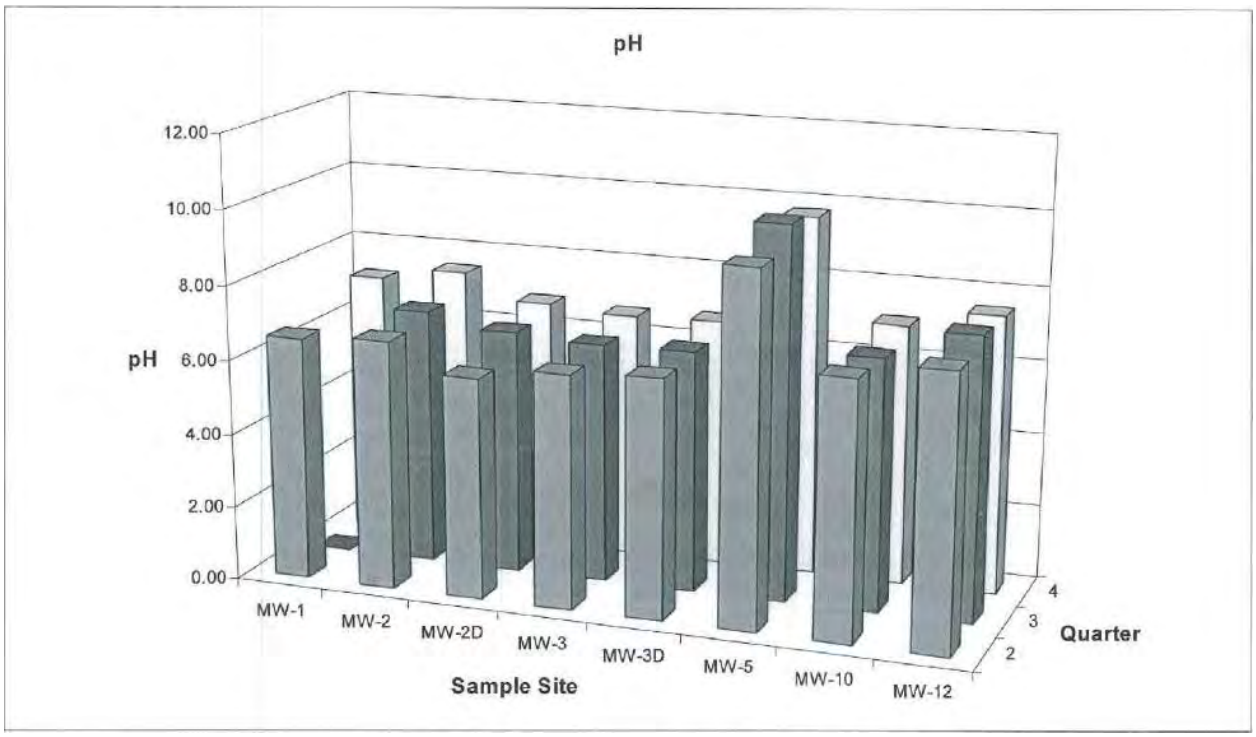


Figure 7-1: Average pH readings from 4PthP quarter addendum to 2005 groundwater monitoring results.

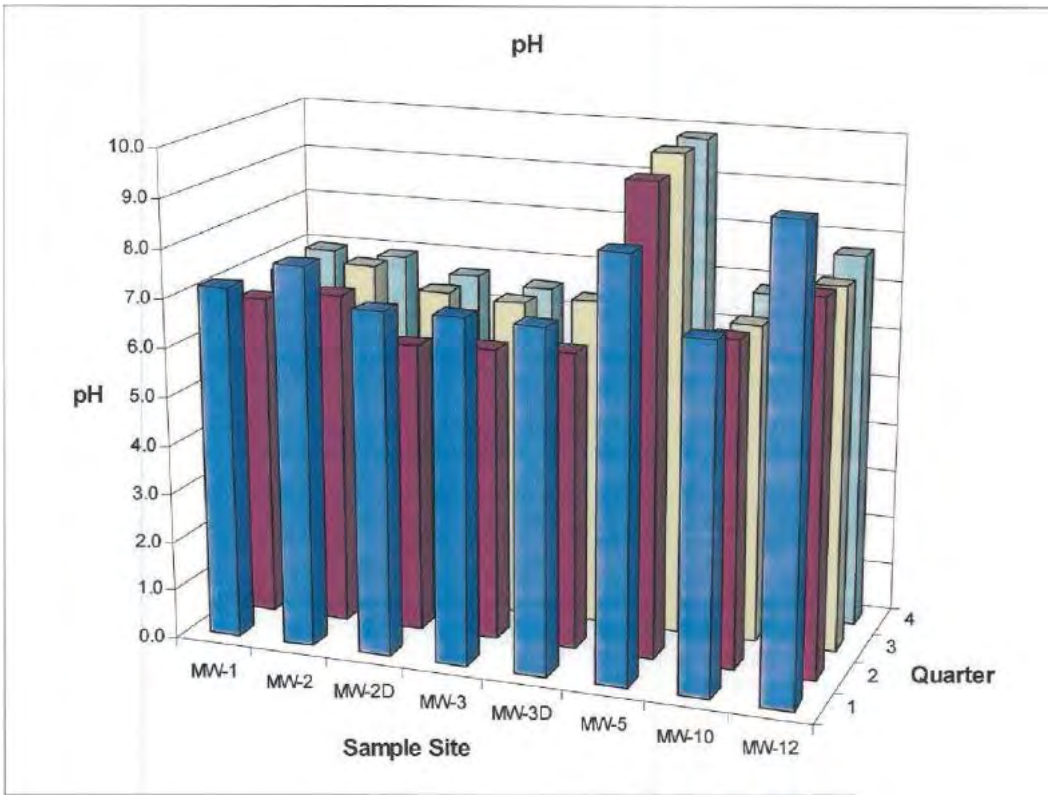


Figure 7-2: Average pH readings from 2006 groundwater monitoring results from 8 sites adjacent to an aerated stabilization basin.

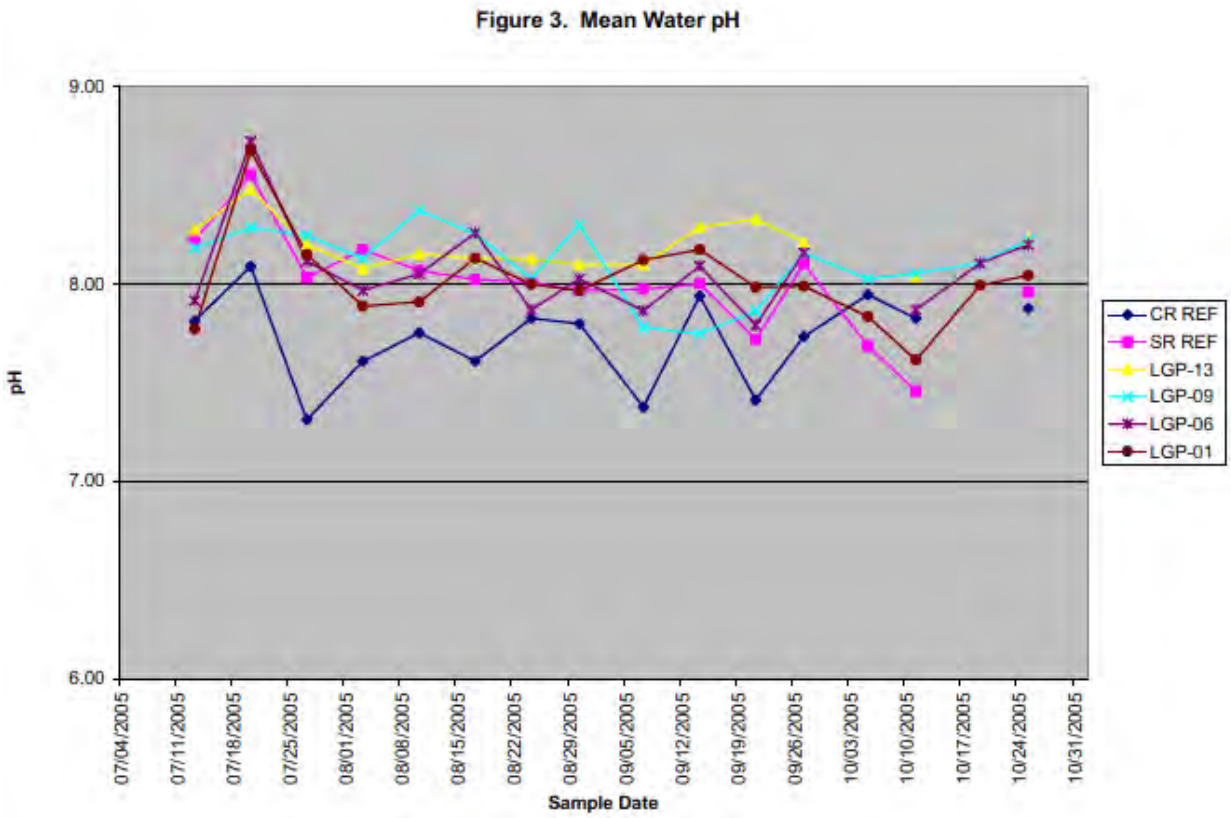


Figure 7-3: Average pH readings from 2005 surface water study at sites above and below facility, and within effluent from facility.

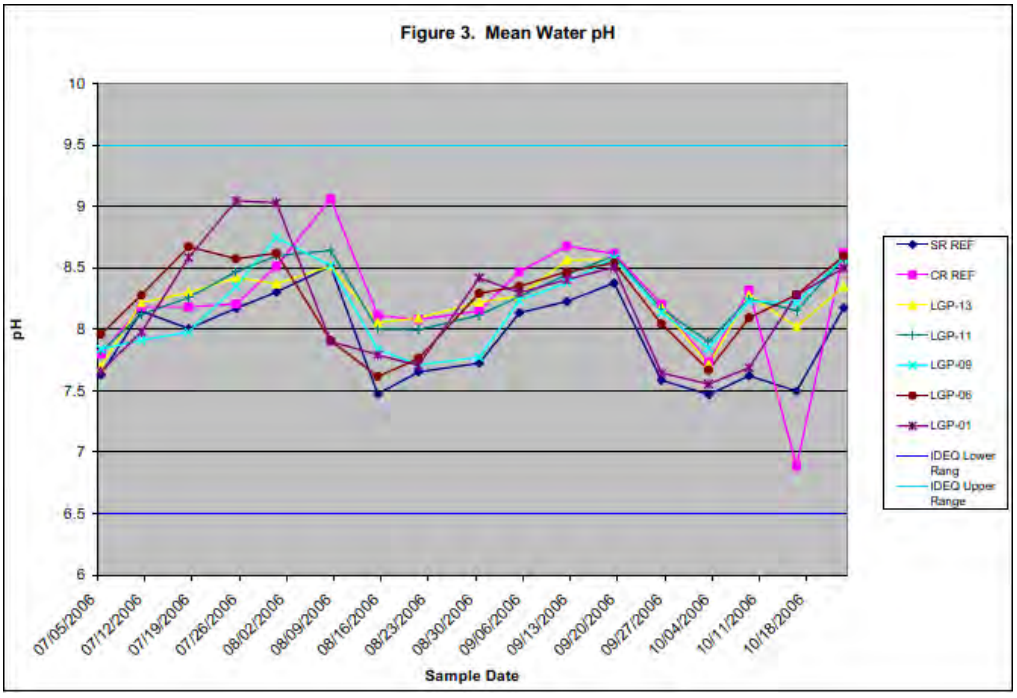


Figure 7-4: Mean pH results from 2006 weekly receiving water monitoring study.

Table 7-2: Average pH readings from 4th quarter addendum to 2005 groundwater monitoring results

Sample Site	2nd Quarter S.U.[2]	No. of Observations	3rd Quarter[1] S.U.[2]	No. of Observations	4th Quarter S.U.[2]	No. of Observations
MW-1	6.56	1	Note 3	0	7.13	2
MW-2	6.71	2	6.93	1	7.48	2
MW-2D	5.95	2	6.58	1	6.80	2
MW-3	6.28	2	6.45	1	6.67	2
MW-3D	6.41	2	6.49	1	6.73	2
MW-5	9.44	1	10.04	1	9.74	2
MW-10	6.88	2	6.77	1	7.04	2
MW-12	7.30	2	7.59	1	7.52	2

Notes:

[1] pH not adjusted for temperature
[2] Standard Units
[3] pH sample not taken due to equipment failure

Table 7-3:2006 Groundwater monitoring results from 8 sites adjacent to an aerated stabilization basin.

Sample Site	1st Quarter S.U.[1]	2nd Quarter S.U.[1]	3rd Quarter S.U.[1]	4th Quarter S.U.[1]
MW-1	7.21	6.62	6.85	6.97
MW-2	7.77	6.84	7.09	6.93
MW-2D	7.01	5.94	6.67	6.66
MW-3	7.04	6.00	6.59	6.53
MW-3D	6.99	6.08	6.75	6.54
MW-5	8.52	9.57	9.84	9.86
MW-10	7.02	6.64	6.54	6.81
MW-12	9.38	7.61	7.45	7.72

Notes:

[1] Standard Units

Table 7-4: Average pH readings from 2005 surface water study at sites above and below facility, and within effluent from facility

Mean pH

Sampling Date	SR REF	CR REF	LGP-13	LGP-09	LGP-06	LGP-01
07/07/2005	8.23	7.81	8.28	8.18	7.92	7.77
07/13/2005	8.56	8.09	8.48	8.29	8.72	8.68
07/20/2005	8.03	7.31	8.20	8.25	8.12	8.15
07/27/2005	8.18	7.61	8.08	8.13	7.97	7.89
08/03/2005	8.07	7.75	8.15	8.37	8.05	7.91
08/10/2005	8.02	7.61	8.13	8.25	8.26	8.13
08/17/2005	8.01	7.83	8.13	8.03	7.87	8.00
08/24/2005	7.97	7.80	8.10	8.30	8.03	7.97
08/30/2005	7.97	7.38	8.10	7.78	7.87	8.12
09/07/2005	8.00	7.94	8.29	7.75	8.09	8.18
09/14/2005	7.72	7.41	8.33	7.86	7.79	7.98
09/21/2005	8.10	7.73	8.21	8.16	8.16	7.99
09/27/2005	7.69	7.95		8.03		7.84
10/05/2005	7.46	7.83	8.04	8.06	7.87	7.62
10/11/2005				8.11	8.11	7.99
10/19/2005	7.96	7.88	8.22	8.22	8.20	8.05
10/25/2005				8.23	8.22	7.56
Average	**8.00**	**7.73**	**8.20**	**8.12**	**8.08**	**7.99**
Median	**8.01**	**7.80**	**8.18**	**8.16**	**8.07**	**7.99**
Min	**7.46**	**7.31**	**8.04**	**7.75**	**7.79**	**7.56**
Max	**8.56**	**8.09**	**8.48**	**8.37**	**8.72**	**8.68**

Table 7-5: Mean pH results from 2006 weekly receiving water monitoring study.

Sampling Date	SR REF	CR REF	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
07/05/2006	7.63	7.80	7.70	7.82	7.84	7.96	7.66
07/11/2006	8.15	8.18	8.22	8.12	7.91	8.28	7.98
07/18/2006	8.01	8.18	8.30	8.26	7.98	8.67	8.59
07/25/2006	8.17	8.21	8.42	8.47	8.35	8.58	9.05
07/31/2006	8.30	8.51	8.37	8.60	8.75	8.62	9.03
08/08/2006	8.51	9.06	8.52	8.64	8.52	7.91	7.90
08/15/2006	7.47	8.11	8.06	8.00	7.84	7.62	7.79
08/21/2006	7.65	8.08	8.09	8.00	7.71	7.76	7.71
08/30/2006	7.73	8.15	8.22	8.12	7.77	8.29	8.42
09/05/2006	8.14	8.47	8.27	8.27	8.24	8.35	8.30
09/12/2006	8.23	8.68	8.56	8.45	8.38	8.47	8.41
09/19/2006	8.38	8.62	8.57	8.60	8.58	8.54	8.51
09/26/2006	7.59	8.20	8.18	8.17	8.14	8.05	7.65
10/03/2006	7.47	7.73	7.72	7.90	7.85	7.67	7.55
10/09/2006	7.62	8.32	8.28	8.25	8.23	8.09	7.69
10/16/2006	7.50	6.89	8.02	8.15	8.22	8.28	8.28
10/23/2006	8.18	8.63	8.35	8.58	8.58	8.60	8.50
Average	**7.92**	**8.23**	**8.23**	**8.26**	**8.17**	**8.22**	**8.18**
Median	**8.01**	**8.20**	**8.27**	**8.25**	**8.22**	**8.28**	**8.28**
Min	**7.47**	**6.89**	**7.70**	**7.82**	**7.71**	**7.62**	**7.55**
Max	**8.51**	**9.06**	**8.57**	**8.64**	**8.75**	**8.67**	**9.05**

7.5.1.2.3 Water Quality Standard

The current Idaho water quality standards require the hydrogen ion concentration (pH) values to be within the range of 6.5 to 9.0 su, based on the goal of protection of aquatic life. The current Washington water quality standards require pH in waters designated for salmonid spawning, rearing, and migration, the aquatic life use, assigned to the Snake River, be within the range of 6.5 to 8.5 su with a human-caused variation within this range of less than 0.5 units.

7.5.1.2.4 Effluent Limitation

Although not a specific pollutant, pH is related to the acidity of alkalinity of a wastewater stream. It is not a linear or direct measure of either acidity or alkalinity, however, it may properly be used as a surrogate to control both excess acidity and excess alkalinity in water.

EPA's technology-based effluent guidelines applicable to the Clearwater Mill discharge specify a pH range of 5.0 to 9.0 su. To protect water quality, the effluent limits in the 2019 draft permit incorporated the more stringent water quality-based minimum limit of 5.6 su as well as a more-stringent water quality-based maximum limit of 8.5 su.

The NPDES regulations (40 CFR section 401.17) concerning pH limits allow for a period of excursion when the effluent is being continuously monitored. These requirements have been incorporated into the draft permit. Excursions from the limited range (5.7 to 8.5) are permitted subject to the following limitations:

- The total time during which the pH values are outside the required range of pH values shall not exceed 7 hours and 26 minutes in any calendar month; and

- No individual excursion from the range of pH values shall exceed 60 minutes.

Although the draft permit allows excursions of pH for the required range, it is unlikely that the discharge will exceed the previous range of the permitted discharge (i.e., 5.5 to

9.0 su) because it would take longer than the allowed 60 minutes to buffer a large variation in the wastewater pH due to the long retention time of the pond (i.e., ~8 days) and the large volume of wastewater in the pond (~347.5 million gallons; Potlatch, 2003). Additionally, the total time during which the pH values would be allowed to exceed the range of pH in any calendar month limits the number of excursions that would occur within a month.

7.5.1.2.5 Benchmarks

Data relevant to pH were obtained from EPA's Quality Criteria for Water (USEPA, 1986). 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.

Extremes of pH or rapid pH changes can cause stressful conditions or kill aquatic life outright. Even moderate changes from acceptable criteria limits of pH are deleterious to some species. The relative toxicity to aquatic life of many materials is increased by changes in the water pH.

The European Inland Fisheries Advisory Commission (1969) reviewed pH toxicity to freshwater fish published by various authors. The Commission concluded:

> *There is no definite pH range within which a fishery is unharmed and outside which it is damaged, but rather, there is a gradual deterioration as the pH values are further removed from the normal range. The pH range that is not directly lethal to the fish is 5 - 9; however, the toxicity of several common pollutants is markedly affected by pH changes within this range, and increasing acidity or alkalinity may make these poisons more toxic. Also, an acid discharge may liberate sufficient CO_2 from bicarbonate in the water either to be directly toxic, or to cause the pH range 5 - 6 to become lethal.*

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 parts per million (ppm) or the water contains iron salts precipitated as ferric hydroxide, a compound of unknown toxicity. Values of pH ranging from 6.0 to 6.5 su are unlikely to harm fish unless free carbon dioxide is present above 100 ppm, while pH values ranging from 6.5 to 9.0 su 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 EPA (1986) to conclude that a pH range of 6.5 to 9.0 su 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.

Mount (1973) conducted 13-month (single generation) bioassays on the fathead minnow, *Pimephales promelas*, at pH levels of 4.5, 5.2, 5.9, 6.6, and a control of 7.5. At the two lowest pH values (4.5 and 5.2), behavior was abnormal, and the fish were deformed. At pH values less than 6.6, egg production and hatchability were reduced compared to the control. Mount (1973) concluded that a pH of 6.6 was marginal for vital life functions.

Bell (1971) performed bioassays using nymphs of caddisflies (two species), stoneflies (four species), dragonflies (two species), and mayflies (one species). All are important fish prey items. The 30-day TL50 pH values ranged from 2.45 to 5.38, with the caddisflies being the least sensitive and the mayflies being the most sensitive. The pH values at which 50 percent of the organisms emerged ranged from 4.0 to 6.6, with increasing emergence occurring with the increasing pH values.

Another study (Ikuta et al., 2003) found that low pH (5.8-6.4) significantly inhibited nest-digging behavior of several salmonid species. Land-locked sockeye salmon were found to be the most sensitive of the test species (Ikuta et al., 2003).

Many researchers specify that the toxic action of hydrogen ions on fish under acidic conditions induces production of mucus on the gill epithelium, which can have several negative effects; there is a precipitation of proteins in the epithelial cells; an interference in the respiratory gas and ion exchange across the gill; and acidosis of the blood is known to occur, which affects oxygen uptake (Ellis 1937; Westfall 1945; AFS 1979; Boyd 1990). Low pH waters (typically below 5.0) tend to be more common than high pH waters (usually 9 and above), but whereas low pH can cause reduced growth rates, high pH and therefore excess hydroxyl ions, can cause destruction of gill and skin epithelium and eye effects as well (Alabaster and Lloyd 1980; Boyd 1990).

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).

At the higher end of the pH scale, even less is known regarding effects on fish. In EPA'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). Benthic invertebrate

populations may be adversely affected by pH levels greater than 9.0; thus, altering the food base for salmonids (ODEQ, 1995).

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 Outfall 001.

> Metals: 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. The activity of pH 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. pH below 5.0 su can cause toxicity from solubilization of metals such as aluminum. 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 (Baker et al., 1990). In summary, reductions in pH below natural levels will tend to increase metal availability and toxicity.
>
> Temperature: pH does not directly affect temperature; however, they both vary on a seasonal and diurnal basis.
>
> Dissolved Oxygen: Dissolved oxygen and pH may affect the toxicity of certain chemical species, but studies to date are inconclusive (ODEQ, 1995).
>
> Ammonia: The acute toxicity of ammonia has been shown to increase as pH increases because un-ionized ammonia (which is more toxic than the ammonium ion) concentrations increase with increasing pH. The very limited amount of data regarding effects of pH on chronic ammonia toxicity also indicates increasing ammonia toxicity with increasing pH. Invertebrates are generally more tolerant than fishes to the acute and toxic effects. Ammonia has been shown to be 10 times more toxic at pH 8.0 su than at pH 7.0 su (EIFAC, 1969).
>
> Pentachlorophenol: The acute and chronic toxicity of pentachlorophenol to freshwater animals increases as pH and dissolved oxygen concentration of the water decrease.

Based on the results of these toxicity studies, EPA estimated that a pH range of 6.5 to 9.0 su appeared to provide adequate protection for the life of freshwater fish and bottom dwelling invertebrates, although the toxicity of other parameters may be affected by

changes within this range. Outside of this range, fish experience physiological toxicity increasing in severity with deviation from this range, ultimately resulting in lethality.

This BE uses a pH range of 6.5 to 9.0 as the direct benchmarks for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead and the indirect benchmark for prey species.

7.5.1.2.6 Effects Analysis

The pH of the Snake River in the Action Area ranges between 7.5 and 9.0 su, and the pH of the Clearwater River in the Action Area ranges between 7.3 and 8.3 su. This is well within the range that is safe for threatened and endangered salmonids (6.5 to 9.0 su).

Since the lower end of the pH effluent range required by the permit (5.5 su) is less than the lower end of the pH range for the toxicity benchmark (6.5 su), this analysis looks at the direct effects within the exposure volume of the effluent (i.e., the area where the concentration of the plume is lower than the toxicity benchmark range) and the effects at and beyond the exposure volume boundary.

7.5.1.2.6.1 Direct and Indirect Effects

The CORMIX model predicted that the water column pH that is safe to threatened and endangered salmonids from direct and indirect exposure (6.5 su) is met within 32.7 meters (107 feet) when a background pH of 7.87 su (which is the 5th percentile pH at station LGP-13) is assumed. EPA concludes that the discharge of pH is **not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.2.6.2 Habitat Effects

At and beyond the exposure volume, the permit limits are designed to protect the water quality standard for pH (6.5 to 9.0). Since the water quality standard is equivalent to the pH range safe for threatened and endangered salmonids, EPA has concluded that the discharge of pH is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.3 Temperature

7.5.1.3.1 Introduction

The suitability of water for total body immersion is greatly affected by temperature. In temperate climates, danger from exposure to low temperatures is more prevalent than exposure to elevated water temperatures. Temperature also affects the self-purification phenomenon in water bodies and therefore the aesthetic and sanitary qualities that exist. Increased temperatures accelerate the biodegradation of organic material both in the overlying water and in bottom deposits that makes increased demands on the dissolved oxygen resources of a given system. The typical situation is exacerbated by the fact that oxygen becomes less soluble as water temperature increases. Thus, greater demands are

exerted on an increasingly scarce resource which may lead to total oxygen depletion and noxious septic conditions. These effects have been described by Phelps (1944), Camp (1963), and Velz (1970). Indicator enteric bacteria, and presumably enteric pathogens, are likewise affected by temperature. It has been shown that both total and fecal coliform bacteria die away more rapidly in the environment with increasing temperatures (Ballentine and Kittrell, 1968).

Temperature changes in water bodies can alter the existing aquatic community. The dominance of various phytoplankton groups in specific temperature ranges has been shown. For example, from 20 to 25˚C, diatoms predominated; green algae predominated from 30 to 35˚C, and blue-greens predominated above 35˚C (Cairns, 1956).

Rivers and streams in the Pacific Northwest naturally warm in the summer due to increased solar radiation and warm air temperature. Human changes to the landscape have magnified the degree of river warming, which adversely affects salmonids and reduces the number of river segments that are thermally suitable for salmonids. Human activities can increase water temperatures by increasing the heat load into the river, by reducing the river's flow and thus capacity to absorb heat, and by eliminating or reducing the amount of groundwater flow which moderates temperatures and provides cold water refugia. EPA has presented specific ways in which human development has caused excess warming of rivers (USEPA, 2003), which are summarized below:

- Removal of streamside vegetation reduces the amount of shade that blocks solar radiation and increases solar heating of streams. Examples of human activities that reduce shade include forest harvesting, agricultural land clearing, livestock grazing, and urban development.
- Removal of streamside vegetation also reduces bank stability, thereby causing bank erosion and increased sediment loading into the stream. Bank erosion and increased sedimentation results in wider and shallower streams, which increases the stream's heat load by increasing the surface area subject to solar radiation and heat exchange with the air.
- Water withdrawals from rivers for purposes such as agricultural irrigation and urban/municipal and industrial use result in less river volume. Some withdrawn water is returned to the river as treated wastewater or irrigation return flow, but often at warmer temperature than it was withdrawn. The temperature of rivers with shallower depth equilibrates faster to surrounding air temperature, which leads to higher maximum water temperatures in the summer when lower flows lead to shallower depths.
- Water discharges from industrial facilities, wastewater treatment facilities and irrigation return flows can add heat to rivers.
- Channeling, straightening, or diking rivers for flood control and urban and agricultural land development may reduce some components of cool groundwater flow into a river that moderates summertime river temperatures. These human actions can reduce two forms of groundwater flow. One form is groundwater that is created during over-bank flooding and is slowly returned to the main river channel to cool the water in the summer. A second form is water that is

exchanged between the river and the riverbed (i.e. hyporheic flow). Hyporheic flow is plentiful in fully functioning alluvial rivers systems. Groundwater that flows into rivers from regional aquifer systems provides most of the cool groundwater to rivers and is unaffected by most stream channel modifications.

- Removal of upland vegetation and the creation of impervious surfaces associated with urban development increases storm runoff and reduces the amount of groundwater that is stored in the watershed and slowly filters back to the stream in the summer to cool water temperatures.
- Dams and their reservoirs can affect thermal patterns in several ways. In some cases, they can increase maximum temperatures by holding waters in reservoirs to warm, especially in shallow areas near shore. In other cases, reservoirs, due to their increased volume of water, are more resistant to temperature change and thus can be cooler than unimpounded rivers. The greater resistance of reservoirs to temperature changes results in reduced diurnal temperature variation and delayed changes in river temperature. For example, dams can delay the natural cooling that takes place in the late summer-early fall, thereby harming late summer-fall migration runs. Reservoirs also inundate alluvial river segments, thereby diminishing the groundwater exchange between the river and the riverbed (i.e., hyporheic flow) that cools the river and provides coldwater refugia during the summer. Further, dams can significantly reduce the river flow velocity, thereby causing juvenile migrants to be exposed to high temperatures for a much longer time than they would under a natural flow regime.

It should also be noted that some human development could create water temperatures colder than an unaltered river. The most significant example of this occurs when cold water is released from the bottom of a thermally stratified reservoir behind a dam.

pH: Temperature does not directly affect pH, however, they both vary on a seasonal and diurnal basis. Algae in the stream give off CO_2 at night when they respire. CO_2 disassociates to form carbonic acid, thus lowering the pH to potentially stressful levels. This pH stress is greatest at night, when temperature is at its coolest and thus least stressful. However, respiration is seasonally greatest during the summer, when algal populations are greatest, and thus coincides with seasonal high temperatures.

Dissolved Oxygen: The saturation concentration of dissolved oxygen in water decreases with increasing temperature: fresh water at a temperature of 0ºC has an oxygen solubility of 14.6 mg/L while that at 30ºC has a solubility of 7.6 mg/L (APHA, 1998).

Ammonia: USEPA (2013) updated their Ambient Water Quality Criteria for Ammonia. The pH and temperature relationship with ammonia established in the 1999 document (USEPA 1999a) still holds. They reviewed the literature and found that, following normalization for pH, the freshwater acute toxicity data concerning temperature dependence show neither large effects nor any clear consistency among or within species or studies. Therefore, the acute ammonia criterion does not change with temperature. However, the acute ammonia criterion is lower when salmonids are present. USEPA (1999a) also looked at the

chronic toxicity of ammonia to fish and concluded that available data suggest minimal dependence of ammonia toxicity on temperature. They stated that although limited available chronic data suggest LC20s might be lower at low temperatures, the effect is small and uncertain (USEPA, 1999a). The chronic ammonia criterion does, however, depend on temperature, pH, and whether early life stages are present. The chronic criterion increases with decreasing temperature and increases with increasing pH.

7.5.1.3.2 Environmental Baseline

The Endangered Species Act Monitoring and NPDES Compliance monitoring report by AMEC Environmental (2006) documents data obtained from temperature, current, and stage meters placed at several locations in July and October 2006 near the Snake and Clearwater Rivers confluence. The data indicated that mean water temperature decreased during the sample period more than likely due to seasonal changes (Figure 7-5).

Temperature data recorded at these stations show seasonal changes over time. Temperature at SR-REF was generally the highest recorded temperature, except on August 8th, 2005 (LGP-01 20.3ºC). This spike in temp at LGP-01 was the only temperature to exceed the 20-degree benchmark for July and August. Temperatures tended to increase moving downstream, with the lowest temperatures closest to the facility.

A Benchmark Temperature of 18º C was used for September and October. During September, the temperature at the Snake River reference location exceeded the benchmark for every sample. The three most downstream sample locations (LGP-09, LGP-06, and LGP- 01) exceeded the benchmark on the first 2 sample dates. The benchmark temperature was never exceeded at station LGP-13, the closet to the facility.

Figure 7-6 shows the mean and range of temperatures measured during July 2005 to October 2005. Table 7-6 shows the mean temperatures measured during July to October 2006.

The environmental baseline temperature indicates that the Snake River is at temperatures exceeding the benchmark, as well as the water quality criterion for protection of cold-water aquatic life, during July through September. Because the background temperature is at or exceeds acceptable temperatures in July through October, EPA has used 0.3˚C in the assessment for temperature because this temperature difference between the background and downstream water would not result in a detectable quantity in the receiving water.

7.5.1.3.3 Water Quality Standard

The current Idaho water quality standards for the Snake River is 22˚C as a daily maximum and a maximum daily average of 19˚C, based on the goal of protection of aquatic life. If temperature criteria for the designated aquatic life use are exceeded in the receiving waters upstream of the discharge due to natural background conditions, then wastewater must not raise the receiving water temperatures by more than three tenths (0.3) degrees C.

The current Washington water quality standard for the Snake River from its mouth to the Washington – Idaho – Oregon border (River Mile 176.1) is 20˚C as a daily maximum temperature. When natural conditions exceed a daily maximum of 20.0°C, no temperature increase will be allowed which will raise the receiving water temperature by greater than 0.3°C.

EPA Region 10 Guidance for Pacific Northwest State and Tribal Temperature Water Quality Standards (EPA, 2003) recommends the criteria in Table 7-7 as safe levels for salmonids.

7.5.1.3.4 Effluent Limitation

The proposed 2019 temperature effluent limitations are the same as the 2005 final permit limits including for October through June a maximum daily effluent limit of 33˚C; the proposed 2019 effluent limit for July is a maximum daily effluent limit of 32ºC, and the proposed 2019 effluent limit for August through September requires a maximum daily effluent limit of 31˚C.

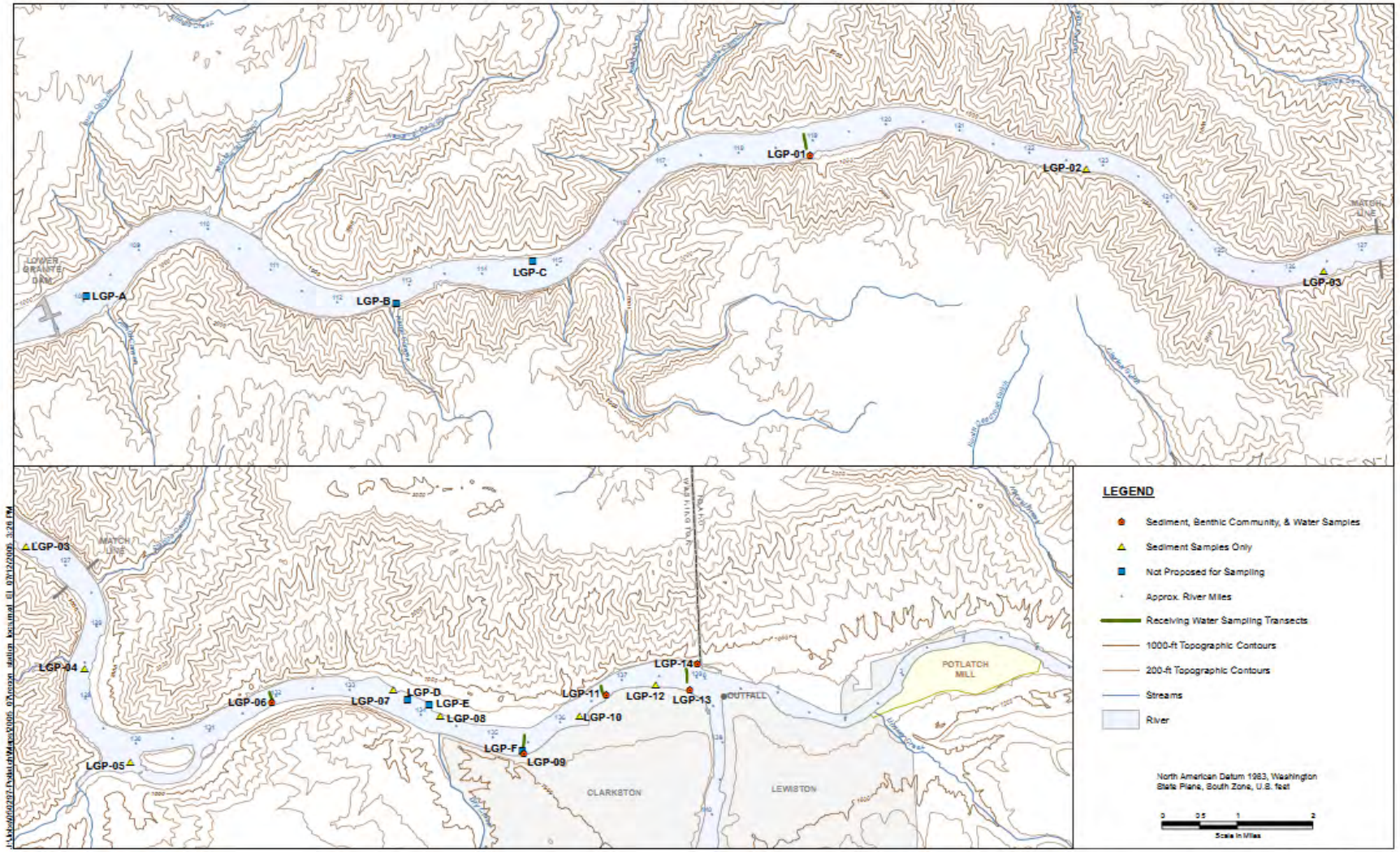


Figure 7-5: Temperature sensor locations for 2005 data collected by AMEC Environmental. (2006)

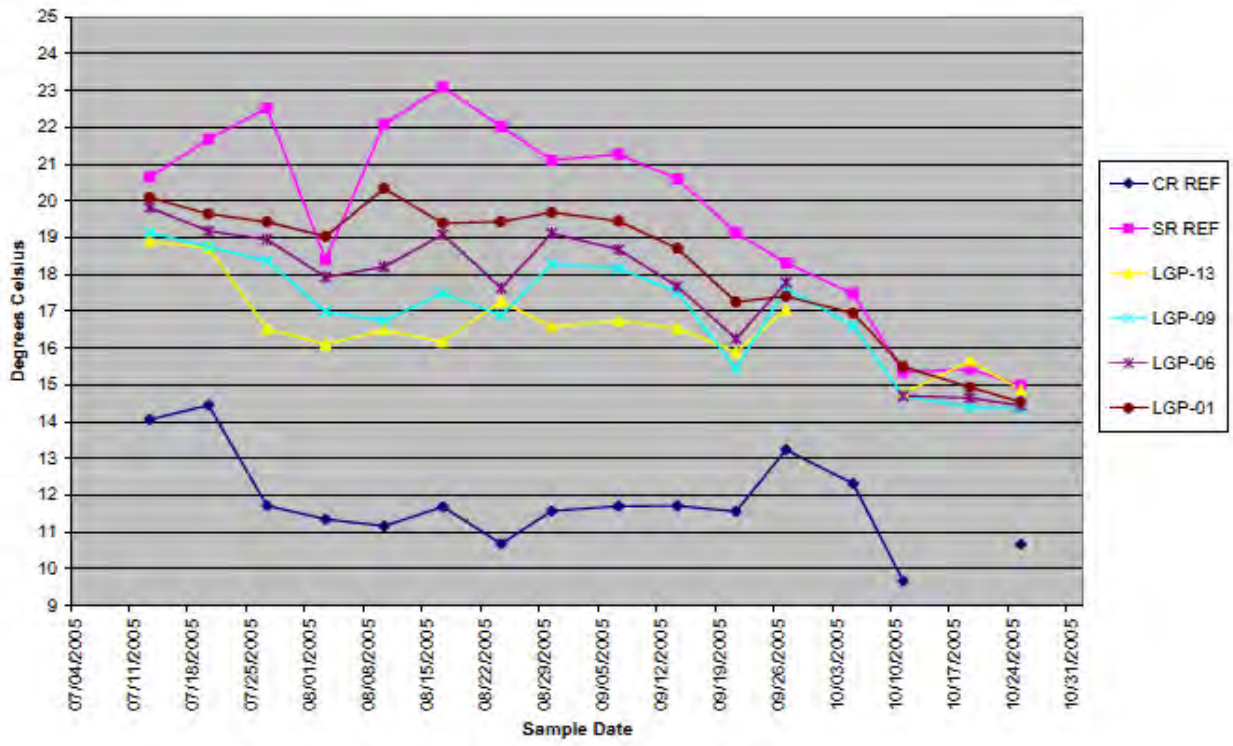


Figure 7-6: Mean and Range of Temperatures at Clearwater River (AMEC, 2006)

Table 7-6: Mean Temperatures in July through October 2006 (AMEC, 2005)

Mean Temperature (degrees Celsius)

Sampling Date	SR REF	CR REF	LGP-13	LGP-09	LGP-06	LGP-01
07/07/2005	20.65	14.05	18.90	19.14	19.82	20.09
07/13/2005	21.67	14.44	18.71	18.74	19.18	19.64
07/20/2005	22.52	11.71	16.51	18.38	18.95	19.42
07/27/2005	18.40	11.34	16.10	16.98	17.92	19.03
08/03/2005	22.08	11.15	16.49	16.75	18.21	20.33
08/10/2005	23.09	11.68	16.17	17.49	19.08	19.39
08/17/2005	22.02	10.67	17.27	16.88	17.63	19.43
08/24/2005	21.10	11.57	16.58	18.28	19.12	19.68
08/30/2005	21.26	11.70	16.74	18.16	18.68	19.45
09/07/2005	20.60	11.70	16.53	17.52	17.67	18.71
09/14/2005	19.13	11.55	15.89	15.44	16.25	17.25
09/21/2005	18.31	13.23	17.04	17.61	17.77	17.40
09/27/2005	17.47	12.31		16.60		16.94
10/05/2005	15.34	9.66	14.76	14.71	14.70	15.49
10/11/2005	15.43		15.65	14.38	14.64	14.93
10/19/2005	14.98	10.66	14.86	14.35	14.43	14.53
10/25/2005	14.07	9.78	14.61	14.31	14.41	15.38
Average	**19.30**	**11.70**	**16.42**	**16.81**	**17.40**	**18.06**
Median	**20.60**	**11.62**	**16.50**	**16.98**	**17.85**	**19.03**
Min	**14.07**	**9.66**	**14.61**	**14.31**	**14.41**	**14.53**
Max	**23.09**	**14.44**	**18.90**	**19.14**	**19.82**	**20.33**

Table 7-7: EPA Region 10 Recommended Temperature Criteria for Salmonid Uses

Salmonid Uses	**Criteria**
Summer Maximum Conditions	
Bull Trout Juvenile Rearing	12˚C 7DADM
Salmon/Trout "Core" Juvenile Rearing (*Salmon adult holing prior to spawning, and adult and sub-adult bull trout foraging and migration may also be included in this category*)	16˚C 7DADM
Salmon/Trout Migration plus Non-Core Juvenile Rearing	18˚C 7DADM
Salmon/Trout Migration	20˚C 7DADM
General Conditions	
Bull Trout Spawning	9˚C 7DADM
Salmon/Trout Spawning, Egg Incubation, and Fry Emergence	13˚C 7DADM
Steelhead Smoltification	14˚C 7DADM
7DADM refers to the Maximum 7-day average of the daily maximums Salmon refers to Chinook and Sockeye Trout refers to Steelhead	

USEPA (2005) conducted a "Temperature Assessment for the Potlatch Mill Discharge through Outfall 001" and the results of that memo indicated:

1) The temperature of the Snake River is below the numeric Idaho water quality criteria for cold water biota in the months of October through June.
2) The natural background criterion applies in the months of July, August, and September because the natural background temperatures of the Snake River in the area of the discharge exceed the numeric water quality criterion most of the time during these months.
3) Based on the 0.3°C increase allowed by the natural background provisions of the Idaho water quality standards and the fact that the actual upstream temperature is below the numeric temperature criterion from October through June, a mixing zone can be applied to the discharge throughout the year.
4) The temperature effluent limits are derived from and compliant with the Idaho water quality standards, and a discharge in compliance with the temperature effluent limits will not cause or contribute to violations of the Idaho water quality criteria outside of a small mixing zone.
5) The temperature mixing zone and the resulting temperature effluent limits will not cause unreasonable interference with or danger to the beneficial used of the Snake River, including salmonid migration.

7.5.1.3.5 Benchmarks

Based on the information in the Region 10 temperature guidance (USEPA, 2003) and the timing of fish use of the Action Area described in Section V, **this BE establishes the following temperature benchmarks for the waterbody as a whole as protective to bull trout, Snake**

River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead:

March - May: 14ºC 7DADM for smoltification
June: 18ºC 7DADM for juvenile rearing
July - August: 20ºC 7DADM for adult migration (This is approximately equivalent to the Idaho water quality criterion of 19ºC average daily temperature)
September - October: 18ºC 7DADM for juvenile rearing
November - February: 16ºC 7DADM for bull trout

From literature reviewed by EPA for the development of the Region 10 temperature guidance (USEPA, 2003), the following adverse effects may result from thermal plumes:

- Exposures of less than 10 seconds can cause instantaneous lethality at 32ºC.
- Thermal shock leading to increased predation can occur when salmon and trout exposed to near optimal temperatures (e.g., 15ºC) experience a sudden increase in temperature within the range of 26-30ºC.
- Adult migration blockage conditions can occur at 21ºC.
- Adverse impacts on salmon and trout spawning, egg incubation, and fry emergence can occur when the temperatures exceed 13ºC.

Therefore, this BE establishes the following benchmarks as protective to bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead to the thermal plume:

- **The maximum temperature within the plume after 2 seconds of plume travel from the point of discharge does not exceed 32ºC.**
- **The thermal plume does not result in more than 5 percent of the Snake River channel cross-section above 25ºC.**
- **The thermal plume does not result in more than 25 percent of the Snake River channel cross-section above 21ºC. When the Snake River channel exceeds 21ºC, the thermal plume does not increase more than 25 percent of the Snake River channel cross-section above ambient conditions by a measurable amount (e.g., less than 0.3ºC).**

7.5.1.3.6 Effects Analysis

This analysis compares the resulting temperature of the thermal plume to the benchmarks described above.

USEPA (2018; Appendix D) conducted another assessment, "Temperature Assessment for the Clearwater Paper Lewiston Mill Discharge through Outfall 001." The memo outlines the evaluation of the mixing properties of the discharge in order to determine compliance with water quality standards and the EPA's thermal plume recommendations. Results are summarized in Table 7-8, below.

Table 7-8: Cormix modeling results for temperature

Month	Ambient T (°C)	Effluent T (°C)	T at 2 s. of Plume Travel (°C) Target: < 32 °C	T at 5% of Cross-Sectional Area (°C) Target: < 25 °C	T at 25% of Cross-Sectional Area (°C) or distance to Target of < 21 °C or < 0.25 °C increase[2]	Dilution Factor at 25% of Stream Width	T at 25% of Stream Width (°C) Criterion: 19 °C and 1 °C increase or 0.3 °C increase	T at Washington Border (°C) Criterion: 20 °C and t = 34÷ (T + 9) increase or 0.3 °C increase[4]
January	4.4	33	8.3	5.8	4.7 °C	52.0	5.0 (0.6 increase)	4.9 (0.5 increase)
February	5.0	33	8.7	6.3	5.2 °C	55.0	5.5 (0.5 increase)	5.5 (0.5 increase)
March	7.9	33	11.0	9.0	8.1 °C	61.4	8.3 (0.4 increase)	8.3 (0.4 increase)
April	11.1	33	13.3	11.8	11.2 °C	83.3[3]	11.4 (0.3 increase)[3]	11.4 (0.3 increase)
May (31.6 mgd)	13.2	33	14.6	13.7	13.3 °C	139.6	13.4 (0.2 increase)[3]	13.4 (0.2 increase)
May (38.6 mgd)	13.2	33	14.8	13.7	13.3 °C	113.3	13.4 (0.2 increase)[3]	13.4 (0.2 increase)
June (31.6 mgd)	18.3	33	19.8	18.8	18.4 °C	85.4	18.5 (0.2 increase)[3]	18.5 (0.2 increase)
June (38.6 mgd)	18.3	33	19.9	18.9	18.4 °C	72.0	18.5 (0.2 increase)[3]	18.5 (0.2 increase)
Early July (31.6 mgd)	20.0	32	21.4	20.5	20.1 °C	68.8	0.20 increase[3]	0.20 increase
Early July (38.6 mgd)	20.0	32	21.4	20.5	20.1 °C	59.0	0.22 increase	0.20 increase
Late July (31.6 mgd)	16.0[5]	32	18.3	16.9	0.21 m	48.5	16.3 (0.3 increase)	16.3 (0.3 increase)
Late July (limit, 38.6 mgd)	22.5[1]	32	23.6	22.9	0.11 °C increase	58.9	0.22 increase	0.20 increase
Late July (Avg., 38.6 mgd)	16.0	29.3	18.0	16.8	16.1 °C	42.1	16.3 (0.3 increase)	16.3 (0.3 increase)
August (31.6 mgd)	16.8[5]	31	19.1	17.7	0.27 m	39.4	17.2 (0.4 increase)	17.1 (0.3 increase)
August (38.6 mgd)	16.8[5]	31	19.0	17.7	0.29 m	36.5	17.2 (0.4 increase)	17.2 (0.4 increase)
Early Sep. (31.6 mgd)	16.0[5]	31	18.5	17.0	0.22 m	36.5	16.4 (0.4 increase)	16.4 (0.4 increase)
Early Sep. (limit, 38.6 mgd)	21.0[5]	31	22.4	21.5	147 m	44.5	0.27 increase	0.24 increase
Late Sep. (limit, 31.6 mgd)	19.0[1]	31	20.7	19.6	0.76	47.9	0.25 increase	19.2 (0.2 increase)
Late Sep. (limit, 38.6 mgd)	19.0[1]	31	20.7	19.6	19.1	44.5	0.27 increase	19.2 (0.2 increase)
Late Sep. (avg., 31.6 mgd)	19.0[1]	25.8	Note 6	Note 6	Note 6	47.9	0.14 increase	Note 6
Late Sep. (avg., 38.6 mgd)	19.0[1]	25.8	Note 6	Note 6	Note 6	44.6	0.15 increase	Note 6

Month	Ambient T (°C)	Effluent T (°C)	T at 2 s. of Plume Travel (°C) Target: < 32 °C	T at 5% of Cross-Sectional Area (°C) Target: < 25 °C	T at 25% of Cross-Sectional Area (°C) or distance to Target of < 21 °C or < 0.25 °C increase[2]	Dilution Factor at 25% of Stream Width	T at 25% of Stream Width (°C) Criterion: 19 °C and 1 °C increase or 0.3 °C increase	T at Washington Border (°C) Criterion: 20 °C and t = 34÷ (T + 9) increase or 0.3 °C increase[4]
October (limit)	18.5[1]	33	20.6	19.3	0.74 m	46.5	18.8 (0.3 increase)	18.8 (0.3 increase)
October (avg.)	18.5[1]	25.0	Note 6	Note 6	Note 6	46.5	18.6 (0.1 increase)	Note 6
November	10.2	33	13.5	11.4	10.4	48.0	10.7 (0.5 increase)	10.6 (0.4 increase)
December	5.8	33	9.8	7.2	6.1	46.9	6.4 (0.6 increase)	6.3 (0.5 increase)

Notes:

1. The ambient temperature is stratified at this time. The ambient temperature listed is the temperature at the surface, because Cormix predicts that the plume will rise to the surface.
2. If Cormix predicts that the plume will not spread such that the plume occupies 25% of the cross-sectional area of the river within 50,000 meters downstream of the discharge, the distance at which the temperature falls to 21 °C or 0.25 °C above ambient is reported. The discharge meets the thermal plume recommendation for migration blockage from late July through October.
3. During April, May, June, and early July, a mixing zone encompassing 25% of the stream width would extend downstream past the Washington border. The State of Idaho cannot authorize a mixing zone that extends into another State. Thus, the conditions at the Washington border (191 meters downstream) are reported.
4. The values of t = 34/(T+9) are: 2.5 °C in January, 2.4 °C in February, 2.0 °C in March, 1.7 °C in April, 1.5 °C in May, 1.2 °C in June, 1.32 °C in August, 1.4 °C in late September, 1.3 °C in October, 1.8 °C in November, and 2.3 °C in December. In July and early September, the allowable temperature increase is 0.3 °C.
5. From late July through early September, the plume traps below the thermocline. For these scenarios, the ambient temperature is listed as the temperature at the lower end of the thermocline. The plume will not affect the warmer water above the thermocline.
6. Additional scenarios evaluating the conditions at 25% of the stream width using the average effluent temperatures instead of the effluent limits were run for late September and October, to determine if the lower effluent temperature would affect the plume's behavior in the stratified receiving water. The lower effluent temperatures had no effect on the plume's behavior, so no other targets were evaluated using the average effluent temperatures.

7.5.1.3.6.1 Instantaneous lethality

Table 7-8 shows that the discharge at the final permit limits would not cause instantaneous lethality to salmon and steelhead. For the months of July, August, and September, the effluent limit is at or below 32ºC, so there will be no exposure time to lethal temperatures above 32ºC.

For other months of the year, the effluent limit is 33ºC, but as shown in Table 7-8, after two seconds of plume travel time, the temperature will have dropped well below 32 °C, with a maximum of 20.6 °C in October.

7.5.1.3.6.2 Thermal shock

Table 7-8 shows that salmon and steelhead will not experience thermal shock from the discharge in compliance with the permit limits. The temperature at the point where the plume spreads to 5% of the stream width is always less than 25 °C, with a maximum of 22.9 °C in late July at the maximum augmented flow.

7.5.1.3.6.3 Migration blockage

The temperature of the Snake River upstream of outfall 001 typically exceeds the water quality standards for Idaho and Washington in the summer months and often exceeds 21ºC, which can impede migration of adult salmon and steelhead. Thus, in the summer months, the baseline conditions for salmon and steelhead are stressful even if the Clearwater discharge did not exist.

Table 7-8 shows that, in July, August, and September, at the point where the plume spreads to 25% of the stream width, the temperature increase caused by the discharge is no more than 0.3 °C. Because the Clearwater discharge results in some portion of the river with a slight increase in temperature as described above and that the river baseline condition is already stressful for salmon and steelhead, EPA concludes that the Clearwater temperature effluent limits are likely to adversely affect salmon and steelhead during the months of July, August, and September. However, the contribution of the Clearwater discharge to the adverse effects associated with elevated temperatures in the Snake River is small.

In other months of the year, the portion of the river that exceeds the benchmarks is negligible.

Based on this analysis, EPA has concluded that the discharge of temperature at the final effluent limits is **likely to adversely affect Snake River sockeye salmon, Snake River summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**, and **may affect, but is not likely to adversely affect Snake River spring Chinook salmon and bull trout**.

7.5.1.3.7 Habitat Effects

Because temperature is an important feature of salmon habitat and the effect of the temperature limits has the potential to cause migratory blockage, EPA has concluded that the discharge of temperature is **likely to adversely modify the critical habitat for Snake River sockeye salmon, Snake River summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead** and is **not likely to adversely modify the critical habitat for Snake River spring Chinook salmon and bull trout**.

7.5.1.4 Biochemical Oxygen Demand (BOD)

7.5.1.4.1 Introduction

Biochemical oxygen demand (BOD) is the quantity of oxygen required for the biological and chemical oxidation of waterborne substances under ambient or test conditions. Materials that may contribute to the BOD include: carbonaceous organic materials usable as a food source by aerobic organisms; oxidizable nitrogen derived from nitrites, ammonia and organic nitrogen compounds which serve as food for specific bacteria; and certain chemically oxidizable materials (e.g., ferrous iron, sulfides, sulfite, etc.) which will react with dissolved oxygen or are metabolized by bacteria. The BOD in most effluents is derived principally from organic materials and from ammonia (which is itself derived from animal or vegetable matter).

The BOD in effluent affects the dissolved oxygen resources of a body of water by reducing the oxygen available to fish, plant life, and other aquatic species. High biochemical oxygen demand lowers the concentration of dissolved oxygen (DO) in water, and toxicity could occur due to insufficient concentrations of DO. The reduction of dissolved oxygen can be detrimental to fish populations, fish growth rate, and organisms used as fish food.

At extreme conditions, all of the dissolved oxygen in the water can be consumed by BOD resulting in anaerobic conditions and the production of undesirable gases such as hydrogen sulfide and methane. A total lack of oxygen due to the exertion of an excessive BOD can result in the death of all aerobic aquatic inhabitants in the affected area.

Water with a high BOD indicates the presence of decomposing organic matter and associated increased bacterial concentrations that degrade its quality and potential uses. A by-product of high BOD concentrations can be increased algal concentrations and blooms which result from decomposition of the organic matter and which form the basis of algal populations (USEPA, 1976).

BOD is measured using an empirical test in which standardized laboratory procedures are used to determine the amount of oxygen that would be consumed in microbiological biodegradation of the effluent (APHA, 1998). The test measures the molecular oxygen utilized during a specified incubation period for the biochemical degradation of organic material (carbonaceous demand) and the oxygen used to oxidize inorganic materials such as sulfides and ferrous iron (APHA, 1998). The biochemical oxygen demand determined by 5 days incubation is called BOD5. Because dissolved oxygen concentrations vary seasonally and diurnally, direct measurement of DO does not accurately indicate the extent to which compounds in water affect the concentration of DO. Consequently, BOD provides a means to measure the potential changes in the concentration of DO in the receiving water body that could occur due to the presence of compounds in effluents. Historically, in pulp effluents, the presence of wood sugars and other readily metabolized organic substrates contributed a substantial amount of BOD (Laws, 1993). Secondary treatment of wastewater reduces the BOD content of effluent (Laws, 1993).

There are several factors affecting DO in receiving waters. The addition of BOD from effluent results in the oxidation of organic substances and a decrease in the oxygen concentration in the receiving water downstream of the discharge. Factors, in addition to the effluent BOD, that tend to decrease DO include aquatic microbial, plant, and animal respiration. Factors that tend to increase DO include:

- The equilibrium between atmospheric oxygen concentrations and the concentration of dissolved oxygen in water; and
- Photosynthesis by aquatic algae and higher aquatic plants.

7.5.1.4.2 Environmental Baseline

In 2005 and 2006, Endangered Species Act Tier 1 studies were undertaken to evaluate effluent and natural waters above and below the facility. Sampling was conducted in shallow and mid-depth surface water and in 2005 the BOD concentrations ranged from non-detect to 7 mg/L among the downstream locations samples. All BOD concentrations were below 2 mg/L in September, except for three measurements taken from shallow surface water at the farthest downstream sampling location. In July and August, concentrations typically stayed near 1 mg/L, and decreased to non-detect by mid-October 2005. In 2006, BOD concentrations ranged from non-detect to 137 mg/L. Samples collected on 7/5/2006 were qualified by the laboratory as exceeding holding times and because upstream samples were elevated, it is unlikely that the Facility contributed to elevated BOD concentrations downstream. Figure 7-7 through Figure 7-10 present BOD concentrations for each sampling day of the monitoring period (AMEC 2006, AMEC 2007).

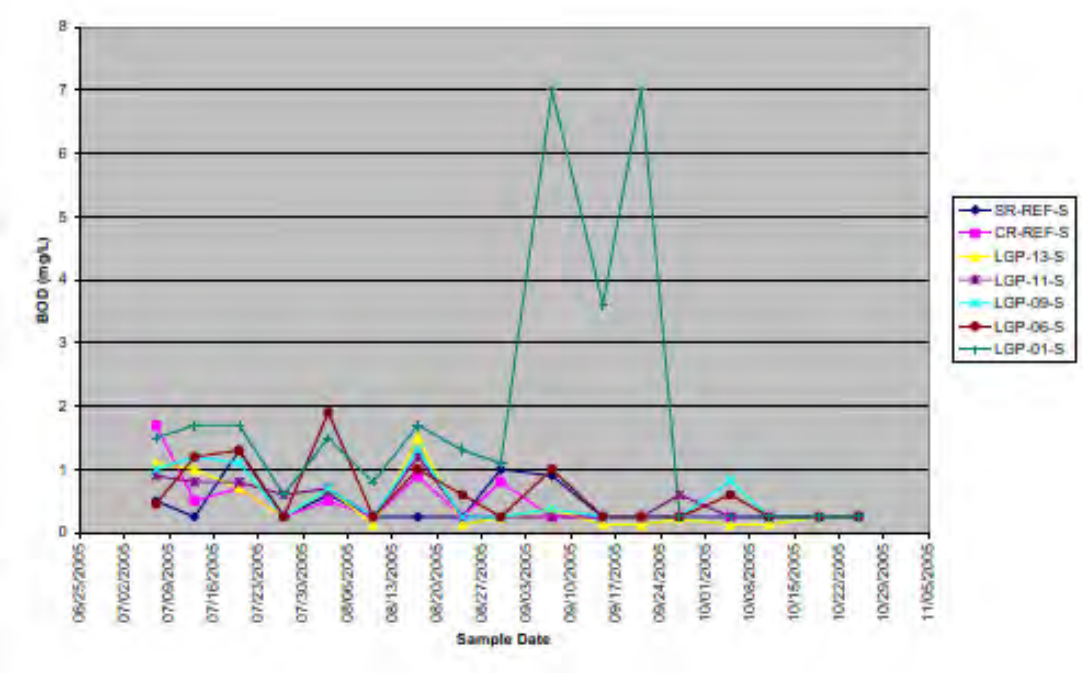


Figure 7-7: Biological oxygen demand in shallow surface water in 2005 (AMEC 2006).

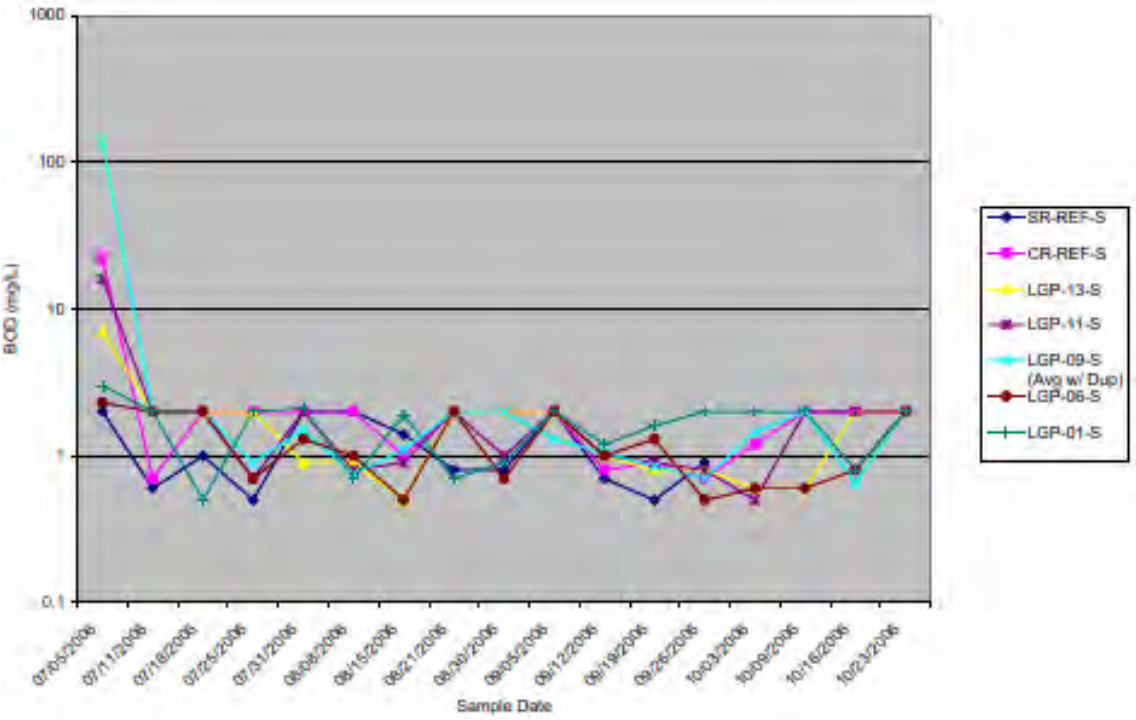


Figure 7-8: Biological oxygen demand in shallow surface water in 2006 (AMEC 2007).

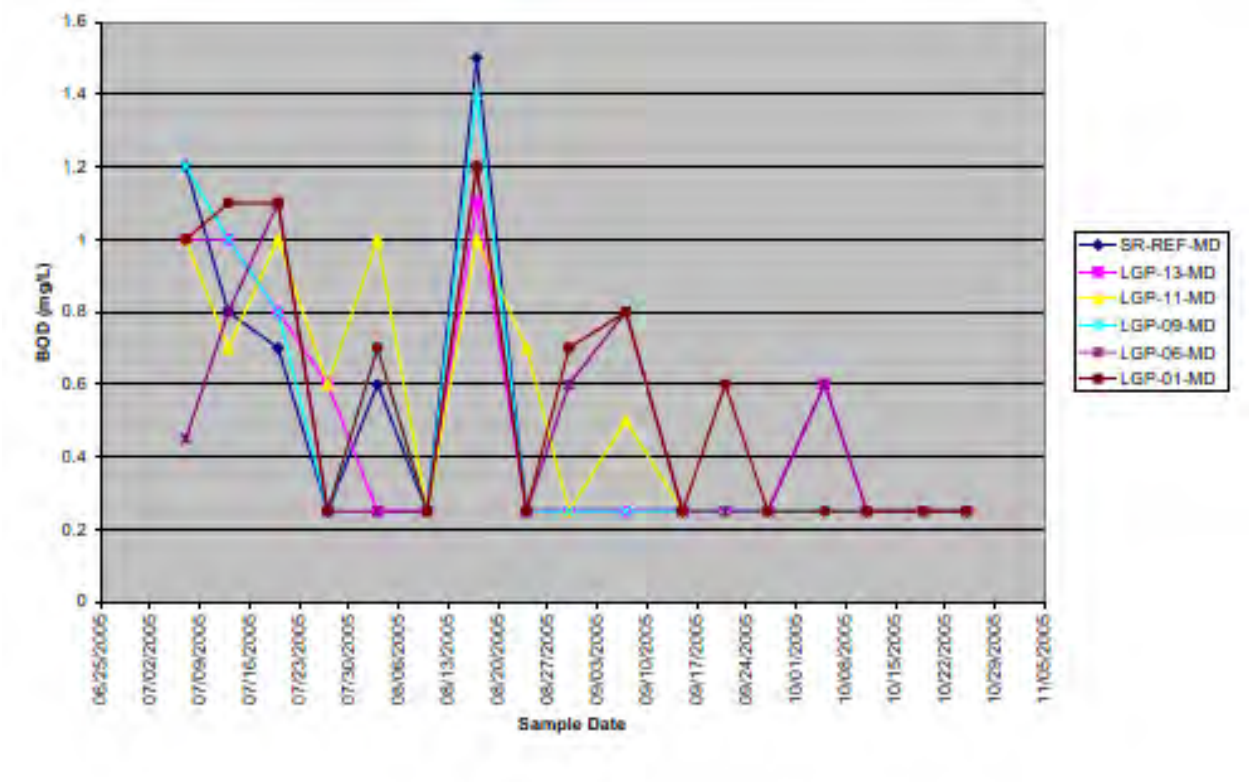


Figure 7-9: Biological oxygen demand in mid-depth surface water in 2005 (AMEC 2006).

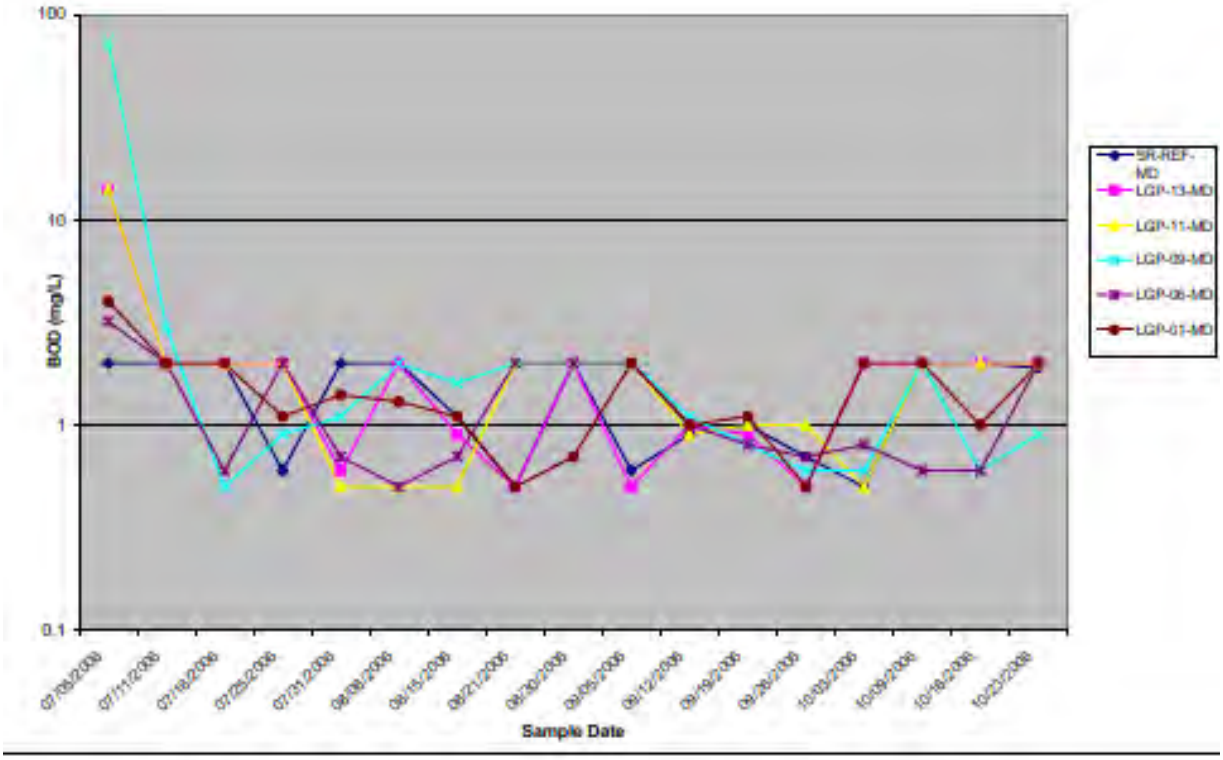


Figure 7-10: Biological oxygen demand in mid-depth surface water in 2006 (AMEC 2007).

7.5.1.4.3 Water Quality Standard

The current Idaho and Washington water quality standards do not specifically address BOD; however, Idaho has a narrative criterion that require waters to be free from oxygen-demanding materials in concentrations that would result in an anaerobic water condition. Additionally, Idaho and Washington have water quality standards for dissolved oxygen (see discussion in section VII.E.3.b, below). Since the BOD decay rate for this system is quite low (0.043 per day), the DO deficit would not measurably occur in Idaho waters, but downstream in Washington waters. In Washington, the applicable DO standard for Class A waters is a minimum of 8.0 mg/l.

7.5.1.4.4 Effluent Limitation

Specific chemical test methods are not readily available for measuring the quantity of many degradable substances and their reaction products. Reliance in such cases is placed on the collective parameter, BOD, which measures the weight of dissolved oxygen used by microorganisms as they oxidize or transform the gross mixture of chemical compounds in the wastewater. The biochemical reactions involved in the oxidation of carbon compounds are related to the period of incubation. The complete biochemical oxidation of a given waste may require a period of incubation too long for practical analytical test purposes. For this reason, the 5-day period has been accepted as standard, and the test results have been designated as BOD_5 (5-day BOD). The BOD_5 test is essentially a bioassay procedure that is used widely to estimate the pollution strength of domestic and industrial wastes in terms of the oxygen that they will require if discharged into receiving streams. The BOD_5 normally measures only 60 to 80 percent of the carbonaceous biochemical oxygen demand of the sample and is used to estimate the gross quantity of oxidizable organic matter.

The measurement of BOD is also an indicator of the total organic load that is being discharged to a receiving stream. Compounds contributing to the total organic waste load found in pulp and paper mill wastes include terpenes, resin acids, fatty acids, phenols, formic acid, acetic acid, sacharinic acids and other small organic acids. These compounds will also contribute to the toxicity of a pulp and paper mill waste (refer to discussion of whole effluent toxicity). A report entitled "Organic Compounds in Aerated Stabilization Basin Discharge" published in TAPPI in October 1975 indicates that biological treatment systems are very successful in eliminating several of the above compounds from kraft mill wastewaters. Resin acids, fatty acids, terpenes, hydrocarbons, and phenols were found to be reduced to the same extent as the overall BOD removal efficiency. Additionally, the appropriate reductions of BOD in the wastewater can effectively lower the toxicity of the effluent.

The June – November BOD_5 effluent limitations in the 2005 permit were modified in April of 2010 for. The BOD_5 effluent limitations effective April 15, 2010 are a daily maximum and average monthly limit of 15,000 and 8,400 lb/day, respectively for June – November. The December – May limits, which were not modified, were maximum daily and average monthly limits of 55,100 and 28,800 lb/day, respectively.

7.5.1.4.5 Benchmarks

The potential toxicity of BOD to aquatic biota has not been studied. However, the potential toxicity of reduced DO concentrations has been studied and is provided in the subsequent discussion of DO in Subsection VII.E.3.b, below. Table 7-9 summarizes the DO benchmarks for

bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

Table 7-9: DO Toxicity Benchmarks for the Biological Evaluation of the Clearwater Mill

	Early Life Stages [1,2]	Other Life Stages
30 Day Mean	NA[3]	6.5
7 Day Mean	9.5 (6.5)	NA
7 Day Mean Minimum	NA	5.0
1 Day Minimum [4]	8.0 (5.0)	4.0

Notes:

1. These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in the parentheses. The 3 mg/L differential is discussed in the criteria document (USEPA, 1986). For species that have early life stages exposed directly to the water column, the figures in the parentheses apply.
2. Includes all embryonic and larval stages and all juvenile forms to 30-days following hatching.
3. NA (not applicable).
4. All minima should be considered as instantaneous concentrations to be achieved at all times.

7.5.1.4.6 Effects Analysis

7.5.1.4.6.1 Direct Effects

The environmental baseline DO concentrations (see section VII.E.3.b) show that the background concentrations in the Snake River (site 1) are between 5.9 and 14.4 mg/L DO with a mean of 8.59 mg/L. This indicates that there are times when the Snake River is below the toxicity benchmark of 8.0 mg/L DO. Therefore, adding oxygen-demanding pollutants into the water body would further decrease the DO concentration in the Lower Snake River. Consequently, this analysis considers the incremental impact of oxygen demand this action has upon the environmental baseline.

There are many factors that could be causing the reduction in the DO concentration in LGR beyond the contribution of the oxygen demanding pollutants from the Clearwater Mill. The location of the Clearwater Mill outfall 001 is at the upper end of the Lower Granite Reservoir. Dissolved oxygen concentrations may change dramatically with lake or reservoir depth. Oxygen production occurs in the top portion of a lake or reservoir, where sunlight drives the engines of photosynthesis. Oxygen consumption is greatest near the bottom of a lake or reservoir, where sunken organic matter accumulates and decomposes. In deeper, stratified reservoirs, this difference may be dramatic when there is adequate oxygen in the epilimnion but deficient in the hypolimnion. If the reservoir is shallow and easily mixed by wind, the DO concentration may be fairly consistent throughout the water column as long as it is windy. When calm, a pronounced decline with depth may be observed.

Seasonal changes also affect dissolved oxygen concentrations. During the summer months stratification in the reservoirs can occur due to water's temperature-dependent density. As water temperatures increase, the density decreases. Thus, the sun-warmed water will remain at the surface of the water body forming the epilimnion, while the denser, cooler water sinks to the bottom (hypolimnion).

At the beginning of the summer, the hypolimnion will contain more dissolved oxygen because colder water holds more oxygen than warmer water. However, as time progresses, an increased number of dead organisms from the epilimnion sink to the hypolimnion and are broken down by

microorganisms. Continued microbial decomposition eventually results in an oxygen-deficient hypolimnion. If the lake or reservoir is in a eutrophic state, this process may be accelerated and the dissolved oxygen in the lake could be depleted before the summer's end.

Warmer temperatures during summer also speed up the rates of photosynthesis and decomposition. When all the plants die at the end of the growing season, their decomposition results in heavy oxygen consumption. Other seasonal events, such as changes in water levels, volume of inflows and outflows, and presence of ice cover, also cause natural variation in DO concentrations.

Mid-summer, the warmer surface water temperature of a lake or reservoir may limit the total amount of oxygen present. If the water becomes too warm, even if 100% saturated, O2 levels may be suboptimal for many fish species. In other words, oxygen can be present in the water, but at too low a concentration to sustain aquatic life. When strong thermal stratification develops, fish may become stressed when the epilimnion strata is too warm for them, while the hypolimnion has too little oxygen. Conditions may become especially serious during a spate of hot, calm weather that could result in the loss of many fish.

The draft permit limit for BOD in the Clearwater Mill discharge limits the effect to less than 0.11 mg/L DO below background conditions in the summer 95 percent of the time (on average, the allowed effect is 0.056 mg/L DO below background conditions in the summer). This means that the maximum summer DO deficit due to the final effluent limits would be 0.2 mg/L. As shown in Figure VII-13, this effect frequently occurs between RM 40 and 70, within Lower Monumental pool, although the RBM10 model predicts the effect to occur near the mouth of the Snake River (at the confluence with the Columbia River) under the most extreme conditions.

In the winter and spring (December through May), the analysis EPA conducted of the technology-based BOD limits in the Clearwater Mill discharge showed a maximum effect of 1.0 mg/L DO decrease from background DO concentrations. This decrease is not expected cause the DO concentrations downstream of the Clearwater Mill outfall 001 to decrease below the benchmark (8 mg/L DO).

After the analysis above, EPA requested that two additional model scenarios be performed. The first scenario represents a Clearwater effluent BOD5 load of 8,400 lb/d with no additional loads present. Figure VII-13 depicts a probability plot of the set of delta DO values resultant from these loading conditions. The 95^{th} delta DO is approximately 0.106 mg/L.

The second scenario includes the addition of BOD loads from three nearby municipalities; City of Lewiston Wastewater Facility (1430 lb/d), City of Clarkston Wastewater Facility (459 lb/d, and City of Asotin Wastewater Facility 41 lb/d. The BOD5 loadings for these 3 facilities were input at the Clearwater effluent location. A BODu to BOD5 ratio of 3.2 was used. For this scenario, the Clearwater effluent BOD5 load is kept at 8,400 lb/d, the same loading specified for scenario 1. Figure VII-13 depicts a probability plot of the set of delta DO values resultant from this scenario loading conditions. The 95th delta DO is approximately 0.121 mg/L.

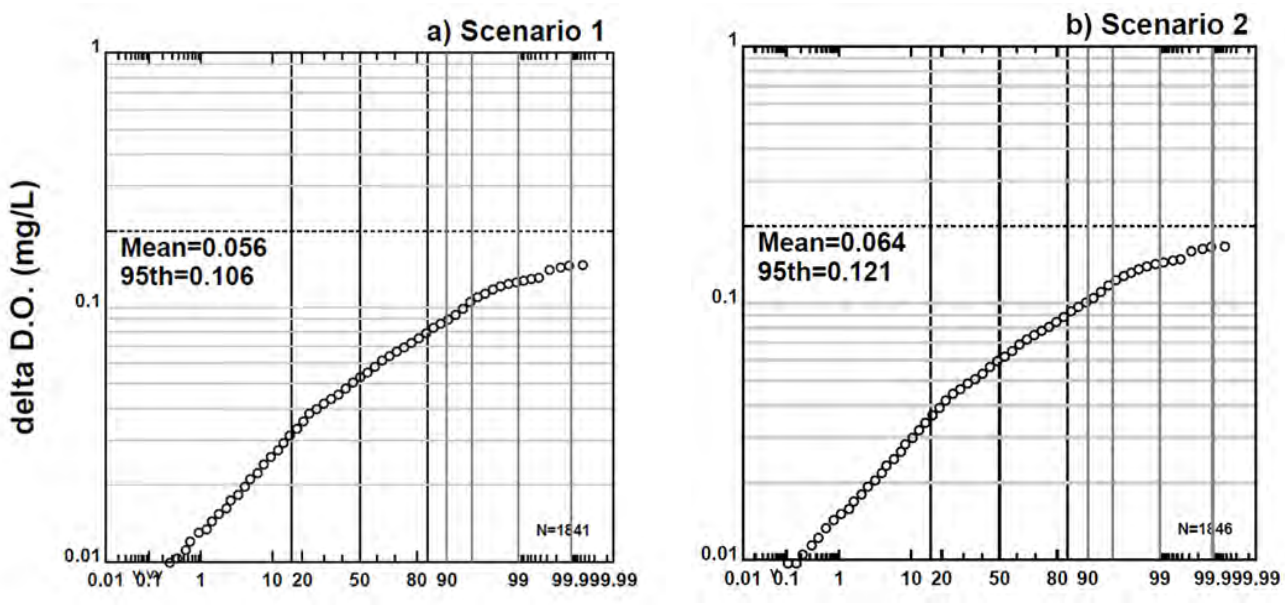


Figure 7-11: Probability plot of predicted impacts to DO in LGR due to background BOD and DO levels (no discharge) and contributions of BOD and DO from the Clearwater Mill discharge using RBM10 model.

Therefore, EPA has concluded that the final BOD effluent limitations in the draft permit **may affect, but are not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.4.6.2 Habitat Effects

Since the maximum effect of the final BOD limits will cause a minimal DO deficit in the water column, EPA has concluded that the discharge of BOD **is not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.5 Total Suspended Solids (TSS)

7.5.1.5.1 Introduction

Total suspended solids (TSS) include both organic and inorganic particulate matter in water and refer to the portion of total solids retained on a 2 μm (or smaller) filter (APHA, 1998). Total solids are the material left from a liquid mixture (e.g., effluent) after evaporation and drying at a defined temperature (APHA, 1998). Turbidity of water is related to the amount of suspended and colloidal matter contain in the water. It affects the clearness and penetration of light that may impair the photosynthetic activity of aquatic plants. The degree of turbidity is only an expression of one effect of suspended solids upon the character of the water. Turbidity is in part

measured by the total suspended solids test and thereby; turbidity is not considered a separate pollutant. Therefore, this analysis is for both TSS and turbidity.

Particulate matter is ubiquitous in natural surface waters, originating from both biological and non-biological sources. Biological particulate matter includes dead cellular material and other organic matter. Non-biological particulate matter is typically sediment washed off the land surface or resuspended from the water-body bottom. Suspended solids concentrations in natural waters vary: the TSS of Lake Superior is about 0.5 mg/L (Chapra, 1997); during floods on the Missouri River in 1993, TSS concentrations exceeded 2000 mg/L (Holmes, 1996). TSS levels in the Snake River are often less than 10 mg/L but may be as high as 60 mg/L (Normandeau, 1999).

Toxicity studies on suspended solids suggest that solids can directly cause toxicity to aquatic biota or can settle to the bottom of the receiving water body and cause toxicity to the benthic community that serves as a prey base for other aquatic biota. Primary treatment in the clarifier has been shown to remove much of the suspended solids in effluent that derive from wood fiber (Laws, 1993). Suspended sediment also reduces the clarity of water (increases turbidity), and thus can interfere with the ability of predator species to find prey. Turbidity refers specifically to the extent to which light is scattered by suspended particulates and soluble material in the water. High turbidity levels would be measured in a cloudy or muddy water body, whereas low turbidity levels would be measured in clear water.

The deposition and accumulation of organic material from municipal, industrial, and agricultural sources can result in a decrease in dissolved oxygen in bottom sediments and other chronic effects which are detrimental to a freshwater ecosystem. The adverse effects of sludge deposits can occur independently of the condition of the overlying water. Anaerobic sediments will kill benthic organisms that require oxygen in the sediments to survive. If bottom deposits become anaerobic, hydrogen sulfide, methane and carbon dioxide gases can be produced. These ebullient gases can affect unstable bottoms and raise mats of decaying organic matter, which are odiferous and aesthetically unpleasing. In addition to sulfides, ammonia is produced from the decomposition of protein and both these materials may be toxic to aquatic life.

Filling in of aquatic environments by sediments and the release of nutrients by decomposition contribute to eutrophication. Low dissolved oxygen concentrations in sediments can kill the eggs of important fish that deposit them on the bottom (such as salmon and trout) or build nests (such as bass and bluegills). Suspension of organic sediments of oxygen demanding sludge during rainfall and increasing river velocities and turbulence, can exert an oxygen demand on the overlying waters and may result in massive fish kills. Sludge deposits also can harbor pathogenic microorganisms that may increase in numbers because of growth supported by organic nutrients in the decaying deposit.

Total suspended solids from aeration stabilization basins (ASBs) largely consist of biological solids generated in the ASB. In a study of the fate of biosolids from biologically treated effluent, it was found that over 95% of the biosolids measured as TSS in a bleached kraft mill full-scale and a laboratory ASB systems were of bacterial origin (NCASI, 1978b). This observation was made upon examination of the final mill effluent and the rediluted solids from the laboratory treatment system via phase microscopy with a hemocytometer, a glass chamber with etched grids used under the microscope as an aid in counting cells.

Samples of effluent from three full-scale ASB treatment facilities (two integrated bleached kraft and one waste paperboard) and a laboratory scale pilot ASB system (dilute kraft liquor) were

analyzed via microscopy to observe the final effluent residual materials, particle morphology, and monitor microbial activity. The scanning electron microscope used for elemental analysis could not directly differentiate elements of lower atomic weight, including the major constituents of organic matter. The presence of these elements as a group was determined from the type and intensity of instrument elemental energy readout. Analysis for mat elemental composition showed a predominance of low atomic weight elements leading to the conclusion that these materials were of organic origin and derived from biological solids generated in the aerated basin (NCASI, 1978a).

Lignin is commonly found in pulp mill effluents as a wood by-product. Degradation products of lignin, along with resin and fatty acids, are component of cooking liquors, such as black liquor, which is the liquid recovered after cooking wood chips at high temperature in the presence of high temperature and pressure in the presence of sodium hydroxide, sodium sulfide, and sodium carbonate (Hodson et al., 1997; Lehtinen et al., 1990). Lignin can accumulate in sediments where it can be used as an indicator of terrestrial woody vegetation and of pulp mill activity (Louchouarn and Lucotte, 1998; Louchouarn et al., 1997). Downstream of a pulp mill in Finland, concentrations of high molecular mass fractions of lignin in sediments ranged from 62 to 97 mg/g dry weight sediment. Combustion of lignin from mill processes has been associated with the release of various hydrocarbons and semi-volatile or volatile compounds (e.g. methane, ethylene, ethane, propylene, acetylene, butane, benzene, toluene, ethylbenzene, styrene, indene, naphthalene, dibenzofuran, phenanthrene, chrysene) (Font et al., 2003).

Suspended solids in pulp mill effluents were previously assumed to be a byproduct of pulp production consisting of residual small wood chips (fiber) that have not been converted to pulp (Laws, 1993). However, more recently, suspended solids in effluent from pulp mills using secondary treatment are believed to be the dead microbes from biological treatment in the aerated stabilization basin (Ellis et al., 2003). The National Council for Air and Stream Improvement (NCASI), as well as other researchers, has studied the composition of suspended solid materials from pulp and paper mill equipped with an aerated stabilization basin (ASB), as is the case for the Clearwater Mill. In a study of a pulp and paper mill in Kawerau, New Zealand, solids in untreated effluent were determined to be wood fiber but were found to be biosolids (bacterial biomass) in treated effluent (Ellis et al., 2003). In the study of the three full-scale ASBs and the one laboratory scale ASB discussed above, very few wood fibers were present. At all facilities, fibrous material was seldom present in the sample analyzed (NCASI, 1978a). This was observed with both the scanning electron and optical microscope. Similarly, in the study of the fate of biosolids from biologically treated effluent, the amount of wood fiber in the final mill effluent and the rediluted solids from the laboratory treatment system was insignificant (NCASI, 1978b). NCASI (1978b) states that this is consistent with other findings where pulp mill effluent biosolids have been examined. In general, post-ASB solids can be characterized as microbial in nature, not fibrous, and primarily comprised of small particles.

Literature sources identify the range of particle sizes of suspended biological solids in mill effluent. Analysis of particles according to size and shape showed dispersed cells to be 1 to 6 microns, and particles up to 15 microns in diameter comprised of agglomerated bacterial cells. The 1-to-6-micron particle size portion was reported to be 73% of the total while the 5-to-15-micron size represented 24% of the total suspended solids. (NCASI, 1977). Table 7-10 shows the types and percentages of biosolids found in the final mill effluent for three sampling rounds. The majority of the biosolids are small (<1.5 microns) in size.

The values given in Table 7-10 should not be interpreted as absolute percentages of each size range in the effluent because very small particles of less than 1 μm are more difficult to see with a microscope than the larger particles. Furthermore, visual acuity limits sizing to an approximate 0.4 μm cutoff. Therefore, the actual percentage of particles in the smallest size range may be somewhat greater than indicated, and the percent of larger size particles may be less than indicated (NCASI, 1978b).

In the study of the three full-scale ASB treatment facilities and the laboratory scale pilot ASB system described above, individual particle size varied from several microns down until particles were no longer visible using the optical microscope. Flocculant solids ranged up to 30 μm

Table 7-10: Types and Number of Biosolids Found in the Final Mill Effluent (NCASI, 1978b)

Size (μ)	Mill ASB (%)		
	6/9/77	6/16/77	6/23/77
< 1.5 (single or double round)	80.4	82.1	78.8
1.5 – 6 (small flow; rod chains; single round)	13.8	15.0	18.5
6 – 10 (spiral; floc; fiber)	3.4	1.9	1.6
10 – 20 (floc; fiber; filament)	2.2	0.6	0.6
> 20 (floc)	0.2	0.3	0.5

(NCASI, 1978a). A sieve analysis was also performed on samples taken from the laboratory scale pilot ASB system (Table 7-11). The sieve analysis shows that most of the particles are less than 8 μm.

Table 7-11: Sieve Analysis (Particle Gradation) of TSS in Laboratory Scale Pilot ASB System - (NCASI, 1978a)

	Percent Retained Filter Pore Size (microns)				TOTAL SS
Sample	8.0	1.0	0.8	0.4	mg/L
ASB effluent	5	84	7	4	110
Settled 5 days	1	82	12	5	81
Settled 10 days	0	82	3	15	66

7.5.1.5.2 Environmental Baseline

7.5.1.5.2.1 Surface Water

Surface water and mid-depth TSS concentrations were measured at one location on the Clearwater River, upstream, near the confluence of the Snake River (just below the diffuser) and at four locations downstream on the Clearwater River as part of the tier 1 endangered species act monitoring and NPDES compliance monitoring studies conducted in 2005 and 2006. During the two-year sampling period, the concentration at the upstream monitoring location, from both

surface water and mid-depth samples, ranged from non-detect (half the detection limit is 1.25 mg/L) to 8.0 mg/L. Out of the total 68 water samples collected from the upstream monitoring location, only seven samples had detectable concentrations of TSS. At the four downstream locations, TSS concentrations during the sample period were similar to those observed closer to the diffuser. The concentration at the four downstream monitoring locations, from both surface water and mid-depth samples, ranged from non-detect (half the detection limit is 2.5 mg/L) to 10.0 mg/L. Only 22 of the total 255 samples from the downstream locations had detectable concentrations of TSS. It should be noted that over the two-year period 15 out of the 22 downstream samples with detectable TSS concentrations came from the site furthest downstream and away from the diffuser. For comparison, TSS concentrations were analyzed, from surface and mid-depth samples, at one location above the diffuser on both the Clearwater and Snake Rivers. A total of 34 samples were collected from the Clearwater River during the sampling period and TSS was not detected in any of the samples (half the detection limit is 2.5 mg/L). The concentration in the Snake River ranged from non-detect (half the detection limit is 2.5 mg/L) to 25.0 mg/L. TSS concentrations for 11 of the total 68 samples from the downstream locations were detectable (Figure 7-12).

7.5.1.5.2.2 Groundwater

TSS concentrations were measured quarterly from eight groundwater sampling wells surrounding the ASB pond in 2005 and 2006 as part of the NPDES groundwater monitoring requirement. During the monitoring period TSS concentrations ranged from non-detect (detection limit not reported) to 236 mg/L. The median TSS concentrations for the 2005 and 2006 sampling periods were 41.0 mg/L and 58.5 mg/L, respectively. It should be noted that the samples from the third quarter in 2005 were not analyzed within the method hold time.

7.5.1.5.3 Water Quality Standard

The current Idaho water quality standards require the turbidity in the Lower Snake River below any applicable mixing zone not to exceed background turbidity by more than 50 NTU instantaneously or more than 25 NTU for more than ten consecutive days based on the goal of protection of aquatic life. The current Washington water quality standards restrict the increase of turbidity in Class A waters, the category assigned to the Snake River, to 5 NTU when background is 50 NTU or less and 10% or 25 NTU, whichever is less, when background is greater than 50 NTU.

7.5.1.5.4 Effluent Limitation

Most suspended solids of mill origin can be removed by proper treatment. The ELGs for TSS are based on production. The ELGs also allows for the addition of limitations from wet barking and log and chip washing operations under Subpart B. The Timber Products ELGs does not allow the discharge of process wastewater from mechanical barking, sawmills, planing mills, and finishing operations, but does provide effluent limitations for hydraulic barking.

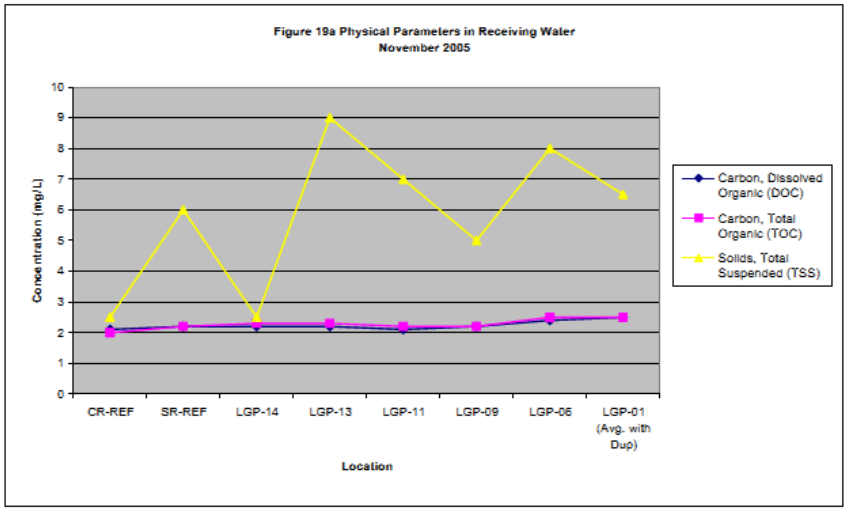
The effluent limitations for TSS specified in the draft permit are based upon technology rather than water quality because there is not a specific water quality criterion for this parameter, nor is it feasible to develop one since the composition of TSS can vary greatly amongst industries and dischargers. Therefore, EPA relies on turbidity to ensure protection of the water quality standard.

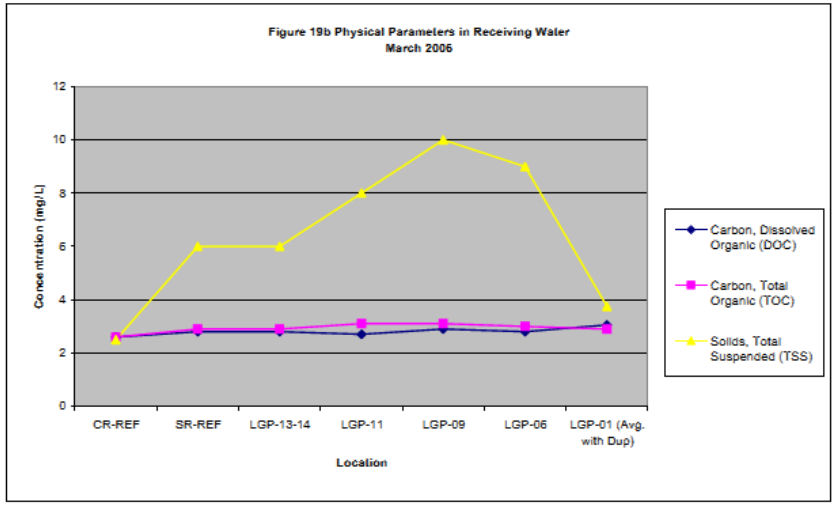
The proposed 2019 TSS effluent limitations require a maximum daily effluent limitation of 86,694 lb/day and an average monthly effluent limitation of 46,436 lb/day. These limits are based upon 2011 production of unbleached kraft market pulp and are equivalent to an effluent concentration of **274 mg/L** and **147 mg/L**, respectively, based upon an effluent flow rate of 31.6 mgd.

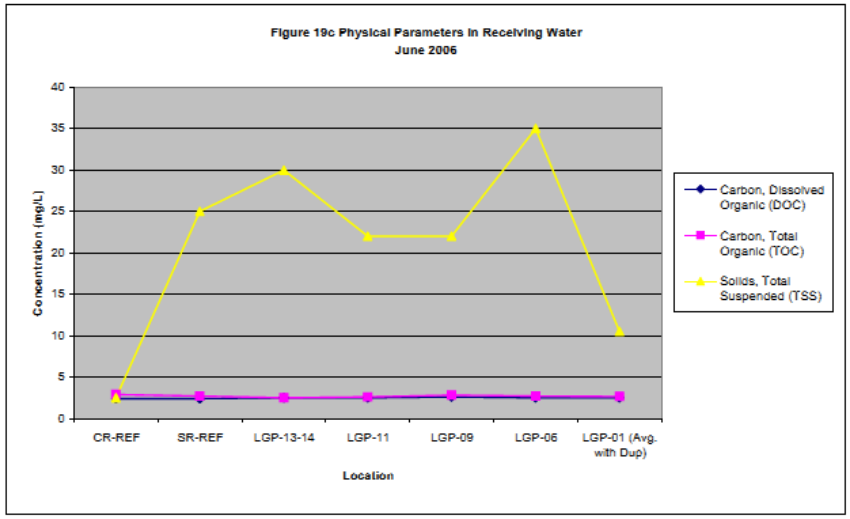
The 2005 permit included a maximum daily effluent limitation of 94,400 lb/day and an average monthly effluent limitation of 50,600 lb/day. These limits were based upon production of unbleached kraft market pulp from 2000 to 2005 and are equivalent to an effluent concentration of **283 mg/L** and **152 mg/L**, respectively, based upon an effluent flow rate of 40 mgd.

The 2005 permit also stated, in footnote #14 to Table 1, that "By May 1, 2008 the permittee will reduce TSS by 25% determined by comparing a 12-month rolling average to the 2002 annual average discharge level." The 2002 annual average effluent loading of TSS (calculated as the average of the monthly average loadings reported in 2002) was 18,723 lb/day. A 25% reduction from this loading is 14,042 lb/day. The EPA considers footnote #14 to Table 1 in the 2005 permit to be an enforceable effluent limitation.

In the 2019 draft permit, the EPA has stated this effluent limit directly in Table 1 as a 12-month rolling average effluent limit of 14,042 lb/day, instead of a footnote. This is equivalent to a concentration of **44 mg/L**, based upon an effluent flow rate of 31.6 mgd.







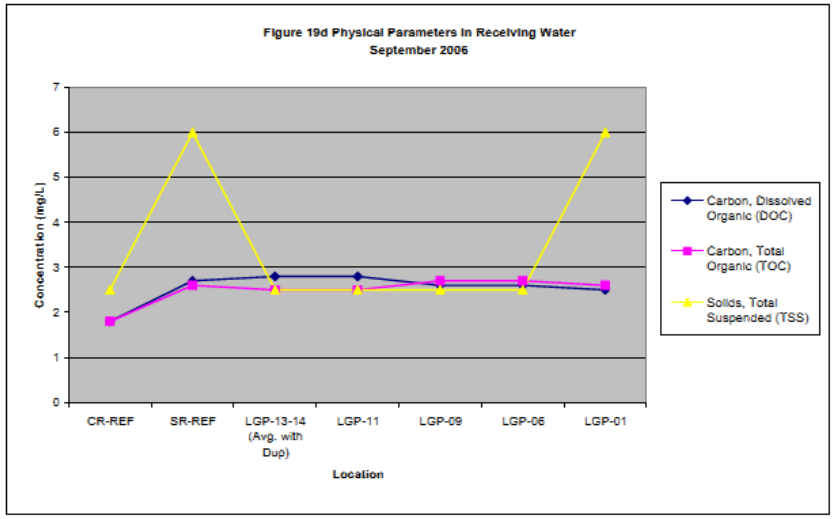


Figure 7-12: Total suspended solids (TSS), total organic carbon (TOC), and dissolved organic carbon measured in surface waters for Snake and Clearwater Rivers including the Lower Granite Pool (AMEC 2007).

7.5.1.5.5 Benchmarks

The TSS in the pulp mill effluent is comprised of dead or dying microbes (biomass) adhered to residual pulp fibers (lignins) from the clarifier. Little relevant toxicity information is available for lignin. Studies have been conducted using compound mixtures such as BKME or black liquor, which may contain lignin. However, the findings of these studies cannot be attributed to lignin itself, because many other compounds were present in the exposure medium. Adsorbable organic halides (AOX) are also attached to the biomass through biosorption. Since AOX is discussed in another section of this BE (section VII.E.1.g), the benchmarks established here are for the biomass portion of the TSS.

At extremely high concentrations, TSS can be associated with habitat changes such as reduced light penetration. Severely reduced light penetration can cause reductions in photosynthesis in bottom vegetation is light is prevented from reaching the river bottom. Although little quantitative information is available regarding the high concentrations of solids that would be required to cause light blockage severe enough to reduce photosynthesis in bottom vegetation, it is all but certain that the concentrations of TSS in effluent and observed in the Snake River downstream of the confluence are lower than would be required to induce such effects.

USEPA's (1986) Quality Criteria for Water includes qualitative assessment of the potential toxicity of total suspended solids (TSS). Four types of toxicity were observed in toxicity tests using suspended solids (USEPA, 1986):

- Mortality, reduced growth rate, and reduced resistance to disease in fish;
- Reduced success in development of fish eggs and larvae;
- Changes in natural movement and migration of fish; and
- Reduced abundance of prey items for fish.

Although USEPA reports that these types of toxicity were observed, the report does not identify the concentrations that caused the toxicity. A review of the literature on the potential toxicity of TSS was conducted to identify studies evaluating TSS comprised of particle sizes similar to that in the Mill's effluent. As noted in Newcombe and Jensen (1996), particle size is a major determinant of potential toxicity associated with TSS. The authors found that large particle sizes (> 75 μm) were associated with more severe toxicity than smaller particle sizes (< 75 μm). Because the majority of the TSS in the Mill's effluent is smaller than 100 μm (Klopping, P., personal communication, September 22, 2003), studies using TSS with particle sizes less than 100 μm were evaluated to identify toxicity benchmarks for TSS for use in the BE.

Whitman et al. (1982) studied the effects of ash from the eruption of Mount Saint Helens on the homing behavior of adult Chinook salmon. Measurements of the size of the ash particles indicated a range of particle sizes of 3 to 60 microns (μm). Whitman and coworkers created a Y-shaped apparatus in which the fish's "home" water was placed in one branch and city water was placed in the other branch. Fish would choose which branch to continue migrating. When no ash was added to either branch, 80% of fish preferred the home water branch, and 20% preferred the non-home water branch. In the first experiment, ash was added to achieve a river concentration of 650 mg ash/L water in the home water and no ash was added to the non-home water. 55% of fish preferred non-turbid non-home water, and 45% of fish preferred the turbid home water. However, when ash was added to both home and non-home branches to achieve a concentration of 650 mg/L, 89% of fish preferred the home water branch and 11% of fish

preferred the non-home water branch. Therefore, a concentration of 650 mg/L suspended solids, because it does not alter migratory behavior, is a NOEC for homing behavior of Chinook salmon.

The potential effects of suspended solids on rainbow trout survival, gill health, and fin health were studied by Herbert and Merkens (1961). Suspended solids of size ranging from 0.46 μm to 17.5 μm were added to aquarium water at various concentrations. Concentrations of 270 mg/L were found to result in more fin rot and much lower survival of rainbow trout, compared to controls. Somewhat lower survival compared to controls was observed at a concentration of 90 mg/L, but no effects on gill health or fin health were observed at this concentration. At 30 mg/L suspended solids, survival did not differ from controls and gill effects were not observed. From these data, a NOEC of 30 mg/L can be established for survival, gill effects, and fin effects.

Herbert et al. (1961) studied the potential effects of suspended solids on brown trout abundance at several stations. At each station, the concentrations of total suspended solids and particle size distribution were measured throughout the duration of the study (approximately one month). At the majority of the stations, the median concentrations of TSS ranged from 934 mg/L to 7470 mg/L; however, at one station, the median concentration was 58.6 mg/L. Particle sizes were generally less than 60 μm. Fish counts at each station were made using a cat-effort method and a recapture of introduced fish method. The results of the survey indicated that concentrations of TSS above 1000 mg/L were associated with markedly reduced abundance of brown trout, whereas concentrations of about 60 mg/L had no adverse effect on brown trout abundance. Therefore, 60 mg/L represents a NOEC for survival in brown trout.

Adult grayling were studied by McLeay et al. (1987). Both inorganic and organic suspended solids were evaluated, at a wide range of concentrations as high as 100,000 mg/L, and particle sizes ranging from 38 μm to 200 μm. Because the majority of the inorganic TSS had particle sizes less than 38 μm, whereas only 3% of the organic TSS had particle sizes less than 38 μm, the results of toxicity studies using the inorganic TSS is more relevant to the TSS in the Mill's effluent.

Servizi and Martens (1991) exposed juvenile Coho salmon to TSS concentrations ranging from 1000 mg/L to 40,000 mg/L. 90% of the particles were less than 5 μm. Following 96 hours of exposure, LC50 concentrations were reported for small fish (3.8 to 7.3 cm) and larger fish. The lowest LC50 observed was 1300 mg/L at fish kept at 16 ºC. Notably, this population of fish was determined to have a kidney infection. Therefore, the LOEC from this study is 1300 mg/L.

In 1992, Servizi and Martens (1992) reported the results of a similar study, in which biological and behavioral indicators were measured in fish exposed to 20 mg/L to 2550 mg/L TSS. At concentrations of 240 mg/L to 2550 mg/L, cough frequency and avoidance were greater than controls. At concentrations of 530 mg/L to 1360 mg/L, glucose levels were not different from controls, but at concentrations of 1530 to 1630, glucose levels were greater than controls. At 20 mg/L, cough frequency was not different from controls. From these findings, a NOEC of 20 mg/L of TSS is selected.

Shaw and Mago (1942) studied the effects of mining silt on salmon eggs and fry. At concentrations of 860 mg/L to 2020 mg/L TSS, the survival of eggs and fry was reduced. From these findings, a LOEC of 860 mg/L is selected.

NOECs from the studies described above range from 20 mg/L to 650 mg/L, and the LOECs range from 100 mg/L to 1300 mg/L. Applying an uncertainty factor of 10 to the LOEC, concentrations result in estimated NOEC concentrations of 10 mg/L to 130 mg/L. Therefore, the lowest measured or estimated NOEC for TSS is 10 mg/L. **This BE uses 10 mg/L TSS as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.5.6 Effects Analysis

7.5.1.5.6.1 Direct Effects

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the chronic mixing zone, where the State authorizes acute criteria to be exceeded, would be 36.5. However, the background concentration of TSS in the Snake River is frequently above the toxicity benchmark. The 90th percentile TSS concentration observed at the Washington State Department of Ecology's monitoring station on the Snake River at the interstate bridge (station #35A150) during the term of the previous permit is 31.2 mg/L, and 32% of samples had a concentration greater than the toxicity benchmark of 10 mg/L. Therefore, the exposure concentration is greater than the toxicity benchmark.

EPA has concluded that the discharge of TSS at the effluent limitation **is likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.5.6.2 Habitat Effects

The potential for the suspended solids in the Clearwater Mill's effluent to contribute to bedload sedimentation was also evaluated. Particulates that are heavier than water tend to settle from the water column due to gravity, however this tendency can be counteracted by turbulence. As a result, particle settling is greater in quiescent waters than in turbulent waters. Thus, the slow-moving waters of lakes and reservoirs usually have less suspended sediment than the fast-moving waters of rivers. In rivers, impounded sections create conditions that favor particle settling and the removal of suspended sediments.

The ability of reservoirs to remove suspended sediment is captured in a measure known as the "trapping efficiency," which is the percentage of the incoming particulate matter that remains in the reservoir. Dendy (1974) provides the following empirical formula to estimate the trapping efficiency of a reservoir:

$$TE = 100 \times 0.97^{0.19^{\log C/I}}$$

where,

TE is the trapping efficiency (%);
C is the reservoir storage volume; and
I is the annual average inflow volume to the reservoir in consistent units with C.

Applying the trapping efficiency formula to the Snake River reservoirs predicts trapping efficiencies of 99% for Lower Granite and Little Goose Reservoirs, and 98% to Lower Monumental and Ice Harbor Reservoirs. The cumulative trapping efficiency through all four

reservoirs is 100%. Therefore, it is unlikely that any suspended particles from the Clearwater Mill's effluent will travel beyond Ice Harbor Dam.

Recent evidence has shown that downstream from a pulp and paper mill effluent discharge, particles form, coagulate and flocculate into larger particles faster than predicted by current sediment transport models (Krishnappan, 1996). Existing sediment transport models have failed to consider the phenomenon of pulp mill effluent induced coagulation and flocculation (PMEICF). Conventional models assume that all particles behave as individual particles and flocculation does not occur. The microbial involvement in biological floc formation is well documented (Pavoni, 1972; Paerl, 1974; Rao et al., 1991; Mueller, 1996). Bacteria excrete polymeric substances, which may be significant in the floc formation. However, only some bacterial species are considered "floc-formers" (Friedman and Dugan, 1968). The physiological changes of the bacteria could influence the observed induced flocculation.

PMEICF can prohibit sufficient degradation of chemical constituents in the effluent causing a build-up of organic material on the river bottom. Some of the chemical constituents may cause adverse conditions, may be toxic or induce anoxic or toxic conditions. Additionally, they may still possess a significant BOD, resulting in low DO in the river. Changed conditions near the river bottom may cause harm to benthic organisms, which could have adverse effects on the entire food chain.

The sediment data collected by Clearwater (1997a, 1998a, 1999a, 2000a, 2001a, 2002a) indicate that settling of the suspended solids in the effluent may be occurring between Snake River Miles 137 and 131. Since the maximum effluent concentration of TSS is quite high (264 mg/L) compared with the benchmark of 10 mg/L and the findings of several studies listed above, the discharge at the TSS effluent limits concentrations are expected to contribute to bedload sedimentation, especially in upper LGR. Since the habitat in LGR does not support spawning of threatened and endangered species, the bedload sedimentation will have no effect on eggs and emergent fry. However, the LGR does support juveniles, subyearlings, yearlings, and adults, and holding habitat for bull trout and steelhead.

Additionally, the potential for decreased visibility resulting from the maximum effluent concentration of TSS as discussed in the direct effects analysis increases the potential for predation and migratory blockage. Therefore, EPA has concluded that the discharge of TSS at the proposed effluent limitation is **likely to adversely modify critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6 Chlorinated Organic Compounds (COCs)

7.5.1.6.1 Introduction

Chlorinated organic compounds (COCs) represent a group of commercially produced, substituted phenols and cresols referred to as chlorophenols and chlorocresols. Chlorinated phenols are used as intermediates in the synthesis of dyes, pigments, phenolic resins, pesticides and herbicides. Certain chlorophenols also are used directly as flea repellents, fungicides, wood preservatives, mold inhibitors, antiseptics, disinfectants, and anti-gumming agents for gasoline. COCs in the pulp and paper mill effluents are formed as byproducts of the bleaching process.

The chlorinated phenols represent a group of substituted phenols and cresols prepared by direct chlorination or the hydrolysis of the higher chlorinated derivatives of benzene. COCs include phenols, guaiacols, catechols, and vanillins substituted with from one to five chlorine atoms per molecule. Phenols contain a hydroxyl group in the number one position. Substituted catechols are mono-ortho hydroxy substituted phenols with additional chlorines as indicated in the name. Substituted guiaiacols are mono-ortho-methoxy substituted phenols with additional chlorines as indicated in the name.

Purified COCs exist as colorless crystalline solids, except for 2-chlorophenol that is a clear liquid, while the technical grades may be light tan or slightly pink due to impurities (Bennett, 1962; Kirk and Othmer, 1964; Heilbron et al., 1975; Sax, 1975; Weast, 1975; Windholz, 1976; Hawley, 1975). As a group, the COCs are characterized by an odor that has been described as unpleasant, medicinal, pungent, phenolic, strong, or persistent (Kirk and Othmer, 1964; Sax, 1975; Lange, 1952). A summary of the various pertinent physical properties of COCs of concern with this action is provided in Table 7-12 and Figure 7-13 provides the chemical structures of the COCs discussed in this BE.

In general, the volatility of the compounds decreases and the melting and boiling points increase as the number of substituted chlorine atoms increases. The solubility of the COCs ranges from soluble to very soluble in relatively non-polar solvents such as benzene and petroleum ether. Chlorophenols behave as weak acids, and the acidity increases with increased chlorination, as shown by the dissociation constants (pKa). This behavior implies that ionization of higher chlorophenols in aqueous solutions occurs over a wider pH range (i.e., Pentachlorophenol begins to dissociate at a pH of about 3.5, but 2,4,5-trichlorophenol does not dissociate below a pH of 7).

Dissociation of the chemicals influences their sorption on colloids and their toxicological properties. The toxicity of COCs to fungi is decreased as the degree of dissociation increases. Volatility and water solubility of COCs decrease with increasing degree of chlorination. Although their solubility in water is low they are readily soluble in many organic solvents. Partition coefficients (K_{ow}) in favor of the organic solvents facilitate isolation of the compounds for analysis.

It is generally accepted that COCs will undergo photolysis in aqueous solutions due to ultraviolet irradiation and that photodegradation leads to the substitution of hydroxyl groups in place of the chlorine atoms with subsequent polymer formation. Studies by Grabowski (1961) and Joschek and Miller (1966) indicated that UV irradiation of 2-chlorophenol produced catechol and/or 2,2-dihydroxydiphenyl. Omura and Matsuura (1971) reported that UV irradiation (290 mμ) of 2-chlorophenol produced a complex mixture of products, including a large quantity of resinous material while the photolysis of 3-chlorophenol produced a high yield of resorcinol. Photolysis of 2,4-dichlorophenol in dilute aqueous solutions at a peak wavelength of 253.7 mμ was virtually complete within 2 to 40 minutes depending upon the pH (Aly and Faust, 1964).

Other studies have demonstrated the photodegradation of 2,4-dichlorophenol following five hours of daily solar irradiation for 10 days (Crosby and Tutass, 1966). They observed the formation of the intermediates 4-chlorocatechol and 1,2,4-benzenetriol. The principal product of degradation recovered was a dark brown residue tentatively identified as a mixture of dechlorinated polyquinoids. Although it has been speculated that photolysis of chlorophenols may produce dibenzo-p-dioxins, no 2,3,7,8-TCDD was detected during the riboflavin-

sensitization photo-oxidation of 2,4-dichlorophenol to tetrachlorinated diphenol ethers (Plimmer and Klingebiel, 1971).

Effective microbial degradation of COCs has been demonstrated in activated sludge, lagoon effluent, and enrichment cultures. Thus, effective waste treatment of wastewater containing COCs occurs when appropriate bacterial populations are present. Bacteria capable of metabolizing COCs have been found in soil, water, pentachlorophenol-treated wood, and sewage treatment plants exposed to Pentachlorophenol-containing effluents. Certain soil bacteria can detoxify Pentachlorophenol by methylation, forming pentachloroanisole. Other bacterial strains isolated from continuous-flow enrichment cultures can metabolize PCPs

Table 7-12: Physical Properties of Chlorinated Phenols Regulated by the Draft Permit

Compound	MW	K_{ow}	log K_{ow}	MP (ºC)	BP (ºC)	Water Sol. (mg/L)	Vapor Pressure (atm)	Volatilization Half Lives (hr)	Minimum of Half Lives in Surface Water (hr)	Maximum of Half Lives in Surface Water (hr)	Geometric Mean of Half Lives in Surface Water (hr)
2,4,5-trichlorophenol	197.5	5012	3.7	66-67	245-246	1,070	2.78E-05	768	0.5	336	13
2,4,6-trichlorophenol	197.5	6310	3.69	68	246	800	1.53E-05	480	2	96	14
2,3,4,6-tetrachlorophenol	231.9	25119	4.45	69-70	150 (a)	100 @25˚C	2.76E-06	1032	1	336	18
Pentachlorophenol	266.3	794328	5.12	190	309.5 (b)	14 @20˚C	1.97E-07	3120	1	110.4	11
3,4,5-trichlorocatechol	213.5	5012	3.7			143.3 (c)					
3,4,6-trichlorocatechol	213.5	5012	3.7	275.4		143.3 (c)					
Tetrachlorocatechol	213.5	15849	4.2	184-186		46.0 (c)	1.36E-06				
3,4,5-trichloroguaiacol	227.5	12589	4.1			61.5 (c)					
3,4,6-trichloroguaiacol	227.5	7943	3.9			96.9 (c)					
4,5,6-trichloroguaiacol	227.5	631	2.8			1181.8 (c)					
Tetrachloroguaiacol	261.9	39811	4.6			26.0					
Trichlorosyringol	257.5	15849	4.2			55.5 (c)					

(a) Decomposes at 16 mmHg.
(b) Pentachlorophenol decomposes at its boiling point
(c) Formula from Lymann et al. (1990): log S = -0.9874* K_{ow} –0.0095*Tm+0.7178 for halobenzenes. [reference states to use 25˚ if temperature is below 25˚C]
Sources: NCASI (1992), CRC Handbook of Chemistry and Physics (75th Ed.) (1995), Merck Index (10th Ed.), Mackay et al. (1992), Texas NRCC (2000), HSDB (2000), Howard et al. (1991)

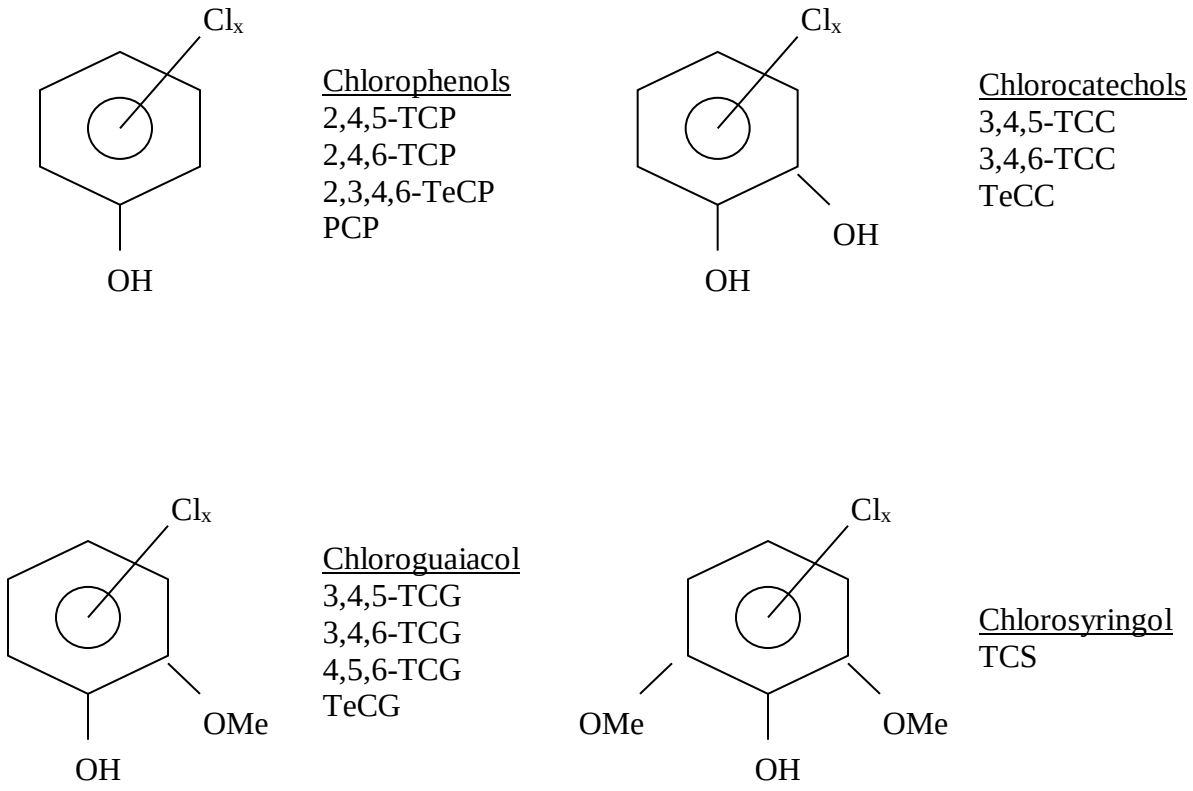


Figure 7-13: Chlorinated Organic Compounds (COC). Figure reproduced (with modification) from AMEC (2006).

quantitatively with the release of chloride, quantitative disappearance of the substrate, and almost quantitative oxygen uptake. Most bacterial strains capable of degrading COCs have been isolated from areas where the compound is commonly found or have been artificially developed in the laboratory, utilizing gradual enrichment and acclimation of the bacteria to increasing levels of COCs. The extent to which COC-metabolizing bacteria are present in the environment is not known. In most cases, rapid COC metabolism depends on gradual acclimation. Thus, the hazard posed by COCs in environments where these compounds have not previously been present may increase.

7.5.1.6.1.1 2,4,5-Trichlorophenol

2,4,5-trichlorophenol is slightly soluble in water, has an ionization constant (pKa) of 7.0 to 7.4 at 25˚C (Ahlborg and Thunberg, 1980; Doedens, 1963; USEPA, 1980), and a log n-octanol/water partition coefficient (log K_{ow}) of 3.70 (Hansch and Leo, 1979). 2,4,5-trichlorophenol is used as an algicide, fungicide, and bactericide and as an antimildew and preservation agent in cooling towers, pulp mills and in hide and leather processing (Ahlborg and Thunberg, 1980; USEPA, 1980). It is also used in the production of the pesticides erbon, fenchlorphos, fenoprop (2,4,5-TP), hexachlorophene, and 2,4,5-trichlorophenoxyacetic acid (2,4,5-T) (Ahlborg and Thunberg, 1980; Buikema et al., 1979; Doedens, 1963; Kozak et al., 1979; Stolzenburg and Sullivan, 1984).

Contamination of waters with 2,4,5-trichlorophenol and other chlorophenols has resulted from the use of chlorophenoxyacetic acid herbicides containing chlorophenolic impurities, from the chlorination of waste treatment plant effluents, and from pulp bleaching (Ahlborg and Thunberg, 1980; Buikema et al., 1979; Jolley et al., 1976; Rockwell and Larsen, 1978). Residues have been detected in fish and other organisms collected downstream from pulp mills (Paasivirta et al., 1985).

7.5.1.6.1.2 2,4,6-Trichlorophenol

2,4,6-trichlorophenol is slightly soluble in water, has an ionization constant (pKa) of 7.4 at 25˚C (Drahonovsky and Vacek, 1971), and a log n-octanol/water partition coefficient (log K_{ow}) of 3.69 (Hansch and Leo, 1985). 2,4,6-trichlorophenol has previously been used as a fungicide, herbicide, and defoliant, though most uses have been cancelled within the U.S. (Lewis, 1997). It is also useful as an ingredient in the preparation of insecticides and soap germicides.

7.5.1.6.1.3 2,3,4,6-Tetrachlorophenol

2,3,4,6-trichlorophenol (2,3,4,6-tetrachlorophenol) is soluble in water up to 100 mg/L (USEPA, 1979), and a log n-octanol/water partition coefficient (log K_{ow}) of 4.45 (Hansch and Leo, 1985). 2,3,4,6-tetrachlorophenol is used as a fungicide (Lewis, 1997), as a germicide for the preservation of wood, latex, and leather, and as insecticides (Doedens, 1963). As of 2016, many of the uses are discontinued.

7.5.1.6.1.4 Pentachlorophenol

Pentachlorophenol was one of the most widely used biocides. In 1986, approximately 28 million pounds were used in the United States. It was registered for use as a molluscide, fungicide, herbicide, insecticide, disinfectant, wood preservative, slimicide, and paint preservative. In 1984, EPA restricted its use; consequently, it is no longer available for home and garden use (ATSDR, 1993). Approximately 80% of the total technical grade pentachlorophenol use is for wood preservation. Most wood treated with pentachlorophenol is done so commercially, using pressurized treatment. Treatment with PCP results in a 5 to 8-fold increased useful life of wood products. The aqueous form, sodium pentachlorophenate (NaPCP) has been used in pressboard, insulation, and industrial cooling water, among other uses (Crosby, 1981; Eisler, 1989). At pulp and paper mills, pentachlorophenol is formed as byproducts of the bleaching process or from the use of biocides and slimicides.

Pentachlorophenol is slightly soluble in water, while its alkaline salts, such as sodium pentachlorophenate (Na-PCP), are highly soluble in water (Weast, 1975). Pentachlorophenol is soluble in water up to 1,000 mg/L at 25˚C (Scow et al., 1980). The chemical properties of pentachlorophenol, however, are closely related to the pH of the aqueous solution. Pentachlorophenol has a pK_A of 4.7, which means that at a pH of 4.7, aqueous solutions will contain 50% ionized PCP. At a pH 6.7, that of many natural waters, pentachlorophenol is 99% ionized. This ionization makes pentachlorophenol more water soluble, and therefore more mobile, in soil at neutral pH (Crosby, 1981; ATSDR, 1993). Pentachlorophenol has a log n-octanol/water partition coefficient (log K_{ow}) of 5.12 (Hansch and Leo, 1985).

Once released to water, the half-life of pentachlorophenol ranges from less than one day to 15 days. The degree of degradation is controlled by the amount of incident radiation (sunlight penetration), dissolved oxygen, and pH of the water. Photolysis and degradation by

microorganisms are considered the major mechanisms by which pentachlorophenol is degraded in water. Degradation of pentachlorophenol in water forms other compounds, primarily pentachloroanisole, 2,3,4,5-tetraclorophenol, 2,3,4,6-tetrachlrophenol, and 2,3,5,6-tetrachlrophenol (ATSDR, 1993). The toxicity of pentachlorophenol increases with decreasing pH. However, since pentachlorophenol is rarely present in pure form, accurate measurement is difficult and raises questions regarding pentachlorophenol toxicity tests.

Pentachlorophenol was shown to undergo photochemical degradation in aqueous solutions by ultraviolet irradiation and sunlight, with the formation of several chlorinated benzoquinones, 2,4,5,6-tetrachlororesorcinol, and chloranilic acid (Mitchell, 1961; Hamadmad, 1967). Wong and Crosby (1977) reported the degradation by sunlight or UV light of dilute solutions (100 mg/L) of pentachlorophenol to lower chlorophenols, tetrachlorodihydroxybenzenes, and non-aromatic fragments such as dichloromelaeic acid. Subsequent irradiation of the tetrachlorodiols produced hydroxylated trichlorobenzoquinones, trichlorodiols, dichloromaleic acid, and non-aromatic compounds. The irradiation of dichloromaleic acid produced chloride ions and carbon dioxide.

One of the primarily modes of action of pentachlorophenol, and other chlorophenols, is inhibition of oxidative phosphorylation, which causes a decrease in the production of adenosine triphosphate (ATP) in plants and animals. One consequence of this impairment is increased basal metabolism, resulting in increased oxygen consumption and high fat utilization. The effects of pentachlorophenol may reduce the availability of energy for maintenance and growth, thus reducing survival of larval fish and ability of prey to escape from a predator (Johansen et al. 1985, Brown et al. 1985, Eisler 1989). Pentachlorophenol is known to cause several types of adverse effects in animals including dysfunction of the reproductive, nervous, and immune systems, hormone alterations, and impaired growth. In general, fish growth and behavioral endpoints have been shown to be sensitive indicators of pentachlorophenol exposure (Webb and Brett 1973, Hodson and Blunt 1981, Dominguez and Chapman 1984, Brown et al. 1985). This pesticide is also considered a probable human carcinogen.

In general, fish are more sensitive to pentachlorophenol than are other aquatic organisms. Salmonids have been found to be the most sensitive fish species tested under acute exposure conditions (Choudhury et al., 1986; Eisler, 1989; USEPA, 1980, 1986, 1995, 1996). Warmwater species are generally less sensitive than coldwater species in acute lethal toxicity tests (USEPA, 1995a). Evaluation of threatened or endangered salmonid species against the rainbow trout, a typical test organism, found that the Apache trout (*Oncorhynchus apache*) was more sensitive than the rainbow trout in acute lethality tests with pentachlorophenol, indicating an additional margin of safety may be needed to protect listed salmonids when using rainbow trout test data in toxicity assessments (USEPA, 1995a).

7.5.1.6.2 Environmental Baseline

Evidence has accumulated that the various chlorophenols are formed as intermediate metabolites during the microbiological degradation of the herbicides 2,4-dichlorophenol (2,4-D) and 2,4,5-trichlorophenoxy acetic acid (2,4,5-T), and pesticides Silvex®, Ronnel®, lindane, and benzene hexachloride (Kearney and Kaufman 1972; Steenson and Walker 1957; Fernley and Evans 1959; Loos et al. 1967; Aly and Faust 1964; Crosby and Tutass 1966; Watts and Stonherr 1973; Crosby and Wong 1973; Gotto et al. 1971; Leng 1976).

Chlorophenols may be produced inadvertently by chlorination reactions that take place during the disinfection of wastewater effluents or drinking water sources. The formation of 2- and 4-chlorophenol and higher phenols has been reported under conditions similar to those employed during the disinfection of wastewater effluents (Aly 1968; Bernhart and Campbell, 1972) and the synthesis of 2-chlorophenol took place in one hour in aqueous solutions containing as little as 10 mg/L phenol and 20 mg/L chlorine (Bernhart and Campbell, 1972). Other studies have demonstrated the formation of up to 1.7 µg/L 2-chlorophenol during the chlorination of sewage effluents and cooling tower waters (Jolly, 1973, 1975).

Chlorinated organic compounds (COCs) were analyzed in samples collected from the Snake and Clearwater Rivers, Lower Granite Pool, and Effluent in August and November 2005 and January, March, June, and September 2006. Table 7-14 shows the results of the sample analysis. As shown in the table, no COCs were detected in any of the samples collected in January 2006.

Over the course of 2005 and 2006 COCs were sporadically detected, all in concentrations too low to quantify, so all values were estimated. 3,4,6-trichlorocatechlor (0.001 µg/L) and tetrachlorocatechol (0.001 µg/L) were detected in the Clearwater reference site and LGP-14 respectively (November 2005). No COCs were detected March and June 2006. 2,4,6-Triclorophenol (0.00057 µg/L) was detected at LGP-6 in September 2006 (AMEC, 2007).

7.5.1.6.3 Water Quality Standard

Because the toxicity of chlorinated phenols to various aquatic life forms is structure-dependent, giving rise to wide variability, it would be inappropriate to derive a criterion for these chemicals as a group. Instead, criteria should be derived for individual chemicals, when sufficient information becomes available (USEPA, 1980). Criteria have only been developed for 2,4,5-trichlorophenol, 2,4,6-trichlorophenol, and pentachlorophenol.

- The most stringent water quality criterion developed by EPA (USEPA, 1986) for 2,4,5-trichlorophenol is 1 µg/L for organoleptic (taste and odor) effects.
- The most stringent water quality standard in Idaho and Washington for 2,4,5-trichlorophenol is Idaho's criterion of 140 µg/L as a long-term average, for the protection of human health.
- The most stringent water quality standard in Idaho and Washington for 2,4,6-trichlorophenol is Washington's criterion of 0.25 µg/L as a long-term average, for the protection of human health. The most stringent water quality criterion developed by EPA (USEPA, 2015) for 2,4,6-trichlorophenol is 1.5 µg/L for the protection of human health.
- The most stringent water quality standard in Idaho and Washington for pentachlorophenol is Washington's criterion of 0.002 µg/l as a long-term average for the protection of human health.

For all other COCs, the Idaho and Washington water quality standards have a narrative criterion to limit toxic material concentrations to levels below those which have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic conditions to the most sensitive biota dependent upon those waters, or adversely affect public health.

7.5.1.6.4 Effluent Limitation

Nearly all of the 28 COCs for which samples were analyzed (by Method 1653) were found in bleach plant and final effluents from chemical pulp mills that bleach. Typically, bleaching processes that result in the formation of 2,3,7,8-TCDD and 2,3,7,8-TCDF also generate the higher substituted tri-, tetra-, and penta-chlorinated compounds. Of the detected COCs, 12 of the higher substituted chlorinated compounds are associated with the presence of 2,3,7,8-TCDD and 2,3,7,8-TCDF.

EPA established effluent limitation guidelines for the following 12 COCs: 2,4,5-trichlorophenol, 2,4,6-trichlorophenol, 2,3,4,6-tetrachlorophenol, Pentachlorophenol, 3,4,5-trichlorocatechol, 3,4,6-trichlorocatechol, Tetrachlorocatechol, 3,4,5-trichloroguaiacol, 3,4,6-trichloroguaiacol, 4,5,6-trichloroguaiacol, Tetrachloroguaiacol, and Trichlorosyringol. Secondary treatment can generally achieve about 50% removal of these compounds.

Because bleaching conditions at pulp and paper mills that favor formation of COCs also favor formation of 2,3,7,8-TCDD and 2,3,7,8-TCDF, limiting COCs will ensure further progress toward reducing formation and discharge of 2,3,7,8-TCDD and 2,3,7,8-TCDF below currently measurable conditions.

7.5.1.6.4.1 2,4,5-Trichlorophenol

The 2019 draft permit requires a maximum daily fiber line limit of 2,4,5-trichlorophenol be below the minimum level of 2.5 µg/L. The maximum equivalent 2,4,5-trichlorophenol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **0.955 µg/L**. This is the same as the 2,4,5-trichlorophenol limitation in the 2005 permit.

7.5.1.6.4.2 2,4,6-Trichlorophenol

The 2019 draft permit requires a maximum daily fiber line limit of 2,4,6-trichlorophenol be below the minimum level of 2.5 µg/L. The maximum equivalent 2,4,6-trichlorophenol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **0.955 µg/L**. This is the same as the 2,4,6-trichlorophenol limitation in the 2005 permit.

7.5.1.6.4.3 2,3,4,6-Tetrachlorophenol

The draft permit requires a maximum daily fiber line limit of 2,3,4,6-tetrachlorophenol be below the minimum level of 5 µg/L. The maximum equivalent 2,3,4,6-tetrachlorophenol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **1.91 µg/L**. This is the same as the 2,3,4,6-tetrachlorophenol limitation in the 2005 permit.

7.5.1.6.4.4 Pentachlorophenol

The 2019 draft permit requires a maximum daily fiber line limit of pentachlorophenol be below the minimum level of 5 µg/L. The maximum equivalent pentachlorophenol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **1.91 µg/L**.

The EPA determined that the fiber line limits for pentachlorophenol would not ensure that the applicable Washington water quality criterion for protection of human health would be met at the

border. Therefore, the permit proposes more stringent water quality-based effluent limits for pentachlorophenol, for the final effluent.

7.5.1.6.4.5 3,4,5-Trichlorocatechol

The 2019 draft permit requires a maximum daily fiber line limit of 3,4,5-trichlorocatechol be below the minimum level of 5 µg/L. The maximum equivalent 3,4,5-trichlorocatechol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **1.91 µg/L**. This is the same as the 3,4,5-trichlorophenol limitation in the 2005 permit.

7.5.1.6.4.6 3,4,6-Trichlorocatechol

The 2019 draft permit requires a maximum daily fiber line limit of 3,4,6-trichlorocatechol be below the minimum level of 5 µg/L. The maximum equivalent 3,4,6-trichlorocatechol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **1.91 µg/L**. This is the same as the 3,4,5-Trichlorocatechol limitation in the 2005 permit.

7.5.1.6.4.7 Tetrachlorocatechol

The 2019 draft permit requires a maximum daily fiber line limit of tetrachlorocatechol be below the minimum level of 5 µg/L. The maximum equivalent tetrachlorocatechol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **1.91 µg/L**. This is the same as the tetrachlorocatechol limitation in the 2005 permit.

7.5.1.6.4.8 3,4,5-Trichloroguaiacol

The 2019 draft permit requires a maximum daily fiber line limit of 3,4,5-trichloroguaiacol be below the minimum level of 2.5 µg/L. The maximum equivalent 3,4,5-trichloroguaiacol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **0.955 µg/L**. This is the same as the 3,4,5-trichloroguaiacol limitation in the 2005 permit.

7.5.1.6.4.9 3,4,6-Trichloroguaiacol

The 2019 draft permit requires a maximum daily fiber line limit of 3,4,6-trichloroguaiacol be below the minimum level of 2.5 µg/L. The maximum equivalent 3,4,6-trichloroguaiacol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **0.955 µg/L**. This is the same as the 3,4,6-trichloroguaiacol limitation in the 2005 permit.

7.5.1.6.4.10 4,5,6-trichloroguaiacol

The 2019 draft permit requires a maximum daily fiber line limit of 4,5,6-trichloroguaiacol be below the minimum level of 2.5 µg/L. The maximum equivalent 4,5,6-trichloroguaiacol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **0.955 µg/L**. This is the same as the 4,5,6-trichloroguaiacol limitation in the 2005 permit.

7.5.1.6.4.11 Tetrachloroguaiacol

The 2019 draft permit requires a maximum daily fiber line limit of tetrachloroguaiacol be below the minimum level of 5 µg/L. The maximum equivalent tetrachloroguaiacol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **1.91 µg/L**. This is the same as the tetrachloroguaiacol limitation in the 2005 permit.

7.5.1.6.4.12 Trichlorosyringol

The 2019 draft permit requires a maximum daily fiber line limit of trichlorosyringol be below the minimum level of 2.5 µg/L. The maximum equivalent trichlorosyringol concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **0.955 µg/L**. This is the same as the trichlorosyringol limitation in the 2005 permit.

7.5.1.6.5 Toxicity Benchmarks

Chlorinated organic compounds have varying degrees of toxicity. The available freshwater data for chlorinated phenols indicate that toxicity generally increases with increasing chlorination, and that acute toxicity occurs at concentrations as low as 30 µg/L for 4-chlor-3-methylphenol to greater than 500,000 µg/L for other compounds. Chronic toxicity occurs at concentrations as low as 5.67 µg/L (rainbow trout) for pentachlorophenol. Acute and chronic toxicity would occur at lower concentrations among species that are more sensitive than those tested.

Acute values for *Daphnia magna* range from 290 µg/L for 2,3,4,6-tetrachlorphenol to 6,040 µg/L for 2,4,6-trichlorophenol. The 96-hour LC50 values for fathead minnows range from 30 µg/L for 4-chloro-3-methylphenol (USEPA, 1972) to 9,040 µg/L for 2,4,6-trichlorophenol (Phipps et al., 1981). Since many chlorinated phenols are only slightly soluble in water, and since some of the chemical could be expected to be absorbed by the animals and the testing environment, the above conditions could result in a low estimate of the toxicity.

Chronic toxicity for 2,4,6-trichlorophenol was observed at 720 µg/L from an early life stage test with the fathead minnow.

7.5.1.6.5.1 2,4,5-Trichlorophenol

7.5.1.6.5.1.1 Direct Effects

The range of LC50s for rainbow trout and brown trout was 260 µg/l to 900 µg/L 2,4,5-trichlorophenol (Hattula et al., 1981; Knott and Johnston, 1971; Spehar, 1986). A 48-hour exposure of rainbow trout to 2,4,5-trichlorophenol at 1,000 µg/L resulted in 100% mortality (Shumway and Palensky, 1973).

Neville (1995) found a LOEC ranging from 34 µg/L to 125 µg/L and a NOEC of 62.5 µg/L of 2,4,5-trichlorophenol inhibited the growth of rainbow trout. Neville (1995) also reported a NOEC for physiological changes of 211 µg/L and a LOEC of 438 µg/L.

McKim et al. (1985) reported a NOEC of 4.6 µg/L 2,4,5-trichlorophenol for respiratory effects (ventilation rate, bentilation volume, oxygen uptake efficiency) in rainbow trout; however, this was the only concentration tested.

The lowest indirect toxicity concentration (LOEC) was 34 µg/L (Neville, 1995) and the lowest reported NOEC was 62.5 µg/L. The NOEC of 4.6 µg/L reported by McKim et al. (1985) was not used because it was the only concentration tested. Since the lowest LOEC is less than the lowest NOEC, a safety factor of 10 is applied to the lowest LOEC of 34 µg/L to obtain a NOEC of 3.4 µg/L. **This BE uses 3.4 µg/L 2,4,5-trichlorophenol as a direct benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.1.2 Indirect Effects

Non-salmonid fish have been found to have LC50s ranging from 450 µg/L to 50,000 µg/L 2,4,5-trichlorophenol and a maximum acceptable toxicant concentration (MACT) of 497 µg/L (Knott and Johnston, 1971a; Kobayashi et al., 1979; Buccafusco et al., 1981; Benoit-Guyod et al., 1984; Shigeoka et al., 1988; Norberg-King, 1989; Kishino and Kobayashi, 1995).

Mortality data using the invertebrate Daphnia magna resulted in LC50s or EC50s ranging from 780 µg/L to 3,800 µg/L 2,4,5-trichlorophenol (LeBlanc, 1980; LeBlanc et al., 1988, Spehar, 1986). Duckweed had an LC50 of 1,700 µg/L 2,4,5-trichlorophenol (Blackman et al., 1955).

In 7-day and 28-day renewal and flow-through laboratory tests, fathead minnows (*Pimephales promelas*) were found to have NOECs ranging 297 µg/L to 536 µg/L, MATCs ranging from 344 µg/L to 623 µg/L, and LOECs ranging from 398 µg/L to 725 µg/L for growth rate decreases (Norberg-King, 1989; Arthur and Dixon, 1994). Yoshioka et al. (1985) observed changes in growth rates in the ciliate *Tetrahymena pyriformis* at an EC50 of 680 µg/L 2,4,5-trichlorophenol.

The EC50s for invertebrates (*Daphnia magna*) ranged from 550 µg/L to 1,040 µg/L 2,4,5-trichlorophenol (Hattori et al., 1984; Steinberg et al., 1992), though Heinonen et al. (1997) observed unspecified behavioral changes in *Sphaerium corneum* at concentrations of 42 µg/L.

For prey species, the lowest reported NOEC was 297 µg/L (Norberg-King, 1989). **This BE uses 297 µg/L 2,4,5-TCP as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.2 2,4,6-Trichlorophenol

7.5.1.6.5.2.1 Direct Effects

The range of LC50s (the concentration that is expected to be lethal to 50% of the organisms tested) for the salmonids *Oncorhynchus mykiss* and *Salmo trutta* was 730 µg/L to 3,304 µg/L (Hattula et al. 1981; Holcombe et al., 1987; Kennedy, 1990).

The lowest direct toxicity concentration (LC50) for 2,4,6-trichlorophenol is 730 µg/L in brown trout (Huttula et al., 1981). Since neither a LOEC nor a NOEC were cited for direct toxicity, a NOEC was established by application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC) resulting in a value of 7.3 µg/L for this study. **This BE uses 7.3 µg/L 2,4,6-trichlorophenol as a direct toxicity benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.2.2 Indirect Effects

Non-salmonid fish have been found to have LC50s ranging from 180 to 70,000 µg/L and LOECs ranging from 1,335 to 1,760 µg/L, and an EC50 of 2,600 µg/L (Buccafusco et al., 1981; Phipps et al., 1981; Geiger et al., 1985, 1988).

Mortality tests using the invertebrates *Daphnia magna, Tanytarsus dissimilis, Aplexa hypnorum, Moina macrocopa, and Dugesia japonica* have resulted in LC50s ranging from 270 to 15,000 µg/L (LeBlanc, 1980; Dence et al., 1980; Yoshioka et al., 1986; Kukkonen and Oikari, 1987; Holcombe et al., 1987; Virtanen et al., 1989).

Smith et al. (1991) found a LOEC of 750 µg/L of 2,4,6-trichlorophenol inhibiting the growth of flagfish (*Jordanella floridae*). In a 7-day laboratory test, flatworms (*Dugesia japonica*) exposed to 2,4,6-trichlorophenol were found to have an EC50 of 850 µg/L for growth (Yoshioka et al., 1986).

Bitton et al. (1996) reported an EC_{50} for changes in feeding behavior in Ceriodaphnia dubia at a concentration of 4200 µg/L 2,4,6-trichlorophenol.

Clowes (1951) reported that 2,4,6-trichlorophenol affected oxygen consumption and cell division in fertilized sea urchin eggs (*Arbacia punctulata*). At a limiting concentration of 6.2 mg/L, a decreased rate of cell division was initiated in treated eggs; at a concentration of 39 mg/L, cell division ceased entirely.

Schultz and Riggin (1985) determined an acute IC50 (the concentration expected to cause a 50% inhibition of the biological process examined) of 3,990 µg/L 2,4,6-trichlorophenol in the ciliate *Tetrahymena pyriformis* under static conditions. In a lentic system in the field, Schauerte et al. (1982) found that, at a concentration of 5,000 µg/L 2,4,6-trichlorophenol, population abundances of invertebrates in general increased, though abundances of water flea (*Daphnia pulex*), plankton, and the diatom *Nitzschia acicularis* decreased.

For prey species, the lowest indirect toxicity concentration (LOEC) was 320 µg/L in bluegill (Buccafusco et al., 1981). Since a NOEC was not cited for indirect toxicity, a NOEC was established by application of a safety factor of 10 to obtain a NOEC of 32 µg/L for non-salmonid prey. **This BE uses 32 µg/L 2,4,6-trichlorophenol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.3 2,3,4,6-Tetrachlorophenol

7.5.1.6.5.3.1 Direct Effects

For salmonid Oncorhynchus mykiss, Kennedy (1990) reported a range of mean LC50s (the concentration that is expected to be lethal to 50% of the organisms tested) of 334 to 506 µg/L. For brown trout (*Salmo trutta*), Hattula et al. (1981) reported a mean LC50 of 500 µg/L 2,3,4,6-tetrachlorophenol.

The lowest direct toxicity concentration (LC50) for 2,3,4,6-tetrachlorophenol is 334 µg/L in rainbow trout (Kennedy, 1990). Since neither a LOEC nor a NOEC were cited for direct toxicity, a NOEC was established by application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC) would generate a value of 3.3 µg/L for this study. **This BE uses 3.3 µg/L 2,3,4,6-tetrachlorophenol as a direct toxicity benchmark for bull trout, Snake River**

sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.5.3.2 Indirect Effects

Non-salmonid fish have been found to have mean LC50s ranging from 140 to 10,000 µg/L 2,3,4,6- tetrachlorophenol (Buccafusco et al., 1981; Smith et al., 1991).

Mortality tests using the invertebrates (rotifer and Daphnia sp.) have resulted in mean LC50s ranging from 10 to 16,000 µg/L 2,3,4,6-tetrachlorophenol (LeBlanc, 1980; Virtanen et al., 1989; Oikari et al., 1992; and Liber and Solomon, 1994).

Smith et al. (1991) found a LOEC of 1,035 µg/L of 2,3,4,6-tetrachlorophenol inhibiting the growth of flagfish, *Jordanella floridae*.

Behavioral responses in the water flea, (*Daphnia* sp.) from acute (short term) exposure to 2,3,4,6-tetrachlorophenol ranged from 1,400 to 2,300 µg/L (Shigeoka et al., 1988).

Shigeoka et al. (1988) reported that maximum acceptable threshold concentrations (MATCs) in the water flea ranged from 650 to 1200 µg/L.

Liber et al. (1992) reported NOEC and EC50 values for a variety of zooplankton for population-level toxicity. These values were based upon 2-day or 7-day tests in a freshwater aquatic mesocosm system. In general, the NOEC and EC_{50} values were similar across the invertebrates tested. For the copepods, the NOECs ranged from 210 to 510 µg/L, and the EC_{50} values ranged from 270 to 590 µg/L 2,3,4,6-tetrachlorophenol (Liber et al., 1992). In the rotifers, the NOECs ranged from 110 to 200 µg/L, and the EC50 values ranged from 280 to 650 µg/L (Liber et al., 1992). For Daphnia, the EC50 values ranged from 500 to 750 µg/L (Liber et al., 1992).

For prey species, the lowest indirect toxicity concentration (NOEC) was 10 µg/L in Daphnia less than 24-hours old (LeBlanc, 1980). **This BE uses 10 µg/L 2,3,4,6-tetrachlorophenol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.4 Pentachlorophenol

7.5.1.6.5.4.1 Direct Effects

Eisler (1989) reviewed the effects of pentachlorophenol on invertebrates' growth, survival, and reproduction at levels of 3-100 µg/L, while fish are affected at concentrations from 1-68 µg/L. Chronic values for rainbow trout are 5.67-14.46 µg/L at pH values of 6.5-7.4. However, concentrations as low as 0.035-1 µg/L have been correlated with elevated tissue residues in rainbow trout. A 96-hour LC50 was determined for carp larvae at 9.5 µg/L at a pH of 7.2 (Eisler, 1989).

LC50s for Chinook salmon were reported to be 31 and 68 µg/L pentachlorophenol (Johnson and Finley, 1980 and Saarikoski and Viluksela, 1981). LC50 values for 96- hour exposures ranged from 18 to 3,000 µg/L for rainbow trout (Bentley et al., 1975; Johnson and Finley, 1980; Saarikoski and Viluksela, 1981; Hodson et al., 1984; Dominguez and Chapman, 1984; Van Leeuwen et al., 1985; Thurston et al., 1985; Douglas et al., 1986; Kennedy, 1990). The LC50 values for the remaining exposure intervals (4 to 72 hours) generally fell into this interval as well (Bentley et al., 1975; Slooff et al., 1983; Thurston et al., 1985; McKim et al., 1987; Kennedy, 1990). A NOEC for mortality to rainbow trout was reported as 11 µg/L (Dominquez and

Chapman, 1984). Another study (Van Leeuwen et al. 1985) determined the LC50 for early fry of rainbow trout to be 18 µg/L (pH 7.2). The 95% confidence interval ranged from 10 to 32 µg/L, indicating a severe mortality response below the acute and chronic criteria level.

Studies on fish found that they responded to concentrations from 1 to 68 µg/L. Chronic values for rainbow trout range from 5.7 to 14.5 µg/L at pH values of 6.5 to 7.4. Several other studies showing adverse effects on fish at pentachlorophenol concentrations in the low µg/L range are summarized in Eisler (2000).

Behavioral effects in rainbow trout were examined in one study at exposure concentrations that were from 10 to 100 times less than the CMC of 20 µg/L (Little et al. 1990). A statistically significant reduction in the percent survival by salmon that were preyed on by large mouth bass (*Micropterus salmoides*) occurred at an exposure concentration of 0.2 µg/L. Survival of salmon was from 32% to 55% in these predation studies compared to the control at 72%, representing reductions of 28% to 55% in treatment concentrations compared to the control. Statistically significant reductions were also observed in the number of Daphnia consumed and swimming activity when fish were exposed to a pentachlorophenol concentration of 2 µg/L. There was also a significant decrease in the strike frequency by salmon on Daphnia at 20 µg/L.

The exposures in Little et al. (1990) were conducted for 96 hours, static (not flow-through), and concentrations were based on nominal concentrations. Static conditions may underestimate the true exposure concentration because the fish will deplete the concentration in solution over time causing a lack of steady-state exposure. The authors also expressed some concern about contaminants in the formulation used (technical grade pentachlorophenol); however, no reliable data exists on the ability of the contaminants found in the technical grade pentachlorophenol to cause behavioral effects.

The Little et al. (1990) study used acetone as a carrier for pentachlorophenol exposure in treatments and controls, which is very common in such experiments. The concentration of acetone was 41 µg/L, which is considered very low. Acetone produces very low toxicity in salmonids (Majewski et al. 1978) and it is volatized or biodegraded in a matter of hours (Rathbun et al. 1982), implying that acetone was not likely a factor in the observed results.

One study found that juvenile Chinook salmon (*Oncorhynchus tshawytscha*) exposed to 3.9 µg/L exhibited altered blood urea and glucose (Iwama et al. 1986) and Nagler et al. (1986) found significant effects on oocyte impairment at 22 µg/L (pH 7.5).

The lowest direct toxicity concentration for pentachlorophenol is 0.2 µg/L for behavioral effects (Little et al. 1990) and mortality (adjusted) in rainbow trout (Van Leeuwen et al., 1985). Other studies considered for the toxicity benchmark included an NEC of 11 µg/L for mortality/growth changes in embryo and juvenile steelhead (Dominquez and Chapman, 1984). The lowest reported endpoint in AQUIRE was an LC50 of 18 µg/L for early life stage rainbow trout (Van Leeuwen et al., 1985). Since neither a LOEC nor a NOEC were cited for direct toxicity, a NOEC was established by application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC) would generate a value of 0.18 µg/L for this study. **This BE uses 0.18 µg/L pentachlorophenol as a direct toxicity benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.4.2 Indirect Effects

LC50s for the fathead minnow were 95 to 8,000 µg/L based on an 8-day exposure period (Phipps et al., 1981); Adema and Vink (1981) found LC50s ranging from 40 to 1,442 µg/L pentachlorophenol for the guppy (*Poecilia reticulata*) exposed for 7 days; and the lowest LC50 for largemouth bass was 54 µg/L for a 120- day exposure period (Johansen et al., 1985). A 96-hour LC50 was determined for carp larvae at 9.5 µg/L at a pH of 7.2 (Eisler, 1989).

A recent study (Dwyer et al. 2000) found that Atlantic sturgeon (Acipenseridae) were generally the most sensitive species when compared to 15 other fish species from 6 families, including 4 species of the Salmonidae family. This conclusion was based on acute exposure and LC50 values for 5 different compounds (copper, carbaryl, nonylphenol, pentachlorophenol, and permethrin). The 96-hour LC50 for Atlantic sturgeon was less than 40 µg/L.

Data for invertebrates were available for four genera of amphipods (*Gammarus, Crangonyx, Hyalella, and Pontoporeia*), seven genera of snails (*Aplexa, Biomphalaria, Gillia, Helisoma, Physa, Lymnaea, and Viviparus*), two species of rotifers (*Brachionus calyciflorus* and *B. rubens*), and three genera of water fleas (*Daphnia, Simocephalus, and Ceriodaphnia*). LC50 values for amphipods ranged from 92 to 3,120 µg/L pentachlorophenol, for exposures ranging from 24 hours to 30 days (Call et al., 1983; Slooff, 1983; Spehar et al., 1985; Ewell et al., 1986; Hedtke et al., 1986; Graney and Giesy, 1986, 1987; Landrum and Dupuis, 1990; OPP, 1995). Static, flow-through and renewal systems were used. The range of LC50 values were generally similar across the species. LC50 values for the rotifer Brachionus calyciflorus ranged from 1,410 to 16,000 µg/L (Ferrando et al., 1992; Crisinel et al., 1994; Liber and Solomon, 1994), while a lower value (160 µg/L) was calculated for the related species B. rubens (Halbach et al., 1983). The pond snail *Lymnaea acuminata* was the most sensitive species reported in AQUIRE, having LC50 values ranging from 0.16 to 0.293 µg/L across different exposure periods (Gupta and Rao, 1982). For the three water flea genera, LC50 values for 96-hour exposures ranged from 320 to 800 µg/L (Adema and Vink, 1981; Ewell et al., 1986).

Eisler (1989) reviewed the effects of pentachlorophenol on invertebrate growth, survival, and reproduction and reported adverse effects in the range of 3 to 100 µg/L.

Reproductive toxicity data for invertebrates were available for two species of snail (*Lymnaea stagnalis* and *Physa gyrina*) and three genera of water fleas (*Daphnia, Simocephalus, and Ceriodaphnia*). The Maximum Acceptable Toxicant Concentration (MATC) for Ceriodaphnia was 80 µg/L pentachlorophenol, based on either a 4- or 7-day exposure period (Masters et al., 1991). The EC50 for a 16-day exposure in Daphnia was 130 µg/L (Hermens et al., 1984). Unspecified reproductive changes were reported in invertebrates after exposure to concentrations ranging from 4.1 to 340 µg/L pentachlorophenol for periods up to 3 weeks (Adema and Vink, 1981; Slooff and Canton, 1983; Hedtke et al., 1986). The lowest value was reported for the water flea Ceriodaphnia reticulata; the closely related species C. dubia showed an unspecified reproductive effect at 161 µg/L pentachlorophenol (Hedtke et al., 1986).

For prey species, the lowest indirect toxicity concentration for pentachlorophenol is 0.16 µg/L, which was reported as a LC50 in pond snails (Gupta and Rao, 1982). Using a factor of 10 to convert from an effect concentration to a no effect level, the toxicity benchmark is 0.02 µg/L for non-salmonid prey. **This BE uses 0.02 µg/L pentachlorophenol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.5 3,4,5-Trichlorocatechol

7.5.1.6.5.5.1 Direct Effects

No data on the direct effects of 3,4,5-trichlorocatechol to listed species or other salmonids were found in the literature.

Since there are no direct toxicity data available for this parameter, the direct toxicity benchmark for 2,4,5-trichlorophenol was used as a surrogate for 3,4,5-trichlorocatechol. This benchmark was chosen because it had the lowest direct effect of compounds with similar chemical structure and properties. **This BE uses 2.6 µg/L 3,4,5-trichlorocatechol as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.5.2 Indirect Effects

Dence et al. (1980) measured the waterborne concentration of 3,4,5-trichlorocatechol required to cause mortality in the water flea (Daphnia magna). The LC50 (the concentration that is expected to be lethal to 50% of the organisms tested) for Daphnia magna was 339 µg/L (Dence et al., 1980). The life stage of daphnia was not reported.

Since neither a LOEC nor a NOEC were cited for indirect toxicity, a NOEC was established using a factor of 10 to convert from an effect concentration to a no effect level. **This BE uses 34 µg/L 3,4,5-trichlorocatechol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.6 3,4,6-trichlorocatechol

7.5.1.6.5.6.1 Direct Effects

No data on the direct effects of 3,4,6-trichlorocatechol to listed species or other salmonids were found in the literature.

Since there are no direct toxicity data available for this parameter, the direct toxicity benchmark for 2,4,5-trichlorophenol was used as a surrogate for 3,4,6-trichlorocatechol. This benchmark was chosen because it had the lowest direct effect of compounds with similar chemical structure and properties. **This BE uses 2.6 µg/L 3,4,6-trichlorocatechol as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.6.2 Indirect Effects

Kuivasniemi et al. (1985) examined the physiological toxicity of 3,4,6-trichlorocatechol on algae in freshwater systems. Algae were exposed to 0.01 mM of 3,4,6-trichlorocatechol (about 2,135 µg/L) over 20 hours, although whether this concentration was a LOEC, NOEC, or some other type of measurement was not reported. In an acute, 96-hour static laboratory test, Kuivasniemi et al. (1985) reported an EC50 of 0.00092 mM (about 196 µg/L) based on population-level effects in green algae (*Selenastrum capricornutum*).

Since neither a LOEC nor a NOEC were cited for indirect toxicity, a NOEC was established using a factor of 10 to convert from an effect concentration to a no effect level. **This BE uses 19.6 µg/L 3,4,6-trichlorocatechol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.7 Tetrachlorocatechol

7.5.1.6.5.7.1 Direct Effects

A mean LC50 (the concentration that is expected to be lethal to 50% of the organisms tested) of 1,100 µg/L was reported for the salmonid species, *Salmo trutta* (brown trout) (Hattula et al., 1981).

Since neither a LOEC nor a NOEC were cited for direct toxicity, a NOEC was established by application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC) to generate a value of 11 µg/L for this study. **This BE uses 11 µg/L tetrachlorocatechol as a direct toxicity benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.8 Indirect Effects

Geiger et al. (1985) reported a mean LC50 of 1,270 µg/L tetrachlorocatechol for the fathead minnow (Pimephales promelas). Mortality tests using Daphnia magna have resulted in a mean LC50 of 2,230 µg/L tetrachlorocatechol (Dence et al., 1980).

For prey species, the lowest indirect toxicity concentration for tetrachlorocatechol is 1,270 µg/L, which is based on the LC50 value reported by Geiger et al. (1985). Using a factor of 10 to convert from an effect concentration to a no effect level, the toxicity benchmark is 127 µg/L for non-salmonid prey. **This BE uses 127 µg/L tetrachlorocatechol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.9 3,4,5-Trichloroguaiacol

7.5.1.6.5.9.1 Direct Effects

The mean LC50 (the concentration that is expected to be lethal to 50% of the organisms tested) for salmonids Oncorhynchus mykiss was 750 µg/L 3,4,5-trichloroguaiacol (Leach and Thakore, 1975).

Since neither a LOEC nor a NOEC were cited for direct toxicity, a NOEC was established by application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC) to generate a value of 7.5 µg/L for this study. **This BE uses 7.5 µg/L 3,4,5-trichloroguaiacol as a direct toxicity benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.9.2 Indirect Effects

Mortality tests done using Daphnia magna have resulted in mean LC50s ranging from 450 to 730 µg/L 3,4,5-trichloroguaiacol (Virtanen et al., 1989; Oikari et al., 1992).

Since neither a LOEC nor a NOEC were cited for indirect toxicity, a NOEC was established using a factor of 10 to convert from an effect concentration to a no effect level. **This BE uses 45 µg/L 3,4,5-trichloroguaiacol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.10 3,4,6-trichloroguaiacol

7.5.1.6.5.10.1 Direct Effects

No data on the direct effects of 3,4,6-trichloroguaiacol to listed species or other salmonids were found in the literature.

Since there are no direct toxicity data available for this parameter, the direct toxicity benchmark for 2,4,5-trichlorophenol was used as a surrogate for 3,4,6-trichloroguaiacol. This benchmark was chosen because it had the lowest direct effect of compounds with similar chemical structure and properties. **This BE uses 2.6 µg/L 3,4,5-trichloroguaiacol as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.10.2 Indirect Effects

No data on the indirect effects of 3,4,6-trichloroguaiacol to listed species or other salmonids were found in the literature.

Since there are no indirect toxicity data available for this parameter, the indirect toxicity benchmark for 2,4,5-trichlorophenol was used as a surrogate for 3,4,6-trichloroguaiacol. This benchmark was chosen because it had the lowest direct effect of compounds with similar chemical structure and properties. **This BE uses 3.4 µg/L 3,4,6-trichloroguaiacol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.11 4,5,6-trichloroguaiacol

7.5.1.6.5.11.1 Direct Effects

No data on the direct effects of listed species or other salmonids exposed to 4,5,6-trichloroguaiacol were found in the literature.

Since there are no direct toxicity data available for this parameter, the direct toxicity benchmark for 2,4,5-trichlorophenol was used as a surrogate for 4,5,6-trichloroguaiacol. This benchmark was chosen because it had the lowest direct effect of compounds with similar chemical structure and properties. **This BE uses 2.6 µg/L 4,5,6-trichloroguaiacol as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.11.2 Indirect Effects

Mortality tests using the invertebrates have resulted in LC50s of 580 to 22,000 µg/L (*Daphnia magna*), 1,800 µg/L (*Ceriodaphnia dubia*), and 50 µg/L (*Hydropsyche siltalai*) 4,5,6-trichloroguaiacol (Dence et al., 1980; Kukkonen and Oikari, 1987; Petersen and Petersen, 1988; Neilson et al., 1990).

Neilson et al. (1990) derived an EC50 of 5,900 µg/L 4,5,6-trichloroguaiacol in a 48-hour laboratory test for the copepod *Nitocra spinipes*.

For prey species, the lowest indirect toxicity concentration for 4,5,6-trichloroguaiacol is 50 µg/L, which is based on the LC50 value reported by Peterson and Peterson (1988). Using a factor of 10 to convert from an effect concentration to a no effect level, the toxicity benchmark is 5 µg/L

for non-salmonid prey. **This BE uses 5 µg/L tetrachlorocatechol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.12 Tetrachloroguaiacol

7.5.1.6.5.12.1 Direct Effects

LC50s (the concentration that is expected to be lethal to 50% of the organisms tested) for salmonid Oncorhynchus mykiss ranged from 320 to 370 µg/L tetrachloroguaiacol (Leach and Thakore, 1975, 1977; Johansen et al., 1994). Johansen et al. (1994) reported a 96-hour LC50 of 370 µg/L. Johansen et al. (1994) also reported a statistically significant increase in mortality in fish in the presence of a pathogenic bacteria when exposed to 200 µg/L of tetrachloroguaiacol for 25 days.

Johansen et al. (1994) reported physiological changes in juvenile Oncorhynchus mykiss (rainbow trout) exposed to tetrachloroguaiacol for 25 days. At exposure concentrations of 20 µg/L, an increase in the percentage leucocrit count was observed in addition to a large decrease in cortisol levels in plasma (2.8 times lower). These same parameters were also affected at each higher exposure concentration (100 and 200 µg/L). The LOEC for these responses is 20 µg/L.

Yang and Randall (1996) examined osmoregulation in Oncorhynchus kisutch (Coho/silver salmon). Only one dose was tested in this experiment (mean water concentration of 100 µg/L). Statistically significant changes were observed in plasma sodium, muscle moisture content, and gill ATPase indicating effects on the fish to osmoregulate normally. Because only one dose was examined an LOEC or NOEC could not be determined. It is not known if lower exposure concentrations would also cause these biological responses.

Johansen at al. (1994) examined swimming behavior in Oncorhynchus mykiss (rainbow trout) exposed to tetrachloroguaiacol. A statistically significant reduction in critical swimming speed was observed at 100 µg/L after a 24-hour exposure. A reduction was also found at 200 µg/L, but not 300 or 400 µg/L. The LOEC value for this response is 100 µg/L.

The lowest direct toxicity concentration for tetrachloroguaiacol is 100 µg/L, which is based on the LOEC value reported by Johansen (1994). The LOEC of 20 µg/L was not used to establish the direct toxicity benchmark because the physiology change due to increased leucocrit and decreased cortisol has not been established as an adverse effect. A NOEC was established by application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC) to generate a value of 10 µg/L for this study. **This BE uses 10 µg/L tetrachloroguaiacol as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.12.2 Indirect Effects

High mortality (90%) was observed in embryos of fathead minnows (Pimephales promelas) at 200 µg/L tetrachloroguaiacol (Woodland and Maly, 1997). The survival of embryos in the next higher dose of 100 µg/L was 90%, indicating that the LC50 for this compound was between 100 µg/L and 200 µg/L.

Static 48-hour mortality tests using Daphnia magna have resulted in mean LC50s ranging from 140 to 370 µg/L (Virtanen et al., 1989; Oikari et al., 1992). A static 24-hour mortality test using Daphnia magna resulted in a mean LC50 of 4,960 µg/L tetrachloroguaiacol (range of 4,190 to

6,210 µg/L) (Dence et al., 1980). Oikari (1987) examined mortality in *Alburnus alburnus* (bleak) and reported a 96-hour LC50 of 110 µg/L tetrachloroguaiacol.

Woodland and Maly (1997) reported a significant reduction in the proportion of fathead minnows (Pimephales promelas) embryos that hatched at 100 µg/L of tetrachloroguaiacol.

For prey species, the lowest indirect toxicity concentration for tetrachloroguaiacol is 100 µg/L, which is based on several LC50 values. Using a factor of 10 to convert from an effect concentration to a no effect level, the toxicity benchmark is 10 µg/L for non-salmonid prey. **This BE uses 10 µg/L tetrachlorocatechol as an indirect toxicity benchmark for prey species.**

7.5.1.6.5.13 Trichlorosyringol

7.5.1.6.5.13.1 Direct Effects

No data on the direct effects of listed species or other salmonids exposed to trichlorosyringol were found in the literature.

Since there are no direct toxicity data available for this parameter, the direct toxicity benchmark for 2,4,5-trichlorophenol was used as a surrogate for trichlorosyringol. This benchmark was chosen because it had the lowest direct effect of compounds with similar chemical structure and properties. **This BE uses 2.6 µg/L trichlorosyringol as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.5.13.2 Indirect Effects

In an acute, 96-hour static laboratory test, Kuivasniemi et al. (1985) reported an EC50 of 0.0033 mM (~850 µg/L) based on population-level effects in green algae (*Selenastrum capricornutum*).

Since neither a LOEC nor a NOEC were cited for indirect toxicity, a NOEC was established using a factor of 10 to convert from an effect concentration to a no effect level. **This BE uses 85 µg/L trichlorosyringol as an indirect toxicity benchmark for prey species.**

7.5.1.6.6 Effects Analysis

7.5.1.6.6.1 2,4,5-Trichlorophenol

Trichlorophenols may be present in the aquatic environment either in dissolved form, associated with suspended matter or bottom sediments, or absorbed by organisms. Metal salts of these compounds have greater water solubility, and if introduced, they would exist primarily in the dissolved form. The tendency of chlorophenols to ionize depends on the pH of the system. They are nonionized in aqueous solutions with pH lower than 5 and become increasingly dissociated as the pH rises. The degree of dissociation could determine the extent of sorption of trichlorophenols by colloids in aquatic systems; however, specific information is not available. Hydrological factors such as current patterns and mixing as well as sorption, degradation, and migration of organisms affect the movement of these chemicals.

The half-life for 2,4,5-trichlorophenol is 13-hours; therefore, half the 2,4,5-trichlorophenol added to the system from the discharge would decrease by one-half within 6 miles (assuming a river

flow rate of 0.2 m/s), which is one-sixth the distance to the Lower Granite Dam. For a first order rate, the equation relating the rate to half-life is:

$$k = \frac{0.693}{t^{0.5}}.$$

This results in a decay rate (k) of 1.28 per day for 2,4,5-trichlorophenol.

7.5.1.6.6.1.1 Direct Effects

The maximum effluent concentration of 2,4,5-trichlorophenol (0.955 µg/L) is below the direct toxicity benchmark (3.4 µg/L); therefore, EPA concludes that the discharge of this compound **may directly affect but is not likely to adversely affect to bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.1.2 Indirect Effects

The maximum effluent concentration of 2,4,5-trichlorophenol (0.955 µg/L) is below the indirect toxicity benchmark (297 µg/L); therefore, EPA concludes that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 2,4,5-Trichloropenol in all samples collected. 2,4,5-Trichloropenol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

7.5.1.6.6.1.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA concludes that the discharge of 2,4,5-trichlorophenol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.2 2,4,6-Trichlorophenol

Trichlorophenols may be present in the aquatic environment either in dissolved form, associated with suspended matter or bottom sediments, or absorbed by organisms. Metal salts of these compounds have greater water solubility, and if introduced, they would exist primarily in the dissolved form. The tendency of chlorophenols to ionize depends on the pH of the system. They are nonionized in aqueous solutions with pH lower than 5 and become increasingly dissociated as the pH rises. The degree of dissociation could determine the extent of sorption of trichlorophenols by colloids in aquatic systems; however, specific information is not available. Hydrological factors such as current patterns and mixing as well as sorption, degradation, and migration of organisms affect the movement of these chemicals.

The half-life for 2,4,6-trichlorophenol is 14-hours; therefore, half the 2,4,6-trichlorophenol added to the system from the discharge would decrease by one-half within 6 miles (assuming a river flow rate of 0.2 m/s), which is one-sixth the distance to the Lower Granite Dam. For a first order rate, the equation relating the rate to half-life is:

$$k = \frac{0.693}{t^{0.5}}.$$

This results in a decay rate (k) of 1.19 per day for 2,4,6-trichlorophenol.

7.5.1.6.6.2.1 Direct Effects

The maximum effluent concentration of 2,4,6-trichlorophenol (0.955 µg/L) is below the direct toxicity benchmark (7.3 µg/L); therefore, EPA concludes that the discharge of 2,4,6-trichlorphenol **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.2.2 Indirect Effects

The maximum effluent concentration of 2,4,6-trichlorophenol (0.955 µg/L) is below the indirect toxicity benchmark (32 µg/L); therefore, EPA concludes that the discharge of 2,4,6-trichlorphenol **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 2,4,6-Trichloropenol in all samples collected. 2,4,6-Trichloropenol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

7.5.1.6.6.2.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the discharge of 2,4,6-trichlorphenol at the maximum effluent concentration is **not likely to adversely affect the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.3 2,3,4,6-Tetrachlorophenol

Tetrachlorophenols may be present in the aquatic environment either in dissolved form, associated with suspended matter or bottom sediments, or absorbed by organisms. Metal salts of these compounds have greater water solubility, and if introduced, they would exist primarily in the dissolved form. The tendency of chlorophenols to ionize depends on the pH of the system. They are nonionized in aqueous solutions with pH lower than 5 and become increasingly dissociated as the pH rises. The degree of dissociation could determine the extent of sorption of tetrachlorophenols by colloids in aquatic systems; however, specific information is not available. Hydrological factors such as current patterns and mixing as well as sorption, degradation, and migration of organisms affect the movement of these chemicals.

The half-life for 2,3,4,6-tetrachlorophenol is 18-hours; therefore, half the 2,3,4,6-tetrachlorophenol added to the system from the discharge would decrease by one-half within 8 miles (assuming a river flow rate of 0.2 m/s), which is one-fifth the distance to the Lower Granite Dam. For a first order rate, the equation relating the rate to half-life is:

$$k = \frac{0.693}{t^{0.5}}.$$

This results in a decay rate (k) of 0.924 per day for 2,3,4,6-tetrachlorophenol.

7.5.1.6.6.3.1 Direct Effects

The maximum effluent concentration of 2,3,4,6-tetrachlorophenol (1.91 µg/L) is below the direct toxicity benchmark (3.3 µg/L); therefore, EPA concludes that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.3.2 Indirect Effects

The maximum effluent concentration of 2,3,4,6-tetrachlorophenol (1.91 µg/L) is below the indirect toxicity benchmark (10 µg/L); therefore, EPA concludes that the discharge of 2,3,4,6-tetrachlorphenol **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 2,3,4,6-Tetrachlorophenol in all samples collected. 2,3,4,6-Tetrachlorophenol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

7.5.1.6.6.3.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the discharge of 2,3,4,6-tetrachlorphenol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.4 Pentachlorophenol

7.5.1.6.6.4.1 Direct Effects

In the aquatic environment, pentachlorophenol may be in dissolved form, associated with suspended matter or bottom sediments, or absorbed by organisms. Metal salts of the compound have much greater water solubility and therefore would exist primarily in the dissolved form. The tendency of pentachlorophenol to ionize depends on the pH of the system. It is nonionized in aqueous solutions with pH lower than 5 and becomes increasingly dissociated as the pH rises. The degree of dissociation could determine the extent of sorption of colloids present in aquatic systems; however, specific information is not available. Hydrological factors such as current patterns and mixing as well as sorption, degradation, and migration of organisms affect the movement of the chemical.

There is limited evidence of microbiological degradation of pentachlorophenol in aquatic environments. Photodecomposition and volatilization from water also occur. The half-life for pentachlorophenol is 11-hours; therefore, half the pentachlorophenol added to the system from the discharge would decrease by one-half within 5 miles (assuming a river flow rate of 0.2 m/s), which is one-seventh the distance to the Lower Granite Dam. For a first order rate, the equation relating the rate to half-life is:

$$k = \frac{0.693}{t^{0.5}}.$$

This results in a decay rate (k) of 1.51 per day for pentachlorophenol.

Since the maximum effluent concentration of pentachlorophenol allowed under this permit (1.91 µg/L) is greater than the direct water column toxicity benchmark (0.18 µg/L), this analysis looks at the effects within the exposure volume of the effluent (i.e., the area where the concentration of the plume exceeds the toxicity benchmark) and the effects at and beyond the exposure volume boundary.

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the chronic mixing zone would be 36.5, therefore the maximum exposure concentration at the edge of the chronic mixing zone would be the maximum daily limit of 1.91 µg/L divided by the available dilution of 36.5, or 0.052 µg/L. The calculated maximum exposure concentration of 0.052 µg/L is less than the toxicity benchmark of 0.18 µg/L.

At and beyond the exposure volume, the permit limits are designed to protect the water quality standard for pentachlorophenol (0.002 µg/L). Since the water quality standard is less than the toxicity benchmark (0.18 µg/L), it is not likely that threatened or endangered salmonids would be exposed to unsafe levels of pentachlorophenol.

Therefore, EPA has concluded that the discharge of pentachlorophenol at the maximum effluent concentration is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.6.4.2 Indirect Effects

As predicted by the model, the indirect toxicity benchmark will not be met within the jet action of the plume; therefore, it is likely that prey species would be exposed to unsafe levels of pentachlorophenol.

At and beyond the exposure volume, the permit limits are designed to protect the water quality standard for pentachlorophenol (0.28 µg/L). Since the water quality standard is almost 10 times the indirect toxicity benchmark (0.18 µg/L), prey species will not be exposed to unsafe levels of pentachlorophenol.

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for pentachlorophenol in all samples collected. Pentachlorophenol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

Therefore, EPA has concluded that the discharge of pentachlorophenol at the maximum effluent concentration is not likely to indirectly affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.6.4.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is greater than the established benchmarks, EPA concludes that the discharge of pentachlorophenol at the maximum effluent concentration **is likely to adversely modify the**

critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.6.5 3,4,5-Trichlorocatechol

7.5.1.6.6.5.1 Direct Effects

The maximum effluent concentration of 3,4,5-trichlorocatechol (1.91 µg/L) is below the direct toxicity benchmark (2.6 µg/L); therefore, EPA has concluded that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.5.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 3,4,5-Trichloroguaiacol in all samples collected. 3,4,5-Trichloroguaiacol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of 3,4,5-trichlorocatechol (1.91 µg/L) is below the indirect toxicity benchmark (34 µg/L); therefore, EPA has concluded that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.5.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA concludes that the discharge of 3,4,5-trichlorocatechol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.6 3,4,6-Trichlorocatechol

7.5.1.6.6.6.1 Direct Effects

The maximum effluent concentration of 3,4,6-trichlorocatechol (1.91 µg/L) is below the direct toxicity benchmark (2.6 µg/L); therefore, EPA has concluded that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.6.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 3,4,6-Trichlorocatechol in all samples collected. 3,4,6-Trichlorocatechol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of 3,4,6-trichlorocatechol (1.91 µg/L) is below the indirect toxicity benchmark (19.6 µg/L); therefore, EPA has concluded that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River**

sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.6.6.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA concludes that the discharge of 3,4,6-trichlorocatechol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.7 Tetrachlorocatechol

7.5.1.6.6.7.1 Direct Effects

The maximum effluent concentration of tetrachlorocatechol (1.91 µg/L) is below the direct toxicity benchmark (11 µg/L); therefore, EPA has concluded that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.7.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for tetrachlorocatechol in all samples collected. Tetrachlorocatechol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of tetrachlorocatechol (1.91 µg/L) is below the indirect toxicity benchmark (127 µg/L); therefore, EPA has concluded that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.7.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA concludes that the discharge of tetrachlorocatechol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.8 3,4,5-trichloroguaiacol

7.5.1.6.6.8.1 Direct Effects

The maximum effluent concentration of 3,4,5-trichloroguaiacol (0.955 µg/L) is below the direct toxicity benchmark (7.5 µg/L); therefore, EPA concludes that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.8.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 3,4,5-trichloroguaiacol in all samples collected. 3,4,5-Trichloroguaiacol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of 3,4,5-trichloroguaiacol (0.955 µg/L) is below the indirect toxicity benchmark (45 µg/L); therefore, EPA concludes that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.8.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the discharge of 3,4,5-trichloroguaiacol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.9 3,4,6-trichloroguaiacol

7.5.1.6.6.9.1 Direct Effects

The maximum effluent concentration of 3,4,6-trichloroguaiacol (0.955 µg/L) is below the direct toxicity benchmark (2.6 µg/L); therefore, EPA concludes that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.9.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 3,4,6-trichloroguaiacol in all samples collected. 3,4,6-Trichloroguaiacol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of 3,4,6-trichloroguaiacol (0.955 µg/L) is below the indirect toxicity benchmark (3.4 µg/L); therefore, EPA concludes that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.10 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the discharge of 3,4,6-trichloroguaiacol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.11 4,5,6-trichloroguaiacol

7.5.1.6.6.11.1 Direct Effects

The maximum effluent concentration of 4,5,6-trichloroguaiacol (0.955 µg/L) is below the direct toxicity benchmark (2.6 µg/L); therefore, EPA concludes that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.11.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for 4,5,6-trichloroguaiacol in all samples collected. 4,5,6-Trichloroguaiacol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of 4,5,6-trichloroguaiacol (0.955 µg/L) is below the indirect toxicity benchmark (5 µg/L); therefore, EPA concludes that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.11.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the discharge of 4,5,6-trichloroguaiacol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.12 Tetrachloroguaiacol

7.5.1.6.6.12.1 Direct Effects

The maximum effluent concentration of tetrachloroguaiacol (1.91 µg/L) is below the direct toxicity benchmark (10 µg/L); therefore, EPA concludes that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.12.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for tetrachloroguaiacol in all samples collected. Tetrachloroguaiacol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of tetrachloroguaiacol (1.91 µg/L) is below the indirect toxicity benchmark (10 µg/L); therefore, EPA concludes that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.12.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the discharge of tetrachloroguaiacol at the maximum effluent concentration **is not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.6.13 Trichlorosyringol

7.5.1.6.6.13.1 Direct Effects

The maximum effluent concentration of trichlorosyringol (0.955 µg/L) is below the direct toxicity benchmark (2.6 µg/L); therefore, EPA has concluded that the discharge of this compound **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.13.2 Indirect Effects

A 2007 fish tissue study conducted by Anchor Environmental determined a consistent "Non-detect" for trichlorosyringol in all samples collected. Trichlorosyringol is not likely to bioaccumulate in salmonid or other fish species (Anchor, 2008).

The maximum effluent concentration of trichlorosyringol (0.955 µg/L) is below the indirect toxicity benchmark (85 µg/L); therefore, EPA has concluded that the discharge of this compound **may indirectly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.1.6.6.13.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the discharge of trichlorosyringol at the maximum effluent concentration is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.7 2,3,7,8-TCDD and 2,3,7,8-TCDF

7.5.1.6.7.1 Introduction

Polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs) commonly occur as complex mixtures in the environment. They are persistent bioaccumulative contaminants that are found ubiquitously in environmental matrices, including tissues of fish, birds and mammals. The most well studied chemical in this group of compounds being 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD).

PCDDs and PCDFs are chlorinated tricyclic aromatic compounds which are released into the environment as a result of the production of paper products from chlorine bleached wood pulp, chemical manufacturing processes, metal smelting, municipal and industrial waste incineration plants, burning of coal, wood or oil for home heating and production of electricity, domestic

fires, motor vehicle exhausts (gasoline combustion), and disposal of municipal sewage treatment plant sludge. 2,3,7,8-TCDD is found as a contaminant in the forestry herbicide 2,4,5-trichlorophenoxy propionic acid (Silvex) and the industrial chemical 2,4,5-trichlorophenol. Unwanted trace amounts of some of the higher-chlorinated dioxins, especially the hexa and octa isomers, have also been associated with the wood preservative, pentachlorophenol.

Industrial processes do not produce the PCDDs and PCDFs intentionally. Rather, most PCDDs and PCDFs are generated in very small amounts as unwanted impurities during the manufacture of several chlorinated chemicals and consumer products, including certain wood treatment chemicals, some metals, and paper products. When the wastewater, sludge, or solids from these processes are released into waterways or soil, the sites may become contaminated with PCDDs and PCDFs. The various processes that create PCDDs and PCDFs are either being slowly phased out or are strictly controlled. It is currently believed that PCDD and PCDF emissions associated with incineration and combustion activities are the predominant environmental source of these contaminants (USEPA, 2002).

PCDDs consist of two benzene rings connected by two oxygen bridges. PCDFs also consist of two benzene rings connected by two bridges, but only one of the two bridges is an oxygen atom. There are eight positions where substitution of hydrogen atoms by other atoms can occur. In addition, toxicity of PCDD/Fs tends to decrease with chlorination. The 17 most toxic congeners (Table 7-13) all include chlorine substitutions on the 2,3,7, and 8 positions, with 2,3,7,8-TCDD being the most toxic (Fletcher and McKay, 1993; Feeley, 1995) of all possible PCDD/F congeners.

According to information in the Hazardous Substances Data Base (HSDB, 2000), the volatilization half-life for 2,3,7,8-TCDD in surface water ranges from 46 days to 50 years. Information in Howard et al. (1991) indicates that the half-life of 2,3,7,8-TCDD in surface water ranges from 1.15 years to 1.65 years (or 10,074 hours to 14,191 hours). The half-life of dioxins in anaerobic soils is estimated to be 10 to 12 years; in sediments it may be decades or centuries (WDOE, 1992).

Table 7-13: Chlorinated Dioxin and Furan Compounds Analyzed

7 Chlorinated Dioxins Analyzed	10 chlorinated Furans Analyzed
2,3,7,8-TCDD	2,3,7,8-TCDF
1,2,3,7,8-PeCDD	1,2,3,7,8-PeCDF
1,2,3,4,7,8-HxCDD	2,3,4,7,8-PeCDF
1,2,3,6,7,8-HxCDD	1,2,3,4,7,8-HxCDF
1,2,3,7,8,9-HxCDD	1,2,3,6,7,8-HxCDF
1,2,3,4,6,7,8-HpCDD	1,2,3,7,8,9-HxCDF
OCDD	2,3,4,6,7,8-HxCDF
	1,2,3,4,6,7,8-HpCDF
	1,2,3,4,7,8,9-HpCDF
	OCDF

The high hydrophobicity and lipophilicity of PCDDs accounts for their very low solubility in water, though they do adsorb to organic material in the particulate and dissolved phases. 2,3,7,8-TCDF is often found in fish tissue because of its affinity for lipids and because of its formation as a by-product in the industrial processes, especially pulp and paper mills (USEPA, 2002). Also, PCDD/Fs are immobile once they become incorporated into sediment, where they

are highly resistant to environmental and biological degradation and persist for decades (reviewed in Fletcher and McKay 1993; Clark et al., 1996).

Demonstrated toxic effects of 2,3,7,8-TCDD in fish, birds, and mammals include immunotoxicity; adverse effects on reproduction, development and endocrine functions; wasting syndrome; and mortality. Several PCDDs, PCDFs, and PCBs have been shown to cause toxic responses similar to 2,3,7,8-TCDD, in both laboratory and field situations. For further information regarding effects observed specifically in wildlife species, refer to USEPA (1993a, 2001) and references therein. Presently, evidence is sufficient to conclude that a common mechanism of action, involving binding of the chemicals to the aryl hydrocarbon receptor (AhR) as the initial step, underlies 2,3,7,8-TCDD-like toxicity elicited by these PCDDs, PCDFs, and PCBs (Van den Berg et al., 1998; Hahn, 1998). PCDDs, PCDFs, and PCBs present in the environment are generally found as complex mixtures such that assessment of ecological risk requires a means of quantifying their cumulative effects.

7.5.1.6.7.1.1 Bioaccumulation

Bioaccumulation is the net accumulation of a substance by an organism due to uptake from all environmental sources. PCDDs, PCDFs, and PCBs are nonpolar compounds that cannot be easily excreted unless they are first transformed into polar compounds with the introduction of a polar functional group through metabolism. These compounds do not biomagnify via the diet within invertebrate food chains and are not metabolized at a significant rate by invertebrates. Therefore, invertebrate tissues tend to be at equilibrium with water and sediments (Thomann, 1989; Gobas, 1993). PCDD, PCDF and PCB concentrations in contaminated sediments often exceed values expected for equilibrium conditions with surface waters. Thus, organisms whose food chain are linked to contaminated sediments through benthic invertebrates will have greater exposures than those with food chains linked to surface water through pelagic invertebrates.

Unlike invertebrates, vertebrates metabolize PCDDs and PCDFs. PCDDs and PCDFs that do not possess chlorines at all four 2, 3, 7, and 8 positions do not bioaccumulate in vertebrates. Although metabolism of PCDDs and PCDFs with chlorine substitution at the 2,3,7, and 8 positions (the most toxic congeners) occurs to a lesser extent than those without, it is sufficient to significantly reduce bioaccumulation with the same degree of chlorination (Endicott and Cook, 1994).

The most important chemical property that controls bioavailability from water, sediment, or soils is hydrophobicity, which can be measured by the octanol-water partition coefficient, K_{ow}. PCDDs and PCDFs for which dioxin-like toxicity is established have log K_{ow}s, increasing with degree of chlorination, from approximately 6 to 9. This high degree of hydrophobicity makes measurement of concentrations in water very difficult, especially for PCDDs and PCDFs. Conversely, concentrations in surficial sediments or soils are often measurable and can be used effectively to reference each chemical's distribution to abiotic and biotic components of the ecosystem. In aquatic ecosystems, concentrations measured in surficial sediments can be used to estimate average concentrations in water.

Properties such as bioavailability, bioaccumulation, metabolism and biomagnification also differ among PCDDs and PCDFs such that the relative concentration of the individual chemicals vary with species and trophic level. Therefore, concentrations of individual PCDDs and PCDFs, and in abiotic media often do not reflect the chemical concentration profile observed in the tissues of wildlife. TEFs and Relative Potency Factors should only be applied based on the specific

chemical mixtures in the exposures of the organisms for which risks are being assessed. Thus, it is imperative that chemical concentrations in abiotic media be converted to concentrations in either the tissues of organisms being assessed or their food through use of appropriate bioaccumulation factors prior to applying TEFs for calculating TECs. TECs should generally not be directly based on water, sediment, or soil since these media are inconsistent with the dosimetry basis for the toxicity equivalence model.

One method for estimating bioaccumulation is by using bioconcentration factors (BCFs), but BCFs have poor applicability to PCDDs and PCDFs. BCFs, which are measured under laboratory conditions, involve uptake of the chemical by aquatic organisms only from water through respiration (i.e., through gills). Thus, for very hydrophobic chemicals, BCFs tend to underestimate bioaccumulation, which is the net uptake and retention of a chemical through all routes of exposure, uptake and elimination. Complicating factors for PCDDs and PCDFs in aquatic food chains are metabolism rates that may be sufficient to greatly reduce the impact of dietary exposure. Laboratory studies to estimate BCFs were not conducted, therefore, other exposure methodology was used.

Bioaccumulation factors (BAFs) and biota-sediment accumulation factors (BSAFs) are obtained from direct measurements or prediction of uptake and elimination of the chemical through all routes of exposure. A bioaccumulation factor is the ratio of the concentration of a substance in tissue of an organism to its concentration in the ambient exposure media (e.g., water or soil) in situations where both the organism and its food are exposed, and the ratio does not change substantially over time. For aquatic organisms, the factor is the ratio of the concentration of chemical in the organism to its concentration in water, expressed in L/kg. For terrestrial organisms, the factor is the ratio of the chemical concentration in the organism to its concentration in soil.

Typically, BAFs and BSAFs are determined and applied for conditions that approximate steady-state of the organism with respect to water and sediments, respectively. Thus, BAFs and BSAFs are the appropriate quantitative expressions for the relationships between concentrations of PCDDs, PCDFs, and PCBs in the environment (water, sediment, soil) and concentrations in an organism's tissues. For a visualization and sensitivity analysis of the critical determinants of site-specific BAF and BSAF values, see Burkhard et al. (2003).

Because physical, chemical, and biological properties vary among the individual PCDDs and PCDFs, bioaccumulation factors must also be congener-specific and species-specific. Bioaccumulation factors (BAFs and BSAFs) are the essential connectors of concentrations of PCDDs, PCDFs, and PCBs in the environment with concentrations in the diet or relevant tissues of organisms of concern, which are then used to calculate TECs. Bioaccumulation factors can be incorporated within a time dependent multi-media mass balance simulation model, as has been applied to 2,3,7,8-TCDD (Gobas et al., 1998). Bioaccumulation factors also have been used explicitly to define water quality standards, as in the *Great Lakes Water Quality Initiative* (USEPA, 1995a) and the *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA, 2000).

7.5.1.6.7.1.2 Biota to Sediment Accumulation Factors

As summarized in the *Environmental Baseline* section, 2005 permit requirements included high-volume effluent sampling and sampling resident fish tissue for constituents of bioaccumulation concern including dioxins and furans. Caged bivalve tissue and sediment were also sampled for

the concentrations of bioaccumulative constituents including dioxins and furans. Table 7-14, Table 7-15, Table 7-16, and Table 7-17 summarize the measured concentrations of dioxins and furans in each media.

BSAFs were developed from organic-carbon normalized concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF measured in the sediment and lipid-normalized concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF measured in both whole body and fillet with skin on of largescale sucker and smallmouth bass. The calculated BSAFs indicated that 2,3,7,8-TCDD and 2,3,7,8-TCDF are not bioaccumulating at high rates and tend to be approximately the same throughout the LGR Table 7-18).

Table 7-14: Summary of dioxins/furans measured in the high-volume sampling of effluent from the Clearwater Mill.

Chemical	Detection Frequency	Percent Qualified as Estimated	Result Average	Max Detect Result	Min Detect Result	Max MDL Limit	Min MDL Limit	Detected Standard Deviation
Dioxin/Furans (pg/L)								
1,2,3,4,6,7,8-HpCDD	78%	36%	0.104	0.798	0.00853			0.171
1,2,3,4,6,7,8-HpCDF	56%	53%	0.0210	0.182	0.00779			0.0409
1,2,3,4,7,8,9-HpCDF	3%	3%	0.00672	0.0668	0.0668			
1,2,3,4,7,8-HxCDD	6%	6%	0.00639	0.0148	0.00689			0.00559
1,2,3,4,7,8-HxCDF	6%	6%	0.00548	0.0374	0.00895			0.0201
1,2,3,6,7,8-HxCDD	11%	11%	0.00872	0.0377	0.0132			0.0112
1,2,3,6,7,8-HxCDF	3%	3%	0.00539	0.0497	0.0497			
1,2,3,7,8,9-HxCDD	8%	8%	0.00773	0.0287	0.00700			0.0109
1,2,3,7,8,9-HxCDF	6%	6%	0.00303	0.0249	0.00723			0.0125
1,2,3,7,8-PeCDD	3%	3%	0.00609	0.00699	0.00699			
1,2,3,7,8-PeCDF	6%	6%	0.00521	0.0113	0.00848			0.00199
2,3,4,6,7,8-HxCDF	6%	3%	0.00568	0.0725	0.00973			0.0444
2,3,4,7,8-PeCDF	8%	8%	0.00556	0.0232	0.0130			0.00523
2,3,7,8-TCDD	6%	6%	0.00300	0.0148	0.00498			0.00694
2,3,7,8-TCDF	39%	33%	0.00747	0.108	0.00172			0.0289
OCDD	100%	50%	0.679	5.47	0.0223			1.03
OCDF	53%	53%	0.0744	0.330	0.0162			0.0812

Table 7-15: Summary of caged-bivalve tissue concentrations of dioxins and furans at reference locations in the Snake and Clearwater Rivers as well as multiple points in the LGR from the Clearwater Mill diffuser to the Lower Granite Dam.

Chemical	Detection Frequency	Percent Qualified as Estimated	Result Average	Max Detect Result	Min Detect Result	Max MDL Limit	Min MDL Limit	Detected Standard Deviation
Dioxin/Furans (ng/kg)								
1,2,3,4,6,7,8-HpCDD	88%	94%	0.325	0.430	0.245			0.0515
1,2,3,4,6,7,8-HpCDF	0%	0%	0.145					
1,2,3,4,7,8,9-HpCDF	0%	0%	0.158					
1,2,3,4,7,8-HxCDD	31%	31%	0.191	0.222	0.179			0.0165
1,2,3,4,7,8-HxCDF	0%	0%	0.0566					

1,2,3,6,7,8-HxCDD	0%	0%	0.169					
1,2,3,6,7,8-HxCDF	0%	0%	0.0524					
1,2,3,7,8,9-HxCDD	0%	0%	0.168					
1,2,3,7,8,9-HxCDF	0%	0%	0.0698					
1,2,3,7,8-PeCDD	0%	0%	0.103					
1,2,3,7,8-PeCDF	0%	0%	0.113					
2,3,4,6,7,8-HxCDF	0%	0%	0.0586					
2,3,4,7,8-PeCDF	0%	0%	0.100					
2,3,7,8-TCDD	0%	0%	0.0562					
2,3,7,8-TCDF	100%	69%	0.294	0.458	0.155			0.0939
OCDD	100%	100%	1.69	2.74	1.24			0.414
OCDF	0%	0%	0.308					

Table 7-16: Summary of dioxins and furans measured in sediments at reference locations in the Snake and Clearwater Rivers, as well as multiple points in the LGR from the Clearwater Mill diffuser to the Lower Granite Dam.

					Diffuser →					Downstream
Analyte	**Units**	**SR-REF**	**CR-REF**	**CR-REF/Note**	**LGP-13**	**LGP-14**	**LGP-11**	**LGP-09**	**LGP-06**	**LGP-01**
Total Organic Carbon	%	2.8	0.905		2.22	2.245	3.925	4.285	2.68	1.488
2,3,7,8-TCDD	ng/kg	ND	ND		ND	ND	ND	ND	ND	ND
2,3,7,8-TCDF	ng/kg	0.142	0.035	U	0.127	0.334	0.613	0.201	0.158	0.247
2,3,7,8-TCDF-OC	ng/kg-OC	0.051	0.039		0.057	0.149	0.156	0.047	0.059	0.166

Table 7-17: Summary of dioxins and furans measured in resident fish tissue in the Snake and Clearwater Rivers as well as multiple points in the LGR from the Clearwater Mill diffuser to the Lower Granite Dam.

Chemical	Detection Frequency	Percent Qualified as Estimated	Result Average	Max Detect Result	Min Detect Result	Max MDL Limit	Min MDL Limit	Detected Standard Deviation
Dioxin/Furans (ng/kg)								
1,2,3,4,6,7,8-HpCDD	25%	25%	0.149	0.599	0.0556			0.121
1,2,3,4,6,7,8-HpCDF	5%	5%	0.0608	0.396	0.126			0.108
1,2,3,4,7,8,9-HpCDF	0%	0%	0.0757					
1,2,3,4,7,8-HxCDD	4%	4%	0.0980	0.344	0.243			0.0534
1,2,3,4,7,8-HxCDF	4%	0%	0.240	6.03	5.39			0.277
1,2,3,6,7,8-HxCDD	8%	8%	0.172	2.16	0.0767			1.04
1,2,3,6,7,8-HxCDF	4%	4%	0.0826	1.52	1.45			0.0320
1,2,3,7,8,9-HxCDD	3%	3%	0.103	0.289	0.232			0.0300
1,2,3,7,8,9-HxCDF	0%	0%	0.0409					
1,2,3,7,8-PeCDD	11%	8%	0.212	3.88	0.0589			1.80
1,2,3,7,8-PeCDF	4%	0%	0.234	4.98	4.11			0.359
2,3,4,6,7,8-HxCDF	4%	4%	0.0621	0.984	0.825			0.0723
2,3,4,7,8-PeCDF	12%	8%	0.573	15.0	0.0477			6.54
2,3,7,8-TCDD	8%	4%	0.667	17.4	0.0343			8.95
2,3,7,8-TCDF	92%	34%	1.16	25.1	0.0398			4.71
OCDD	28%	27%	0.399	4.77	0.130			0.825
OCDF	2%	2%	0.227	0.650	0.253			0.281

Table 7-18:Lipid Normalized BSAFs for 2,3,7,8-TCDD and 2,3,7,8-TCDF for two species in the Snake and Clearwater River Reference locations and multiple locations in the LGR from the Clearwater Mill diffuser to the Lower Granite Dam.

CDD/CDF	Species (Type)	CR-REF	SR-REF	LGP-01	LGP-02	LGP-03	LGP-04	LGP-05	LGP-06
2,3,7,8-TCDD	Largescale Sucker (WB)	0.189	0.557	0.213	0.491	0.504	1.517	0.364	0.331
	Largescale Sucker (FS)	0.293	0.874	0.888	0.926	1.166	4.176	0.996	0.708
	Smallmouth Bass (WB)	0.181	0.731	0.740	0.980	0.608	1.917	0.390	0.242
	Smallmouth Bass (FS)	0.480	2.159	2.524	4.610	3.366	15.368	1.177	1.845
2,3,7,8-TCDF	Largescale Sucker (WB)	0.887	2.158	1.361	1.756	1.211	3.769	0.654	1.585
	Largescale Sucker (FS)	1.334	2.219	0.706	1.561	0.697	2.442	0.708	2.012
	Smallmouth Bass (WB)	0.620	1.688	0.444	0.863	0.810	2.096	0.627	1.481
	Smallmouth Bass (FS)	0.902	1.825	0.425	1.005	0.682	1.748	0.300	1.260

WB – whole body; FS – Fillet with skin on.

7.5.1.6.7.1.3 Toxicity Equivalence Concentrations of Chlorinated Dioxins and Furans

Chlorinated dioxins and furans are found in the environment together with other structurally related chlorinated chemicals, such as some of the various dioxin-like PCB congeners. Therefore, people and other organisms are generally exposed to mixtures of these structurally similar compounds, rather than to a single chlorinated dioxin or furan, or dioxin-like PCB congener.

To estimate risks for exposure to dioxin-like chemicals or chemicals which have a similar mechanism of action through the aryl hydrocarbon receptor a method was developed to estimate a toxicity equivalence concentration (Van den Berg et al., 1998). In this methodology, the toxicity equivalence factor (TEF) for 2,3,7,8-TCDD is equal to 1; all other dioxin, furan, and dioxin-like PCB congeners are calculated as some relative percent of 1. When the TEF is multiplied by the congener concentration, a toxicity equivalence concentration (TEC) is obtained. The sum of the toxicity equivalence concentration for each congener is the toxicity equivalence quotient (TEQ). The toxicity equivalence factors (Table 7-19) were derived by a panel of experts using careful scientific judgment after considering all available relative potency data (Van den Berg et al., 1998).

Table 7-19: Toxicity Equivalence Factors (TEF) for dioxins, and furans (from Van den Berg et al., 1998).

Congener	TEF		
	Mammals	Birds	Fish
2,3,7,8-TCDD	**1**	**1**	**1**
1,2,3,7,8-PeCDD	1	1	1
1,2,3,4,7,8-HxCDD	0.1	0.05	0.5
1,2,3,6,7,8-HxCDD	0.1	0.01	0.01
1,2,3,7,8,9-HxCDD	0.1	0.1	0.01
1,2,3,4,6,7,8-HpCDD	0.01	<0.001	0.001

OCDD	0.0001	0.0001	<0.0001
2,3,7,8-TCDF	**0.1**	**1**	**0.05**
1,2,3,7,8-PeCDF	0.05	0.1	0.05
2,3,4,7,8-PeCDF	0.5	1	0.5
1,2,3,4,7,8-HxCDF	0.1	0.1	0.1
1,2,3,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,7,8,9-HxCDF	0.1	0.1	0.1
2,3,4,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,4,6,7,8-HpCDF	0.01	0.01	0.01
1,2,3,4,7,8,9-HpCDF	0.01	0.01	0.01
OCDF	0.0001	0.0001	<0.0001

The toxicity equivalence concentration for each congener is the product of the toxicity equivalence factor multiplied by the concentration for an individual dioxin-like congener as shown in the following equation:

$$TEC_i = TEF_i \times [\text{congener fish tissue concentration}]_i$$

where,

TEF = Toxicity equivalence factor
TEC = toxicity equivalence concentration.

The total toxicity equivalence concentration is derived by summing the individual TEC of the congeners. The summed concentration is referred to as "2,3,7,8-TCDD TEQ" or as "TEQ."

7.5.1.6.7.2 Environmental Baseline

7.5.1.6.7.2.1 NPDES Permit Surface Water Sampling

As part of the Clearwater facilities required quarterly monitoring requirements (2005 and 2006), surface water samples were monitored for 2,3,7,8-TCDD, 2,3,7,8-TCDF, as well as various dioxin and furan congeners (Appendix C). In general, the results indicate that the Clearwater facility should have no influence on downstream parameter measurements and no detrimental effect on listed species. No meaningful differences between reference conditions and downstream conditions were observed. With just a few exceptions, all measurements complied with EPA toxicity benchmarks and state water quality standards. Those measurements that did not meet benchmarks occurred at reference locations and locations farthest downstream of the Clearwater facility. Required NPDES surface water sampling from 2007 indicated similar results as those found in 2005 and 2006. Water column benchmarks exist for only two of the dioxin/furan congeners, but all measurements fell below these benchmarks at all reference and sample sites.

7.5.1.6.7.2.2 Groundwater

Another requirement of Clearwater's NPDES permit is quarterly monitoring of groundwater in the vicinity of the facility. Monitoring in 2005 and 2006 from eight stations resulted in non-

detects for dioxins and furans for all sampling events. These results indicate that groundwater is not contributing dioxins/furans to the surrounding area.

7.5.1.6.7.2.3 NPDES Permit Sediment Sampling

Sediment sampling in 2005 in accordance with the renewal of Clearwater's NPDES permit indicated no concentrations exceeding their benchmarks in either single replicate or four-sample arithmetic averages (Table 7-16). Most of the analytes that were detected at sample stations downstream of the Clearwater diffuser were also detected at the reference stations on both the Clearwater and Snake Rivers. Sediment chemistry concentrations tended to be lower at the reference stations as compared to the downstream sample stations. This sampling effort included a reconnaissance survey to identify areas within the Clearwater and Snake rivers, as well as the Lower Granite Dam, that have a sufficient amount of fine-grained sediment to have the potential to be affected by or accumulate organics. A total of 20 locations where bottom sediments consisted of fine-grained sediments were identified in the Snake River. One reference location, in Swallow's Nest Park at River Mile (RM) 142 on the west side of the Snake River, was also selected. Numerous attempts to locate fine-grained sediments farther upstream and in the Snake River channel near the shoreline yielded only sand.

7.5.1.6.7.2.4 NPDES Permit Resident Fish Tissue Testing

In accordance with Clearwater's NPDES permit, resident fish were sampled within the vicinity and downstream of Outfall 001 from July 10 to July 19, 2007. Resident fish tissue analysis is required to support the effort to characterize any potential effects of Outfall 001 on endangered and listed species, and the overall environment in general. Fish sampling occurred from one sampling area each in the Clearwater and Snake Rivers upstream of their confluence, and six sampling locations downstream of the confluence in the pool of the Lower Granite Dam.

Results of testing indicate that concentrations of dioxins/furans were below their respective benchmark criteria for all samples (Table 7-17). Most of the analytes that were detected in fish from sample stations downstream of the Clearwater diffuser were also detected at the reference stations on both the Clearwater and Snake Rivers, although tissue concentrations tended to be lower at the reference stations as compared to the downstream sample stations.

7.5.1.6.7.2.5 NPDES Permit Bivalve Tissue Testing

In accordance with Clearwater's NPDES permit, caged bivalve tissue monitoring studies were conducted within the vicinity and downstream of Outfall 001 from April to May 2007. Caged bivalve tests are required to support the characterization of the potential effects of discharges from Outfall 001 on endangered and listed species, as well as to the overall environment in general. Sampling occurred at two upstream reference locations and five locations distributed between Outfall 001and the Lower Granite Dam.

Results of testing indicated that most of the dioxins/furans found in bivalve tissue from sample stations downstream of Outfall 001 were also found from bivalve tissue from reference stations on both the Clearwater and Snake Rivers (Table 7-15). Data analysis indicated a high level of similarity in type and concentration of dioxin/furan found between reference and sample stations. No toxicity benchmarks for bivalves were included in the NPDES permit for Outfall 001, therefore no comparison was made.

7.5.1.6.7.3 Water Quality Standard

The most stringent water quality standard in Idaho and Washington for dioxin is 0.013 picograms per liter (pg/l) as a long-term average, for the protection of human health. This concentration was used as the basis for the 1991 Columbia River Total Maximum Daily Load (USEPA, 1991). In the TMDL, Clearwater was given a wasteload allocation of 0.39 mg/day as an annual average.

The current Idaho and Washington water quality standards do not have 2,3,7,8-TCDF numeric criteria, however they both have narrative criteria to limit toxic material concentrations to levels below those which have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic conditions to the most sensitive biota dependent upon those waters, or adversely affect public health.

7.5.1.6.7.4 Effluent Limitation

The 2019 draft permit requires a maximum daily fiber line limit of 2,3,7,8-TCDD be below the minimum level of 10 pg/L. The maximum equivalent 2,3,7,8-TCDD concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (31.6 mgd) flow ratio of 0.382 would be **3.84 pg/L**. Additionally, the 2019 draft permit proposes an average monthly limit of 0.091 mg/day 2,3,7,8-TCDD and a maximum daily effluent limit of 0.132 mg/day 2,3,7,8-TCDD that are equivalent to 2,3,7,8-TCDD concentrations in the effluent of **0.63 pg/L** and **0.92 pg/L**, respectively.

The 2005 permit included a maximum daily fiber line limit of 2,3,7,8-TCDD be below the minimum quantitation level of 10 pg/L. The maximum equivalent 2,3,7,8-TCDD concentration in the effluent due to the fiberline limitation and a fiberline (13.1 mgd) to effluent (40 mgd) flow ratio of 0.365 would be **3.65 pg/L**. Additionally, the final permit included an average annual limit of 0.39 mg/day 2,3,7,8-TCDD and a maximum daily effluent limit of 1.1 mg/day 2,3,7,8-TCDD that is equivalent to a 2,3,7,8-TCDD concentration in the effluent of **2.58 pg/L** and **7.26 pg/L**, respectively.

The 2019 draft permit proposes a maximum daily fiber line limit of 31.9 pg/L 2,3,7,8-TCDF, which is equivalent to a maximum effluent concentration of **12.2 pg/L**. This is the same as the 2,3,7,8-TCDF limitation in the 2005 permit.

7.5.1.6.7.5 Toxicity Benchmarks

Extensive data on the toxicity of 2,3,7,8-TCDD (and some of the other PCDD/F congeners) to fish are available and have been summarized by USEPA (1993a). With few exceptions, these find that early life stages are more sensitive to the effects of PCCD/Fs than later life stages (USEPA, 1993a). These studies have examined many species of fish using a variety of exposure methods (e.g., waterborne, interperitoneal injection, egg injection and dietary exposure) (USEPA, 1993a). Most studies have been conducted in a laboratory setting but some have been conducted in more natural settings (USEPA, 1993a).

The discussion below focuses on studies with salmonids, since they are the fish species of concern in the BE, and on waterborne and dietary exposures, since these represent more relevant pathways of exposure for fish downstream of the diffuser. Note as well that the discussion in the BE is not intended to present a comprehensive review of PCDD/F toxicity to fish, rather it

supplements the information contained in USEPA (1993a) by summarizing the results of a few key studies that were not included in USEPA (1993a).

Of the various fish species tested, salmonids have been found to be the most sensitive to dioxins and related compounds (Walker et al., 1990; Walker and Peterson 1991; USEPA 1993a; Guiney et al., 1996; Elonen et. al., 1998). Presented below are tissue, water-borne, and sediment exposure concentrations of TCDD that have been reported to cause adverse effects in salmonids.

7.5.1.6.7.5.1 Tissue benchmark

Salmonid eggs have been demonstrated to be relatively sensitive to 2,3,7,8-TCDD. For example, Helder (1981) exposed rainbow trout eggs for 96 hours to 1 ng TCDD/L, which elicited a reduction in survival in the resulting yolk sac fry, and Spitsbergen et al. (1991) demonstrated that lake trout eggs, when exposed to ~10 ng/L for 48 hours, accumulated 40 ng TCDD/kg and underwent significantly increased mortality at hatching or at the sac fry stage. Increased mortality at the sac fry and swim-up stage following exposure of salmonid eggs to TCDD has also been reported at higher water or tissue concentrations. For example, in lake trout (Walker et al., 1991), brook trout (*Salvelinus fontinalis*) (Walker and Peterson, 1994; Johnson et al., 1998), rainbow trout (Eisler, 1986), or lake herring (*Coregonus artedii*) (Elonen et al., 1998), lethal effects have been seen at water concentrations of 8 to 31 ng/L or whole body tissue concentrations of 55 to 270 ng/kg, after exposure times of 20 minutes to 48 hours.

In a study by Giesy et al. (2002), adult female rainbow trout that were exposed via the diet to 2,3,7,8-TCDD and produced eggs that suffered from decreased survival and contained 0.11 ng TCDD/kg egg. However, in addition to containing 2,3,7,8-TCDD added to the diet by the investigators, the diet also contained background 2,3,7,8-TCDD TEQ activity that was more than 10-fold higher (at the lowest dose) than the added 2,3,7,8-TCDD TEQ. Because the source of this background TEQ activity was neither identified nor measured in fish tissue or eggs, the tissue and organ 2,3,7,8-TCDD concentrations cannot be used as benchmarks. A detailed discussion of the Giesy et al. (2002) study is presented in Appendix F.

Sublethal effects have also been reported following exposure of eggs to 2,3,7,8-TCDD. Johnson et al. (1988), exposed adult brook trout to 2,3,7,8-TCDD via the diet, and noted that the spawned eggs, which had tissue concentrations of 41 ng TCDD/kg egg, produced embryos with increased edema and exophthalmia (protrusion of the eyeball). Alternatively, Helder (1981) directly exposed rainbow trout eggs to 0.1 ng TCDD/L for 96 hours, wherein the resulting fry exhibited significant growth retardation for 72 days.

Mehrle et al. (1988) exposed rainbow trout swim-up fry for 28 days to water with a range of 2,3,7,8-TCDD concentrations followed by a 28-day depuration period. They report 0.038 ng/l as a LOAEL and 0.0011 ng/l as a NOAEL with associated egg concentrations of 765 ng/kg and 21 ng/kg 2,3,7,8-TCDD. Other studies (summarized in USEPA 1993a) have exposed early life stages of salmonids to higher waterborne concentrations and reported adverse effects and also no adverse effects (e.g., Walker et al. (1991) report a NOAEL in lake trout eggs of 34 ng/kg 2,3,7,8-TCDD).

Miller et al. (1973) exposed juvenile Coho salmon (*Oncorhynchus kisutch*) for twenty-four hours to 0.056 ng TCDD/L in a water-born exposure study, which resulted in a tissue concentration of 54 ng TCDD/kg whole body wet weight and reduced survival over sixty days. Additional studies on juvenile salmonids report adverse effects at somewhat higher water or tissue

concentrations. For example, increased mortality was observed within ten days in juvenile rainbow trout exposed for ninety-six hours to 10 ng TCDD/L (Helder, 1981), and studies which employed diet or injection as a route of exposure reported delayed mortality at whole body tissue concentrations of 5–1,380 µg TCDD/kg (Hawkes and Norris, 1977; Kleeman et al., 1988; van der Weiden et al., 1990).

Sublethal effects in juvenile salmonids include decreases in food consumption and weight gain by juvenile Coho salmon when exposed for 24-48 hours to 5.6-10.53 ng/L 2,3,7,8-TCDD, followed by a twenty or fifty-six-day post-exposure period (Miller et al., 1979). The reported whole-body concentration after one hundred fourteen days was ~478 ng TCDD/kg. Additional studies on rainbow trout (Helder, 1981; van der Weiden et al., 1990) or Coho salmon (Mehrle et al., 1988), based on water-born exposure or injection, reported growth retardation, slight edema, and/or congestion, lymphoid atrophy, and histopathological changes in the spleen due to short term (1-4 days) exposure to 10 or 56 ng/L TCDD, and whole body concentrations of 13,300 or 500,000 ng/kg.

Few studies have exposed adult rainbow trout to 2,3,7,8-TCDD either through food or water though several studies have examined adults using interperitoneal injection (USEPA, 1993a). Based upon a review of these data, and other studies on earlier life stages described above, U.S. EPA concluded that 50 ng 2,3,7,8-TCDD/kg body weight represents a concentration in fish associated with low risk to sensitive fish.

Sublethal effects in adult salmonids exposed to 2,3,7,8-TCDD include a study with Brook trout in which a body burden of 1,200 ng TCDD/kg, resulting from dietary exposure for one hundred eighty-two days, was associated with minor behavioral effects and a delay in onset of spawning by thirteen days (Tietge et al., 1998).

The interim report on risks to aquatic life and associated wildlife (USEPA, 1993a) from exposure to 2,3,7,8 TCDD recommends a level of 50 ng/kg in fish eggs associated with low risk for lake trout. This translates to about 90 ng/kg maternal tissue for lake trout. The level of 80 ng/kg in fish eggs is associated with high risk to lake trout and that translates to 140-200 ng/kg maternal tissue for lake trout. Bull Trout are more sensitive than lake trout thus an interspecies factor of 10x is applied to the 90 ng/kg to reach 9 ng/kg maternal tissue for Bull Trout. Although data are not available regarding the comparative sensitivity of lake trout and other salmonids evaluated in the BE, other salmonids were assumed to have a sensitivity equal to bull trout (that is, more sensitive than lake trout).

The benchmark for salmonid toxicity in fish tissue is 9 ng 2,3,7,8-TCDD/ kg bodyweight (measured as the toxicity equivalence concentration) for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.7.5.2 Waterborne and sediment benchmarks

The waterborne and sediment benchmark for salmonids in this BE are based on the USEPA Ambient Water Quality Criteria Derivation Methodology Human Health Technical Support Document (EPA/822/B-98/005, July 1998).

Concentrations in biota, sediments, and water are defined to accommodate variability in bioavailability conditions and express bioaccumulation on a thermodynamic basis (degree of equilibrium between biota, water, and sediments). The concentration of the chemical in the

organism's tissues (C_t) is normalized to lipid content (C_P) with the fraction lipid (f_P) in the organism's tissues. The concentration of the bioavailable chemical in water is defined as the concentration of freely dissolved chemical (C^{fd}) which is calculated with the fraction of chemical that is freely dissolved (f^{fd}) as estimated from concentrations of particulate organic carbon (POC) and dissolved organic carbon (DOC) in the water (USEPA, 1995a and 2000):

$$BAF_{\ell}^{fd} = \frac{C_{\ell}}{C_{\ell}^{fd}} = \frac{C_t \cdot {}^{1}\!/\!_{f_{\ell}}}{C_w^t \cdot f^{fd}}$$

Because of the extreme difficulty in measuring site-specific BAFs, a recommended method for deriving BAFs is to use site-specific BSAFs and the relationship between BSAFs and BAFs (EPA/822/B-98/005 - Section 2.4.4.2):

"...the BSAF method appears to work well not only for predicting BAFs using data from the same system (Lake Ontario) but also for predicting BAFs between systems (Green Bay vs. Lake Ontario). These evaluations support the use of the BSAF method for predicting BAFs."

The relationship between BAFs and BSAFs is characterized by (Π) which is the ratio of the BAF (L/kg freely dissolved, lipid basis) to the BSAF ([kg-contaminant sediment/fraction OC]/[kg-contaminant fish/fraction lipid]). Π/ K_{ow} is a measure of the disequilibrium in the system between sediment and water. This section of the BE uses this approach to estimate water and sediment benchmarks that should be met where fish are exposed following initial dilution. Transport and dilution modeling is not part of this evaluation.

Steps to estimating the waterborne and sediment concentrations protective of sensitive fish species in this BE are: (1) Calculate site-specific BSAFs; (2) Use the relationship between Π and K_{ow} (e.g., $\Pi/ K_{ow} = 5$) to estimate Π; (3) Use the relationship between Π, BAF, and BSAF (Π = BAF/BSAF) to estimate BAF; (4) Use BSAF and BAF to estimate protective sediment and water concentrations; (5) Describe the sensitivity of this approach to BSAFs, Π/ K_{ow}, and the ratio of TCDF/TCDD. The sensitivity analysis uses the approach used by Burkhard et al. (2003).

(1) Select calculated site-specific BSAFs:

From Table 7-18 the calculated BSAFs for the largescale sucker and smallmouth bass were evaluated and a default median TCDF BSAF of 1.1 was selected for both TCDD and TCDF. The TCDD estimates were considered to be too influenced by detection limit issues (all sediment concentrations were non-detects). Although TCDD BSAFs are likely to be higher than TCDF BSAFs, this BE set them equal. The sensitivity analysis presented below shows that water and sediment benchmarks are sensitive to BSAFs, correctly emphasizing the prime need for good site-specific measurements of BSAFs.

Default BSAF: 1.1 (kg-contaminant sed/fraction OC)/(kg-contaminant fish/fraction lipid)
Range: 0.3 – 3.8 (Table 7-18)

(2) Use the relationship between Π and K_{ow} (octanol-water partition coefficient of the chemical) to estimate Π.

Π/ K_{ow} is approximately 5 for the Fox River (Burkhard et al., 2003) representing a system with continual water column inputs and vertical mixing. This was selected as a reasonable default for

the receiving water. To evaluate sensitivity to Π/ K_{ow}, a value of 20 was also evaluated – this represents a system where the sediments are acting more as a source to the water column and organisms than in the Π/ K_{ow} =5 system. The sensitivity analysis shows that increasing Π/ K_{ow} lowers water benchmark but not the sediment benchmark.

K_{owTCDD} = 10,471,285 for TCDD (Log K_{ow} = 7.02)
K_{owTCDF} = 3,162,277 for TCDF (Log K_{ow} = 6.5)

Default Π/Kow: 5; so $\Pi=5(K_{ow})$
Range: 5 - 20

(3) Use the relationship between Π, BAF, and BSAF to estimate BAF; Since Π = BAF/BSAF, then BAF= Π * BSAF. These are calculated from the values selected in (1) and (2) above.

(4) Use BSAF and BAF to estimate protective sediment and water concentrations; The following equation calculates fish tissue TEQ concentration from TCDD and TCDF (subscripts D and F refer to TCDD and TCDF respectively).

$$TEQ_{\ell ipid}^{fish-tissue} = (Water_{D\,fd})(BAF_D)(TEF_D) + (Water_{F\,fd})(BAF_F)(TEF_F)$$

The ratio of freely dissolved TCDF to TCDD in the effluent can be used to convert the Water TCDD to TCDF as follows:

$$Water_{F\,fd} = (Water_{D\,fd})(Effluent\frac{(TCDF)}{(TCDD)})\ (fraction_{fd}TCDF/\ fraction_{fd}TCDD), \text{ where:}$$

$fraction_{fd}$ = 1/(1 + [DOC] x Kow x 0.08 + [POC] x Kow)

For DOC = 0.0000029 kg/L and POC= 0.0000005 kg/L; national averages, and similar to Snake River (Jack Harrison HDR Engineering, ID, personal communication)

$fraction_{fd}$ = 0.1154 for TCDD
$fraction_{fd}$ = 0.3017 for TCDF

And so:

$(fraction_{fd}TCDF/\ fraction_{fd}TCDD)$ = 2.614

Substituting, yields:

$$TEQ_{\ell ipid}^{fish-tissue} = (Water_{D\,fd})(BAF_D)(TEF_D) + (Water_{D\,fd})(Effluent\frac{(TCDF)}{(TCDD)})(BAF_F)(TEF_F)\ (2.614)$$

Solving for $(Water_{D\,fd})$:

$$\text{Water}_{\text{TCDD}\,fd} = \frac{\text{TEQ}^{fish-tissue-protective}_{\ell ipid}}{(\text{BAF}_{TCDD})(\text{TEF}_{TCDD}) + (\text{BAF}_{TCDF})(\text{TEF}_{TCDF})(\text{Effluent}\,\frac{(TCDF)}{(TCDD)})(2.614)}$$

$$\text{Water}_{\text{TCDF}\,fd} = (\text{Water}_{D\,fd})(\text{Effluent}\,\frac{(TCDF)}{(TCDD)})\ (2.614) \qquad \text{[from above]}$$

Note that converting back from freely dissolved to totals is done by dividing by the fraction that is freely dissolved (i.e., divide the $TCDD_{fd}$ concentration by 0.1154 and the $TCDF_{fd}$ concentration by 0.3017).

Sediment concentrations are calculated from fish tissue concentrations divided by BSAF. Fish tissue concentrations are calculated from water benchmark multiplied by BAF. Sediment concentrations are also the same as the water benchmark multiplied by Π:

$$\text{Sediment}_{TCDD_{oc}} = \frac{(\text{Water}_{TCDD\,fd})(\text{BAF}_{TCDD})}{(\text{BSAF}_{TCDD})} = (\text{Water}_{TCDD\,fd})(\prod), \text{ and}$$

$$\text{Sediment}_{TCDF_{oc}} = \frac{(\text{Water}_{TCDF\,fd})(\text{BAF}_{TCDF})}{(\text{BSAF}_{TCDF})} = (\text{Water}_{TCDF\,fd})(\prod)$$

BAFs are from (2), above.
TEFs are fixed (TCDD=1; TCDF=0.05)
TEQ is fixed = 9 ng TEQ/kg fish / 0.07 lipid = 129 ng TEQ/kg lipid

Default Effluent $\frac{(TCDF)}{(TCDD)}$ *(2.614)*: 8.31 = ratio on freely dissolved basis

Range Effluent $\frac{(TCDF)}{(TCDD)}$ *(2.614)*: 3-8.31; to evaluate greater volatilization of TCDF

The sensitivity analysis shows that if the TCDF/TCDD ratio declines there is a corresponding very slight increase in the TCDD benchmark, but about three-fold decrease in TCDF benchmark.

The fish tissue toxicity benchmark for salmonids was applied as a TEQ (TEQ = 9 ng/kg). Using a Salmonid lipid of 0.07 (personal communication with Philip Cook), a Π/Kow=5, and BSAFs=1.108, and a ratio of TCDF/TCDD=3.2, the calculated water column concentration to achieve a TEQ of 9 ng/kg (=129 ng/kg lipid normalized) is 0.0020 pg/L freely dissolved for TCDD and 0.0165 pg/L freely dissolved for TCDF. These are converted back to totals by dividing by the fraction freely dissolved calculated previously (0.1154 for TCDD, and 0.3017 for TCDF).

The calculated water concentration to achieve a TEQ of 9 ng/kg (=129 ng/kg lipid normalized) is 0.017 pg/L total for TCDD and 0.055 pg/L total for TCDF.

The calculated sediment concentration to achieve a TEQ of 9 ng/kg (=129 ng/kg lipid normalized) is 103.4 ng/kg-oc for TCDD and 261.1 ng/kg-oc for TCDF.

The waterborne benchmark for all life stages (i.e., egg to adult) of bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook

salmon, and Snake River steelhead is 0.0020 pg/L for 2,3,7,8 TCDD and 0.0166 pg/L for 2,3,7,8 TCDF.

The sediment benchmark for all life stages (i.e., egg to adult) of bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead is 114.5 ng/kg-oc for 2,3,7,8 TCDD and 262.5 ng/kg-oc for 2,3,7,8 TCDF.

7.5.1.6.7.6 Effects Analysis

7.5.1.6.7.6.1 2,3,7,8-TCDD

Since the maximum effluent concentration for 2,3,7,8-TCDD allowed under this permit (0.92 pg/L) is greater than the water column toxicity benchmark (0.0020 pg/L), this analysis looks at the effects within the exposure volume of the effluent (i.e., the area where the concentration of the plume exceeds the toxicity benchmark) and the effects at and beyond the exposure volume boundary.

7.5.1.6.7.6.1.1 Water column exposure

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the chronic mixing zone would be 36.5. Therefore, the maximum exposure concentrations at the edge of the chronic mixing zone would be 0.026 pg/L from July – September and 0.041 pg/L. The calculated maximum exposure concentration of 0.063 pg/L is more than the toxicity benchmark of 0.0020 pg/L. At and beyond the exposure volume, the permit limits are designed to protect the water quality standard for 2,3,7,8-TCDD (0.013 pg/L). Since the water quality standard is higher than the toxicity benchmark (0.002 pg/L), it is likely that threatened or endangered salmonids would be exposed to unsafe levels of 2,3,7,8-TCDD outside the exposure volume.

Therefore, EPA has concluded that the discharge of 2,3,7,8-TCDD at the maximum effluent concentration **is likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.7.6.1.2 Sediment exposure

The environmental baseline in the Action Area shows that the sediment concentrations of 2,3,7,8-TCDD are below 1 ng/kg (Table 7-16). Since this is significantly lower than the sediment toxicity benchmark (103.4 ng/kg)**, EPA concludes that the discharge of 2,3,7,8-TCDD at the maximum effluent concentration may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.7.6.1.3 Habitat Effects

Since the concentration that listed species, or their prey would be exposed to is greater than the established benchmarks, EPA concludes that the discharge 2,3,7,8-TCDD at the maximum effluent concentration is **likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.7.6.2 2,3,7,8-TCDF

Since the maximum effluent concentration for 2,3,7,8-TCDF allowed under this permit (12.2 pg/L) is greater than the water column toxicity benchmark (0.0166 pg/L), this analysis looks at the effects within the exposure volume of the effluent (i.e., the area where the concentration of the plume exceeds the toxicity benchmark) and the effects at and beyond the exposure volume boundary.

7.5.1.6.7.6.2.1 Water column exposure

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the chronic mixing zone would be 36.5, therefore the maximum exposure concentration at the edge of the chronic mixing zone would be the maximum daily limit of 12.2 pg/L divided by the available dilution of 36.5, or 0.33 pg/L. The calculated maximum exposure concentration of 0.33 pg/L is more than the toxicity benchmark of 0.0165 pg/L.

Therefore, EPA has concluded that the discharge of 2,3,7,8-TCDF at the maximum effluent concentration **is likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.7.6.2.2 Sediment exposure

The environmental baseline in the Action Area shows that the sediment concentrations of 2,3,7,8-TCDF are below 1 ng/kg (Table 7-16). Since this is more than 10 times lower than the sediment toxicity benchmark (261.1 ng/kg), EPA concludes that the discharge of this compound at the maximum effluent concentration **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.7.6.2.3 Habitat Effects

Since the concentration that listed species, or their prey would be exposed to is greater than the established benchmarks, EPA concludes that the discharge of 2,3,7,8-TCDF at the maximum effluent concentration is **likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.1.6.8 Adsorbable Organic Halides (AOX)

7.5.1.6.8.1 Introduction

Adsorbable Organic Halides (AOX) refers to a class of organic compounds with similar chemical and physical properties. AOX is a measure of the total amount of halogens (chlorine, bromine, and iodine) bound to dissolved or suspended organic matter in a wastewater sample. Relatively few specific chlorinated compounds contributing to AOX have been isolated (Kringstad et al., 1984). Both low- and high-molecular weight chlorinated compounds are measured by the AOX test. High-molecular weight chlorinated material comprising AOX is persistent in the aquatic environment and a portion of the high-molecular weight material is bioaccumulative and toxic (Paasivirta, 1991; Higashi et al., 1992). Specific tests to measure the

fraction of AOX that may be bioaccumulative (e.g., EOX-extractable organic halogens; EPOX-extractable persistent organic halogens) have not been standardized and there is no substantial database for these fractional measures of AOX upon which to establish protective levels.

Suntio et al. (1988) published a list of about 250 compounds, most of which are chlorinated, found in pulp mill effluent. Each one was present at a low concentration, but the number of chemicals present was numerous. The major categories of chlorinated compounds are organic acids, phenols, catechols, guaiacols, benzene derivatives, aldehydes, acetone derivatives and aliphatics. A switch from elemental chlorine (Cl_2) to chlorine dioxide (ClO_2) greatly reduces the amount of chlorine by-products in the effluent (Gifford, 1994). Characteristics of these by-products range from water-soluble and rapidly biodegradable substances to persistent and highly bioaccumulative substances such as dioxins and furans (Elliott et al., 1994).

7.5.1.6.8.2 Environmental Baseline

AOX has not been tested for in either the Snake or the Clearwater Rivers in the Action Area; therefore, the environmental baseline is unknown. To account for the uncertainty of the environmental baseline, this assessment assumes that the AOX environmental baseline is 0.33 TU, which is one-third the benchmark of 1 TU. Therefore, the environmental baseline is assumed to be below water column levels that are considered safe for threatened and endangered salmonids. Table 3-2 summarizes the range and average concentrations of various parameters monitored within effluent discharged from the Clearwater facility, and has been updated to reflect new data.

Beginning in 2005, and running through 2006, a groundwater monitoring program collected samples from 8 different sites adjacent to an aerated stabilization basin for the facility. Measurements made in 2005 indicated a maximum of 1280 ppb, with a minimum of 83 ppb (µg/L) (Table 7-20). In 2006, the maximum and minimum recorded concentrations of AOX were 911 and 12 ppb (µg/L), respectively (JUB Engineers 2006 a & b and 2007). Figure 7-14 and Figure 7-15 show the readings from each site in 2005 and 2006, respectively. In 2005 and 2006, as part of the NPDES annual monitoring report, both 2,3,7,8 TCDD and TCDF measurements were collected during the weekly receiving water monitoring study; 2,3,7,8 TCDD was not detected in any downstream or upstream location, but was detected in the solid fraction of effluent at a level below the toxicity benchmark (0.06 pg/L) set by the EPA.

Table 7-20: AOX readings from 4th quarter addendum to 2005 groundwater monitoring results.

Sample Site	2nd Quarter ug/L	3rd Quarter ug/L	4th Quarter ug/L
MW-1[1]	1280	ND	ND
MW-2	254	375	327
MW-2D	981	873	892
MW-3	781	706	847
MW-3D	665	641	584
MW-5	612	677	721
MW-10	811	779	820
MW-12	112	96.5	83

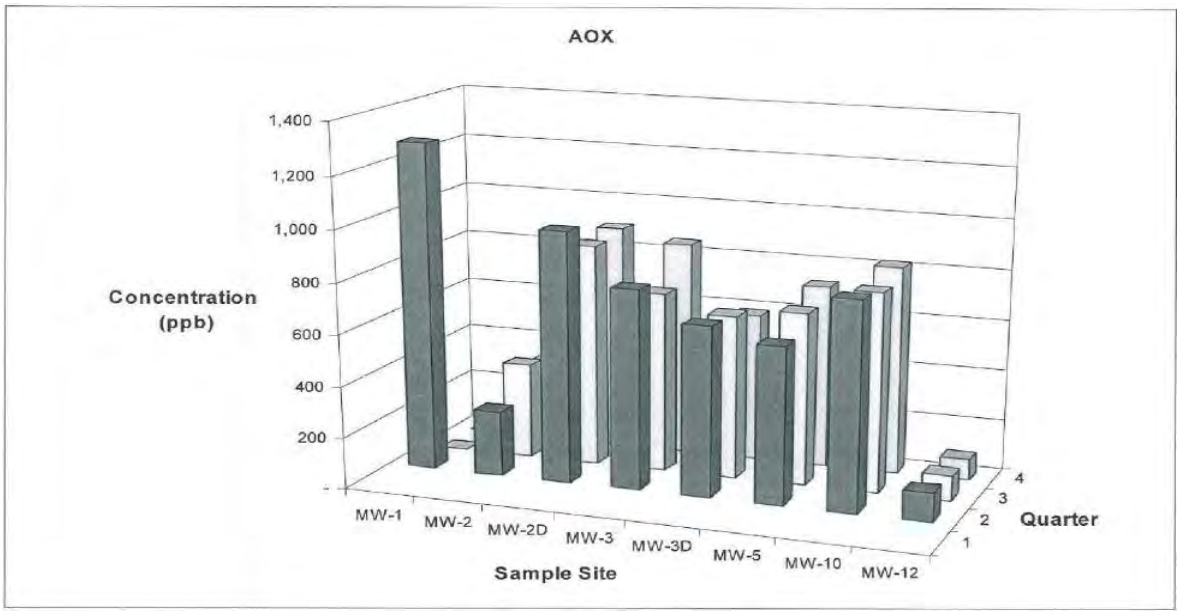


Figure 7-14: AOX readings from 4th quarter addendum to 2005 groundwater monitoring results.

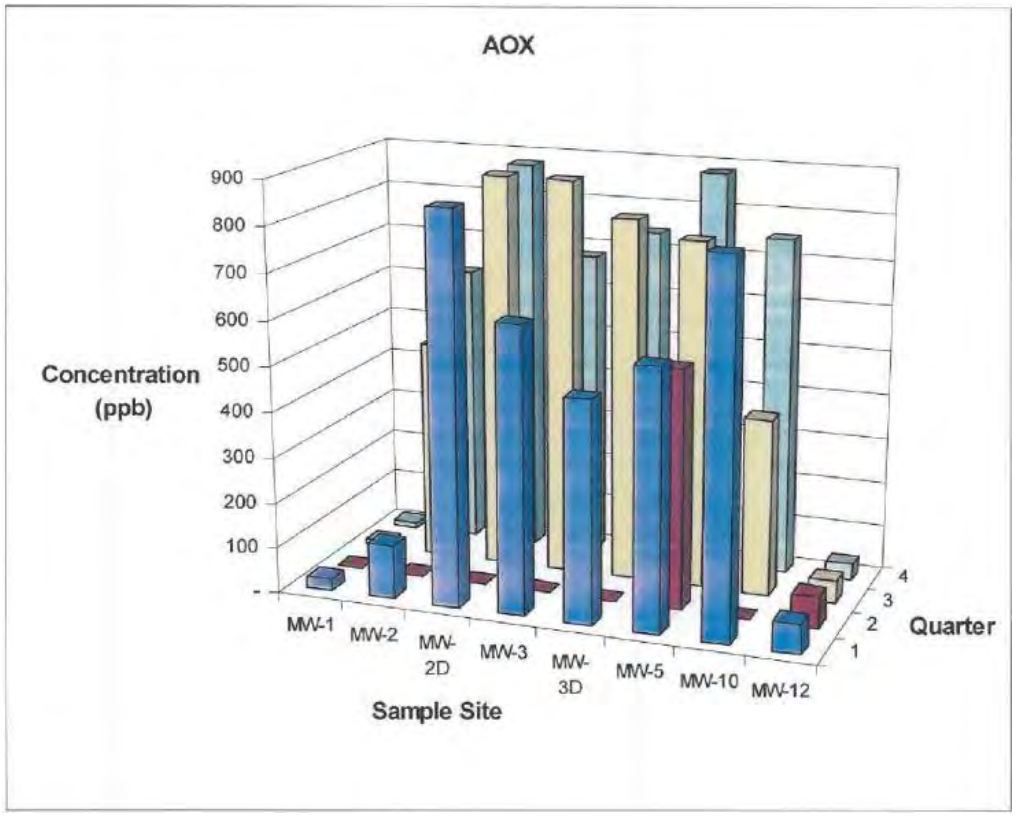


Figure 7-15: AOX readings from 2006 groundwater monitoring results from 8 sites adjacent to an aerated stabilization basin.

Table 7-21: AOX readings from 2006 groundwater monitoring results from 8 sites adjacent to an aerated stabilization basin.

Sample Site	1st Quarter ug/L	2nd Quarter ug/L	3rd Quarter ug/L	4th Quarter ug/L
MW-1[1]	25	18	14	12
MW-2	119	369	487	622
MW-2D	857	911	875	875
MW-3	629	696	874	679
MW-3D	487	585	800	744
MW-5	569	526	764	882
MW-10	807	834	391[1]	748
MW-12	64	69	50	37

7.5.1.6.8.3 Water Quality Standard

The current Idaho and Washington water quality standards do not specifically address AOX, however they both have narrative criteria to limit toxic material concentrations to levels below those which have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic conditions to the most sensitive biota dependent upon those waters, or adversely affect public health.

7.5.1.6.8.4 Effluent Limitation

In the effluent, essentially all of the AOX is chlorinated compounds formed during bleaching with chlorine and other chlorinated bleaching agents. Inefficient application of chlorine-containing bleaching chemicals can generate increased levels of AOX. Statistically valid relationships between AOX and specific chlorinated organic compounds have not been established. It is unlikely that correlations for a macro constituent such as AOX, which is measured at the mg/L level, with micro constituents such as chlorinated phenolics measured at the μg/L level, can be made. However, further data gathering and more refined statistical

analysis may establish relationships among AOX and certain chlorinated pollutants or groups of pollutants.

The data EPA used to develop ELGs demonstrate a correlation between the presence of AOX, and the amount of chlorinated bleaching chemical, used in relation to the residual lignin in the pulp (expressed as the kappa factor). The record further shows that there is a correlation between the kappa factor and the formation of dioxin and furan. Therefore, EPA concluded that reducing AOX loadings has the effect of reducing the mass of dioxin, furan, and other chlorinated organic pollutants discharged by this industry. Minimizing AOX will usually have the effect of reducing the generation of chloroform, 2,3,7,8-TCDD, 2,3,7,8-TCDF, and chlorinated phenolic compounds. Additionally, some AOX is biodegraded during secondary treatment in the ASB.

EPA's decision to regulate AOX is also based on the fact that AOX, unlike most of the chlorinated organic compounds regulated today, is comparatively inexpensive to monitor for and is easily quantified by applicable analytical methods. Thus, while EPA could have decided to control the formation of dioxin, furan, chloroform, and the 12 regulated chlorinated phenolic pollutants by requiring mills to monitor for those pollutants daily, EPA also recognizes that testing for those pollutants is expensive and time consuming. In contrast, daily monitoring for AOX as required by the ELGs is considerably less expensive. Moreover, the presence of AOX can be readily measured in the Mill's effluent, in contrast to the presence of many of the chlorinated organic compounds regulated in the ELGs, which for the most part are likely to be present at levels that cannot be reliably measured by the current analytical methods in 40 CFR 136.

The effluent limitations for AOX specified in the draft permit are based upon technology rather than water quality because there is not a specific water quality criterion for this parameter, nor is it feasible to develop one since the composition of AOX can vary greatly amongst industries and dischargers. Therefore, EPA relies on whole effluent toxicity to ensure protection of the narrative water quality standard for toxics.

The proposed 2019 effluent limitations require a maximum daily effluent limitation of 2,979 lb AOX/day and an average monthly effluent limitation of 1,951 lb AOX/day. These limits are based upon the last five years production of unbleached kraft market pulp and are equivalent to an effluent concentration of **9.40 mg/L** and **6.16 mg/L**, respectively, based upon an effluent flow rate of 31.6 mgd.

7.5.1.6.8.5 Toxicity Benchmarks

Since the composition of AOX varies greatly amongst facilities and industries, depending upon the raw materials used and the specific bleaching and wastewater treatment processes at the facilities, there are no studies in the literature that have evaluated the toxicity of AOX to either terrestrial or aquatic species. Toxicity data for some of the specific chlorinated phenolic compounds that compose AOX and are pollutants of concern for this discharge are provided in the specific discussions of those compounds. These compounds include 12 chlorinated phenolic compounds, 2,3,7,8-TCDD and 2,3,7,8-TCDF.

Environmental persistence data indicate that several removal mechanisms may be responsible for reducing concentrations of COCs in river systems. Information in HSDB (2000) indicates that volatilization, photolysis, and biodegradation have been identified as removal mechanisms. For

many COCs, photolysis is an important removal mechanism. Measured half-lives in surface water are not available for many of the COCs. Table 7-12 presents the available published half-live values. As shown in Table 7-12, half-lives for COCs in surface water range from a few hours to a few hundred hours with a geometric mean between 10 and 18 hours (HSDB, 2000; Howard et al., 1991). Half-lives that combine photodegradation, biodegradation, and volatilization tend to be shorter than half-lives that only consider volatilization.

Since many of these compounds are hydrophobic, they will associate with sediment and organic particles in the water column. Such associations will cause the chemicals to persist in the water and not be eliminated from the system. Organisms ingest sediment and small organic particles (including plankton) that likely contain these hydrophobic contaminants leading to bioaccumulation and potential toxic effects.

An adequate toxicological assessment of AOX must consider the additive effects of chlorophenols, chlorocatechols, chloroguaiacols, dioxins and furans. Because these compounds likely act by a common mode of action, they should be considered together with the toxic unit approach. A conservative assumption is that the toxicity of many toxicants is additive, which is supported by studies and review articles (McCarty and Mackay, 1993; Escher and Hermans, 2002). One recent study has demonstrated additivity for phenolic compounds and in some cases, the interaction was synergistic (Escher et al., 2001). For example, the binary mixture of 3,4,5-trichlorophenol and 2,4-dinitrophenol were shown to be synergistic in their ability to cause toxicity. For COCs, additivity is a reasonable assumption. Hence a simple toxic-unit approach would be valuable in protecting listed species from multiple toxicants that are at, or close to toxic levels.

The effluent of 2 bleached kraft paper mills in Ontario was evaluated to determine toxicity, in relation to AOX concentrations. One site, which had AOX concentrations in its effluent ranging from 21.6-34.6 mg/L, caused lethal toxicity in rainbow trout (*Oncorhynchus mykiss*); LC50 values ranged from 39-71% effluent (Craig et al. 1999). Chronic values (producing reproductive inhibition) ranged from 2-25% effluent, for *Ceriodaphnia dubia* (Craig et al. 1999).

For each toxicant found in a water body, its potential for toxicity can be determined by the equations:

$$ToxicUnit(TU) = \sum \frac{[effluent]}{[benchmark]}$$ within the mixing zone

or

$$ToxicUnit(TU) = \sum \frac{[water]}{[benchmark]}$$ at and beyond the mixing zone

where,

[effluent] = the maximum effluent concentration
[water] = the maximum concentration at and beyond the mixing zone
[benchmark] = the direct toxicity NOEC.

The TU value of 1.0 is considered the combined NOEC toxicity benchmark value. If the TU calculation is below 1.0, then the combined water concentrations for COCs must be below the

level considered likely to cause toxicity. As TU values increase above 1.0, the potential for toxic effects increases.

This BE uses a TU value of 1.0 as the combined NOEC toxicity benchmark value for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.8.6 Effects Analysis

The fate of AOX compounds is largely dependent on their physical and chemical properties, resulting in various end products and final accumulation sites. When AOX is discharged into the river, the organic components in the effluent tend to accumulate with organic substances, such as the sediments or biological tissues, or volatilize into the air (Gifford, 1994). The hydrophilic components will likely remain in solution. The intermediates formed during the degradation process may be more biodegradable substances or more persistent compounds (Gifford, 1994). Organisms in the sediment can take up the hydrophobic compounds, initiating accumulation in the food chain (Gifford, 1994). AOX compounds have been reported to accumulate in sediments downstream of pulp mills (Jokela et al., 1993). Chloroguaiacols and chlorocatechols have been reported to have high sedimentation near the mill, while chlorophenols are not as strongly affected (Kukkonen et al., 1996). A discussion of the sedimentation of AOX compounds is discussed in the TSS effects analysis (see Subsection 1.e).

Table 7-22 provides the results of the additive toxicity equivalency of compounds contributing to AOX in the effluent. However, it should be known, that it would be unlikely that all of these pollutants would be discharged simultaneously at their maximum daily effluent limits.

Since the maximum toxicity equivalent for AOX allowed under this permit (90 TU) is greater than the water column toxicity benchmark (1 TU), this analysis looks at the effects within the exposure volume of the effluent (i.e., the area where the concentration of the plume exceeds the toxicity benchmark) and the effects at and beyond the exposure volume boundary.

7.5.1.6.8.6.1 Direct Effects

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the chronic mixing zone would be 36.5, therefore the maximum exposure concentration at the edge of the chronic mixing zone would be the maximum AOX toxicity equivalency of 90 divided by the available dilution of 36.5, or 2.47. The calculated maximum AOX toxicity equivalency of 6.62 is higher than the toxicity benchmark of 1.0.

Table 7-22: AOX Toxicity Equivalency in the Clearwater Mill Effluent

Compound	Maximum Effluent Concentration (µg/L)	Direct Toxicity Benchmark (µg/L)	TU
2,4,5-trichlorophenol	0.95	2.6	0.37
2,4,6-trichlorophenol	0.95	7.3	0.13
2,3,4,6-tetrachlorophenol	1.91	3.3	0.58
Pentachlorophenol	1.91	0.18	10.6
3,4,5-trichlorocatechol	1.91	2.6	0.73
3,4,6-trichlorocatechol	1.91	2.6	0.73
Tetrachlorocatechol	1.91	11	0.17
3,4,5-trichloroguaiacol	0.95	7.5	0.13
3,4,6-trichloroguaiacol	0.95	2.6	0.36
4,5,6-trichloroguaiacol	0.95	2.6	0.36

Tetrachloroguaiacol	1.91	10	0.19
Trichlorosyringol	0.95	2.6	0.36
2,3,7,8-TCDD (pg/L)	0.92	0.063	14.6
2,3,7,8-TCDF (pg/L)	12.2	0.20	61
AOX toxicity equivalency			**90**

Therefore, EPA has concluded that the discharge of this compound at the maximum effluent concentration is likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.1.6.8.6.2 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is greater than the established benchmarks, EPA concludes that the discharge of AOX at the maximum effluent concentration is **likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2 Parameters Monitored in Permit without Effluent Limitations

7.5.2.1 Ammonia

7.5.2.1.1 Introduction

Ammonia is a colorless gas with a very sharp, pungent odor. It is about one-half as dense as air at ordinary temperatures and pressures. Ammonia forms a minute portion of the atmosphere; it is produced naturally in soil by bacteria, decaying plants and animals, animal wastes, and volcanic gases; and it occurs in surface and ground waters due to the decomposition of nitrogenous organic matter. It is one of the constituents of the complex nitrogen cycle and is essential for many biological processes. Ammonia in surface waters may also result from the discharge of industrial and municipal wastes. Most of the ammonia produced in chemical factories is used to make fertilizers. The remaining is used in textiles, plastics, explosives, pulp and paper production, food and beverages, household cleaning products, refrigerants, and other products. It is also used in smelling salts. The amount of ammonia produced by humans every year is almost equal to that produced by nature every year.

The melting and boiling points of ammonia are -77.7°C and -33.5°C, respectively. It dissolves easily in water and evaporates quickly. In water, ammonia occurs in two forms, which together are called the total ammonia nitrogen. Chemically, these two forms are represented as NH_4^+ and NH_3. NH_4^+ is called ionized ammonia because it has a positive electrical charge, and NH_3 is called unionized ammonia since it has no charge. Ammonia is a weak base while ammonium ions are a weak acid in aqueous solution, where some of the ions dissociate into ammonia and hydrogen ions. Most of the ammonia in water transforms to ammonium, an odorless liquid. Ammonia and ammonium can transform back and forth in water given the proper conditions. Water temperature and pH will decide which form of ammonia is predominant at any given time in an aquatic system. These speciation relationships are important, since NH_3, un-ionized ammonia, is the form that is most toxic to fish. This is mainly because it is a neutral molecule and thus can diffuse across the epithelial membranes of aquatic organisms much more readily than the charged ammonium ion. High external un-ionized ammonia concentrations reduce or reverse diffusive gradients and cause the buildup of ammonia in gill tissue and blood.

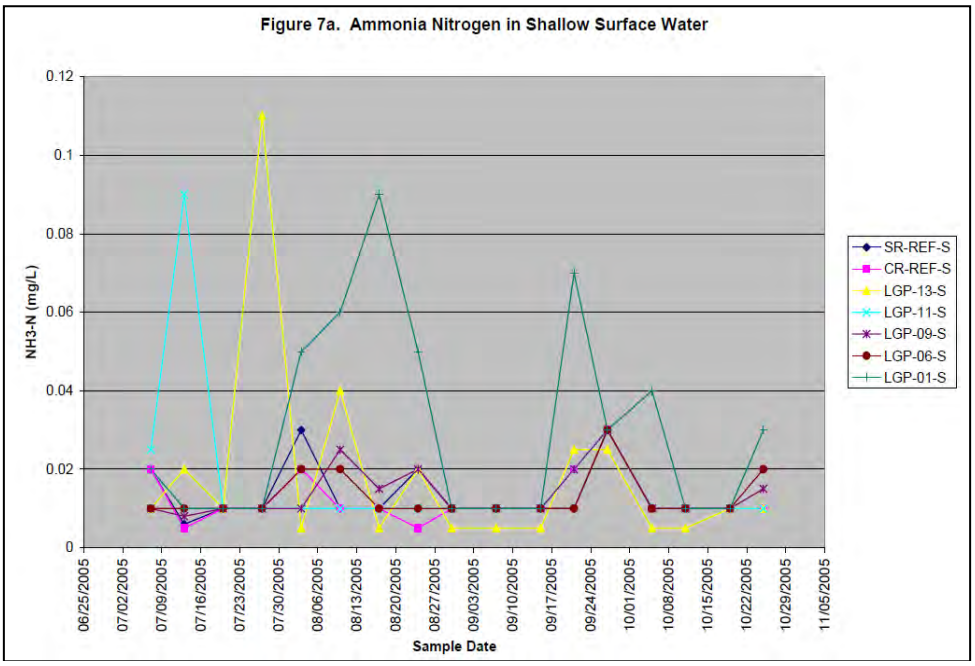
Ammonia exists in its un-ionized form only at higher pH levels and is most toxic in this state. The lower the pH, the more ionized ammonia is formed, and its toxicity decreases. Ammonia, in the presence of dissolved oxygen, is converted to nitrate (NO_3) by nitrifying bacteria. Nitrite (NO_2), which is an intermediate product between ammonia and nitrate, sometimes occurs in quantity when depressed oxygen conditions permit. Ammonia can exist in several other chemical combinations including ammonium chloride and other salts.

Ammonia does not last very long in the environment. Plants and bacteria rapidly take up ammonia from soil and water; therefore, ammonia does not build up in the food chain, but serves as a nutrient source for plants and bacteria.

7.5.2.1.2 Environmental Baseline

Because ammonia occurs naturally, it is found throughout the environment in soil, air, and water at varying concentrations. As part of the re-issuance of the facility's NPDES permit, non-discretionary monitoring to characterize conditions in the effluent, receiving water, sediment, and biological media near the facility was initiated. One of the parameters measured was ammonia. Data from the 2005 sampling effort indicated ammonia nitrogen concentrations ranging from non-detect to 0.11 mg/L (Figure 7-16). The maximum value of 0.11 mg/L was measured at site LGP-13-S, just downstream of the outfall. Among the upstream samples, ammonia nitrogen concentrations ranged from ND to 0.03 mg/L. Among the downstream samples, ammonia nitrogen concentrations ranged from ND to 0.11 mg/L. In both shallow and mid-depth samples, concentrations were typically below 0.04 mg/L. During each week, the location of the maximum concentration varied among mid-depth samples. In shallow samples, the maximum concentrations were often detected at LGP-01 (the farthest downstream sample location).

Similar to the data from 2005, data from the 2006 sampling effort indicate nitrogen concentrations in the range of non-detect to 0.11 mg/L (Figure 7-17). The maximum concentration was measured in a sample collected at station SR-REF-S on 7/18/06.



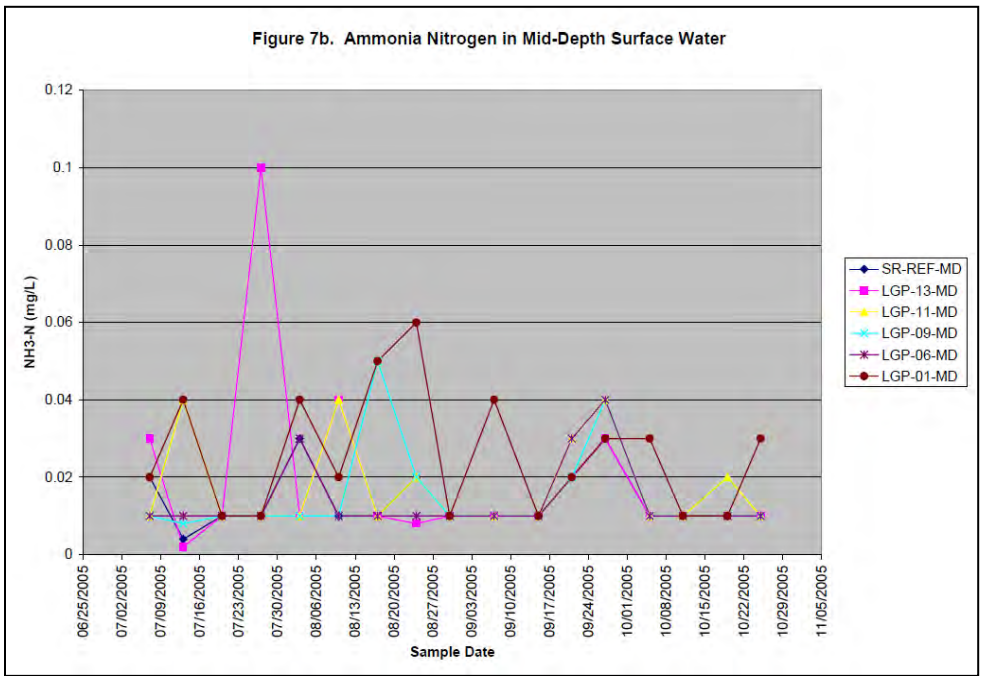
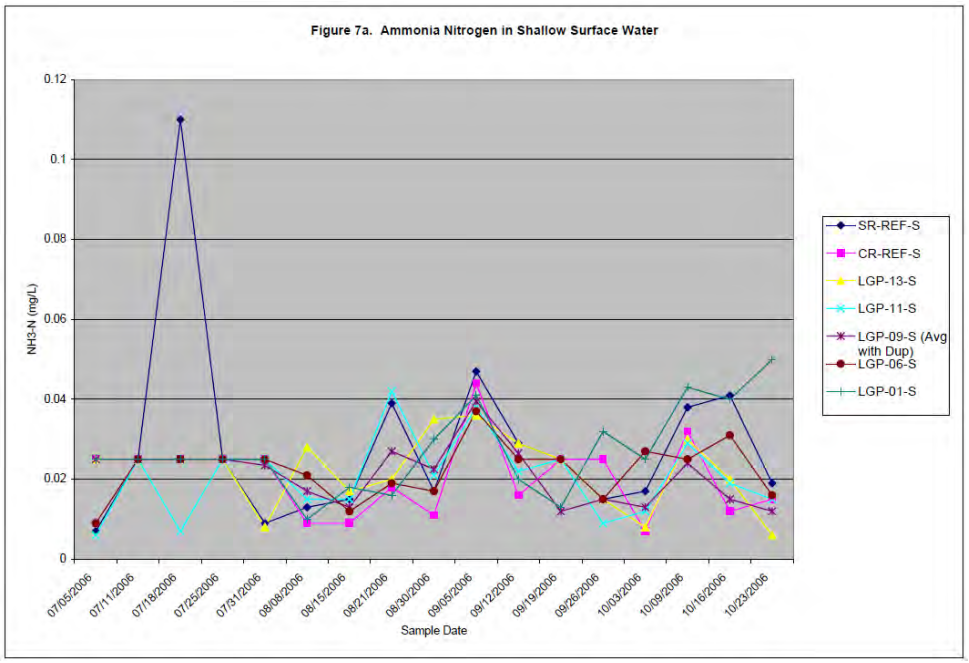


Figure 7-16: Receiving water ammonia concentrations at surface and mid-depth within the vicinity of the Clearwater facility (2005).



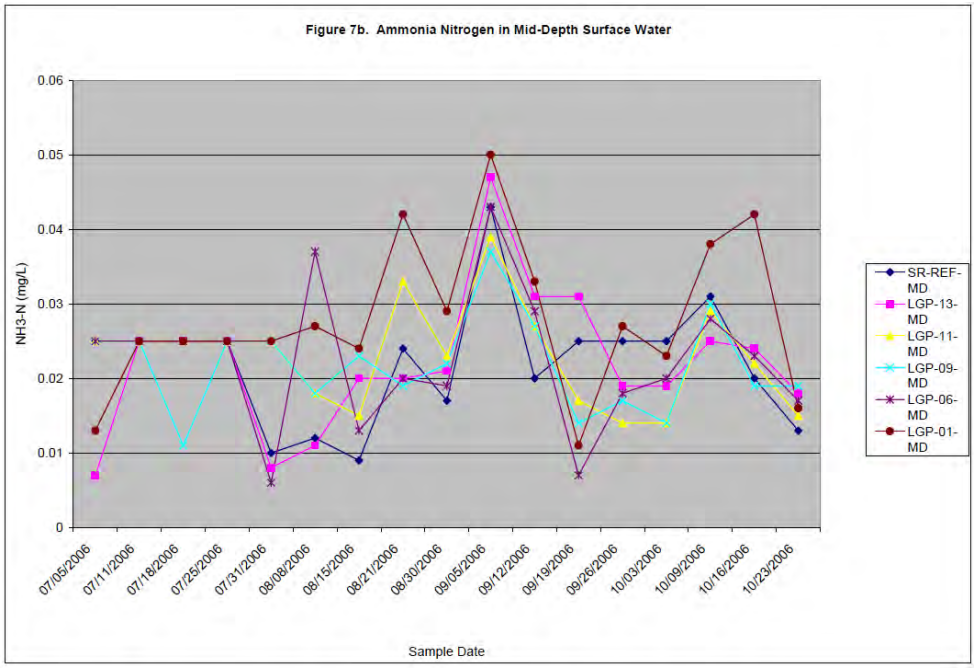


Figure 7-17: Receiving water ammonia concentrations at surface and mid-depth within the vicinity of the Clearwater facility (2006).

Among the downstream samples, ammonia nitrogen concentrations ranged from ND to 0.05 mg/L (LGP-01-S; 10/23/06 and LGP-01-MD; 9/05/06). In both shallow and mid-depth samples, concentrations were typically below 0.03 mg/L. During each week, the location of the maximum concentration was often detected at LGP-01-S (the farthest location from the Facility). In shallow samples, the location of maximum concentration varied over the monitoring period.

These results indicate that the facility's effluent has little likelihood of affecting listed and endangered species in terms of ammonia toxicity given the low levels found in monitoring.

As part of the Biological Opinion (BO) completed by NOAA Fisheries, one non-discretionary requirement for permit issuance was the implementation of a monitoring and assessment plan to characterize the effluent and receiving water near the facility. As part of this monitoring, ammonia was analyzed on a quarterly basis at surface and mid-depth. Ammonia nitrogen concentrations during the 2005 monitoring effort indicated ammonia levels in the range of ND (non-detect) to 0.11 mg/L. Downstream levels were assessed in the range of ND to 0.11 mg/L. Ammonia concentrations during the 2006 monitoring effort indicated ammonia levels in the range of ND to 0.11 mg/L. Downstream levels were assessed in the range of ND to 0.05 mg/L indicating that it is not likely that the ammonia will have negative effects on listed species given the low levels found in monitoring.

Required groundwater monitoring performed quarterly in 2005 and 2006 within the vicinity of the facility indicated ammonia levels in the range of 0.45 and 18.5 mg/L with a mean of 5.2 mg/L in 2005 (Table 7-23 and Figure 7-18); and 0.64 to 17.3 mg/L with a mean of 4.97 mg/L in 2006, indicating higher levels of ammonia in groundwater than surface water (Figure 7-19).

Table 7-23: Ammonia as Nitrogen, 2005 Annual Groundwater Monitoring Report Addendum.

Site	2nd Quarter	3rd Quarter	4th Quarter
MW-1[1)]	18.50	1.08	1.65
MW-2	0.45	0.99	1.08
MW-2D	3.04	2.81	2.70
MW-3	4.46	4.08	3.70
MW-3D	3.90	4.06	3.70
MW-5	13.50	13.80	9.70
MW-10	1.82	8.04	8.00
MW-12	9.80	1.72	1.50
(1) – Data from separate well locations, see 2005 Groundwater Report.			

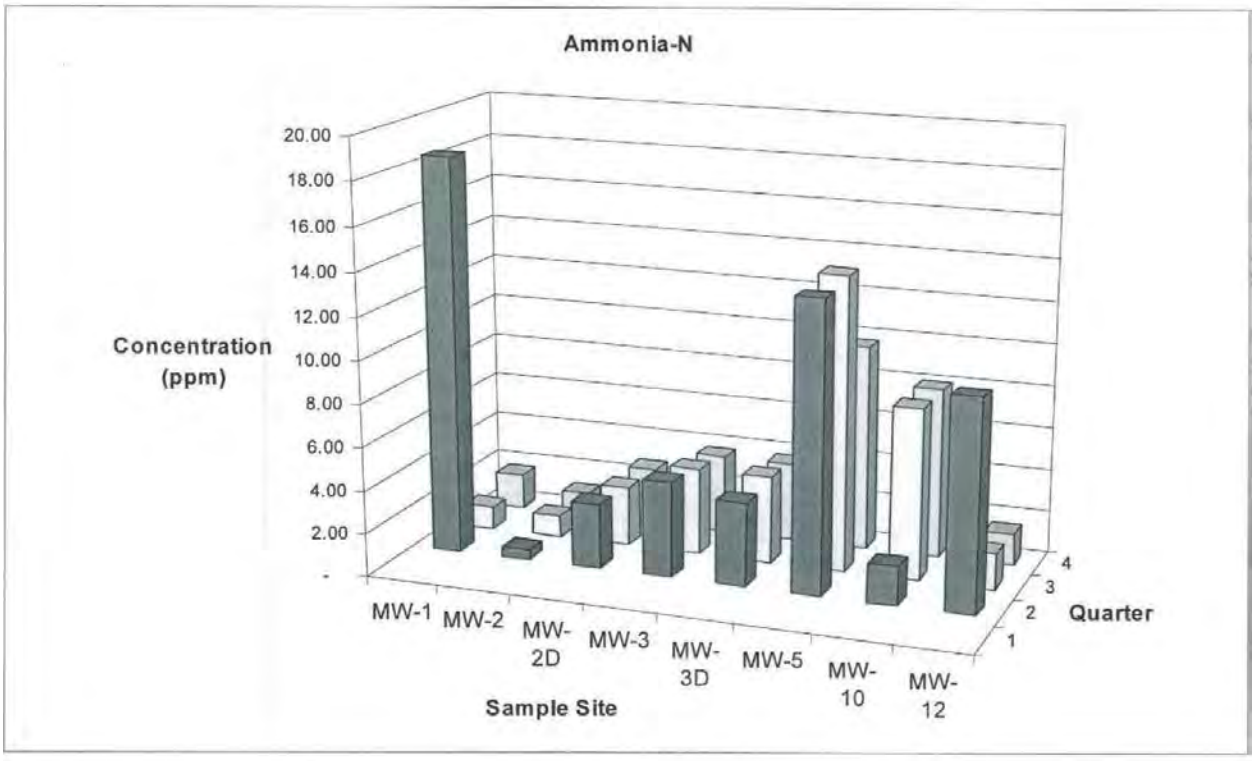


Figure 7-18: Concentration of ammonia in groundwater at multiple groundwater wells at the Clearwater facility in 2005 (AMEC, 2006).

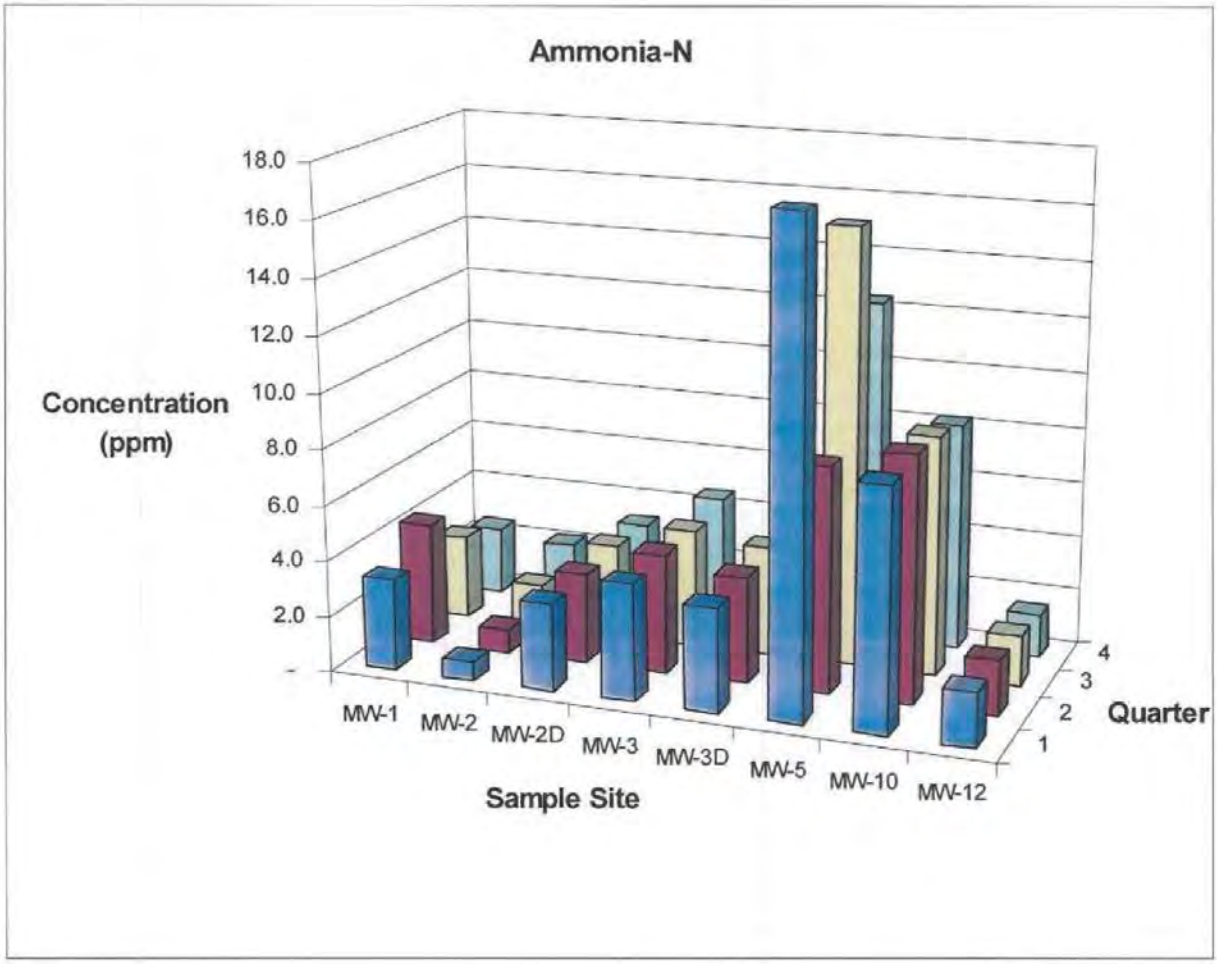


Figure 7-19: Concentration of ammonia in groundwater at multiple groundwater wells at the Clearwater facility in 2006 (AMEC, 2007).

7.5.2.1.3 Water Quality Standard

The most stringent Idaho and Washington water quality standards require the ammonia criteria not to be exceeded dependent upon the temperature and pH of the waterbody for the protection of aquatic life. The criteria are as follows:

The one-hour average concentration of total ammonia nitrogen (in mg/L N) is not to exceed, more than once every three years, the value calculated using the following equation:

$$CMC = \frac{0.275}{1+10^{7.204-pH}} + \frac{39.0}{1+10^{pH-7.204}} .$$

Based on the water quality standards for pH (6.5 – 9.0) and temperature (22˚) that apply to this waterbody for this duration, the 1-hour ammonia water quality criterion would result in the range of 0.88 to 33 mg/L (as total ammonia). [Note that the lower criterion results from the higher pH.] Since the average pH of the Snake River is 7.6, the CMC used for the analysis of the potential effects in this BE is 11 mg/L total ammonia.

The thirty-day average concentration of total ammonia nitrogen (in mg/L N) is not to exceed, more than once every three years, the value calculated using the following equation when early life stages are likely present:

$$CCC = \left(\frac{0.0577}{1+10^{7.688-pH}} + \frac{2.487}{1+10^{pH-7.688}}\right) \times \min(2.85, 1.45 \cdot 10^{0.028 \cdot (25-T)})$$

Based on the water quality standards for pH (6.5 – 9.0) and temperature (19˚) that apply to this waterbody for this duration, the 30-day ammonia water quality criterion would result in the range of 0.30 to 5.0 mg/L (as total ammonia). [Note that the lower criterion results from the higher pH.] Since the average pH of the Snake River is 7.6, the CCC used for the analysis of the potential effects in this BE is 2.9 mg/L total ammonia.

The highest four-day average within the thirty-day period should not exceed 2.5 times the CCC. Based on the water quality standards for pH (6.5 – 9.0) and temperature (22˚) that apply to this waterbody for this duration, the 4-day average ammonia water quality criterion would result in the range of 0.75 to 12.5 mg/L (as total ammonia). [Note that the lower criterion results from the higher pH.] The four-day average CCC used for the analysis of potential effects in this BE is 7.3 mg/L total ammonia.

7.5.2.1.4 Effluent Limitation

Pulp and paper mills normally contain only minor concentrations of ammonia; however, higher concentration can be observed when ammonia is added to provide desired biological waste treatment efficiencies. The 1992 permit contained limits of 5.4 and 3.0 mg/L (daily maximum and monthly average, respectively) for ammonia because Clearwater occasionally added ammonia to the treatment system influent to provide nutrients for the treatment system. Clearwater has since discontinued this practice; thus, the maximum effluent ammonia concentration reported by Clearwater is now 8.5 mg/L. Based on this concentration, there is no reasonable potential to cause or contribute to an exceedance of water quality standards and permit limits are no longer necessary.

7.5.2.1.5 Toxicity Benchmark

Evidence exists that ammonia exerts a toxic effect on all aquatic life depending upon the pH, dissolved oxygen level, and the total ammonia concentration in the water. A significant oxygen demand can result from the microbial oxidation of ammonia. Approximately 4.5 grams of oxygen are required for every gram of ammonia that is oxidized. Ammonia can add to eutrophication problems by supplying nitrogen to aquatic life. Ammonia exerts an oxygen demand, contributes to eutrophication and can be toxic.

Studies of ammonia exposure to early life stages (ELSs) of salmonids have yielded conflicting information. For example: the ELS tests by Calamari et al. (1977, 1981) with rainbow trout produced a total ammonia nitrogen LC_{20} of 1.34 mg N/L at pH 8. Solbe and Shurben (1989) indicated that the LC_{20} might be even lower for this species. In contrast, both Thurston et al. (1984) and Burkhalter and Kaya (1977) found no indication of severe mortality in cutthroat trout during 21- and 42-day exposures until higher concentrations of total ammonia were reached (e.g., 18.7-22.0 mg N/L). When Koch et al. (1980) exposed *Oncorhynchus clarkii henshawi* (Lahontan cutthroat trout) to various levels of ammonia over a 103-day period, several endpoints were reached. There were no successful hatches of the trout embryos at exposure levels of 148 mg/L, or more, of total ammonia nitrogen; and EC20 values of 17.89, and 25.83, at pH 7.57 and 7, respectively (USEPA, 2013). A more recent ELS study by Brinkman et al. (2009) exposed O. mykiss to various concentrations of total ammonia nitrogen, and found that survival, growth, and

biomass of the fry were reduced at 16.8 mg/L, with no effect at 7.44 mg/L (USEPA, 2013). The authors of that study calculated an EC20 for biomass of 15.60 mg/L at pH 7, of total ammonia nitrogen (USEPA, 2013). These values were pulled from an EPA document which also formulated a species mean chronic value (SMCV), based on several studies, for O. mykiss, of 6.663 mg/L total ammonia nitrogen, at pH 7 (USEPA, 2013). In the Thurston et al. (1984) study, exposure was continuous for several generations, whereas, exposure began within 24 hours of fertilization in the tests conducted by the three other research teams.

An important factor in studies assessing ammonia toxicity may be the specific ELS. Alevins, fry, and even eggs may appear to be "tolerant" to ammonia, relative to the criteria. In rainbow trout studies, Solbe and Shurben (1989) demonstrated that testing during the time period between fertilization and exposure indicated a certain level of sensitivity (i.e., increased mortality). When exposure began within 24 hours of fertilization, 26 mg N/L killed 98 percent of the embryos; however, when exposure began 24 days after fertilization, 26 mg N/L killed only three percent of the embryos. After 49-days, 26 mg N/L killed 40 percent of the embryos.

A review of other ammonia toxicity tests using rainbow trout did not show ammonia sensitivity at total ammonia concentrations near the acute or chronic criteria (most tests were on fish weighing less than 10 g). However, in many of the tests (e.g., ammonia effects on growth), a substantial drop in dissolved oxygen during the test confounded the results. Some test results also indicated acclimation and recovery. Daoust and Ferguson (1984) reported that swimming and feeding of some fish were affected for a period, followed by recovery. Smith and Piper (1975) found abnormal tissue in exposed fish; yet fish placed in clean water for 45 days at the end of the test had normal tissues.

Few studies useful to this evaluation were found for effects on salmon. Rice and Bailey (1980) exposed embryos and alevins of pink salmon for 61 days to ammonia at a relatively low pH of 6.4. Adjusting the results to a pH of 8 gives a total ammonia concentration of 11.2 mg N/L where the weight of emerging alevins was significantly reduced relative to the controls (although the large extrapolation for the low pH makes this concentration somewhat uncertain). Size at emergence was said to be important because smaller fry are less capable of surviving due to lower swimming endurance and higher susceptibility to predation. In sockeye salmon exposed to ammonia from the embryo stage to hatching, Rankin (1979) found an EC_{20} of less than 4.4 mg N/L total ammonia (adjusted to pH 8). Burrows (1964) exposed fingerlings to low ammonia concentrations at pH 7.8 for six weeks at 6º and 14º C to study the effects on gills; the fish did not recover after three weeks in clean water at 6º C, but they did recover at 14º C. However, compared to the first 24 hours of an embryo's life, fingerlings are probably not a particularly sensitive life stage (Solbe and Shurben, 1989).

It is well documented that un-ionized ammonia has the potential to be toxic to fish. Among fish, Chinook salmon (an applicable surrogate species) are moderately sensitive to ammonia; the acute sensitivity of Chinook salmon to ammonia ranks nine of 27 among freshwater fish genera (i.e., top 1/3). Servizi and Gordon (1990) found the 96-h LC50 for fingerling Chinook salmon weighing from one to seven grams to be 25.98 mg/L total ammonia nitrogen (TAN) at pH 8; whereas Thurston and Meyn (1984) found the 96-h LC50 for juvenile Chinook salmon weighing from 14.4 to 18.1 grams ranged from 14.50 to 19.53 mg/L.

Arillo et al. (1981) studied the biochemical effects of ammonia on rainbow trout tissues. The researchers report their data as un-ionized ammonia. Without adequate information to convert

the data to total ammonia, the following discussion uses ammonia as un-ionized. The test fish were not exposed to ammonia during the sensitive post-fertilization period, they were exposed as large fry. After a 48-hour exposure to 20 μgN/L of un-ionized ammonia, the biochemical data reflect induced alterations in various parameters. The biochemical compounds tested are, in many cases, involved in the primary toxicity mechanism of ammonia in trout. The researchers believe that the biochemical alterations represent more than the effect of an adaptive strategy and are probably the expression of physiological damage caused by a failure to maintain biochemical homeostasis. Arillo et al. (1981) point out that concentration values lower than 20 μgN/L of un-ionized ammonia caused an increased predisposition to disease and induced histopathological phenomena in gill epithelium. In addition, *Oncorhynchus mykiss* (formerly *Salmo gairdneri*) embryos exposed to 25 μgN/L of un-ionized ammonia show, at hatching, evident epithelial alterations (Arillo et al. 1981).

Potential indirect effects to the Pacific salmon would include loss of prey items, when those prey items are more sensitive to inorganics and potential loss of habitat as described for non-salmonid fish (Section 6.2.2). Additional potential indirect effects include olfactory impairment at relatively low concentrations resulting in an impaired avoidance response to predators (San Francisco Estuary Institute, 2013), as well as the potential impairment of survival and migratory success of wild salmonids (Baldwin et al., 2003).

EPA has conducted a full literature review of ammonia toxicity in the development of the 2013 update of ambient water quality criteria for ammonia (USEPA, 2013). From this information and using the 95th percentile pH and temperature observed at station LGP-13(8.54 standard units and 19.4°C respectively), **this BE uses the 2013 freshwater ammonia chronic criterion of 0.338 mg/L as the ammonia benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2.1.6 Effects Analysis

7.5.2.1.6.1 Direct Effects

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the chronic mixing zone would be 36.5. Based on this dilution factor and a background ammonia concentration of 51 μg/L, the maximum exposure concentration at the edge of the chronic mixing zone is 0.282 mg/L. The calculated maximum exposure concentration of 0.282 mg/L is less than the toxicity benchmark of 0.338 mg/L and the water quality standard of 0.743 mg/L.

Therefore, EPA concludes that the discharge of this compound at the maximum effluent concentration **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2.1.6.2 Indirect Effects

The EPA believes the 2013 ammonia criteria will be protective of indirect effects as well as direct effects. The maximum exposure concentration of ammonia (0.282 mg/L) is below the water quality standard for ammonia (0.743 mg/L) and the toxicity benchmark (0.338 mg/L). Therefore, the discharge of this compound **may indirectly affect, but is not likely to adversely**

affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.2.1.6.3 Habitat Effects

Since the concentration that could affect listed species, their prey, or benthic invertebrates is less than the established benchmarks, EPA has concluded that the ammonia in the discharge is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2.2 Chemical Oxygen Demand (COD)

7.5.2.2.1 Introduction

Chemical oxygen demand (COD) is the measure of the oxygen equivalent of the organic matter in a sample that is susceptible to oxidation by a strong chemical oxidant. The result is expressed as a concentration of oxygen consumed. The COD is a purely chemical oxidation test devised as an alternative method of estimating the total oxygen demand of a wastewater. Since the method relies on the oxidation-reduction system of chemical analyses rather than on biological factors, it is more precise, accurate, and rapid than the BOD test. The COD test is widely used to estimate the total oxygen demand (ultimate rather than 5-day BOD; BOD_5) to oxidize the compounds in a wastewater. It is based on the fact that strong chemical oxidizing agents under acid conditions can oxidize organic compounds, with a few exceptions, with the assistance of certain inorganic catalysis.

The COD test measures the oxygen demand of compounds that are biologically degradable and of many that are not. Pollutants measured by the BOD_5 test will be measured by the COD test. In addition, pollutants that are more resistant to biological oxidation will also be measured as COD. COD is a more inclusive measure of oxygen demand than is BOD_5 and will result in higher oxygen demand values than will the BOD_5 test.

The compounds which are more resistant to biological oxidation are of concern not only because of their slow but continuing oxygen demand on the resources of the receiving water, but also because of their potential health effects on aquatic life and humans. Many of these compounds result from industrial discharges and some have been found to have carcinogenic, mutagenic, and similar adverse effects, either singly or in combination. Concern about these compounds has increased due to demonstrations that their long life in receiving waters (the result of a slow biochemical oxidation rate) allows them to contaminate downstream waters. The commonly used systems of water purification are not effective in removing these types of materials and disinfection such as chlorination may convert them into even more hazardous materials.

Several studies have investigated the effects of COD discharged by pulp and paper mills upon aquatic life. Folke (1995) asserted that COD provides a useful indication of the sub-lethal toxicity of elemental chlorine free and totally chlorine free effluents, but that other factors, such as the source of the COD within the pulp mill, are important as well.

NCASI (1996) pointed out that Folke's paper did not receive peer review. NCASI and Archibald et al. (1998) also asserted that the regression analysis presented as Figure 7 in Folke is misleading because one data point with relatively low COD and a relatively high response index

was excluded without explanation, and a second data point with relatively high COD and the highest response index has a large impact on the regression analysis.

NCASI pointed out that the authors of one of the studies that produced some of the data presented in Folke's Table 7 attributed much of the effluents' effects to chlorate, rather than COD, and specifically the toxicity of chlorate to the macro brown macroalga *Fucus vesiculosus* (bladder wrack) (Lehtinen et al. 1991). Bladder wrack is an important component of the ecosystem of the Baltic Sea (Lehtinen et al. 1988). However, while chlorate is highly toxic to certain macro brown algal species such as bladder wrack, chlorate is non-toxic to most aquatic species (Van Wijk and Hutchinson 1995).

In a study of Japanese pulp and paper mill effluents, Araki (1997) observed a correlation with an r^2 of 0.78 between COD and toxicity as measured by the Microtox test using *Photobacterium phosphoreum*.

Verta et al. (1996) observed a correlation with an r value of 0.928 between COD and a calculated "toxicity index" (a combination of the results of toxicity tests using *Pseudomonas putida*, *Vibrio fisheri*, zebra fish (*Brachydanio rerio*), and the green alga *Selenastrum capricornutum* as the test organisms) for treated and untreated effluents from two pulp mills. However, the authors observed that "secondary treatment eliminated toxicity almost totally."

7.5.2.2.2 Environmental Baseline

Available data for COD within the Action Area are summarized below.

Table 7-24: Ambient COD Concentration in the Action Area (mg/L)

	Snake River Below Lower Granite Dam (USGS Station #13343600)	Snake River Below Ice Harbor Dam (Ecology Station #33A070)	Snake River near Pasco (Ecology Station #33A050)
Minimum	0	2	7
10th Percentile	3	4	9
Median	10	9	10
Average	12	14	13
90th Percentile	22	19	20
Maximum	52	276	29
Standard Deviation	10	27	7
Count	32	113	7
Earliest Sample Date	7/21/1975	10/2/1978	12/3/1990
Latest Sample Date	7/13/1978	11/5/1990	9/2/1991

7.5.2.2.3 Water Quality standard

The current Idaho and Washington water quality standards do not specifically address COD; however, Idaho has a narrative criterion that requires waters to be free from oxygen-demanding materials in concentrations that would result in an anaerobic water condition. Additionally, Idaho and Washington have water quality standards for dissolved oxygen (see discussion in section VII.E.3.b, below).

7.5.2.2.4 Effluent Limitation

The EPA is not proposing effluent limits for COD in the Clearwater Paper permit. The permit includes BMP requirements that are intended to reduce the discharge of wood extractives, and, in turn, COD. In addition, the permit includes seasonal water quality-based effluent limits for BOD5 that require reductions in the facility's discharges of organic material beyond what is required by the technology-based effluent limits. Further, the permit has WET provisions to address toxicity.

There is some evidence that the toxicity of pulp and paper effluents may be correlated to the COD concentration (Araki 1997, Folke 1995). However, pulp and paper effluents receiving secondary treatment generally have both a low COD concentration and low toxicity (Verta et al. 1996, Martel and Kovacs 1997). The effect of chlorate upon bladder wrack in mesocosom experiments was a confounding factor in some of the experiments showing an apparent correlation between COD and toxicity (NCASI 1996, Lehtinen et al. 1991).

Because the link between COD concentration and toxicity is unclear, and because the permit contains other conditions which address both toxicity and COD, no effluent limit is proposed.

7.5.2.2.5 Toxicity Benchmarks

It is important to note that studies investigating the effects of COD in pulp mill effluents exposed organisms to whole effluents and not the COD component in isolation. However, Verta et al. (1996) observed no toxicity to *Pseudomonas putida*, *Vibrio fisheri*, zebrafish (*Brachydanio rerio*), or the green alga *Selenastrum capricornutum* when these organisms were exposed to effluents containing 310 mg/L COD or less. Therefore, the EPA has used 310 mg/L as the toxicity benchmark for COD.

7.5.2.2.6 Effects Analysis

The chronic toxic effects are associated at least in part with families of non-chlorinated organic materials that are measured by the existing COD analytical method. Some of these materials, including several wood extractive constituents found in pulping liquors, are refractory (i.e., resistant to rapid biological degradation) and thus are not measurable by the five-day biochemical oxygen demand (BOD_5) analytical method. Therefore, the proposed permit limits for BOD may not, by themselves, control the chemical oxygen demand of the effluent.

However, the impact of the COD discharged by the facility is likely to be small. The maximum daily effluent concentration of COD reported by Clearwater Paper is 665 mg/L. The 90^{th} percentile ambient concentration of COD measured at USGS Station #13343600 (Snake River below Lower Granite Dam) was 22 mg/L. Thus, at the edge of the chronic mixing zone, the maximum expected concentration of COD is 66 mg/L. This is less than the toxicity benchmark of 310 mg/L. Therefore, EPA has concluded that the discharge of this compound **may affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2.2.6.1 Habitat Effects

Since the maximum effect of the final BOD limits will cause a minimal DO deficit in the water column, EPA has concluded that the COD of the discharge **is not likely to adversely modify the**

critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.2.3 Nutrients

7.5.2.3.1 Summary

Nutrients are essential to the health and diversity of our surface waters. However, in excessive amounts, nutrients cause hypereutrophication, which results in overgrowth of plant life and decline of the biological community. Chronic nutrient over-enrichment of a waterbody can lead to the following consequences: low dissolved oxygen, fish kills, algal blooms, overabundance of macrophytes, likely increased sediment accumulation rates, and species shifts of both flora and fauna. The principal adverse impact of nutrient enrichment is to change the trophic state of a waterbody. Excessive nutrients promote excessive algae and plant growth that leads to the depletion of DO because of nighttime respiration and bacterial decomposition and increased pH due to removal of dissolved carbon dioxide from the water during photosynthesis.

Cultural eutrophication is a term used to describe the undesirable effects in the water quality of a lake that is unnaturally enriched by fertilizers and other sources of nutrients from human activities. The most important nutrient relative to eutrophication of fresh waters is phosphorus, an essential element for the growth of aquatic plants and algae. When phosphorus is overabundant, it can lead to an excessive growth of algae and aquatic weeds, and an accompanying reduction in water quality.

The trophic state of a lake reflects the availability of nutrients for the growth of algae. Lakes range from oligotrophic (nutrient poor—literally "poorly fed") to eutrophic (nutrient rich—"well fed"). In between these extremes, lakes are termed mesotrophic while extremely eutrophic conditions are called hypereutrophic. Lower Granite Reservoir (LGR) is currently mesotrophic to eutrophic (Falter, 2001; Normandeau, 1999). Typically, lakes begin as oligotrophic and, over geologic time, become progressively more and more eutrophic. The rate of this transition depends upon the quantity of organic matter supplied to the lake by its drainage basin and the nutrients recycled within the lake itself.

The process described above, proceeding imperceptibly through geologic time, would hardly seem sufficiently worrisome to provoke concern. However, a lake's natural course can be greatly accelerated by human activities. Fertilizer from lawns or farmlands; wastewater from residential septic systems, sewage treatment plants, and industrial sources; and urban runoff are likely to supply nutrients at a far greater rate than natural processes. The resulting acceleration of the lake's nutrient enrichment hastens the arrival of the eutrophic condition, the process called cultural eutrophication.

Nutrients (particularly phosphorus, but also nitrogen) play a critical role in lake eutrophication. Although algae and aquatic plants require many chemical elements for growth and life processes, the major nutrients are phosphorus, nitrogen, and carbon. On average, plant tissue contains these three elements in the ratio of one-part phosphorus (by weight) to seven parts nitrogen to forty parts carbon (Wetzel, 2001). This ratio must be roughly preserved in the plant's nutrient intake as well. According to Liebig's Law of the Minimum, the growth of an organism will be limited by that nutrient which is least abundant relative to the organism's needs. In most lakes, this limiting nutrient is the element phosphorus (Wetzel, 2001), and thus, studies of lake eutrophication focus on phosphorus.

Phytoplankton (algae) derive their nutrients from the water column, and thus are directly affected by nutrient inflows to the lake. Macrophytes (aquatic plants) generally derive their nutrients from the bottom sediments and are much less directly affected by nutrient inflows. Generally, macrophyte growth is limited by physical factors including the amount of light, water depth, temperature, and bottom sediment composition. Only free-floating plants respond directly to water-column nutrient concentrations. Rooted macrophyte problems are worsened, however, by the attendant growth of attached algae, which depend upon water-column nutrients for growth. These algae are often filamentous and cause mats and other unaesthetic conditions.

To understand the dynamics of phosphorus in a lake, it is also necessary to understand the lake's thermal structure and hydrodynamics. Deep lakes and reservoirs in temperate climate zones show a distinct seasonal cycle in temperature structure with depth. At the end of winter, a lake is typically mixed throughout its depth and shows a vertically isothermal temperature profile—the water is at a constant temperature of approximately 4 °C from top to bottom. As the sun and atmosphere warm the lake surface through the spring, the shallowest water warms relative to the deeper water. Soon, a distinct layer of warmer and lighter surface water floats atop the cold, heavier deep water. The surface layer is known as the epilimnion; the deep water as the hypolimnion. The intervening layer, in which temperature decreases rapidly with depth, is known as the thermocline. (Technically, the thermocline is defined as the zone in which the change in temperature with depth exceeds 1 °C per meter of depth.)

Through the summer, the thermocline becomes stronger (that is, the change in temperature over the vertical distance of the thermocline increases). Finally, with surface cooling in the fall, the temperature stratification weakens until the fall overturn, when the lake mixes throughout its depth and temperature is once again isothermal. This annual stratification pattern is much weaker in a run-of-the-river reservoir like Lower Granite Reservoir. In this type of reservoir, the through-flow is vigorous enough to prevent the formation of a strong stratification. This is particularly the case in reservoirs with deep outlets. Such an arrangement leads to a persistent strong current from shallow at the upstream end to deep at the downstream end which counteracts the formation of vertical stratification.

In lakes that form a strong stratification, the period of stratification is important to lake water quality. The thermocline is a strong barrier to mixing: the configuration of heavy, cold water beneath a layer of much warmer and less dense surface water is highly stable. Consequently, there is little mixing between the epilimnion and the hypolimnion, and the two layers can develop distinctly different water quality.

The epilimnion receives sunlight through the water surface and thus supports algae, which require sunlight for photosynthesis and growth. Typically, in the epilimnion, dissolved oxygen is high from atmospheric input and nutrient concentrations low due to algal consumption. In contrast, algal growth is limited in the cooler, darker hypolimnion. Dissolved oxygen must diffuse through the thermocline to reach the hypolimnion and thus is reduced there. Organic matter that settles into the hypolimnion and chemical constituents diffused from the lake bottom increase nutrient and other constituent concentrations, creating a dramatically different water quality than in the epilimnion. In particular, bacterial degradation of organic matter in the hypolimnion may consume oxygen faster than it is replenished and cause the hypolimnion to become anaerobic (without dissolved oxygen). In this case, the water chemistry changes so as to enhance the release of the phosphorus from the lake sediments, further increasing the nutrient load to the lake.

The mixing of water from the hypolimnion to the epilimnion may be an important factor in the water quality of the lake. Mixing across a well-established thermocline or within the hypolimnion is very limited. Indeed, lake modeling studies by Wang and Harleman (1982) show that diffusion across and below the thermocline of stratified lakes is at or near the rate of molecular diffusion. Hypolimnetic mixing may be higher if there is significant flow or other motion within the hypolimnion. For example, there may be flow in a reservoir from stream inflows to a deep dam outlet. Another source of motion is an internal seiche, the back-and-forth oscillation of the thermocline in a type of motion similar to sloshing in a bathtub.

The epilimnion of a lake or reservoir typically is well mixed owing to a nearly constant input of mixing energy from the wind. Occasional strong winds will cause the surface layer to mix into the thermocline and become deeper. This is a far more important mechanism for transport from the hypolimnion to the epilimnion than diffusion across the thermocline.

The separation between epilimnion and hypolimnion is much weaker and less influential in run-of-the-river reservoirs like Lower Granite Canyon. Rather than forming two layers of distinctly different temperature and with little interaction, temperature changes only gradually from surface to bottom in a reservoir and there remains mixing from top to bottom.

Other processes that affect the lake's water quality and trophic status are chemical, physical, and biological reactions involving phosphorus.

7.5.2.3.1.1 Phosphorus

Phosphorus is a non-metallic element, which occurs in nature only as phosphate compounds. Phosphorus as phosphate is one of the major nutrients required for plant nutrition and is essential for life. Phosphorus in the aquatic environment exists in either a particulate phase or a dissolved phase. Particulate phosphorus includes living and dead plankton, precipitates of phosphorus and phosphorus adsorbed to particulates. The dissolved phosphorus includes inorganic phosphorus and organic phosphorus excreted by organisms.

Much of the particulate phosphorus falls to the lake bottom where it joins a large pool of phosphorus in the sediment. Orthophosphate concentrations in the sediment often reach very high levels. Stumm and Stumm-Zollinger (1972) report concentrations in the interstitial water in lake sediments as much as 1000 times greater than typical water column concentrations. Under aerobic conditions, sediment phosphorus is effectively sealed by an oxidized microlayer at the sediment surface. If the water column becomes anaerobic, however, phosphorus in the sediment is released to the water column. For this reason, lake eutrophication may be significantly worsened in lakes in which there develops a strong summer stratification accompanied by oxygen depletion in the hypolimnion. The absence of oxygen also creates adverse conditions for aquatic life and may cause the water to acquire unpleasant taste and odor. Anaerobic conditions are not a concern in Lower Granite Reservoir because it does not form the intense stratification of a typical lake.

In excess of a critical concentration, phosphates stimulate plant growths. Increasing supplies of phosphorus frequently causes increasing plant growths. Such phenomena are associated with a condition of accelerated eutrophication or aging of waters. Generally, it is recognized that phosphorus is not the sole cause of eutrophication but there is substantiating evidence that frequently it is the key element of all of the elements required by freshwater plants, and generally, it is present in the least amount relative to need. Therefore, an increase in phosphorus

allows use of other already present nutrients for plant growth. Further, of all of the elements required for plant growth in the water environment, phosphorus is the most easily controlled by man.

Phosphates enter waterways from several different sources. The predominant point sources of phosphorus in waterbodies include the use of phosphate detergents and other domestic products, sewage treatment plants, industrial discharges (e.g., potato processing), stormwater runoff, and cattle feed lots. Crop, forest, idle, and urban land contribute varying amounts of phosphorus-diffused sources in drainage to watercourses. This drainage may be surface runoff of rainfall, effluent from tile lines, or return flow from irrigation. Concentrations of domestic duck or wild duck populations, tree leaves, and fallout from the atmosphere are also contributing sources.

Evidence indicates that: (1) high phosphorus concentrations are associated with accelerated eutrophication of waters, when other growth-promoting factors are present; (2) aquatic plant problems develop in reservoirs and other standing waters at phosphorus values lower than those critical in flowing streams; (3) reservoirs and lakes collect phosphates from influent streams and store a portion of them within consolidated sediments, thus serving as a phosphate sink; and (4) phosphorus concentrations critical to noxious plant growth vary and nuisance growths may result from a particular concentration of phosphate in one geographical area but not in another. The amount or percentage of inflowing nutrients that may be retained by a lake or reservoir is variable and will depend upon: (1) the nutrient loading to the lake or reservoir; (2) the volume of the euphotic zone; (3) the extent of biological activities; (4) the detention time within a lake basin or the time available for biological activities; and (5) the level of discharge from the lake or of the penstock from the reservoir.

7.5.2.3.1.2 Nitrogen

Two gases (molecular nitrogen and nitrous oxide) and five forms of nongaseous, combined nitrogen (amino and amide groups, ammonium, nitrite, and nitrate) are important in the nitrogen cycle. The amino and amide groups are found in soil organic matter and as constituents of plant and animal protein. The ammonium ion either is released from proteinaceous organic matter and urea or is synthesized in industrial processes involving atmospheric nitrogen fixation. The nitrite ion is formed from the nitrate or the ammonium ions by certain microorganisms found in soil, water, sewage, and the digestive tract. The nitrate ion is formed by the complete oxidation of ammonium ions by soil or water microorganisms; nitrite is an intermediate product of this nitrification process. In oxygenated natural water systems nitrite is rapidly oxidized to nitrate. Growing plants assimilate nitrate or ammonium ions and convert them to protein. A process known as denitrification takes place when nitrate-containing soils become anaerobic and the conversion to nitrite, molecular nitrogen, or nitrous oxide occurs. Ammonium ions may also be produced in some circumstances.

Among the major point sources of nitrogen entry into waterbodies are municipal and industrial wastewaters, septic tanks, and feed lot discharges. Diffuse sources of nitrogen include farm-site fertilizer and animal wastes, lawn fertilizer, leachate from waste disposal in dumps or sanitary landfills, atmospheric fallout, nitric oxide and nitrite discharges from automobile exhausts and other combustion processes, and losses from natural sources such mineralization of soil organic matter (NAS, 1972). Water reuse systems in some fish hatcheries employ a nitrification process for ammonia reduction; this may result in exposure of the hatchery fish to elevated levels of nitrite (Russo et al., 1974). Wise and Johnson (2011) assessed surface-water nutrient conditions

and sources in the Pacific Northwest and found that annual nutrient yields were greater on the west side of the Cascade Range than the east side. For total nitrogen stream load, forest land was generally the largest source. The combined input from agriculture, point sources, and developed land was responsible for most of the nutrient load discharged.

7.5.2.3.2 Environmental Baseline

The concentrations of phosphorus and nitrogen were measured in several forms (total phosphorus, orthophosphate, nitrate nitrogen, nitrite nitrogen, ammonia nitrogen, and total Kjeldahl nitrogen) at upstream locations in the Snake and Clearwater Rivers and at five downstream locations in the Snake River as part of the Receiving Water Studies conducted by Clearwater from 1997 through 2002. In 2005 and 2006, Endangered Species Act Tier 1 studies were undertaken to evaluate effluent and natural waters above and below the facility. The following paragraphs describe the findings from the Receiving Water Studies and ESA Tier 1 studies for each form of phosphorus and nitrogen.

During 1997 through 2002, the concentration of total phosphorus (TP) at the upstream monitoring location on the Snake River (Site 2) ranged from non-detect (one half the detection limit = 8 µg/L) to 130 µg/L with a mean of 60.1 µg/L. At the upstream location on the Clearwater River (Site 1), the mean TP concentration ranged from non-detect (one half the detection limit = 8 µg/L) to 98 µg/L, with a mean of 32 µg/L. At the five downstream locations, the mean TP concentration did not vary significantly from the downstream monitoring location closest to the diffuser (Site 3) to the downstream monitoring location furthest from the diffuser (Site 7), with mean TP concentrations ranging from 51.3 µg/L at Site 4 to 57.6 µg/L at Site 6. However, maximum concentrations increased with downstream distance from the diffuser (106 µg/L at Site 3 to 156 µg/L at Site 7).

In 2005 and 2006, total phosphorus concentrations were measured during ESA Tier 1 studies. TP concentrations ranged from non-detect to 0.13 mg/L in 2005 and non-detect to 0.16 mg/L in 2006. Measurements were taken for both shallow and mid-depth surface water. Among upstream samples, total phosphorous ranged from non-detect to 0.12 mg/L in 2005 and non-detect to 0.10 mg/L in 2006. Among downstream samples, the measured range was 0.02 to 0.13 mg/L in 2005 and 0.02 to 0.16 mg/L in 2006. In both years, concentrations in the Snake River tended to increase through September and then decrease in October, suggesting either a seasonal fluctuation or the existence of a non-point source contribution to phosphorus (AMEC 2006, AMEC 2007). Figure 7-20 through Figure 7-23 show TP concentrations measured during the studies.

Washington Department of Ecology maintains a monitoring station on the Snake River at Interstate Bridge (Site 35A150) where various parameters are measured monthly. Included in these analyses are data for phosphorus and nitrogen. Total phosphorus measurements from 1990 to 1999 and 2007 to 2015 range from 0.01 (non-detect) to 0.2 mg/L, with a mean of 0.06 mg/L. Dissolved phosphorus concentrations from 1990 to 2015 range from 0.003 (non-detect) to 0.095 mg/L, with a mean of 0.037 mg/L. Nitrate + Nitrite values range from 0.02 to 3.35 mg/L with a mean of 0.61 mg/L for the time period of 1990 to 2015.

Falter (2001) has evaluated historical concentrations of nutrients upstream and downstream of the confluence of the Snake and Clearwater Rivers. In the Snake River upstream of the confluence near Anatone, Washington, where the river is free-flowing, the mean concentration of

TP from June to August was 40 µg/L in the years 1975 through 1977 and was 46 µg/L in the years 1997 through 1998. Falter (2001) also describes mean concentrations of TP in "the impounded reach" of 35 µg/L during 1975 through 1977, and 37 µg/L during 1997 through 1998.

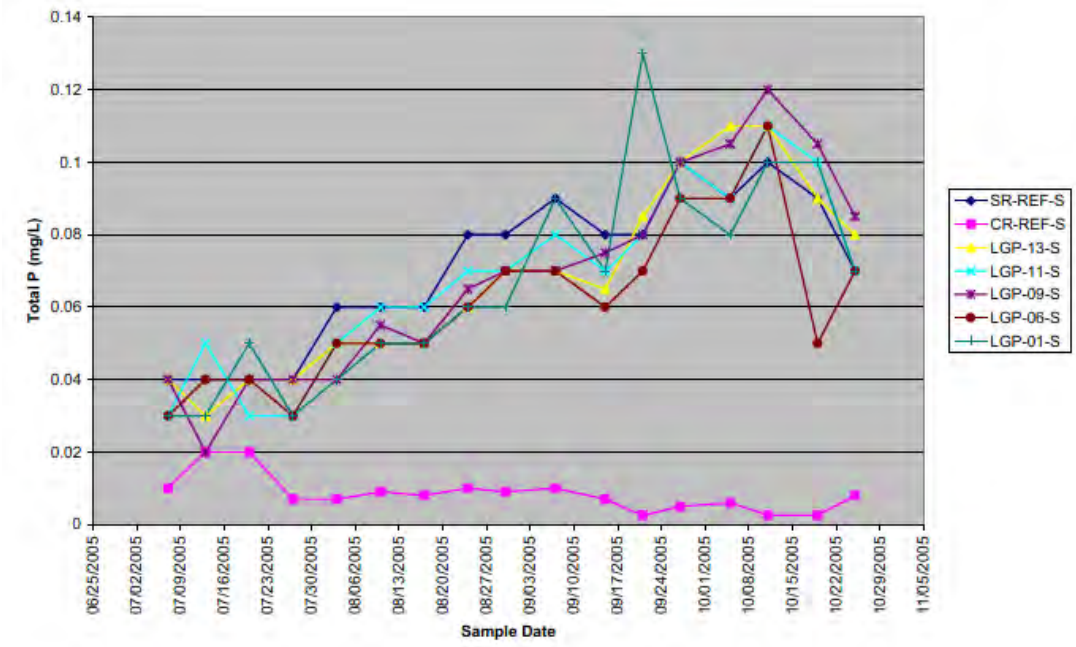


Figure 7-20: Total Phosphorus in Shallow Surface Water measured during ESA Tier 1 Study (AMEC 2006).

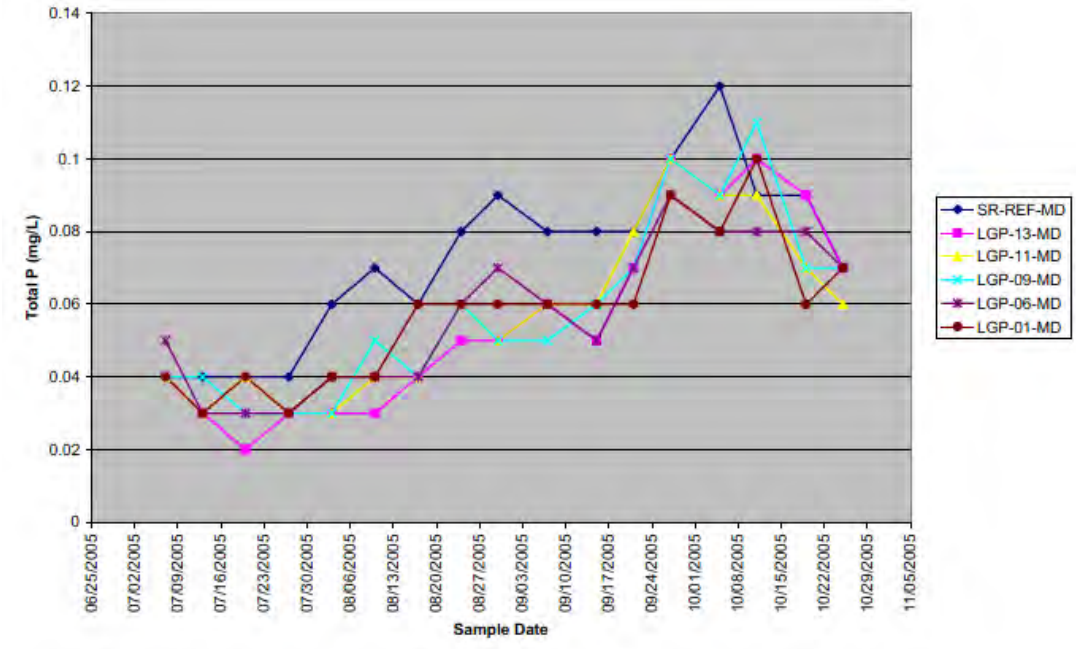


Figure 7-21: Total Phosphorus in Mid-Depth Surface Water measured during ESA Tier 1 Study (AMEC 2006).

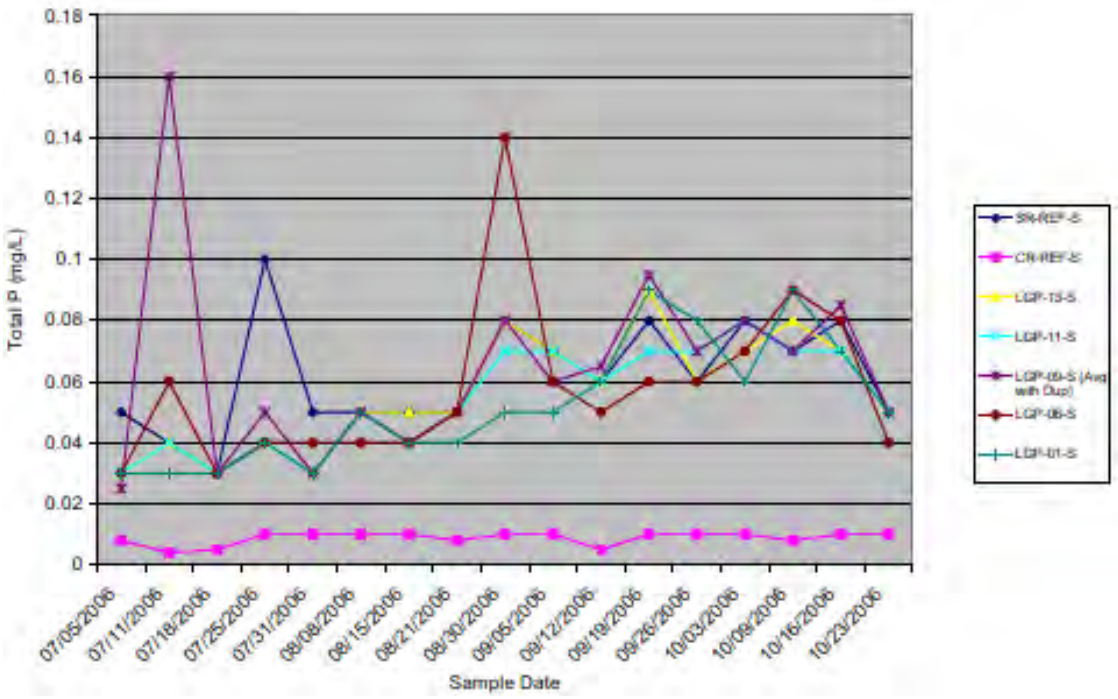


Figure 7-22: Total Phosphorus in Shallow Surface Water measured during ESA Tier 1 Study (AMEC 2007).

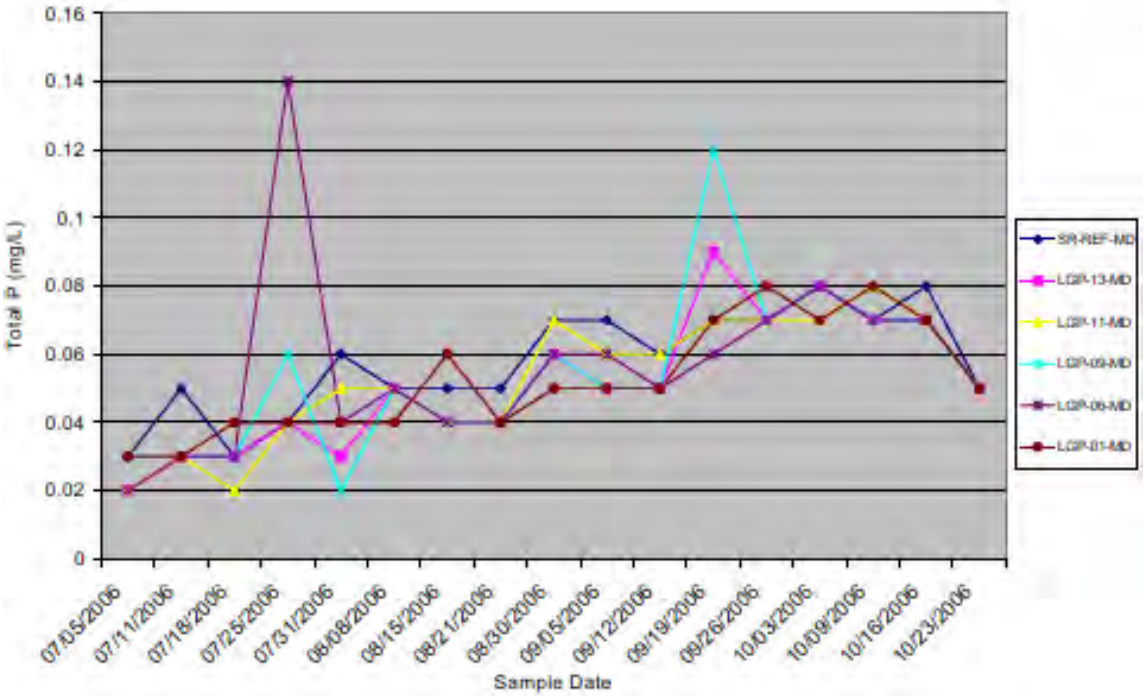


Figure 7-23: Total Phosphorus in Mid-Depth Surface Water measured during ESA Tier 1 Study (AMEC 2007).

The Washington State Department of Ecology monitors water quality in the Snake River from monitoring station 35A150, located at the Washington-Idaho Interstate Bridge on U.S. Highway 12 (river mile 139.6). Finalized data are available from 1962 through September of 2015. Total

phosphorus concentrations were measured by method EPA200.8M from 2005 through September of 2007 and by method EPA365.1 from October of 2007 through 2015. Nitrate/nitrite nitrogen concentrations were measured by method SM4500NO3I. From January 2005 to September 2015, nitrate/nitrite nitrogen concentrations ranged from 0.024 to 3.350 mg/L, with an average of 0.632 mg/L. Over the same time period, total phosphorus ranged from 0.005 to 0.159 mg/L, with an average of 0.057 mg/L. Total phosphorus and nitrate/nitrite nitrogen concentrations from the years 2005 to 2015 are summarized below in Table 7-25.

In 2005 and 2006, nitrate/nitrite nitrogen concentrations measured during ESA Tier 1 studies. Concentrations ranged from non-detect to 0.62 mg/L in 2005 and 0.019 to 0.61 mg/L in 2006. Measurements were taken for both shallow and mid-depth surface water. In both years, concentrations generally increased over the duration of the monitoring period. The concentration profile in the Snake River samples suggested either a strong seasonal influence or the existence of a non-point source contribution (AMEC 2006, AMEC 2007). Figure 7-24 through Figure 7-27 show nitrate/nitrite nitrogen concentrations measured during the studies.

Table 7-25: Nitrate/Nitrite Nitrogen and Total Phosphorus Data from the Washington State Department of Ecology Monitoring Station 35A150, January 2005 – November 2018.

Year	Nitrate/Nitrite Nitrogen			Total Phosphorus		
	Min (mg/L)	Mean (mg/L)	Max (mg/L)	Min (mg/L)	Mean (mg/L)	Max (mg/L)
2005	0.113	0.593	1.070	0.018	0.062	0.109
2006	0.105	0.566	1.150	0.037	0.071	0.159
2007	0.148	0.611	1.050	0.017	0.049	0.082
2008	0.122	0.830	3.350	0.005	0.050	0.087
2009	0.146	0.655	1.230	0.049	0.071	0.095
2010	0.212	0.682	1.240	0.033	0.056	0.126
2011	0.097	0.577	1.160	0.016	0.063	0.107
2012	0.118	0.573	1.110	0.022	0.048	0.076
2013	0.158	0.679	1.370	0.027	0.055	0.081
2014	0.024	0.571	1.190	0.025	0.051	0.105
2015	0.242	0.609	1.160	0.027	0.043	0.071
2016	0.158	0.669	1.310	0.037	0.054	0.088
2017	0.109	0.661	1.420	0.026	0.058	0.152
2018	0.145	0.592	1.350	0.021	0.049	0.100
2005-2015	0.024	0.632	3.350	0.005	0.057	0.159

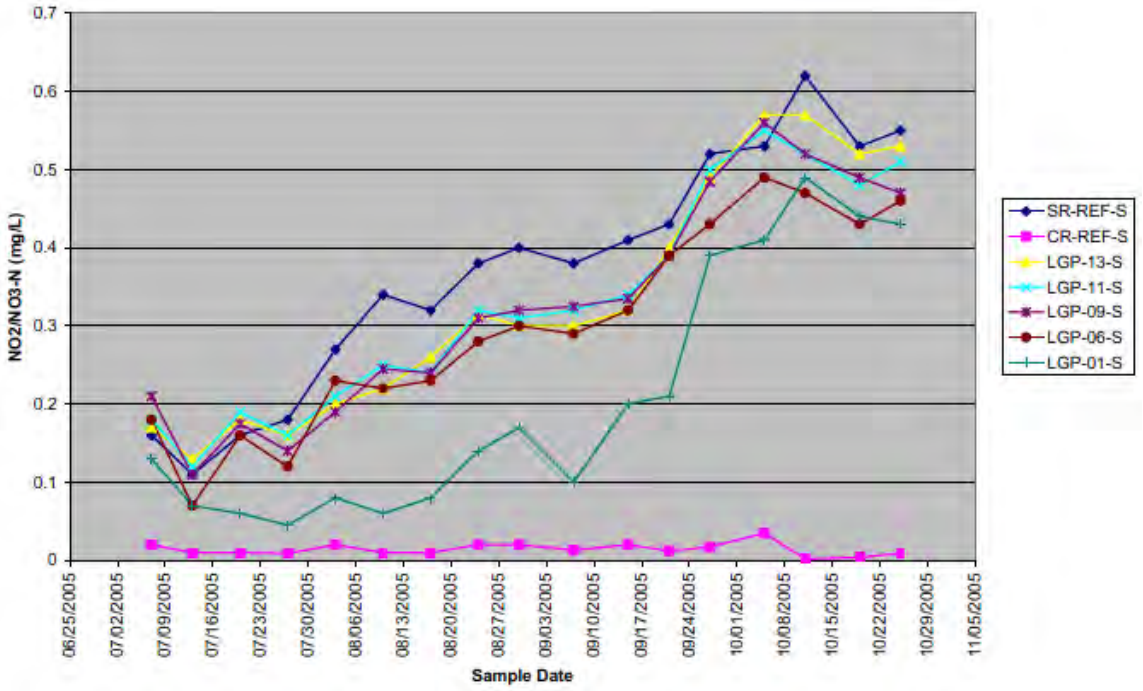


Figure 7-24: Nitrate/Nitrite Nitrogen in Shallow Surface Water measured during ESA Tier 1 Study (AMEC 2006).

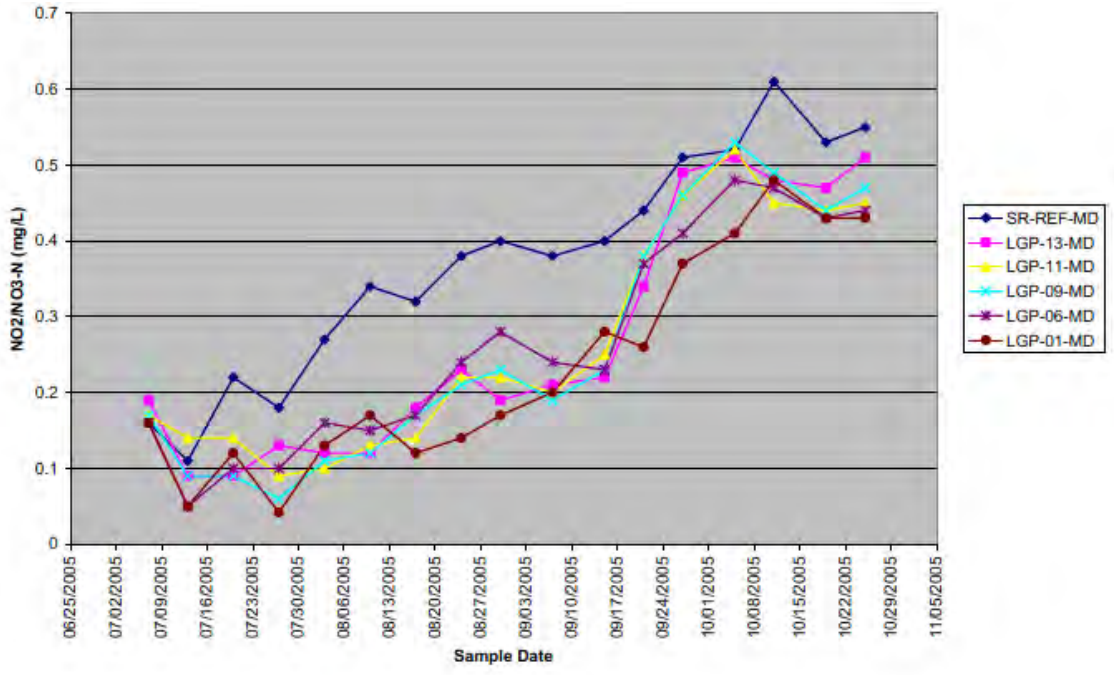


Figure 7-25: Nitrate/Nitrite Nitrogen in Mid-Depth Surface Water measured during ESA Tier 1 Study (AMEC 2006).

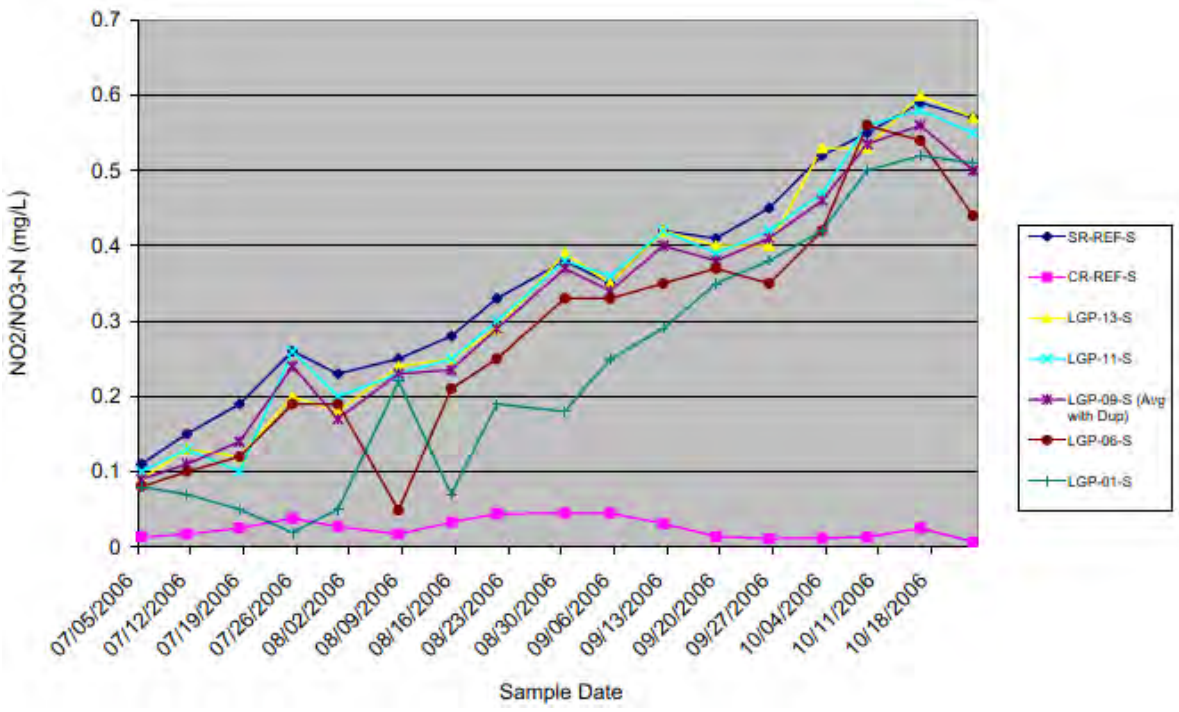


Figure 7-26: Nitrate/Nitrite Nitrogen in Shallow Surface Water measured during ESA Tier 1 Study (AMEC 2007).

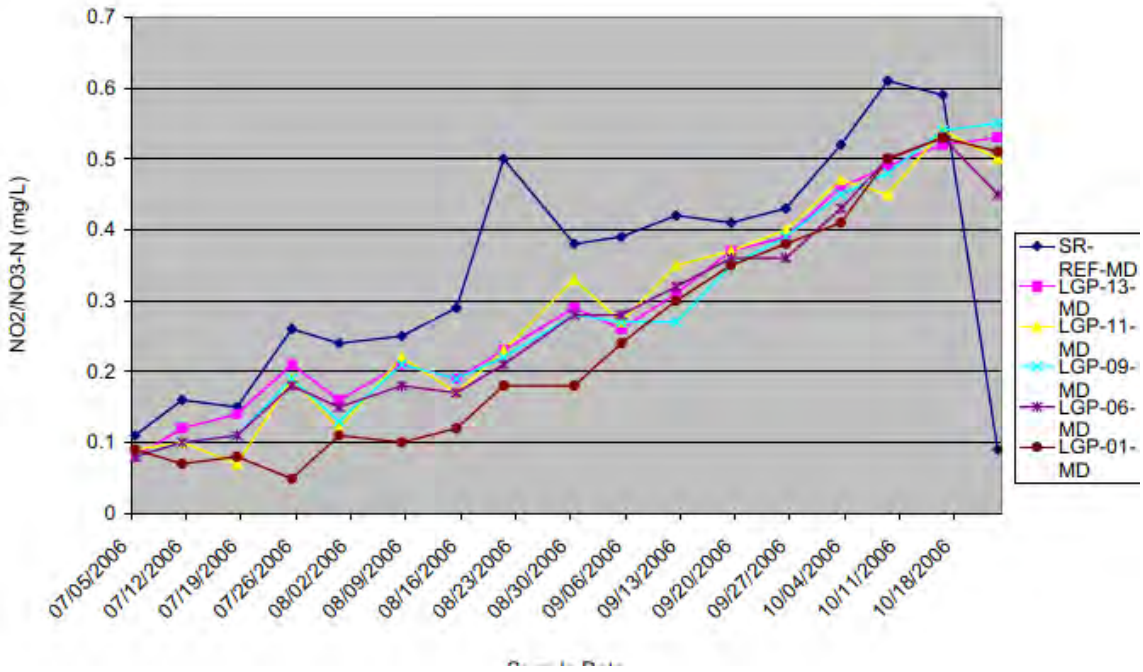


Figure 7-27: Nitrate/Nitrite Nitrogen in Mid-Depth Surface Water measured during ESA Tier 1 Study (AMEC 2007).

7.5.2.3.3 Water Quality Standard

The current Washington water quality standards do not specifically address nutrients; however, Idaho has a narrative criterion that requires surface waters to be free from excess nutrients that can cause visible slime growths or other nuisance aquatic growths impairing designated beneficial uses. Neither Idaho nor Washington has numeric criteria for phosphorus or nitrogen in their water quality standards.

7.5.2.3.4 Effluent Limitation

The 2019 draft permit does not limit nutrients because there is not enough information to determine if the nutrient contribution from the discharge has the reasonable potential to violate water quality standards.

7.5.2.3.5 Benchmark

Because the effect of nutrients is eutrophication in the Lower Granite Reservoir, rather than toxicological effects to aquatic and terrestrial species, the benchmarks for this parameter focus on levels that protect the LGR from eutrophication rather than toxicity to species. However, EPA does have some information regarding nitrate and nitrite toxicity to aquatic life and has presented that information in this discussion.

7.5.2.3.5.1 Nitrates/Nitrites

Westin (1974) determined that the respective 96-hour and 7-day LC50 values for Chinook salmon, *Oncorhynchus tshawytscha*, were 1,310 and 1,080 mg/L nitrate nitrogen in fresh water. For fingerling rainbow trout, *Salmo gairdneri*, the respective 96-hour and 7-day LC50 values were 1,360 and 1,060 mg/L nitrate nitrogen in fresh water. Trama (1954) reported that the 96-hour LC50 for bluegills, *Lepomis macrochirus*, at 20˚C was 2,000 mg/L nitrate nitrogen (sodium nitrate) and 420 mg/L nitrate nitrogen (potassium nitrate).

The 96-hour and 7-day LC50 values for Chinook salmon, *Oncorhynchus tshawytscha*, were found to be 0.9 and 0.7 mg/L nitrite nitrogen in fresh water (Westin, 1974). Smith and Williams (1974) tested the effects of nitrite nitrogen and observed that yearling rainbow trout, *Salmo gairdneri*, suffered a 55 percent mortality after 24 hours at 0.55 mg/L; fingerling rainbow trout suffered a 50 percent mortality after 24 hours of exposure at 1.6 mg/L; and Chinook salmon, *Oncorhynchus tshawytscha*, suffered a 40 percent mortality with 24 hours at 0.5 mg/L. There were no mortalities among rainbow trout exposed to 0.15 mg/L nitrite nitrogen for 48 hours. These data indicate that salmonids are more sensitive to nitrite toxicity than are other fish species, e.g., minnow, *Phoxinus*, that suffered a 50 percent mortality within 1.5 hours of exposure to 2,030 mg/L nitrite nitrogen, but required 14 days of exposure for mortality to occur at 10 mg/L (Klingler, 1957), and carp, *Cyprinus carpio*, when raised in a water reuse system, tolerated up to 1.8 mg/L nitrite nitrogen (Saeki, 1965).

Russo et al. (1974) performed flow-through nitrite bioassays in hard water (hardness = 199 mg/L $CaCO_3$; alkalinity = 176 mg/L $CaCO_3$; pH = 7.9) on rainbow trout, *Salmo gairdneri*, of four different sizes, and obtained 96-hour LC50 values ranging from 0.19 to 0.39 mg/L nitrite nitrogen. Duplicate bioassays on 12-gram rainbow trout were continued long enough for their toxicity curves to level off, and asymptotic LC50 concentrations of 0.14 and 0.15 mg/L were reached in 8 days; on day 19, additional mortalities occurred. For 2-gram rainbow trout, the minimum tested level of nitrite nitrogen at which no mortalities were observed after 10 days was

0.14 mg/L; for the 235-gram trout, the minimum level with no mortality after 10 days was 0.06 mg/L.

The lowest direct toxicity concentration (LC50) for nitrate nitrogen is 1,310 mg/L in Chinook salmon (Westin, 1974). The lowest indirect toxicity concentration (LOEC) was 420 mg/L nitrate nitrogen (potassium nitrate) in bluegill (Trama, 1954). Since neither a LOEC nor a NOEC were cited for direct toxicity, a NOEC was established by applying two safety factors (10 for the ACR and 10 for the LOEC to NOEC) would generate a value of 13.1 mg/L for this study. **This BE uses 13.1 mg/L nitrate nitrogen as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

The lowest direct toxicity concentration (NOEC) for nitrite nitrogen is 0.06 mg/L in rainbow trout (Russo et al., 1974). **This BE uses 0.06 mg/L nitrite nitrogen as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2.3.6 Effects Analysis

7.5.2.3.6.1 Direct Effects

Pulp and paper industry wastewaters are typically deficient in nitrogen and phosphorus and cannot be effectively treated using conventional biological treatment processes without the addition of supplementary nutrients, such as urea and phosphoric acid. The microorganisms that consume the dissolved organic constituents in pulp and paper wastewater (typically lignin and other wood-based molecules) require a nutrient source. Supplementation is a difficult step to manage efficiently, requiring extensive post-treatment monitoring and some degree of overdosing to ensure sufficient nutrient demand under all conditions. As a result, treated wastewaters usually contain excess amounts of both nutrients, leading to potential impacts on the receiving waters such as eutrophication.

The mean nitrite + nitrate in effluent from 2005 through 2016 is about 0.05 mg/L, with a maximum of 0.49. Comparing this to the upstream nitrate/nitrite of 0.025 mg/L in the Clearwater River (maximum = 0.035 mg/L) and 0.35 mg/L in the Snake River (maximum 0.59 mg/L), it appears that the discharge's nitrite/nitrate contribution to downstream is insignificant. Therefore, EPA has concluded that the discharge of nutrients **may affect, but are not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2.3.6.2 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the nutrients in the discharge is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.2.4 Whole Effluent Toxicity (WET)

7.5.2.4.1 Summary

EPA's whole effluent toxicity (WET) approach to toxics control for the protection of aquatic life involves the use of acute and chronic toxicity tests to measure the toxicity of wastewaters. Whole effluent toxicity is a useful parameter for assessing and protecting against impacts upon water quality and designated uses caused by the aggregate toxic effect of the discharge of pollutants (Wang, 1990). Whole effluent toxicity tests employ the use of standardized, surrogate freshwater plants, invertebrates, and vertebrates. EPA has published extensive written protocols listing numerous freshwater species for toxicity testing.

It is important to recognize that toxicity caused by contaminants in the effluent, as measured by the whole effluent toxicity tests, is only one of many influences that determine the health of a biological community. Impact from toxics would only be suspected where effluent concentrations, after dilution, are at or above the toxicity effect concentrations. Influences from substrate differences and physical conditions, such as dissolved oxygen, temperature, channelization, flooding and weather cycles, also can affect the biological community adversely. These other types of influences may be better evaluated by using a bioassessment approach.

The value of the toxicity test is its ability to assess the impact of discharged toxicants independent of effects from other factors. This allows the identification and ability to control the portion of the impact caused by the discharge. Biological, physical, and chemical factors of the community can influence the actual effects that effluent toxicity may cause in the receiving water.

An acute toxicity test is defined as a test of 96-hours or less in duration in which lethality is the measured endpoint. A chronic toxicity test is defined as a long-term test in which sublethal effects, such as fertilization, growth, and reproduction, are usually measured, in addition to lethality. Traditionally, chronic tests are full life-cycle tests or a shortened test of about 30 days known as an early life stage test. However, the duration of most of the EPA chronic toxicity tests have been shortened to 7 days by focusing on the most sensitive life-cycle stages. For this reason, the EPA chronic tests are called short-term chronic tests.

In a laboratory acute toxicity test, an effluent sample is collected, diluted, and placed in test chambers with the chosen species. After 24, 48, 72, and 96 hours, the number of live organisms remaining in each test concentration and in a control is recorded. At test termination, the number of dead organisms is recorded and an LC50 is calculated.

In a laboratory chronic toxicity test, an effluent sample is collected, diluted, and placed in test chambers. The test organisms are placed in these test chambers for specified periods of time. At various times during the exposure period, the organisms in each chamber are observed. At test termination, the lowest effluent concentration that causes a significant adverse impact on the most sensitive endpoint for that test is calculated (this endpoint can be mortality, reduced fertilization, lower fecundity, reduced growth, etc.).

Dilution water is an important part of toxicity testing. Dilution water may either be standard laboratory water and/or the receiving water. The receiving water is used to dilute the effluent in some cases because it more closely simulates effluent and receiving water interactions. The EPA methods manuals recommend six dilutions, including the control to determine the magnitude of

toxicity. An example of a dilution series used in whole effluent toxicity tests is 100, 50, 25, 12.5 and 6.25 percent effluent, and a control.

Quality control and quality assurance are an integral part of whole effluent toxicity testing. Use of a standard control water and a reference toxicant test are both recommended to ensure quality assurance in chronic testing. It is important to understand that each of the chronic tests has minimum criteria of acceptability for each endpoint that is measured in the controls (i.e., 80 percent survival and minimum criteria for growth, reproduction, and fertilization). The acute tests also have criteria of acceptability measured in the controls.

EPA conducted the Complex Effluent Toxicity Testing Program (CETTP) that examined sites in both freshwater and saltwater systems to investigate whether or not an evaluation of effluent toxicity, when adequately related to receiving water conditions (i.e., temperature, pH, salinity), can give a valid assessment of receiving system impacts on waters that support aquatic biota (Bergman et al., 1985; USEPA, 1987; Schimmel et al., 1989; and Schimmel et al., 1989a). EPA evaluated the results of these studies (Dickson et al., 1991) that, when linked together, clearly show that if toxicity is present after considering dilution, impact will also be present. Impact correlations will be higher where higher toxic impact occurs and lower where impacts are expected to be minimal. Such a response is expected given the complexity of ecosystems and that biological communities and species have different sensitivities to toxicants and many respond differently. Also, higher river dilution will reduce the potential instream impact from effluent toxicity.

Even though the CETTP study sites were randomly chosen and were not selected to represent a statistically valid sampling of all types of waterbodies in the United States, EPA believes that it is reasonable to assume in the absence of data showing otherwise that this relationship is basically independent of the waterbody type. The CETTP studies also did not investigate replication of results over time because toxicity results cannot be expected to be replicated over time in waters where river flow and other time-variant factors change the degree of ambient toxicity.

Clearwater's effluent has been required by the 2005 NPDES permit to conduct whole effluent toxicity testing. It is important to note that these studies used moderately hard reconstituted water for dilution rather than receiving water. Therefore, the studies do not account for any potential toxic effects due to background sources.

7.5.2.4.2 Environmental Baseline

There have been no toxicity tests conducted on the ambient receiving waters; therefore, the environmental baseline is unknown.

7.5.2.4.3 Water Quality Standard

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."

There are no national criteria for whole effluent toxicity. Where a state criterion for toxicity is expressed as a narrative statement, EPA uses recommendations in the *Technical Support Document for Water Quality-based Toxics Control* (USEPA, 1991a) to develop effluent limits

protective of the narrative criteria. EPA's recommended magnitudes for WET are 1 TU_c and 0.3 TU_a for the chronic and acute criteria, respectively. TU are toxicity units where the toxicity units are defined as the ratio of the exposure concentration to the 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 (e.g. LC50). TU_c is equal to the ratio of undiluted effluent (100% effluent) to the IC25 (where 25% of the test organisms are affected) determined through WET testing. TU_a equals the ratio of undiluted effluent (100% effluent) to the LC50, the concentration that results in 50% mortality as determined through WET testing.

7.5.2.4.4 Effluent Limitation

The 2019 draft permit does not establish effluent limitations for whole effluent toxicity because the WET data collected under the current permit shows that there is not reasonable potential to violate water quality standards (see calculations in Appendix B). However, EPA believes that it is important to have current data when reissuing the permit in the future; therefore, the draft permit requires Clearwater to conduct twice yearly chronic whole effluent toxicity testing using water flea (*Ceriodaphnia dubia*), fathead minnow (*Pimephales promelas*), and green alga (*Selenastrum capricornutum*) and twice yearly acute whole effluent toxicity testing using rainbow trout (*Oncorhynchus mykiss*). These data will be analyzed during the permit reissuance process to determine whether a limit should be included in future permits.

Table 7-26 presents the available results of the quarterly toxicity tests conducted during the period from 2005 to 2009. These tests evaluated effects on growth, reproduction, and survival using *Ceriodaphnia dubia* and Fathead minnows (*Pimephales promelas*). Five concentrations of effluent (1, 3, 10, 30, and 100 % effluent) were tested against a control comprised of 100 % dilution water. With the one exception of the fourth quarterly 2005 test, no TUc value exceeded 1 for the *P. promelas* whole effluent toxicity tests conducted during 2005 through the first quarter of 2009. However, all *C. dubia* tests conducted during this period had TUc values greater than 1 except for the first quarter 2009 test.

Table 7-26: Summary of Results of Quarterly Toxicity Tests Conducted Using Effluent from Clearwater's Lewiston Mill

Year	Quarter	Ceriodaphnia dubia	Pimephales promelas
		TU_c	**TU_c**
2005	3rd Quarter	3.3	1
	4th Quarter	10	10
2006	1st Quarter	10	1
	2nd Quarter	10	1
	3rd Quarter	3.	1
	4th Quarter	3.3	1
2007	1st Quarter	3.3	1
2008	2nd Quarter	10	1
	3rd Quarter	3.3	1
	4th Quarter	3.3	1
2009	1st Quarter	1	1
Notes: TU_c – Chronic Toxicity Units.			

7.5.2.4.5 Toxicity Benchmark

Whole effluent toxicity is facility-specific so no data from literature can help to evaluate the level of protection provided by the permit limits. Therefore, this BE uses EPA's recommended magnitude of **1 TU_c for WET as a direct toxicity benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead and an indirect toxicity benchmark for prey species.**

7.5.2.4.6 Effects Analysis

Whole effluent toxicity addresses the cumulative impact due to the toxicity of the whole effluent, including all the parameters for which the permit provides limits and does not limit. The potential toxicity of BKME to aquatic species has been studied extensively in recent years using effluent from many pulp and paper mills, including Clearwater's Mill in Lewiston. A thorough literature review was conducted to identify sources of information regarding the potential toxicity of BKME. Appendix J describes the process used to select toxicity studies for evaluation in the BE, based on the characteristics of the mill used as the source of the effluent evaluated in the study. The following paragraphs describe the potential toxicity observed in toxicity studies conducted using whole effluent from other mills and using whole effluent from Clearwater's Lewiston Mill.

Acute toxicity has been observed in studies in which daphnids were exposed to BKME. The results of several such studies indicate that a range of BKME concentrations in effluent have been associated with acute toxicity to daphnids. EC_{50} values from these studies range from 37% BKME to 100% BKME. The minimum EC50 of 37% was from a study in which *Daphnia magna* were exposed to effluent from an elemental chlorine free (ECF) bleaching process (Ahtiainen et al., 1996).

Both increases in enzyme activity of fish (enzyme induction) and decreases in enzyme activity of fish (enzyme inhibition) have been associated with exposure to BKME. Production of the mixed function oxygenase (MFO) enzyme 7-ethoxyresorufin-o-deethylase, or EROD, has been studied most frequently (Hodson et al., 1996). MFO induction may be correlated with other changes in physiological function, such as alterations in steroid hormone concentrations and reproductive abnormalities (Munkittrick et al., 1998).

The MFO enzyme system is involved in the metabolism of a variety of compounds, including dioxins, dioxin-like PCBs, and PAHs, and can be induced not only by TCDD in BKME, but by a variety of organic compounds, including PAH-related natural products from wood as well. Both field and laboratory studies have shown that the replacement of chlorine in the bleaching of kraft pulp by chlorine dioxide or non-chlorine containing compounds (such as peroxide) does not appear to eliminate EROD induction in exposed fish (Martel et al., 1994; Munkittrick et al., 1992; Haley et al., 1995). In recent studies, several compounds have been identified as potentially responsible for MFO induction in pulp mill effluents that have undergone chlorine substitution and do not have high dioxin levels. The active compounds are typically similar to moderately hydrophobic, planar, aromatic PAHs, and include chlorinated pterostilbene, which belongs to a group of substances occurring naturally in coniferous trees (Burnison et al., 1999), juvabione and dehydrojuvabione, natural extracts from balsam fir (Martel et al., 1997), and retene, a derivative of resin acids from wood that is found in mill effluent (Fragoso et al., 1998). These results suggest that the chloride replacement process may not eliminate the ability of this effluent to induce MFO enzymes, although this process has been shown to reduce potential

toxicity to fish, as evidenced by the results of the pre-conversion and post-conversion Experimental Streams Studies discussed later in this section and in Appendix J.

Reported threshold concentrations of effluent necessary for MFO induction in salmonids are variable, which is not surprising considering that the potency of the effluent may vary depending on test methods and mill processes. In one study, a concentration of 1.5 % BKME was associated with a three-fold increase in EROD activity in fingerling Chinook (Campbell et al., 1996). Although the authors did not specify whether the increase was statistically significant relative to controls, the results are reported as a "dose-dependent response." Wilson et al. (2001) exposed juvenile Chinook salmon for 28 days to treated effluent from an elemental chlorine free mill and found significantly increased EROD activity at effluent concentrations of 2% and above, as well as increased hepatic cytochrome P450 1A (CYP1A) protein at concentrations higher than 2% effluent. Williams et al. (1996) report a threshold concentration for MFO induction in 6-day exposures of 1.12% for effluent from a plant with oxygen delignification, chlorine substitution, and secondary treatment. Martel and Kovacs (1997) exposed rainbow trout to effluent concentrations of 1, 5 and 10% effluent from 13 mills employing both primary and secondary treatment and a variety of pulping techniques to determine their potential to induce mixed function oxygenase activity. Of the seven BKME effluents receiving secondary treatment, five did not produce statistically significant EROD activity at any of the test concentrations. The two effluents that did produce statistically significant EROD activity did so only at the highest effluent concentration (10%).

The lowest concentration of BKME associated with changes in EROD activity among other fish species is 0.008 % BKME (Priha, 1996), using salmonids and effluent from a combined elemental chlorine free (ECF) and totally chlorine free (TCF) bleaching process. Priha (1996) also reports the same minimum BKME concentration (0.008%) is associated with inhibition of EROD activity, using rainbow trout and effluent from an ECF bleaching process. This concentration of BKME is substantially lower than the next higher concentration resulting in changes in EROD activity (0.2% BKME from an ECF bleaching process resulting in EROD inhibition (Priha, 1996).

Among listed species, one study reported only a No-Effect Level of 4% BKME for changes in growth of juvenile Chinook salmon (Owens, 1991). A Low-Effect Level was not reported. Servizi et al. (1993) found no effects on growth in fingerling Chinook salmon that were exposed to 1.5% biologically-treated BKME in the laboratory for 144 d in freshwater, and then held in clean seawater for 66 days.

Among fish species that may be potential prey of listed salmonid species, studies report a range of BKME concentrations (4% to 100%) associated with changes in growth. One study reports that a concentration of 5% BKME from an ECF process was associated with growth changes in Zebrafish larva (Ahtiainen, et al. 1996). Other studies report significant differences in growth in three species (bluegill, channel catfish and largemouth bass) exposed to 4% BKME, compared to controls (Borton et al., 1996). The lowest concentration of BKME associated with changes in growth was 4% BKME.

Among invertebrate and macrophyte species, both growth stimulation and growth inhibition were observed in response to exposure to BKME. Growth stimulation was observed at lower BKME concentrations than growth inhibition. In studies examining growth stimulation in alga species, a broad range of BKME concentrations (0.25% to 100%) were found to cause increased

algal growth. The lowest concentration of BKME associated with growth stimulation was a LOAEL of 0.25%, reported in a study evaluating periphyton exposure to effluent from an ECF bleaching process (Dube and Culp, 1996). An EC50 of 18% BKME was identified for inhibition of growth of *S. capricornutum* using effluent from an ECF process (Ahtiainen et al., 1996).

Among invertebrate species, exposure to BKME was associated with growth stimulation of both mayflies and chironomids at concentrations of 1% to 100%. The lowest concentration of BKME associated with growth stimulation for each species was a LOAEL of 1% (Dube and Culp, 1996; Lowell et al., 1996). Podemski and Culp (1996), Culp et al. (2000), and Podemski (2000) found that BKME concentrations equivalent to levels in the Athabasca River, Alberta, Canada (~one percent BKME) did not cause measurable toxicity for most species (although Podemski, 2000, reported increased mortality for some first instar stages of some mayfly species), but produced enrichment effects, including increases in periphyton and insect biomass, increased growth of stonefly and mayfly species, and invertebrate abundance.

Sibley et al. (2000) assessed the impact of BKME on the distribution and composition of benthic communities at Jackfish Bay, Lake Superior, where effluent concentrations have been estimated in the four percent range (Munkittrick et al., 1994). At this site, discharge of pulp mill effluents led to changes in the benthic community from a typical oligotrophic system with a large variety of benthic species to a system with sediments high in organic matter, dominated by oligochaetes. They also found a small nutrient enriched zone characterized by an abundant and diverse benthic community comprised of benthic harpacticoids, chironomids, and oligochaetes. In a study in Central Portugal at sites receiving BKME that had undergone secondary treatment, Ferreira et al. (2002) observed reduced species diversity of diatoms and invertebrates, and decreased invertebrate densities, at effluent-affected sites, especially in summer when water temperatures were higher, but effluent concentrations associated with these changes were not reported.

In field studies where environmental BKME concentrations were in the 1.7% to 5.2% range for mills using secondary treatment (Munkittrick et al., 1994), changes in 17-ß-estradiol, testosterone, and 11-ketotestosterone were observed. In other studies with effluents from pulp mills with a variety of different processes, including conventional bleaching and chlorine dioxide substitution, rainbow trout embryo survival in a short-term (7-day) test was unaffected (Kovacs and Megraw, 1996).

Among other fish species, BKME concentrations ranging from 1% to 100% were reported as LOECs in studies examining egg hatchability. The lowest LOEC from these studies was 1%, from a study in which zebrafish eggs were exposed to effluent from a TCF process (Ahtiainen et al., 1996). In bioassays testing zebrafish egg hatching and larval survival, LOEC for elemental chlorine free effluents ranged from 10% to 60% for hatching, and from 5% to 50% for larval survival (Ahtiainen et al., 1996). A range of BKME concentrations was reported to be associated with changes in hormone levels, with a LOEC of 3.5% BKME identified from a study in which whitefish were exposed to effluent from a process using chlorine dioxide bleaching (Soimasuo et al., 1998).

A study evaluating the number of eggs per female fathead minnow reported LOECs ranging from 2.5% to 50% BKME (Kovacs and Megraw, 1996). Flink et al. (1994) reported threshold values of 35% for both egg hatching and larval survival for an elemental chlorine free effluent with oxygen delignification. In short-term (7-day) tests with fathead minnow larvae, reductions in survival were seen at effluent concentrations of 16% to 24% for effluents with chlorine

substitution but no oxygen delignification, and no toxicity was found for an effluent with chlorine substitution and oxygen delignification (O'Conner et al., 1994). In other studies, reduced egg production and reduced larval survival and growth were observed in fathead minnow exposed to effluent concentrations of 63% to 75% (Kovacs and Megraw, 1996). Borton et al. (1991) tested effluents from mills with and without oxygen delignification and found even less toxicity (IC25 of 82% to 100%) for effluents from mills with oxygen delignification.

A range of LOECs (2.5% to 25% BKME) was identified from several studies of spawning behavior of fathead minnows exposed to effluent from a chlorine dioxide bleaching process (Kovacs and Megraw, 1996). Whole life cycle tests have been conducted with Ceriodaphnia dubia exposed to BKME (Robinson et al., 1994; O'Conner et al., 1994; Kovacs et al., 1995; Borton et al., 1991; Hall et al., 1996). Results are quite variable, with IC25 values (25% decrease in reproductive output) ranging from seven percent to 100%. Typical values for mills with oxygen delignification and some degree of chlorine substitution are in the 40% to70% range, with one of the lower values being 22% (Kovacs and Megraw, 1996).

BKME concentrations ranging from 0.6% to 97% were associated with reproductive changes in invertebrate species. An IC_{25} of 0.6% BKME was associated with reduced egg fertilization in sand dollars exposed to effluent from an ECF process (Hall et al., 1996).

Among listed species, one study reported only a No-Effect Level of 4% BKME for mortality in juvenile Chinook salmon (Owens, 1991).

In studies of mortality of fathead minnow larvae exposed to effluent from an ECF bleaching process, IC_{25}s ranged from 10% to 100% BKME (O'Connor et al., 1994). Among adult fish, mortality of longnose sucker and whitefish exposed to concentrations of <0.5% to 12% BKME from a chlorine dioxide bleaching process was significantly higher than in controls (Swanson et al., 1994). A concentration of 0.8% BKME was associated with significantly higher mortality in fathead minnow, compared to controls (Robinson et al., 1994).

Among invertebrate species, O'Connor et al. (1994) reports an IC_{25} of 40% to 60% BKME associated with survival of *C. dubia* using effluent from a process using chlorine dioxide substitution and oxygen delignification.

Genetic toxicity was observed by (Easton et al., 1997) in juvenile Chinook salmon exposed to 4% BKME, and an increase in hepatic mRNA was noted at a concentration of 2% BKME (Campbell et al., 1996) in the same species.

Rao et al. (1995) tested extracts from BKME for mutagenicity using several different tests and found evidence of genotoxicity. Wilson et al. (2001) report increased levels of hepatic DNA adducts in juvenile Chinook salmon exposed for 28 days to eight percent and 16% treated effluent from an elemental chlorine free bleached kraft pulp mill. Couillard and Hodson (1996) conducted an epidemiological study on histopathological conditions in white sucker collected downstream of a bleached-kraft pulp mill in the St. Maurice River, Quebec, Canada, and found no increases in pre-neoplastic or neoplastic lesions in fish near the pulp mill site. Couillard and Hodson (1996) also observed higher densities of pigmented macrophage aggregates in liver, spleen, and kidney of white sucker collected downstream of the bleached kraft mill in Quebec, but the effluent concentrations were not reported in this study.

Oikara et al. (1984) exposed rainbow trout for 3, 11 and 30 days to a sulfate soap preparation that resembled unbleached kraft pulp mill effluents, at concentrations approximating 33%, 15%, and

8% BKME, but found no effects on hematocrit, plasma protein levels, or other blood parameters at concentrations below 15%.

Other studies with non-salmonid species suggest that fish may not respond as well to acute stress when exposed to BKME. Lappivaara (2001) exposed juvenile whitefish (*Coregonus lavaretus*) for 6 weeks, to a range of between 4% and 8% untreated and biologically treated BKME and found that normal responses to acute handling stress (e.g., increases in levels of plasma cortisol, blood glucose, hemoglobin, and hematocrit and liver glycogen phosphorylase activity) were attenuated in fish exposed to untreated or treated BKME. Similarly, a study by Hontela et al. (1997) found that perch and pike exposed to BKME in the St. Maurice River, Quebec, were less responsive to handling stress than those from reference sites. Ambient effluent concentrations associated with these effects were not stated.

Hematological changes were observed in other fish species (bluegill, channel catfish, and largemouth bass) at BKME concentrations of 4% (Borton et al., 1996).

Aaltonen et al. (2000) exposed roach (*Rutilus rutilus*) in laboratory to primary- or secondary-treated effluent from a pulp mill using elemental chlorine-free/total chlorine-free bleaching. To study their capability to respond to foreign antigens they were immunized with bovine gamma-globulin prior to exposure. Roach exposed for 21 days to either primary or secondary treated 20% BKME showed alterations in several immunological parameters, and reduced immunoreactivity in response foreign antigens.

The WET test results in Table 7-26 indicate an indirect toxicity effect at 10% effluent (10 TU_c).

Since the whole effluent toxicity for the Clearwater Mill effluent (10 TU_c) is greater than the water column toxicity benchmark (1 TU_c), this analysis looks at the effects within the exposure volume of the effluent (i.e., the area where the concentration of the plume exceeds the toxicity benchmark) and the effects at and beyond the exposure volume boundary.

The CORMIX model results as described in Section VII.A. and Appendix D predicts that the available dilution at the edge of the mixing zone would be 36.5 or 2.74% effluent, therefore the maximum exposure concentration at the edge of the mixing zone would be the maximum toxicity of 10 TUc divided by the dilution factor of 36.5, which is 0.274 TUc. The toxicity at the edge of the chronic mixing zone is less than 1 TUc, thus, there will be no measurable chronic toxicity at the edge of the chronic mixing zone.

7.5.2.4.6.1 Direct Effects

EPA has concluded that the whole effluent toxicity **may affect but is not likely to directly adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.2.4.6.2 Indirect Effects

EPA has concluded that the whole effluent toxicity **may affect but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**.

7.5.2.4.6.3 Habitat Effects

Since the concentration that listed species, their prey, or benthic invertebrates would be exposed to is less than the established benchmarks, EPA has concluded that the whole effluent toxicity is

not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.3 Other Parameters Evaluated

7.5.3.1 Color

7.5.3.1.1 Summary

Color is defined as either "true" or "apparent" color. Standard Methods defines the true color as the color of water from which the turbidity has been removed. Apparent color includes not only the color due to substances in solution, but also due to suspended matter. In the various chemical pulping processes, lignin and lignin derivatives are solubilized and removed from the wood during the cooking process. The spent cooking liquors containing these highly colored compounds are removed from the pulp in a washing sequence following the cooking process. The wash water is highly colored, and large amounts of color are ultimately discharged to the receiving stream despite some recovery operation.

Surface waters may appear colored because of suspended matter that comprises turbidity. Such color is referred to as apparent color and is different from true color caused by colloidal human materials (Sawyer, 1960). Natural color is reported in color "units" which generally are determined using the platinum-cobalt method (Standard Methods, 1971).

There is no general agreement as to the chemical composition of natural color, and in fact the composition may vary chemically from place to place (AWWA, 1971). Light scattering and fluorescence cause color rather than absorption of light energy, and pH affects both particle size of the color-causing colloids and the intensity of color itself.

7.5.3.1.2 Environmental Baseline

Color has not been measured in water samples collected from the Snake and Clearwater Rivers upstream or downstream of the diffuser. Although there are no measurements of color, turbidity has been measured in upstream and downstream samples. Results of turbidity analyses are discussed in Section VII.C.1.h.i.

7.5.3.1.3 Water Quality Standard

The current Idaho and Washington water quality standards do not specifically address color, however EPA has a color criterion that requires water to be virtually free from substances producing an objectionable color for aesthetic purposes; the sources of supply should not exceed 75 color units on the platinum-cobalt scale for domestic water supplies; and increased color (in combination with turbidity) should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonally established norm for aquatic life.

7.5.3.1.4 Effluent Limitation

Color in treated effluents from both unbleached and bleached chemical pulp mills is an easily recognized characteristic of chemical pulp mill wastewaters. The colored material, formed from organic constituents dissolved from the wood during the pulp manufacturing process, is resistant

to conventional biological treatment. Although physicochemical treatment processes are available to remove color, they are unable to achieve this in an economically sustainable fashion. However, pulp and paper mill effluent color is not a major concern in the Lower Granite Reservoir. EPA believes that the combination of TSS effluent limitations and the historical turbidity monitoring has shown that color in combination with turbidity does not violate water quality standards. Therefore, the 2019 draft permit does not require an effluent limitation for color. However, for reference, during the 2009 permit application process, color was measured in the effluent, and determined to be 750 color units.

7.5.3.1.5 Benchmark

The effects of color in water on aquatic life principally are to reduce light penetration and thereby generally reduce photosynthesis by phytoplankton and to restrict the zone for aquatic vascular plant growth.

The light supply necessary to support plant life is dependent on both intensity and effective wavelengths (Welch, 1952). In general, the rate of photosynthesis increases with the intensity of the incident light. Photosynthetic rates are most affected in the red region and least affected in the blue-violet region of the incident light (Welch, 1952). It has been found that in colored waters the red spectrum is not a region of high absorption so that the effective penetration, and therefore the intensity for photosynthesis, is not as restricted as are other wavelengths. It should be emphasized that transmission of all parts of the spectrum is affected by color, but the greatest effect is on the standard or blue-end of the spectrum (Birge and Juday, 1930). In highly colored waters (45 to 132 color units) Birge and Juday (1930) measured the light transmission as a percentage of the incident level and found very little blue, 50 percent or less yellow, and 100 to 120 percent red.

The light intensity required for some aquatic vascular plants to photosynthetically balance the oxygen used in respiration may be 5 percent of full sunlight during maximum summer illumination periods (NTAC, 1968). As much as 10 percent of the incident light may be required for plankton to likewise photosynthetically produce sufficient oxygen to balance their respiration requirements (NTAC, 1968). The depth at which such a compensation point is reached, called the compensation depth, delineates the zone of effective photosynthetic oxygen production. To maintain satisfactory biological conditions, this depth cannot be substantially reduced.

No studies reporting on the potential toxicity of color to salmonid species were found in the scientific literature; therefore, a toxicity benchmark cannot be established for this parameter.

7.5.3.1.6 Effects Analysis.

Color remains one of the more conspicuous properties of pulp and paper discharges. Besides the aesthetic changes in receiving water quality, high levels of color in the wastewater can reduce light penetration and potentially affect benthic plant growth and habitat.

Since the effects of color in water on aquatic life principally are to reduce light penetration, the analysis conducted for light attenuation from TSS shows that the depth at which light equals 1% of the light in the surface changed from 6.5 meters without the Clearwater effluent to 6.4 meters with the Clearwater effluent at the 7Q10 flow rate, a change of about 2%. Greater changes in transparency can be expected during low-flow conditions when the Clearwater discharge is proportionally greater.

7.5.3.1.6.1 Direct and Indirect Effects

Based on the effects analysis of light attenuation for TSS, the color in the discharge is **likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.3.1.6.2 Habitat Effects

Since the listed species, their prey, or benthic invertebrates may be affected by decreased light attenuation, EPA has concluded that the color in the discharge is **likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.3.2 Dissolved Oxygen

7.5.3.2.1 Summary

Dissolved oxygen (DO) refers to the volume of oxygen (gas) that is contained in water. DO concentrations are most often reported in units of milligrams of gas per liter of water (mg DO/L H_2O or mg/L DO). Dissolved oxygen is produced during photosynthesis of aquatic biota and by the transfer of oxygen across the air-water interface and consumed during respiration and decomposition. Oxygen losses readily occur when water temperatures rise, when plants and animals respire, and when microbes aerobically decompose organic matter.

Photosynthesis occurs only during daylight hours because it requires light. Respiration and decomposition, on the other hand, occur 24 hours a day. This difference alone can account for large daily variations in DO concentrations. During the night, when photosynthesis cannot counterbalance the loss of oxygen through respiration and decomposition, DO concentration may steadily decline. It is lowest just before dawn, when photosynthesis resumes.

Other sources of oxygen in water include the air and inflowing streams. Oxygen concentrations are much higher in air, which is about 21% oxygen, than in water, which is a tiny fraction of 1 percent oxygen. Where the air and water meet, this tremendous difference in concentration causes oxygen molecules in the air to dissolve into the water. More oxygen dissolves into water when wind stirs the water because as the waves create more surface area, more diffusion can occur.

The amount of oxygen that can be held by the water depends on the water temperature, salinity, and pressure. Gas solubility increases with decreasing temperature (colder water holds more oxygen) and with decreasing salinity (freshwater holds more oxygen than does saltwater). Both the partial pressure and the degree of saturation of oxygen will change with altitude. However, gas solubility decreases as pressure decreases. Thus, the amount of oxygen absorbed in water decreases as altitude increases because of the decrease in relative pressure.

Another physical process that affects DO concentrations is the relationship between water temperature and gas saturation. Since the saturation of oxygen in water depends on the temperature, cold water can retain more oxygen than warmer water because increasing temperatures decreases the maximum equilibrium of dissolved oxygen with water.

Once absorbed, oxygen is either incorporated throughout the water body via internal currents or is lost from the system. Flowing water is more likely to have high dissolved oxygen levels than is stagnant water because of the water movement at the air-water interface. In flowing water,

oxygen-rich water at the surface is constantly being replaced by water containing less oxygen due to turbulence, creating a greater potential for exchange of oxygen across the air-water interface. Because stagnant water undergoes less internal mixing, the upper layer of oxygen-rich water tends to stay at the surface, resulting in lower dissolved oxygen levels throughout the water column.

Microbes play a key role in the loss of oxygen from surface waters. Microbes use oxygen as energy to break down long-chained organic molecules into simpler, more stable end-products such as carbon dioxide, water, phosphate and nitrate. As microbes break down the organic molecules, oxygen is removed from the system and must be replaced by exchange at the air-water interface. If high levels of organic matter are present in the water, microbes may use all available oxygen.

To the degree that pollution contributes oxygen-demanding organic matter (e.g., sewage, lawn clippings, soils from streambank and shore erosion, and agricultural runoff) or nutrients that stimulate growth of organic matter, pollution causes a decrease in average DO concentrations. If the organic matter is formed in the lake, for example by algal growth, at least some oxygen is produced during growth to offset the eventual loss of oxygen during decomposition. However, in lakes and reservoirs where a large portion of the organic matter is brought in from outside the lake, oxygen production and oxygen consumption are not balanced, and low DO may become even more of a problem.

The development of anoxia in lakes and reservoirs is most pronounced in thermally stratified systems in summer and under the ice in winter when the water mass is cut-off from the atmosphere. Besides the direct effects on aerobic organisms, anoxia can lead to increased release of phosphorus from sediments that can fuel algal blooms when mixed into the upper euphotic zone. It also leads to the buildup of chemically reduced compounds such as ammonium and hydrogen sulfide (H_2S, rotten egg gas) which can be toxic to bottom dwelling organisms. In extreme cases, sudden mixing of H_2S into the upper water column can cause fish kills.

7.5.3.2.2 Environmental Baseline

Endangered Species Act Tier 1 studies were undertaken in 2005 and 2006 to evaluate effluent and natural waters above and below the facility. In the 2005 study, mean DO generally increased at all locations over the monitoring period, with the highest DO concentrations typically observed at the Clearwater River reference location and the lowest DO concentrations observed at the Snake River reference locations. Similar results were observed in the 2006 study. In both years, it was found that all average downstream DO measurements were at least 8 mg/L, which meets the WDOE water quality standard and is evidence that discharge from the Clearwater facility does not affect DO downstream of the facility. Figure 7-28 and Figure 7-29 show mean DO concentrations for the Clearwater River reference location (CR REF), Snake River reference location (SR REF), and locations downstream of the effluent. From nearest to farthest from the effluent, the locations were LGP-13, LGP-11, LGP-09, LGP-06, and LGP-01 (AMEC 2006, AMEC 2007).

7.5.3.2.3 Water Quality Standard

In Idaho, the most restrictive water quality standard for dissolved oxygen that applies to this segment of the Snake River is for the protection of cold water biota. This standard establishes a

minimum dissolved oxygen concentration of 6 mg/l. In Washington, the applicable standard for waters designated for salmonid spawning, rearing and migration is a minimum of 8.0 mg/l.

7.5.3.2.4 Effluent Limitation

The minimum effluent DO concentration reported by Clearwater is 5.0 mg/L. Based on this concentration, there is no reasonable potential to cause or contribute to an exceedance of water quality standards directly from the DO concentration in the effluent and permit limits are not necessary (refer to calculations in Appendix B). The 2005 permit also did not require an effluent limitation for DO. However, there are other pollutants within the effluent that cause an oxygen demand on the receiving water which the permit controls through the effluent limitation for BOD and COD (see section VII.E.1.d and section VII.E.1.b).

7.5.3.2.5 Benchmark

USEPA (1986) evaluated concentrations of DO that have been observed to cause toxicity to embryo and larval salmonids, other salmonid life cycle stages, early life stages of non-salmonids, and other life stages of non-salmonids, which are summarized in Table H- 5 of Appendix H. These DO concentrations are based on a type of toxicity referred to as "production impairment" which is a measure of growth and considers temperature, disease, and pollutant stresses.

EPA determined that a one-day minimum of 4 mg/L DO was protective because the acute lethal limit for salmonids is at or below 3 mg/L DO; however, a significant proportion of the insect species common to salmonid habitats are less tolerant of acute exposures to low DO than are salmonids.

The DO concentrations selected as benchmarks are believed to protect the more sensitive species against potentially damaging production impairment. Because repeated exposure to dissolved oxygen concentrations at or near the acute lethal threshold is expected to be stressful, and because stress can indirectly cause other types of toxicity (such as increased incidence of disease), the selected benchmarks are designed to prevent significant episodes of continuous or regularly recurring exposures to DO concentrations at or near the lethal threshold. This protection has been achieved by setting the daily minimum benchmark for early life stages at the subacute lethality threshold, using a 7-day averaging period for early life stages, and by stipulating a 7-day mean minimum value for other life stages.

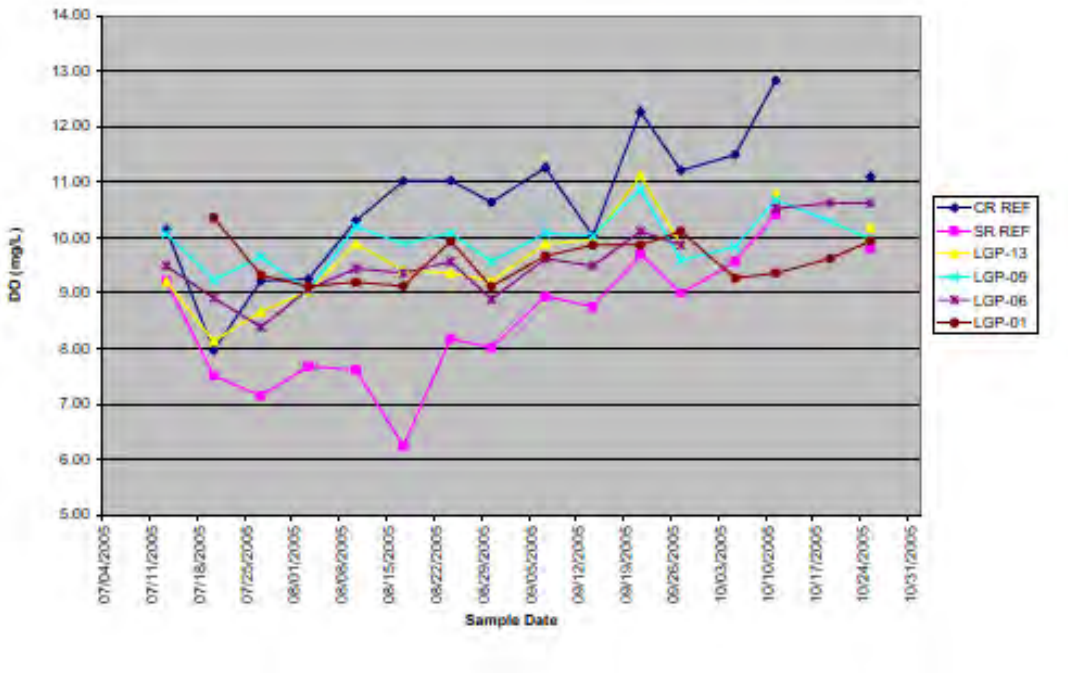


Figure 7-28: Mean DO Concentrations in 2005 Endangered Species Act Tier 1 Study (AMEC, 2006)

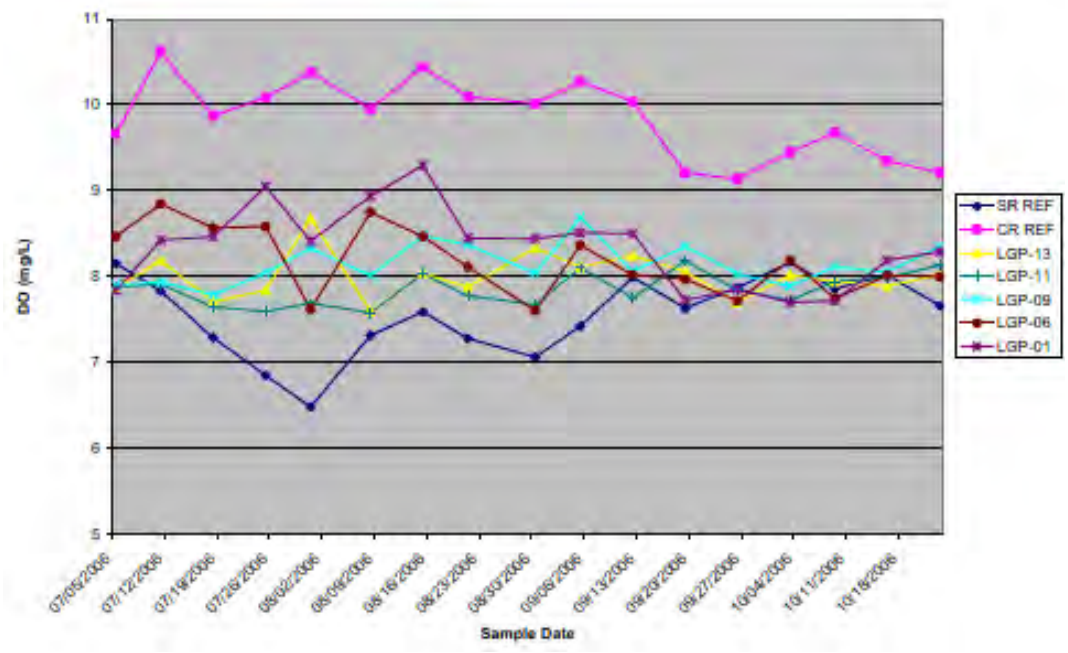


Figure 7-29: Mean DO Concentrations in 2006 Endangered Species Act Tier 1 Study (AMEC, 2007)

Water column DO criteria are established for early life stages (up to 30 days following hatching) that are intended to ensure adequate DO concentrations in the intergravel spaces where early life stages occur. A one-day minimum DO concentration in intergravel DO (IGDO) of 5.0 mg/L and

a seven-day average IGDO of 6.5 mg/L have been identified by USEPA (1986). Based on the assumption that IGDO results from a 3 mg/L reduction in water column DO, EPA determined that water column criteria of 8.0 as a one-day minimum and 9.5 as a seven-day mean DO concentration would be protective of early life stages.

The dissolved oxygen benchmarks are selected to be protective of not only average conditions, but also daily minimum conditions. USEPA (1986) states that both the daily and the seven-day mean minimum dissolved oxygen concentrations "should be considered as instantaneous concentrations to be achieved at all times." Evaluating the extent, duration, and magnitude of an event for comparison to criteria must similarly be a function of the spatial and temporal frequency of the data. Thus, a single deviation below the criterion takes on considerably less significance where continuous monitoring occurs than where sampling is comprised of once-a-week grab samples. The frequency of recurrence is of considerable interest to those modeling dissolved oxygen concentrations because the return period, or period between recurrences, is a primary modeling consideration contingent upon probabilities of receiving water volumes, waste loads, temperatures, etc. It should be apparent that the return period cannot be isolated from the other factors discussed above, and that consideration of the protectiveness should also account for the return period.

From this guidance, EPA has selected the following benchmarks in Table 7-27 for this BE for **bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead**:

Table 7-27: Selected DO Toxicity Benchmarks

	Early Life Stages [1,2]	Other Life Stages
30 Day Mean	NA[3]	6.5
7 Day Mean	9.5 (6.5)	NA
7 Day Mean Minimum	NA	5.0
1 Day Minimum [4]	8.0 (5.0)	4.0

Notes:

1. These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in the parentheses. The 3 mg/L differential is discussed in the criteria document (USEPA, 1986). For species that have early life stages exposed directly to the water column, the figures in the parentheses apply.
2. Includes all embryonic and larval stages and all juvenile forms to 30-days following hatching.
3. NA (not applicable).
4. All minima should be considered as instantaneous concentrations to be achieved at all times.

7.5.3.2.6 Effects Analysis

7.5.3.2.6.1 Water Column

Adequate dissolved oxygen is necessary for the life of fish and other aquatic organisms. Oxygen also is needed by virtually all algae and all macrophytes, and for many chemical reactions that are important to lake functioning. Dissolved oxygen levels of at least 5 - 6 mg/L are usually required for growth, while DO levels below 3 mg/L are stressful to most aquatic organisms.

Dissolved oxygen concentrations may change dramatically with lake or reservoir depth. Oxygen production occurs in the top portion of a lake or reservoir, where sunlight drives the engines of photosynthesis. Oxygen consumption is greatest near the bottom of a lake or reservoir, where sunken organic matter accumulates and decomposes. In deeper, stratified, lakes and reservoirs,

this difference may be dramatic when there is adequate oxygen in the epilimnion but deficient in the hypolimnion. If the lake or reservoir is shallow and easily mixed by wind, the DO concentration may be fairly consistent throughout the water column as long as it is windy. When calm, a pronounced decline with depth may be observed.

Seasonal changes also affect dissolved oxygen concentrations. Warmer temperatures during summer speed up the rates of photosynthesis and decomposition. When all the plants die at the end of the growing season, their decomposition results in heavy oxygen consumption. Other seasonal events, such as changes in water levels, volume of inflows and outflows, and presence of ice cover, also cause natural variation in DO concentrations.

Dissolved oxygen may play a large role in the survival of biota in temperate lakes and reservoirs during the summer months, due to a phenomenon called stratification. Seasonal stratification occurs due to water's temperature-dependent density. As water temperatures increase, the density decreases. Thus, the sun-warmed water will remain at the surface of the water body forming the epilimnion, while the denser, cooler water sinks to the bottom (hypolimnion). The layer of rapid temperature change separating the two layers is called the thermocline.

At the beginning of the summer, the hypolimnion will contain more dissolved oxygen because colder water holds more oxygen than warmer water. However, as time progresses, an increased number of dead organisms from the epilimnion sink to the hypolimnion and are broken down by microorganisms. Continued microbial decomposition eventually results in an oxygen-deficient hypolimnion. If the lake or reservoir is in a eutrophic state, this process may be accelerated and the dissolved oxygen in the lake could be depleted before the summer's end.

Mid-summer, the warmer surface water temperature of a lake or reservoir may limit the total amount of oxygen present. If the water becomes too warm, even if 100% saturated, O2 levels may be suboptimal for many fish species. In other words, oxygen can be present in the water, but at too low a concentration to sustain aquatic life. When strong thermal stratification develops, fish may become stressed when the epilimnion strata is too warm for them, while the hypolimnion has too little oxygen. Conditions may become especially serious during a spate of hot, calm weather that could result in the loss of many fish.

Since low DO concentrations (below 8 mg/L) have been measured in the Snake River upstream of outfall 001 in August and the discharge DO is 5 mg/L, the DO in the effluent plume can cause stress to listed species until the DO levels equilibrate with the ambient DO levels. However, the DO levels equilibrate within 10 feet of the diffuser making it improbable that the fish would be exposed to low DO levels due to the discharge within 10 feet of the diffuser because they would tend to avoid the currents created by the jet action of the plume.

Therefore, EPA has concluded that the DO of the discharge **may directly affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.3.2.6.2 Intergravel

Oxygen is depleted within the gravel bed of a river by the respiration of benthic organisms including bacteria and macroinvertebrates. It is replenished by oxygen that diffuses from the overlying water column or that enters with inflowing water. Water often flows into riverbeds, which are sometimes called the hyporheic zone. The hyporheic zone is the shallow bed area that

represents an interface between the ground water and surface-water systems. Water from the surface flows relatively freely into the hyporheic zone, where it mixes with upwelling ground water, and eventually discharges back to the stream.

Bed topography may have a significant influence on hyporheic exchange and thereby intergravel dissolved oxygen. If the bed surface rises, river flow speeds up because of the reduced cross-sectional area in the stream. Water pressure however, decreases with increasing flow velocity. Thus, there is higher water pressure (and more stream water inflow to the bed) on the upstream side of a rise, and lower water pressure (and bed water outflow) at the top of a rise. Stream bathymetry thus can establish patterns of inflow to and outflow from the gravel bed. This has a significant influence on bed-stream water exchange and thus on intergravel dissolved oxygen.

The complexity of water exchange and oxygen consumption by benthic organisms implies that intergravel dissolved oxygen is highly site specific and not amenable to simple analysis. Water-quality criteria published by USEPA (1986, pg. 9-10) have as a matter of practicality side-stepped these complexities and made a blanket recommendation that intergravel dissolved oxygen be assumed to be 3 mg/L less than the concentration in the overlying water column.

Intergravel dissolved oxygen concentrations are a concern in the Snake River in the tail water reaches immediately below the four Lower Snake River dams. At these locations, water depths are not so great as to preclude anadromous fish spawning. These locations also correspond to zones of generally elevated dissolved oxygen concentrations because water spilled over the dam spillways is highly oxygenated, often to supersaturation. Owing to the high ambient dissolved oxygen concentrations, intergravel dissolved oxygen is not anticipated to be insufficient to support spawning in the dam tailwaters.

Downstream intergravel concentrations of DO will be compared to the toxicity benchmark of 5.0 mg/L for early life stages. If early life stages are present in the area evaluated, Washington's water quality criterion of 8 mg/L for the water column would be protective of this life stage. Since the intergravel DO benchmark of 5.0 mg/L corresponds to the water column DO concentration of 8 mg/L, it is assumed that the intergravel DO concentrations are protected when the water column DO concentration is at or above 8 mg/L. The water column DO concentrations at site 7, the site closest to the dam tailraces, have been measured at concentrations below 8 mg/L in late August through mid-September. Since it has been documented that the fall Chinook salmon spawns in October and emergence occurs in late April to late May (Groves and Chandler, 1999) and there is some re-aeration of the river as it goes through Lower Granite Dam; therefore, EPA has concluded that the DO in the discharge **may affect, but is not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.3.2.6.3 Habitat Effects

Since the concentration of DO that listed species, their prey, or benthic invertebrates would be exposed to is not likely to adversely affect listed threated and endangered species, EPA has concluded that the DO of the discharge is **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.3.3 Metals

7.5.3.3.1 Summary

The important relation between water hardness and lethal toxicity is well documented for some metals. In addition to hardness, hydrogen ion concentration in water (pH) is extremely important in governing the species and solubility of metals and therefore the lethal toxicity. At high pH, many heavy metals form hydroxides or basic carbonates that are relatively insoluble and tend to precipitate. They may, however, remain suspended in the water as fine particles. The toxicity of suspended hydroxides of metal depends on the situation. It is difficult to predict the effect of pH on toxicity. For example, low pH (about 5) as well as high pH (about 9) reduced toxicity of copper and zinc compared to that at neutral pH (Fisheries Research Board of Canada *unpublished data* 1971). There are also numerous other factors, such as dissolved oxygen, temperature, turbidity, carbon dioxide, magnesium salts, and phosphates that affect metals toxicity.

7.5.3.3.1.1 Antimony

Antimony is present in the environment due to natural processes and human activities, mainly in the form of Sb (III) and Sb (V). In non-polluted waters, dissolved antimony is typically present at concentrations below 1.0 µg/L (Filella et al, 2002). Waterborne antimony can result from natural weathering of geologic formations and minerals, as well as anthropogenic sources such as mining effluent, manufacturing and municipal wastes. Antimony oxide is used in various materials as a flame retardant. There are no known biological functions for antimony (USEPA 1988). Antimony does not appear to accumulate in fish and other aquatic animals. In soils, concentrations of antimony are generally less than 1 ppm (ATSDR 1992). Trivalent forms of antimony are known to be more toxic than other chemical species (Nam et al, 2009).

7.5.3.3.1.2 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). The toxicity of arsenic can be altered by several 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 1990, McGeachy and Dixon 1990a) whereas in invertebrates the opposite is true (Bryant et al., 1985). Inorganic forms of arsenic are typically more toxic to aquatic species, particularly the more sensitive early life stages (Eisler, 1988). While evidence does suggest that toxicity of arsenic can be altered by both temperature and phosphorus (two concerns for the mid-Snake River in Idaho), enough information to clearly characterize the relationship between arsenic toxicity and these two factors does not exist.

7.5.3.3.1.3 Chromium VI

Sources of chromium in aquatic systems include electroplating and metal finishing industries, publicly owned treatment plants, iron and steel foundries, inorganic chemical plants, tanneries, textile manufacturing and runoff from urban and residential areas (Towill et al., 1978; Eisler, 1986a). In freshwater environments, hydrolysis and precipitation are the most important processes in determining the environmental fate of chromium, while absorption and

bioaccumulation are considered minor. Chromium (VI) is highly soluble in water and thus very mobile in aquatic systems (Ecological Analysts, 1981).

7.5.3.3.1.4 Copper

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). 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). The toxicity of copper to aquatic organisms is dependent on the speciation of the chemical, water hardness, and the type and life stage of the exposed organisms.

7.5.3.3.1.5 Lead

Lead is a naturally occurring, ubiquitous compound that can be found in rocks, soils, water, plants, animals, and air. Concentrations of lead associated with background freshwater systems are estimated to be <3.0 µg/L (Moore and Ramamoorthy, 1984). It is soluble in water and its bioavailability increases in environments with low pH, low organic content, and low metal salt content (Eisler, 1988a). Lead is most often precipitated to sediments in aqueous environments. The toxicity of lead varies with water hardness. As hardness increases, lead precipitates, and becomes less bioavailable to aquatic organisms. 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, 1988a; Hamir et al., 1982). Aquatic organisms 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. Animals are also more sensitive at younger life stages and when exposure durations are greater.

7.5.3.3.1.6 Nickel

Nickel is a very hard metal that occurs naturally in soils and volcanic dust and is abundant in the earth's crust. Nickel is released to the atmosphere by windblown dust, volcanoes, combustion of fuel oil, municipal incineration, and industries involved in nickel refining, steel production, and other nickel alloy production. Background levels of nickel in soils vary widely depending on local geology and anthropogenic inputs, but concentrations typically range between 4 and 80 ppm. Some areas of the United States may contain natural levels as high as 5,000 ppm. Nickel concentrations in surface water and groundwater range between 3 and 10 µg/L. A lot of nickel released into the environment ends up in soil or sediment where it strongly attaches to particles containing iron or manganese. Under acidic conditions, nickel is more mobile in soil and might seep into groundwater. Nickel does not appear to concentrate in fish and does not bioaccumulate to a great extent in animals. Studies show that some plants can take up and accumulate nickel. (ATSDR 2005)

7.5.3.3.1.7 Zinc

Zinc is naturally introduced into aquatic systems, usually via leaching from igneous rocks. Concentrations of zinc associated with unpolluted freshwater systems are estimated to range between 0.5-15 :g/L (Moore and Ramamoorthy, 1984; Groth, 1971). 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.

7.5.3.3.2 Environmental Baseline

During the 2005/2006 surface water monitoring metals were not measured and therefore, the data presented here was collected during 1997 through 2002, as part of the Receiving Water Studies. Figure 7-30 through Figure 7-32 show the range of the metals measured during the Receiving Water Studies from 1997 to 2002. Table 7-28 below provides monitoring data for metals from the Washington State Department of Ecology at Interstate Bridge just upstream from the confluence with the Clearwater River in 2009. No analytes exceeded their respective standards. Metals are analyzed for on a periodic basis in this program as no more recent data are available from this station.

7.5.3.3.3 Water Quality Standards

The aquatic life criteria for several of the metals of concern are calculated as a function of hardness measured as mg/L of calcium carbonate (CaCO3). As the hardness of the receiving water increases the numerical value of the metal criterion increases and the toxicity of the metal decreases (since the metals tend to precipitate to solids and become less bioavailable to aquatic organisms).

In general, the water quality standards define a hardness range of 25 to 400 mg/l as being applicable to these criteria. The exceptions are Idaho's water quality criteria for cadmium and copper. Idaho's cadmium criteria have a hardness range of 10 to 400 mg/L. Idaho has adopted the EPA's recommended copper criteria, which are not hardness-based but rather based on the biotic ligand model (BLM), however, the BLM criteria have not yet been approved by the EPA, thus, the former hardness-based criteria remain in effect for Clean Water Act purposes (40 CFR 131.21).

Table 7-28: Metals data from the Snake River at Interstate Bridge (Washington State Department of Ecology Site 35A150 – 2009). U – not detected at the reported level; J – estimated value.

Analyte		Range (µg/L)
Silver	Dissolved	0.02(U)
	Total	0.1(U)
Arsenic	Dissolved	1.35 - 4.96
	Total	1.94 – 4.47
Cadmium	Dissolved	0.02(U)
	Total	0.1(U)
Chromium	Dissolved	0.3 – 1.54
	Total	0.5(U) – 1.78
Copper	Dissolved	0.65 – 0.89
	Total	0.79 – 2.18
Nickel	Dissolved	0.22 – 0.78
	Total	0.78 – 1.48
Lead	Dissolved	0.02(U) – 0.051
	Total	0.1(U) – 1.26
Zinc	Dissolved	1.3(J) – 2.7(J)

	Total	5(U) – 6.
Hardness (mg/L)		47 – 143
Mercury		0.002(U) – 0.0064

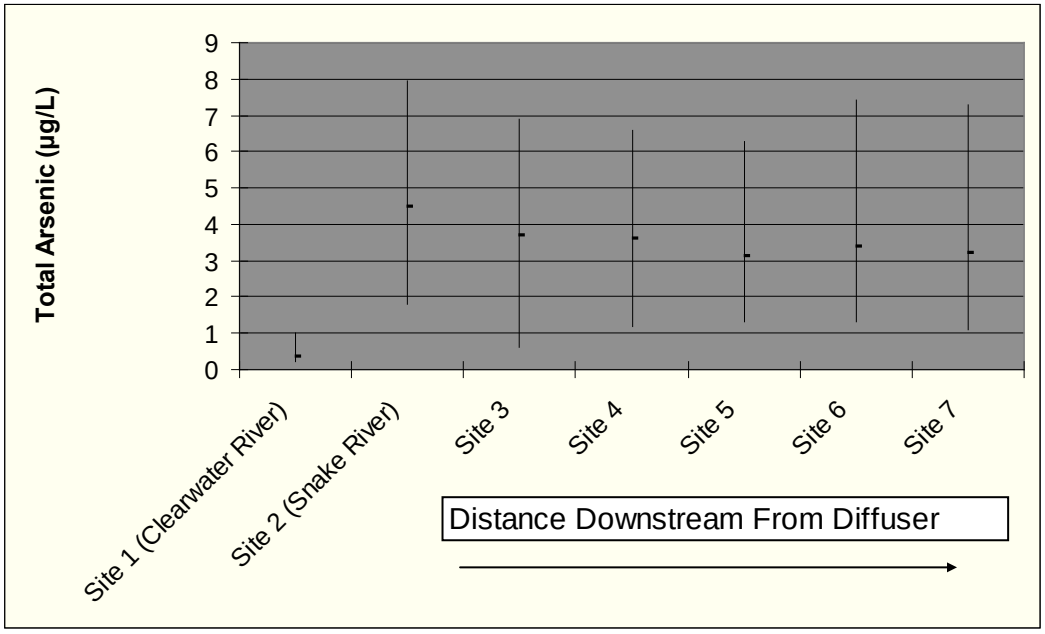


Figure 7-30: Mean and Range of Total Arsenic in the Clearwater and Snake Rivers Measured Upstream and Downstream of the Mill during Receiving Water Studies (Clearwater, 1997, 1998, 1999, 2000, 2001, 2002).

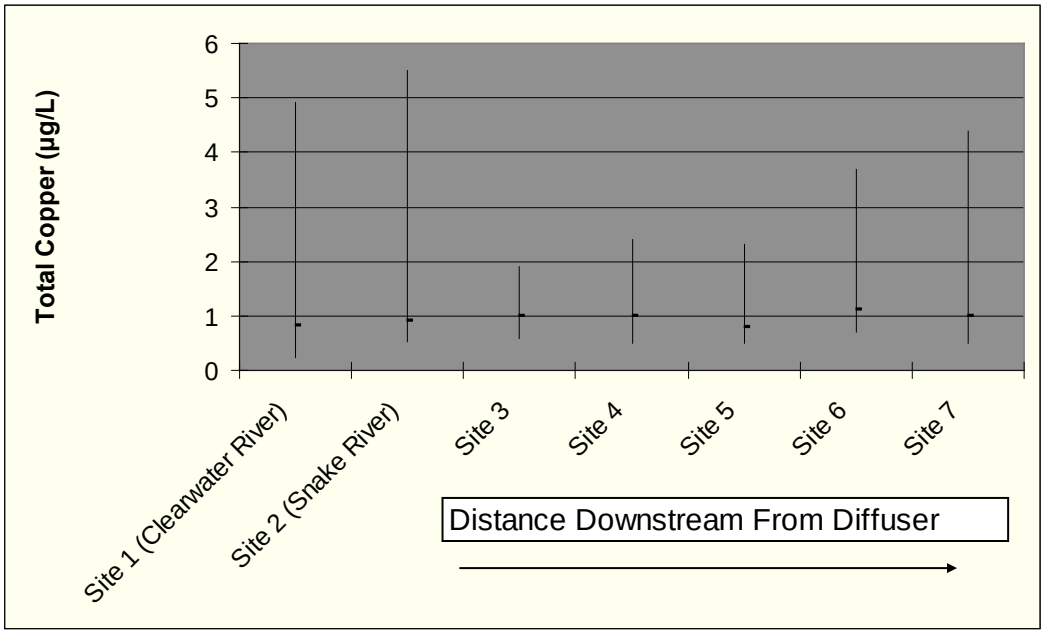


Figure 7-31: Mean and Range of Total Copper in the Clearwater and Snake Rivers Measured Upstream and Downstream of the Mill during Receiving Water Studies (Clearwater, 1997, 1998, 1999, 2000, 2001, 2002).

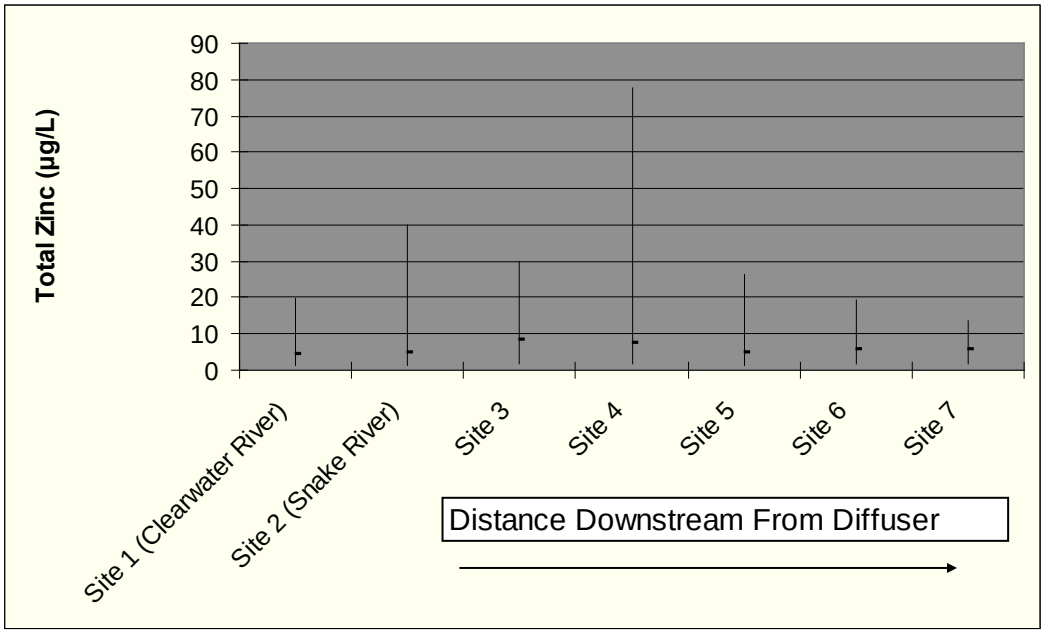


Figure 7-32: Mean and Range of Total Zinc in the Clearwater and Snake Rivers Measured Upstream and Downstream of the Mill during Receiving Water Studies (Clearwater, 1997, 1998, 1999, 2000, 2001, 2002).

The equations used to derive hardness-based criteria are shown in Section 201.02 of the Idaho Water Quality Standards.[10] The 5th percentile of the hardness is normally used to represent the

[10] https://adminrules.idaho.gov/rules/current/58/580102.pdf

reasonable worst-case condition. The 5th percentile hardness values for the Snake River and Clearwater River are 54.9 mg/L and 9 mg/L, respectively. Since the measured hardness falls below the low-end range applicable to the criteria, a hardness of 25 mg/L was used to develop the applicable metals criteria for the Clearwater River.

The current Idaho and Washington water quality standards do not include aquatic life criteria for antimony. The most stringent water quality standard in Idaho and Washington for antimony is 5.2 µg/L for the protection of human health.

The most stringent water quality standards for arsenic are 0.018 µg/l in Washington, and 0.02 µg/L in Idaho, as long-term averages, for the protection of human health.

The current Idaho and Washington water quality standards aquatic life criteria for dissolved copper (Cu) are hardness dependent. The dissolved criteria corresponding to the hardness of the Snake River are 8.35 µg/L (acute) and 5.95 µg/L (chronic). The corresponding total recoverable criteria for copper are 8.70 µg/L (acute) and 6.20 µg/L (chronic), respectively, as the conversion factor for copper is 0.960.

The current Idaho and Washington water quality standards aquatic life criteria for dissolved hexavalent chromium (Cr VI) are 15.7 µg/L (acute) and 10.6 µg/L (chronic). The corresponding total recoverable criteria for hexavalent chromium are 16 µg/L (acute) and 11 µg/L (chronic), respectively, as the conversion factors for hexavalent chromium are 0.982 and 0.962, respectively.

The current Idaho and Washington water quality standards aquatic life criteria for dissolved lead (Pb) are hardness dependent. The dissolved criteria corresponding to the hardness of the Snake River are 28.1 µg/L (acute) and 1.10 µg/L (chronic). The corresponding total recoverable criteria for lead are 31.2 µg/L (acute) and 1.22 µg/L (chronic), respectively, as the conversion factor for lead is 1.46203-(ln hardness x 0.145712), which is equal to 0.901, for the receiving water hardness of 47 mg/L as $CaCO_3$.

The current Idaho and Washington water quality standards aquatic life criteria for dissolved nickel are hardness dependent. The dissolved criteria corresponding to the hardness of the Snake River are 247.2 µg/L (acute) and 27.46 µg/L (chronic). The corresponding total recoverable criteria for nickel are 247.7 µg/L and 27.54 µg/L, respectively, as the conversion factors for nickel are 0.998 and 0.997 µg/L, respectively.

The current Idaho and Washington water quality standards aquatic life criteria for dissolved zinc (Zn) are hardness dependent. The dissolved criteria corresponding to the hardness of the Snake River are 61.8 µg/L (acute) and 62.3 µg/L (chronic). The corresponding total recoverable criteria for zinc are 63.2 µg/L (acute) and 63.2 µg/L (chronic), respectively, as the conversion factors for zinc are 0.978 and 0.986 µg/L, respectively.

7.5.3.3.4 Effluent Limitations

The historical (1992) permit contained effluent limitations for aluminum, arsenic, mercury, lead, and selenium and effluent monitoring for chromium VI, copper, and zinc. Data collected by Clearwater from 1993 to the early 2000s indicate no reasonable potential to cause or contribute to an exceedance of water quality criteria; therefore, no limits for metals were included in the 2005 final permit. The draft (2019) permit also does not require effluent limits for metals (see Appendix B).

7.5.3.3.5 Toxicity Benchmarks

Current literature review and resulting analysis of sublethal and lethal effects for the metals of concern are thoroughly discussed in the Idaho BA document and are not repeated here. Please refer to the Idaho BA (USEPA, 1999) for detailed evaluation of these metals. The applicable water quality criteria were used as the toxicity benchmarks for all species.

7.5.3.3.5.1 Antimony

This BE uses 5.2 µg/L total antimony as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.3.3.5.2 Arsenic

This BE uses 190 µg/L total arsenic as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead. In the 1999 Biological Assessment of the Idaho Water Quality Standards for Numeric Water Quality Criteria for Toxic Pollutants, EPA established a benchmark for arsenic of 190 µg/L (chronic). Given the maximum arsenic concentration from Washington State Department of Ecology station 35A150 of 4.47 µg/L, it is unlikely that arsenic will have an effect on listed species.

7.5.3.3.5.3 Chromium VI

This BE uses 11 µg/L total hexavalent chromium as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.3.3.5.4 Copper

This BE uses 6.20 µg/L total copper as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.3.3.5.5 Lead

This BE uses 1.22 µg/L total lead as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.3.3.5.6 Nickel

This BE uses 27.5 µg/L total nickel as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.3.3.5.7 Zinc

This BE uses 63.2 µg/L total zinc as a benchmark for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.

7.5.3.3.6 Effects Analysis

7.5.3.3.6.1 Direct and Indirect Effects

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). Since the Clearwater Mill is not adding metals to the environmental baseline (i.e., the metals in the discharge are due to pass-through from the intake of Clearwater River water), avoidance behavior would only occur due to interactions of the mill effluent with metals in the receiving water, the only effect from metals would be from interactions with other parameters in the discharge, such as pH or hardness. For migratory fish and for overwintering fish accessing different habitat types, an avoidance response to metals concentration could act as a barrier to fish movement.

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.

By definition, behavioral responses such as avoidance occur at metals concentrations below the chronic criteria. Based on a literature review by IDEQ (2000), the avoidance level for salmonids is 3 µg/L for copper and 14 µg/L for zinc. The aquatic life criteria for copper and zinc exceed these avoidance levels (see Table 4-8). As IDEQ (2000) states in the analysis of avoidance thresholds, currently there is no reliable, accepted method to define the avoidance threshold for mixing zones. Further, there is no current criterion or methodology to establish the avoidance threshold.

In the Idaho BA (USEPA, 1999), EPA made determinations regarding the potential for metals (arsenic, copper, lead, mercury, selenium, and zinc) criteria established by the Idaho water quality standards to adversely affect threatened and endangered species. The affect determinations were made using the following procedures: 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. Table 7-29 summarizes the results of the Idaho BA for metals and cyanide in the Idaho water quality standards at the chronic criterion.

Table 7-29: Summary of Biological Assessment of the Idaho Water Quality Chronic Standards for Metals of Concern (USEPA, 1999)

Parameter	**Species**				
	Salmonids				
	Bull Trout	**Snake River sockeye salmon**	**Snake River spring/summer Chinook salmon**	**Snake River fall Chinook salmon**	**Snake River steelhead**
Arsenic	NL	NL	NL	NL	NL
Chromium VI	NL	NL	NL	NL	NL

Copper	NL	NL	NL	NL	NL
Lead	NL	NL	NL	NL	NL
Nickel	NL	NL	NL	NL	NL
Zinc	NL	NL	NL	NL	NL

Definitions of Acronyms:
NL = not likely to adversely affect
L = may be likely to adversely affect

Since metals in the Clearwater Mill effluent are due to intake water from the Clearwater River, the salmonid species would only be exposed to the levels of metals already in the river system. Therefore, EPA has determined that the metals in the discharge **may affect, but are not likely to adversely affect bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.5.3.3.6.2 Habitat Effects

Since the concentration of metals that listed species, their prey, or benthic invertebrates would be exposed to due to the discharge is not likely to adversely affect listed threated and endangered species, EPA has concluded that the metals in the discharge are **not likely to adversely modify the critical habitat for bull trout, Snake River sockeye salmon, Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, and Snake River steelhead.**

7.6 Effects Determination

7.6.1 Summary of Effects

The previous section of this BE evaluated individual parameters and the potential for exposure to listed threatened and endangered species in the Action Area. Some parameters consider synergistic effects (e.g., AOX and WET) while others only consider the effect of the individual chemical. Table 7-30 summarizes the effects determinations for this BE.

Serious aquatic biology problems have been ascribed to the discharge of pulping wastes into surface waters (Van Horn, 1961 and 1971). Some of these include deoxygenation, toxicity to fish, and interference with spawning. EPA has determined that the reissuance of this permit (with mixing zones and effluent limits) is likely to adversely impact bull trout, steelhead, fall Chinook salmon, spring/summer Chinook salmon, and sockeye salmon. However, reissuance of this permit, with effluent limits and other requirements that are more stringent than the current permit, is seen as a positive step towards maintaining the water quality in LGR.

7.6.2 Issues of Take

The purpose of this section is to assess whether take of a listed species is likely to result from the proposed activity. "Take" is defined as 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. The take prohibition does not extend to proposed or candidate species.

Applying these definitions to the previous analysis, it is likely that an incidental take of bull trout or steelhead, in the form of harm and harassment, could occur. Harm would occur when a fish entered a mixing zone for feeding or breeding but was unable to perform these activities due to physiological alteration from exposure to TSS, chlorinated organic compounds, or WET toxicity. This, of course, is dependent on how long the fish remains within the impact area of these parameters. Harassment is a more likely scenario, which could occur to juveniles out-migrating through the Action Area or rearing species within the Action Area. Exposure to toxic levels of chemical compounds or mixtures at toxic levels or that bioaccumulate to toxic levels would impair development or result in death. Another potential issue is the movement of resident fish that could result in multiple exposures to some individuals.

Table 7-30: Summary of Effects to Listed Species under the ESA from the Clearwater Mill Discharge

Parameter	Bull Trout		Steelhead		Fall Chinook		Spring/Summer Chinook		Sockeye Salmon	
	Species	Habitat	Species	Habitat	Species	Habitat	Species	Habitat	Species	Habitat
BOD	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
COD	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
DO	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Intergravel DO	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
pH	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Temperature	NL	NL	L	L	L	L	NL/L*	NL/L*	L	L
TSS	L	L	L	L	L	L	L	L	L	L
Ammonia	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Color	L	L	L	L	L	L	L	L	L	L
Nutrients	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
AOX	L	L	L	L	L	L	L	L	L	L
Chloroform	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
2,4,5-trichlorophenol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
2,4,6-trichlorophenol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
2,3,4,6-Tetrachlorophenol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Pentachlorophenol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
3,4,5-trichlorocatechol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
3,4,6- trichlorocatechol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Tetrachlorocatechol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
3,4,5-trichloroguaiacol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
3,4,6- trichloroguaiacol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
4,5,6- trichloroguaiacol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Tetrachloroguaiacol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Trichlorosyringol	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
2,3,7,8-TCDD	L	L	L	L	L	L	L	L	L	L
2,3,7,8-TCDF	L	L	L	L	L	L	L	L	L	L
WET	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Antimony	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Arsenic	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Copper	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Hexavalent Chromium	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Lead	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL

Parameter	Bull Trout		Steelhead		Fall Chinook		Spring/Summer Chinook		Sockeye Salmon	
	Species	**Habitat**	**Species**	**Habitat**	**Species**	**Habitat**	**Species**	**Habitat**	**Species**	**Habitat**
Nickel	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Zinc	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL
Overall Effect to Species	**L**	**L**	**L**	**L**	**L**	**L**	**L**	**L**	**L**	**L**

NL = May affect, not likely to adversely affect species or not likely to adversely modify critical habitat
L = Likely to adversely affect species or likely to adversely modify critical habitat
*May affect, but not likely to adversely affect spring Chinook; Likely to adversely affect summer Chinook

7.7 Uncertainty Evaluation

7.7.1 Environmental Baseline

The measurement of chemicals associated with the discharge from the Clearwater facility has been sporadic at best. The station locations were predetermined based on an upstream and downstream scenario with respect to the Clearwater facility. However, there is no discussion of the hydrological or hydraulic conditions which will affect dispersion of the constituents associated with the effluent. Thus, the data reported for sediment concentrations represent estimates of the contaminant load, however, there is a great deal of uncertainty regarding how representative these samples are of the distribution of contaminants in the river. A quality assurance review of the data was not completed; therefore, the accuracy of the data is unknown.

Resident fish tissue monitoring has been conducted in accordance with the Clearwater facilities NPDES permit renewal to support the effort to characterize the potential effects of discharges from the facility on endangered and listed species. As stated previously in this report, none of the concentrations of chemicals analyzed in this sampling exceeded their respective benchmark criteria, although concentrations tended to be lower at reference stations on both the Clearwater and Snake rivers. Fish tissue data from another study (USEPA, 2002) were discussed in this BE.

In contrast, Washington State Department of Ecology (WA DOE 2011) conducted a fish tissue study of the Snake River that included fish sampled from six sites, including area around Lower Granite Dam. In this study sixty samples from ten species of fish were analyzed for mercury, chlorinated pesticides, polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and polychlorinated dibenzo-p-dioxins (PCDD/Fs). Results of this study found that no sites sampled met the state of Washington's water quality standards due to elevated levels of contaminants in one or more species of fish. All sites sampled had fish tissue concentrations that exceeded water quality standards for 4,4'-DDE, 4,4'-DDD, hexachlorobenzene, dieldrin, toxaphene, T-PCB, dioxins/furans, and mercury. The species with the highest number of exceedances included carp, catfish, and the northern pikeminnow. Trends in contaminant levels between sites and sampling years could not be ascertained most likely due to small sample sizes and variability within the populations sampled.

While the accuracy of the available fish tissue data is known, the fish contaminant surveys referenced above were not designed to estimate exposure to chemicals released from the Clearwater facility. Therefore, no definitive statements can be made regarding the chemical contaminants in these fish and the Clearwater facility, or can these fish data be directly correlated with sediment data collected in or near the Clearwater facility.

Other sources of constituents associated with the Clearwater facility include municipal treatment plants and hydroelectric facilities. None of these sources are described in detail nor are there specific data associated with their effluent discharges or operations. Without specific data for these other sources it is difficult to attribute any baseline data to the Clearwater facility.

Cumulative exposures to multiple stressors from other sources were evaluated in this permit by considering background concentrations of pollutants of concern, to the extent that data were available, in the water quality-based permitting analysis (i.e., reasonable potential determination and effluent limit calculation). Other exposures to stressful conditions may render the species more or less sensitive to the constituents in the Mill effluent. They also may alter the behavior of the species such that they are more or less likely to be exposed to releases from the Mill. The

lack of knowledge regarding the likelihood of cumulative exposures increases the uncertainty in the effects determination.

The sediment samples were collected as part of the investigation of chemicals associated with the Clearwater facility. However, they are also subject to the same errors in spatial and temporal variability as well as chemical analytical errors.

7.7.2 Ecological Effects Determinations

7.7.2.1 Parameters of concern

Chemical descriptions are based on general information regarding the physical, chemical, and biological behavior of the Mill effluent constituents in fresh water. Information on certain compounds (e.g., certain COCs) is extremely limited due to the lack of site specific data on these constituents.

The parameters of concern in this BE are those which have been measured in the effluent or which are controlled by effluent limitations guidelines (ELGs) for this industry. Not all of the parameters of concern were considered as having "reasonable potential" in the development of effluent limits for the draft permit. Other pollutants may be present in the effluent discharge, but at concentrations that are well below the applicable water quality standards.

Although the analytical detection levels of many of the pollutants that were not detected are greater than the water quality criteria, EPA did not include water quality-based effluent limits for these parameters because they are believed to be effectively controlled through the limitation of similar pollutants.

Even though other dioxin and furan congeners may be present in the effluent, studies EPA conducted during the development of the ELGs (USEPA, 1993) showed that 2,3,7,8-TCDD and 2,3,7,8-TCDF were the predominant chlorinated dibenzo-*p*-dioxins (CDDs) and chlorinated dibenzo-*p*-furans (CDFs) found in pulp and paper matrices. EPA is proposing to regulate 2,3,7,8-TCDD and 2,3,7,8-TCDF. It is assumed that the control of 2,3,7,8 TCDD and 2,3,7,8 TCDF will effectively minimize generation of the most toxic CDDs and CDFs.

7.7.2.2 Benchmark Determinations

Benchmarks are based on toxicity values or harmful levels of each parameter of concern which may derived from the literature using endangered species or extrapolations from surrogate species. There may be specific studies on the parameters associated with the effluent, and if not, extrapolation from similar chemicals must be made. The benchmark values may have undergone agency review (water quality criteria) or they may be site specific determination based on the judgment of the scientists preparing the Biological Evaluation.

7.7.2.3 Toxicity values

There are numerous uncertainties in the ecological effects assessment. These include uncertainties in (1) the toxicity values (NOELS, LOELS, etc.) used; (2) the toxicity equivalence factors developed for dioxins and furans; (3) chemical mixtures; (5) exposure frequency and duration; (5) cumulative exposures; (6) extrapolation across species; and (7) bioaccumulation.

Confidence in the selected toxicity benchmarks and subsequent confidence in the conclusions of the evaluation that uses the toxicity benchmarks depends upon the quality of the available toxicity data.

Ideally, to predict with the greatest accuracy whether an effect may be adverse, one would use toxicity data from an experiment that measures the type of toxicological response that is of interest, using the species of interest and the experimental design most easily extrapolated to the conditions of interest. However, there are no specific toxicity studies which are appropriate for this particular Biological Evaluation. Studies of mill effluents are discussed in Appendix J. However, these studies have not undergone peer review, therefore the results cannot be reviewed with confidence in the quality or accuracy of the results.

A thorough review of the scientific literature was conducted to identify as many sources of toxicity data for these parameters as possible. In some cases, toxicity data were obtained from a previously compiled collection of toxicity information. In other cases, individual papers published in the scientific literature were reviewed. In still other cases, the toxicity data used by EPA to derive water quality criteria were reviewed.

The quantity and quality of toxicity data available for permitted parameters varies widely in that many parameters contain numerous published studies, while some contain relatively few. Some use listed species, others use non-listed analogs (surrogates).

In some cases, the study reporting the lowest concentration for a parameter did not report the endpoint (e.g., NOEC, LOEC). For these chemicals, the lowest endpoint reported was used and then other endpoints were extrapolated using safety factors.

7.7.2.4 Extrapolation across species

Although using surrogate toxicity data from a similar species, life stage, or parameter increases the uncertainty associated with the BE, this approach is preferable to omitting the evaluation of a species or parameter with no toxicity data.

Actual direct testing of potential toxicity has not been conducted for all chemicals and listed species. While some toxicity data have been collected for nearly all the parameters of concern, toxicity data are generally not available for every life stage of a listed species. In cases where little or no toxicity data are available for a parameter of concern to each life stage of a listed species, toxicity data from a similar parameter, species, or life stage was used as a surrogate.

The surrogate species were selected as the closest related organism for which information was available. When a tested species is more sensitive or about equally sensitive to a non-tested species, the tested species can be considered a suitable surrogate for the non-tested species. Therefore, in this BE rainbow trout often served as a surrogate species to determine the effects of toxic pollutants on salmonids.

7.7.2.5 Extrapolation across chemicals

For some species there is no data on specific chemicals nor are there data for extrapolation across chemicals or species. For avian species there are no data in the literature for the effects of guaiacols, and catecols. A determination of likely to cause adverse effects was made although there are no data to support this position. Since there are some data to suggest that pentachlorophenol is in fact toxic to birds, it was suspected that other semi-volatile chlorinated organic chemicals may also be toxic. However, there are no data to support or refute this statement. The goal of this BE is to set limits which are protective of the endangered species. It was therefore assumed that in lieu of any data to the contrary a determination that there is a

likelihood of adverse effects will be made when there are no toxicity data to make a site-specific determination.

7.7.2.6 Chemical mixtures

Modeled determinations of a toxicity unity were made for the chlorinated organic compounds. In this case these chemicals were assumed to each have the same toxicological endpoint and that the cumulative effects were additive. There are no data in the literature to substantiate this determination. Synergistic effects were determined from the whole effluent toxicity (WET) testing.

The toxicity equivalence procedure was used to assess the mixture of dioxins (TCDD and furans associated with the Clearwater effluent. Toxicity equivalence factors were calculated using all of the available data and were selected to account for uncertainties in the available data and to avoid underestimating risk (Van den Berg et al., 1998). Alternative approaches, including the assumption that all dioxins and furans - carry the toxicity equivalence of 2,3,7,8-TCDD, or that all chlorinated dioxins and furans other than 2,3,7,8-TCDD can be ignored, have been generally rejected as inadequate by EPA and many other countries and international organizations.

7.7.2.7 Indirect effects to prey species

The effects determination relied on the salmonids as a measure of the effect on fish prey species and the benthic community. The impact of smothering to the benthic community was discussed in the section on total suspended solids. Benthic sampling and analysis occurred in 2005 upstream and downstream of the facility to fulfill NPDES permit requirements at the Clearwater facility. Sampling occurred at two upstream reference stations, one in the Clearwater River and one in the Snake River, and six sampling locations downstream of the Clearwater facility. Results of the sampling indicate no influence on the downstream invertebrate community related to the effluent from the Clearwater facility. No significant relationships exist between benthic metrics (i.e. taxa richness, abundance, percent dominant taxa, and tolerance index) and concentrations of chemicals measured in sediment. The only significant relationship observed was between metrics and percent fine sand and water temperature. Therefore, these results support the findings in the various biological opinions that the re-issuance of Clearwater's NPDES permit is not likely to jeopardize the continued existence of Snake River Steelhead, Snake River spring/summer and fall chinook salmon, and Snake River sockeye salmon nor result in the destruction or adverse modification of the designated critical habitat of these species.

7.7.2.8 Exposure

Salmonid exposure to harmful substances in this Biological Evaluation includes direct uptake or contact through absorption across the gills and ingestion of water, food, or sediments. Ingestion of food or prey items by salmonids species is estimated using bioaccumulation factors derived from empirical relationships of sediment and fish tissue data collected from the Columbia Basin near the Clearwater facility.

Caged bivalve tissue monitoring studies were conducted within the vicinity as well as downstream of Outfall 001 in accordance with NPDES permit renewal requirements to support the effort to characterize the potential effects of discharges from the facility on endangered and listed species. The results of this study indicate that tissue concentrations at the reference stations tended to be very similar to concentrations found at downstream sample stations. Most analytes

found at sample stations were also found at reference stations on the Clearwater and Snake rivers. These results indicate no influence from Outfall 001 on endangered and listed species.

7.7.2.9 Frequency and duration

In addition to the toxicity studies the determination of a benchmark involves some evaluation of exposure. The modeling used to estimate effluent concentrations may under or overestimate the concentrations depending on spatial and temporal variability as well as model error.

Exposure duration is defined as the time period over which an organism is exposed to one or more contaminants. The exposure duration used in this BE is a daily average. The model was based on a daily average. It is expected that all actions in this permit were below acute levels at or near the effluent release (jet action). The determination of likelihood of adverse effects is based on an assumption that an average daily exposure is adequate to address the habitat preferences of the endangered species.

7.7.2.10 Bioaccumulation

Ideally, to understand the relationship between the water column concentration and fish tissue, site-specific bioaccumulation factors (BAFs) are needed for each non-ionic organic compound of interest. BAFs account not only for uptake via water exposure, but also uptake through the food chain and the influence of sediment concentrations on both the food and water routes. Thus, site-specific BAFs reflect the disequilibrium of the compound in the major components of the local system (water, sediment, organisms) and should not be transferred from one system to another without considering carefully the underlying assumptions in so doing. In general, one would prefer to develop and run a model for a specific location that reflects back the state of the system (i.e., describing how each compound is partitioning into organic matter in the water column and sediments, and into lipids in the organisms of different trophic states, and how higher-level organisms are obtaining their tissue concentrations through food, water, and sediment exposures).

In 2006, the NPDES permit required Potlatch corporation to conduct resident fish tissue monitoring downstream, and near, Outfall 001. The sampling program was meant to characterize the potential effects of the discharge to the Clearwater and Snake Rivers, on endangered and listed species and the environment (Anchor Environmental 2008a). 8 locations were chosen, and 3 replicates were made per location, per species caught, for both fillet and whole-body samples. After chemical analyses, it was determined that none of the concentrations of targeted chemicals exceeded their benchmark criteria; many were detected in sample stations downstream from the discharge, as well as at the reference stations upstream of it, although concentrations did tend to be lower at those reference locations (Anchor Environmental 2008a). An additional caged bivalve tissue monitoring study, required by the same NPDES permit, was run from April through May 2007. The study consisted of 16 sample collections: 10 composite samples from below the outfall, one field duplicate, 2 baseline pre-deployment composites, and 3 upstream reference site composites (Anchor Environmental 2008b). At the time of the study, there were no toxicity benchmarks for bivalves within the permit, and most analytes detecting in downstream samples were also detected in the upstream reference samples. Upon further examination, it was determined that the same analytes were present in fairly equal quantities in both the upstream reference locations, and downstream locations (Anchor Environmental 2008b).

Such a model and supporting data do not currently exist for the ecosystem near the effluent discharge. The analysis in this document draws upon the work done at the Great Lakes regarding bioaccumulation of TCDD and TCDF into Lake Trout and other fish species, and adjusts, to the extent possible, the Great Lakes BAFs for Snake River conditions using estimated national values for dissolved organic carbon (DOC) and particulate organic carbon (POC) (Burkhard, EPA, personal communication; Oct 30, 2003) that appear to be very similar to values recently measured in the Snake River (Harrison, U of Idaho, personal communication; Oct 30, 2003)Great Lakes BAFs for TCDD and TCDF were adjusted by using Snake River DOC and POC. However, the underlying assumption that trophic structure and sediment contributions to the water column and biota are similar between the Great Lakes and the Snake River remains to be evaluated with site-specific information from the Snake River.

The lack of estimating bioaccumulation for other chlorinated organic chemicals results in an uncertainty in the benchmark estimates for these chemicals. The resultant benchmarks may not be fully protective of the endangered species. This may result in a determination of likely to adversely affect the species.

The site-specific estimate of the bioaccumulation of 2,3,7,8 TCDD and 2,3,7,8 TCDF by salmonids was based on fish tissue data collected during a survey of contaminants in fish from the Columbia River Basin and measurement of these chemicals in sediments by Clearwater Corporation. Combining these two disparate datasets to estimate a biota-sediment accumulation factor (BSAF) and a bioaccumulation factor (BAF) was necessary to estimate a benchmark for tissue levels in fish. However, these datasets were not collected using the same sampling or analytical methods; thus, there is a large uncertainty in combining them to achieve the necessary benchmark. While these data are the best available site-specific information, these surveys were not designed as a synoptic dataset. The site locations were not similar nor were the species necessarily exposed to chemicals from the locations where they were located. Even though largescale suckers and white sturgeon are resident species they do have a rather large home range. Thus, there exposures to chemical contaminants could have occurred in locations far removed from the site where sediment chemical data were collected. While these species may reflect resident fish, they are a surrogate for the salmonids which are the subject of this Biological Evaluation. The life history of large scale suckers and white sturgeon is not the same as the salmonids.

Three parameters that form the basis for the estimate of bioaccumulation benchmarks for 2,3,7,8 TCDD and 2,3,7,8 TCDF (BSAFs, Π/Kow, and the ratio of TCDF/TCDD) were varied in order to understand their influence on the benchmark calculations. As shown in Figure 7-33, Π/Kow defines a line of possible relationships between water and sediment concentrations for a given ecosystem condition. Changing BSAFs or the ratio of Furan/Dioxin in the water column and sediments influences where on the Π/Kow line the paired sediment/water concentrations are located. Increasing Π/Kow lowers the water concentration but not the sediment concentration. Increasing the BSAF lowers both water and sediment (vice-versa for increasing the BSAF). As the ratio of TCDF/TCDD declines there is a very slight increase in TCDD benchmark, but a large decrease in TCDF benchmark. Table 7-31 provides a range in benchmarks due to varying input parameters illustrating the need for site-specific BSAFs and food web parameters to reduce uncertainty.

Table 7-31: Summary Table of Sensitivity in Water and Sediment Benchmark Calculations

	Range	Default
TCDD water-fd (pg/L)	0.0002-0.003	0.0020
TCDF water-fd (pg/L)	0.0019-0.058	0.0165
TCDD sed-oc (ng/kg)	40.6-165	103.4
TCDF sed-oc (ng/kg)	121-919	261.1
Range in benchmarks as a result of varying input parameters illustrating the need for site-specific BSAFs and food web parameters to reduce uncertainty		

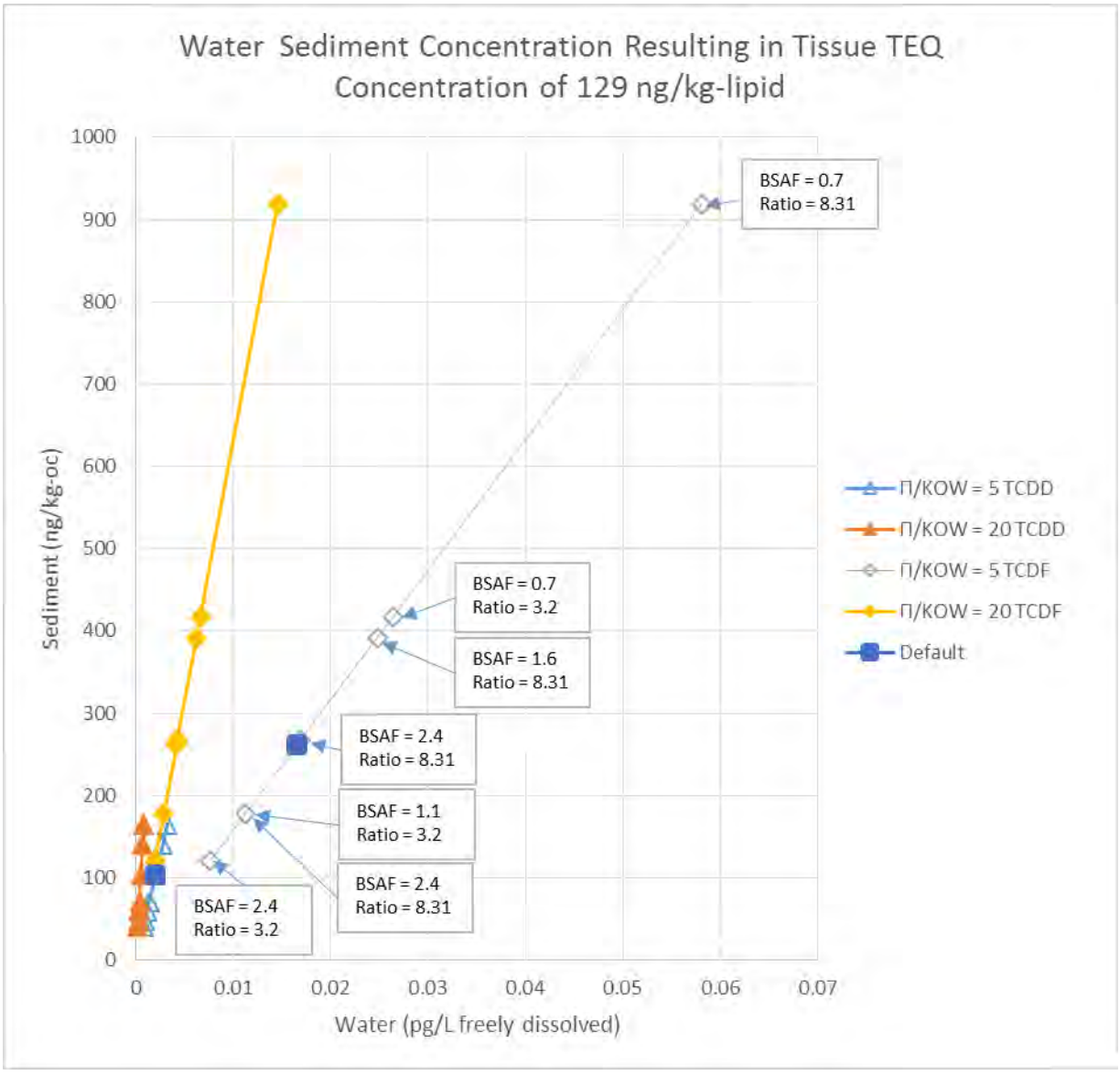


Figure 7-33: II/Kow effects to relationships between water and sediment concentrations for a given ecosystem condition.

7.7.2.11 Species Life history

While the life history of the species is discussed in general terms there may be site specific behavior or habitat factors which limit or alter the species preferences. A review of the scientific literature found that younger life stages of fish are generally more sensitive to chemical toxicants than older fish (e.g., Buhl, 1997; Hutchinson et al., 1998), though this was not always found to be the case. Mayes et al. (1983) did not find fathead minnow fry, juveniles, or adults to vary significantly in sensitivity to nine organic compounds tested. Additionally, Ingersoll et al. (1990) found that the sensitivity of brook trout to aluminum toxicity increased with age.

Relative sensitivity likely varies depending on the substance used in the toxicity test, the toxicological effect observed (e.g., survival, growth) (Pickering et al., 1996), and the endpoint measured (e.g., NOEC, LOEC). This seems to be the case with aquatic invertebrates. Hutchinson et al. (1998) analyzed EC_{50} and NOEC data from the European Centre for Ecotoxicology and Toxicology of Substances (ECETOC) Aquatic Toxicity database and found that based on EC_{50} data, juvenile invertebrates exhibited equal or greater sensitivity than adults to 54% of substances, while they exhibited equal or greater sensitivity than adults to 91% of substances based on NOEC data. While some investigators have found that younger invertebrates are more sensitive to some contaminants than older life stages (Nebeker et al., 1984), others found that older and younger invertebrates exhibited similar sensitivities to acute toxicity (Nebeker et al., 1986).

7.7.2.12 Effluent concentrations

Models and existing data were used to estimate how the proposed effluent limitations could potentially cause exposure (magnitude, frequency, and duration). A discussion of the exposure volume model inputs and assumptions is provided in Appendix D. All effluent values used in this analysis are final effluent concentrations that are calculated from the permit limitations required by this permit. To comply with this requirement, the BE uses the following recently collected data:

- Parameter measurements in effluent from 2003 - 2016;
- Parameter measurements in the Clearwater River and Snake River upstream and downstream of the diffuser collected in 2005 and 2006 to characterize the environmental baseline; and
- The data used in this permit was collected during 2005 and 2006 from the effluent downstream of the diffuser. These data are subject to sampling and analytical uncertainty due to measurement error.

A determination of “may affect, not likely to adversely affect” was made for direct effects when the water column benchmark was met within the jet plume (~10 feet downstream of the diffuser) and a determination of “likely to adversely affect” was made for direct effects when the water column benchmark was met beyond the jet plume and the species orientation in the water column could result in exposure or when no information was available detailing the toxicity of the chemical with regard to the species of concern or an acceptable surrogate. This determination is appropriate because it is improbable that aquatic species would be exposed to concentration in the jet plume of the diffuser due to the velocity and physics of the discharge. However, as the jet momentum of the plume dissipates due to buoyant spreading, the potential for aquatic life exposure increases.

7.8 Conservation Measures

As well as establishing effluent limits for the Outfall 001 discharge, the 2019 draft permit also includes requirements for effluent monitoring, preparation and implementation of a Quality Assurance Plan (QAP) and a Best Management Practices (BMP) Plan. Each of these additional actions is described below.

Implementation of the following actions will allow EPA 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. The following actions are included in the draft permit (see Appendix A).

7.8.1 Effluent Monitoring

The draft permit requires effluent monitoring at Outfall 001 for the limited parameters and for other parameters of concern including flow, production, phosphorus, ammonia, nitrite plus nitrate nitrogen, and whole effluent toxicity (WET). Additionally, the draft permit requires daily monitoring of influent water for organic content, such as COD or total organic carbon (TOC) and seepage from the secondary treatment pond to be monitored in the first and fourth years of the permit, and monitoring of groundwater from seven wells for 2,3,7,8-TCDD, ammonia, nitrite plus nitrate nitrogen and phosphorus. The effluent monitoring requires analysis of each chemical constituent as total recoverable, rather than the dissolved fraction. This allows the analysis of the effects from this discharge for both the water column and the particulate matter. The purpose of effluent monitoring is to measure the quality of the effluent being discharged into the receiving water and to ensure compliance with the permitted effluent limitations. See section I.B of the draft permit located in Appendix A for more details regarding the effluent monitoring requirements.

7.8.2 Internal Monitoring

In order to control chlorinated organics, the permit requires monitoring of the total discharge of process wastewaters from each physical bleach line, designated 011 (chip line) and 021 (sawdust line), of the bleach plant operated at the mill. The monitoring locations are at the effluent from each line prior to commingling with any other waste stream. The purpose of internal monitoring is to monitor the source of chlorinated organics at the source (where they are in a measurable concentration) prior to dilution with other waste streams at the Mill. See section I.F of the draft permit located in Appendix A for more details regarding the internal monitoring requirements.

7.8.3 Whole Effluent Toxicity (WET) Monitoring

In order to determine potential toxic effects of the discharge, the draft permit requires regular chronic WET testing of effluent from Outfall 001. The test species required for this permit include *Ceriodaphnia dubia* and *Pimephales promelas*. 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. These species are currently the best available surrogates for assessing impacts on the listed fish species and, therefore, toxicity testing using these species will give an indication of toxicity from the whole effluent. A caged bivalve study will be done in addition to the whole effluent toxicity testing. See section I.E. of the draft permit located in Appendix A for more details regarding the WET monitoring requirements.

7.8.4 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 permit requires the Clearwater Mill 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.C. of the draft permit located in Appendix A for more details regarding the BMP plan requirements.

7.8.5 Toxicity Reduction Evaluation (TRE)

The draft permit requires the development of an initial investigation Toxicity Reduction Evaluation (TRE) Work Plan. A TRE is an evaluation to reduce an effluent's toxicity or chemical concentration(s) to acceptable levels when it is found to be toxic as a result of a WET test. The TRE Work Plan describes the steps the permittee intends to follow if toxicity is detected above the chronic WET trigger level established in the permit. The TRE Work Plan include, at a minimum, information and data acquisition, a performance evaluation of the facilities wastewater treatment and BMPs, a Toxicity Identification Evaluation (TIE), actions that will be taken to mitigate the impact of the discharge and to prevent the recurrence of toxicity, and a schedule for the TRE process. The TRE may identify a remedial action as simple as improved BMPs or the need to modify the operation of a component of the wastewater treatment system. See section II.A of the draft permit located in Appendix A for more details regarding the TRE requirements.

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8 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. ID0001163 for discharges of process wastewaters and storm water from Clearwater Paper Corporation.

8.1 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.

8.2 Description of the Project/Proposed Activity

EPA is proposing to reissue an NPDES permit to the Clearwater Corporation for the Clearwater Mill in Lewiston, Idaho. The project is described in Section 3, "Description of Action."

8.3 Potential Adverse Effects of the Proposed Project

The Clearwater Paper Action Area is within the designated EFH for Chinook and coho salmon.

Water quality is an important component of EFH. The effects of authorized discharges from Clearwater on Chinook and coho salmon EFH within the action area for this permit are the same as those described for fish species of concern in Sections 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 7, USEPA has determined that issuance of the Clearwater Paper permit is likely to adversely affect Chinook and coho salmon EFH in the vicinity of the discharges.

8.4 EFH Conservation Measures and Conclusion

Clearwater Paper Corporation will monitor the effluent discharges from both outfalls following NPDES requirements. The proposed permit requires Clearwater Paper 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 and coho salmon EFH.

USEPA concludes that the proposed action may adversely affect EFH for Chinook and coho salmon.

8.5 References

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.

Appendix A: Draft NPDES Permit for the Clearwater Mill

Appendix B: Fact Sheet

Appendix C: Surface Water Data

Table C- 1: Mean Results from Weekly Receiving Water Study, 2005

Parameter	Unit	Value	SR-REF-S	SR-REF-MD	CR-REF-S	LGP-13-S	LGP-13-MD	LGP-11-S	LGP-11-MD	LGP-09-S	LGP-09-MD	LGP-06-S	LGP-06-MD	LGP-01-S	LGP-01-MD
TSS	mg/L	**Average**	3.088	3.265	2.5	2.338	2.765	2.5	2.765	2.647	2.5	2.5	2.647	3.676	
		Median	2.5	2.5	2.5	1.25	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	
		Min	2.5	2.5	2.5	1.25	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	
		Max	8	12	2.5	8	7	2.5	7	5	2.5	2.5	5	9	
BOD	mg/L	**Average**	0.556	0.459	0.462	0.431	0.462	0.476	0.515	0.529	0.476	0.606	0.474	1.809	0.556
		Median	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	1.3	0.25
		Min	0.25	0.25	0.25	0.125	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25
		Max	1.3	1.5	1.7	1.5	1.1	1.2	1	1.3	1.4	1.9	1.2	7	1.2
Ammonia Nitrogen	mg/L	**Average**	0.013	0.014	0.012	0.019	0.019	0.017	0.018	0.0137	0.0152	0.013	0.0141	0.031	0.026
		Median	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.02
		Min	0.006	0.004	0.005	0.005	0.002	0.01	0.01	0.008	0.008	0.01	0.01	0.01	0.01
		Max	0.03	0.03	0.03	0.11	0.1	0.09	0.04	0.03	0.05	0.03	0.04	0.09	0.06
Nitrate/Nitrite Nitrogen	mg/L	**Average**	0.37	0.372	0.014	0.331	0.269	0.329	0.265	0.324	0.261	0.298	0.264	0.206	0.233
		Median	0.38	0.38	0.012	0.3	0.21	0.32	0.22	0.32	0.21	0.29	0.24	0.14	0.17
		Min	0.11	0.11	0.002	0.13	0.09	0.12	0.09	0.11	0.06	0.07	0.05	0.045	0.042
		Max	0.62	0.61	0.035	0.57	0.51	0.55	0.52	0.56	0.53	0.49	0.48	0.49	0.48
TKN	mg/L	**Average**	0.294	0.261	0.254	0.268	0.273	0.237	0.253	0.266	0.267	0.282	0.271	0.435	0.3
		Median	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.3	0.2	0.3	0.3
		Min	0.1	0.035	0.035	0.2	0.035	0.035	0.1	0.15	0.035	0.2	0.1	0.2	0.1
		Max	0.5	0.5	0.7	0.45	0.5	0.4	0.5	0.5	0.9	0.5	0.7	1.5	0.5
Total Phosphorous	mg/L	**Average**	0.071	0.072	0.008	0.067	0.056	0.068	0.057	0.068	0.058	0.061	0.057	0.066	0.058
		Median	0.08	0.08	0.008	0.065	0.05	0.07	0.06	0.07	0.05	0.06	0.06	0.06	0.06
		Min	0.04	0.04	0.003	0.03	0.02	0.03	0.03	0.02	0.03	0.03	0.03	0.03	0.03
		Max	0.1	0.12	0.02	0.11	0.1	0.11	0.1	0.12	0.11	0.11	0.09	0.13	0.1
Orthophosphate Phosphorous	mg/L	**Average**	0.056	0.056	0.004	0.053	0.041	0.053	0.045	0.053	0.043	0.048	0.043	0.038	0.042
		Median	0.07	0.06	0.005	0.055	0.04	0.06	0.04	0.055	0.04	0.05	0.05	0.03	0.04

Parameter	Unit	Value	SR-REF-S	SR-REF-MD	CR-REF-S	LGP-13-S	LGP-13-MD	LGP-11-S	LGP-11-MD	LGP-09-S	LGP-09-MD	LGP-06-S	LGP-06-MD	LGP-01-S	LGP-01-MD
		Min	0.01	0.02	0.001	0.02	0.01	0.01	0.02	0.01	0.01	0.01	0.009	0.01	0.01
		Max	0.09	0.09	0.01	0.09	0.085	0.09	0.09	0.1	0.09	0.1	0.08	0.08	0.09
2,3,7,8-TCDD	pg/L	**Average**	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Median	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Min	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Max	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,3,7,8-TCDF	pg/L	**Average**	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Median	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Min	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Max	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table C- 2: Mean Velocity, pH, DO, and Temperature Results from Weekly Receiving Water Study, 2005.

Parameter	Unit	Statistic	SR Ref	CR Ref	LGP-13	LGP-09	LGP-06	LGP-01
Velocity	ft/sec	**Average**	0.22	1.05	0.16	0.22	0.21	0.16
		Median	0.22	0.97	0.14	0.15	0.16	0.19
		Min	0.06	0.39	0.05	0.06	0.03	0.02
		Max	0.4	1.77	0.39	0.55	0.56	0.31
pH	SU	**Average**	8	7.73	8.2	8.12	8.08	7.99
		Median	8.01	7.8	8.18	8.16	8.07	7.99
		Min	7.46	7.31	8.04	7.75	7.79	7.56
		Max	8.56	8.09	8.48	8.37	8.72	8.68
DO	mg/L	**Average**	8.5	10.6	9.6	9.9	9.6	9.6
		Median	8.8	11	9.6	10	9.5	9.5
		Min	6.2	8	8.1	9.1	8.4	9.1
		Max	10.4	12.8	11.1	10.9	10.6	10.4
Temperature	°C	**Average**	19.3	11.7	16.42	16.81	17.4	18.06
		Median	20.6	11.62	16.5	16.98	17.85	19.03
		Min	14.07	9.66	14.61	14.31	14.41	14.53
		Max	23.09	14.44	18.9	19.14	19.82	20.33

Table C- 3: Results from Quarterly Surface Water and Effluent Monitoring Study, 2005.

Parameter	Unit	Type	SR Ref	CR Ref	LGP-13	LGP-09	LGP-06	LGP-01	LGP-11	LGP-14
Phytosterols	µg/L	**B-sitosterol**	0.563	0.46	1.1985	0.679	0.679	3.39	0.632	1.36
	µg/L	**Campesterol**	0.05	0.038	0.0635	0.054	0.054	0.27	0.044	0.059
	µg/L	**Stigmastanol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Stigmasterol**	0.06	ND	0.082	0.07	0.062	0.331	0.06	0.076
	µg/L	**Retene**	ND	ND	ND	ND	ND	ND	ND	ND
Resin Acids	µg/L	**Abietic Acid**	0.009	0.021	0.0265	0.035	0.034	0.02	0.021	0.021
	µg/L	**Dehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Isopimaric Acid**	0.002	0.003	ND	0.006	0.005	0.003	0.003	ND
	µg/L	**Neoabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Palustric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Sandaracopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Total 12/14-Chlorodehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
Chlorophenolics	µg/L	**2,4,6-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,5-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,3,4,6-Tetrachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,6-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,6-Trichoroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Trichlorosyringol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**4,5,6-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pentachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND

Parameter	Unit	Type	SR Ref	CR Ref	LGP-13	LGP-09	LGP-06	LGP-01	LGP-11	LGP-14
	µg/L	**Tetrachloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
Chloroform	µg/L		ND	ND	ND	ND	ND	0.19	ND	ND
DOC	mg/L		2.7	1.9	2.5	2.9	2.8	3.1	11.8	2.5
TOC	mg/L		2.8	2.1	2.6	2.9	2.9	2.9	2.8	2.9
Dioxins	pg/L	**2,3,7,8-TCDD**	ND	0.00125	0.022	ND	ND	ND	ND	ND
	pg/L	**1,2,3,7,8-PeCDD**	ND	0.00125	0.051	ND	0.002125	ND	ND	ND
	pg/L	**1,2,3,4,7,8-HxCDD**	0.003	0.003	0.008	0.003	0.0025	ND	ND	0.003
	pg/L	**1,2,3,6,7,8-HxCDD**	ND	0.002	0.04	0.011	ND	0.012	0.007	0.016
	pg/L	**1,2,3,7,8,9-HxCDD**	0.011	0.004	0.038	0.007	0.0085	0.008	0.007	0.011
	pg/L	**1,2,3,4,6,7,8-HpCDD**	0.214	0.056	0.151	0.203	0.148	0.266	0.128	0.186
	pg/L	**OCDD**	0.81	0.307	0.478	1.01	0.8505	1.66	0.748	0.902
Furans	pg/L	**2,3,7,8-TCDF**	ND	0.00125 U	0.006	0.005	0.005	0.004	0.006	0.006
	pg/L	**1,2,3,7,8-PeCDF**	ND	0.00125 U	ND	ND	0.003	ND	ND	ND
	pg/L	**2,3,4,7,8-PeCDF**	ND	0.00125 U	0.003	0.003	ND	ND	0.003	ND
	pg/L	**1,2,3,4,7,8-HxCDF**	ND	0.00125 U	ND	0.004	0.004	ND	0.005	0.006
	pg/L	**1,2,3,6,7,8-HxCDF**	ND	0.00125 U	ND	0.003	0.0035	ND	0.004	0.003
	pg/L	**1,2,3,7,8,9-HxCDF**	ND	0.00125 U	ND	ND	ND	ND	ND	ND
	pg/L	**2,3,4,6,7,8-HxCDF**	ND	0.00125 U	ND	ND	0.002125	ND	ND	ND
	pg/L	**1,2,3,4,6,7,8-HpCDF**	0.018	0.006	ND	0.022	0.025	0.03	0.018	0.028
	pg/L	**1,2,3,4,7,8,9-HpCDF**	ND	0.00125 U	ND	ND	ND	ND	0.003	ND
	pg/L	**OCDF**	0.02	0.011	0.007	0.037	0.0375	0.045	0.024	0.026

Table C- 4: Mean Velocity, pH, DO, and Temperature Results from Weekly Receiving Water Monitoring Study, 2006.

Parameter	Unit	Type	SR Ref	CR Ref	LGP-13	LGP-09	LGP-06	LGP-01	LGP-11
Velocity	ft/sec	**Average**	0.37	1.51	0.33	0.27	0.27	0.22	0.30
		Median	0.23	0.77	0.25	0.22	0.19	0.18	0.22
		Min	0.15	0.32	0.07	0.10	0.08	0.01	0.07
		Max	1.12	3.67	1.10	0.73	1.23	0.66	0.77

Parameter	Unit	Type	SR Ref	CR Ref	LGP-13	LGP-09	LGP-06	LGP-01	LGP-11
pH	SU	**Average**	7.92	8.23	8.23	8.17	8.22	8.18	8.26
		Median	8.01	8.20	8.27	8.22	8.28	8.28	8.25
		Min	7.47	6.89	7.70	7.71	7.62	7.55	7.82
		Max	8.51	9.06	8.57	8.75	8.67	9.05	8.64
DO	mg/L	**Average**	7.55	9.85	8.00	8.14	8.17	8.34	7.85
		Median	7.63	9.95	8.01	8.05	8.11	8.42	7.84
		Min	6.48	9.13	7.59	7.78	7.60	7.70	7.57
		Max	8.16	10.61	8.67	8.69	8.84	9.29	8.18
Temperature	°C	**Average**	20.23	11.60	17.79	18.19	18.45	19.05	18.56
		Median	21.22	10.95	17.92	18.36	18.62	19.38	18.63
		Min	14.61	9.87	14.38	14.47	14.36	14.59	14.45
		Max	23.72	15.23	21.12	20.42	20.60	22.35	21.08

Table C- 5: Mean Results from Weekly Receiving Water Monitoring Study, 2006

Parameter	Unit	Value	SR-REF-S	SR-REF-MD	CR-REF-S	LGP-13-S	LGP-13-MD	LGP-11-S	LGP-11-MD	LGP-09-S	LGP-09-MD	LGP-06-S	LGP-06-MD	LGP-01-S	LGP-01-MD
TSS	mg/L	**Average**	3.588	4.500	ND	3.235	2.765	ND	ND	ND	ND	2.912	3.588	3.412	3.559
		Median	2.5	2.5	ND	2.5	2.5	ND	ND	ND	ND	2.5	2.5	2.5	2.5
		Min	2.5	2.5	ND	2.5	2.5	ND	ND	ND	ND	2.5	2.5	2.5	2.5
		Max	16	25	ND	7	7	ND	ND	ND	ND	7	8	7	10
BOD	mg/L	**Average**	1.588	1.4	2.782	1.71	2.17	2.27	2.17	9.37	5.63	1.25	1.31	1.61	1.51
		Median	1	1.9	2	2	2	2	2	1.45	1.1	1	0.8	2	1.3
		Min	0.5	0.5	0.7	0.5	0.5	0.5	0.5	0.65	0.5	0.5	0.5	0.5	0.5
		Max	7	2	22	7	14	16	14	137	74	2.3	3.2	3	4
Ammonia Nitrogen	mg/L	**Average**	0.029	0.022	0.02	0.022	0.022	0.021	0.023	0.021	0.022	0.022	0.022	0.027	0.028
		Median	0.025	0.025	0.025	0.025	0.021	0.021	0.025	0.024	0.022	0.025	0.023	0.025	0.025
		Min	0.007	0.009	0.007	0.006	0.007	0.006	0.014	0.012	0.011	0.009	0.006	0.01	0.011
		Max	0.11	0.043	0.044	0.036	0.047	0.042	0.039	0.04	0.037	0.037	0.043	0.05	0.05

Parameter	Unit	Value	SR-REF-S	SR-REF-MD	CR-REF-S	LGP-13-S	LGP-13-MD	LGP-11-S	LGP-11-MD	LGP-09-S	LGP-09-MD	LGP-06-S	LGP-06-MD	LGP-01-S	LGP-01-MD
Nitrate/Nitrite Nitrogen	mg/L	**Average**	0.355	0.341	0.025	0.335	0.292	0.335	0.286	0.321	0.283	0.287	0.276	0.244	0.247
		Median	0.35	0.38	0.025	0.35	0.26	0.36	0.27	0.34	0.27	0.33	0.28	0.22	0.18
		Min	0.11	0.09	0.007	0.09	0.08	0.1	0.07	0.09	0.08	0.049	0.08	0.019	0.049
		Max	0.59	0.61	0.045	0.6	0.53	0.58	0.54	0.56	0.55	0.56	0.53	0.52	0.53
TKN	mg/L	**Average**	0.332	0.324	0.256	0.318	0.385	0.279	0.421	0.363	0.4	0.359	0.362	0.441	0.435
		Median	0.3	0.3	0.3	0.3	0.4	0.3	0.3	0.35	0.4	0.3	0.4	0.4	0.3
		Min	0.05	0.1	0.05	0.2	0.05	0.05	0.05	0.125	0.1	0.1	0.05	0.2	0.2
		Max	0.8	0.5	0.4	0.5	0.7	0.4	1.6	0.65	0.7	1	0.7	1	1.2
Total Phosphorous	mg/L	**Average**	0.062	0.058	0.009	0.056	0.051	0.054	0.052	0.064	0.054	0.058	0.056	0.052	0.052
		Median	0.06	0.06	0.01	0.05	0.05	0.05	0.05	0.06	0.05	0.05	0.05	0.05	0.05
		Min	0.03	0.03	0.004	0.03	0.02	0.03	0.02	0.025	0.02	0.03	0.02	0.03	0.03
		Max	0.1	0.08	0.01	0.09	0.09	0.08	0.08	0.16	0.12	0.14	0.14	0.09	0.08
Orthophosphate Phosphorous	mg/L	**Average**	0.046	0.046	0.005	0.045	0.04	0.043	0.041	0.045	0.042	0.042	0.041	0.035	0.035
		Median	0.05	0.05	0.005	0.05	0.04	0.05	0.04	0.05	0.04	0.05	0.04	0.03	0.03
		Min	0.02	0.02	0.003	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
		Max	0.07	0.07	0.01	0.07	0.07	0.06	0.08	0.075	0.09	0.07	0.07	0.06	0.07
2,3,7,8-TCDD	pg/L	**Average**	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Median	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Min	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Max	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
2,3,7,8-TCDF	pg/L	**Average**	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Median	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Min	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
		Max	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Table C- 6: Results from Quarterly Surface Water and Effluent Monitoring Study, 2006.

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
Nov. 2006	µg/L	**12,14-Dichlorodehydroabietic Acids**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Abietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Dehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Isopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Neoabietic Acid**	ND	ND	ND	ND	0.004	0.001	ND	ND
	µg/L	**Palustric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Sandaracopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Total 12/14-Chlorodehydroabietic Acid**	ND	ND	ND	ND	0.003	ND	ND	ND
Mar. 2006	µg/L	**12,14-Dichlorodehydroabietic Acids**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Abietic Acid**	0.107	ND	ND	ND	0.013	0.039	0.086	0.0265
	µg/L	**Dehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Isopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Neoabietic Acid**	0.001	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Palustric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Sandaracopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Total 12/14-Chlorodehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
Jun. 2006	µg/L	**12,14-Dichlorodehydroabietic Acids**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Abietic Acid**	0.147	0.079	0.073	1.073	0.077	0.145	0.136	0.0625
	µg/L	**Dehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Isopimaric Acid**	0.012	0.013	0.014	0.014	0.013	0.022	0.018	0.0145

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	µg/L	**Neoabietic Acid**	0.003	0.001	0.001	0.001	0.002	0.003	0.002	ND
	µg/L	**Palustric Acid**	ND	ND	ND	ND	0.003	ND	ND	ND
	µg/L	**Pimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Sandaracopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Total 12/14-Chlorodehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
Sept. 2006	µg/L	**12,14-Dichlorodehydroabietic Acids**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Abietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Dehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Isopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Neoabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Palustric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Sandaracopimaric Acid**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Total 12/14-Chlorodehydroabietic Acid**	ND	ND	ND	ND	ND	ND	ND	ND
Nov. 2006	µg/L	**Chloroform**	ND	ND	ND	ND	ND	ND	ND	ND
Mar. 2006	mg/L	**Chloroform**	ND	ND	ND	ND	ND	ND	ND	ND
Jun. 2006	mg/L	**Chloroform**	ND	ND	ND	ND	ND	ND	ND	ND
Sept. 2006	mg/L	**Chloroform**	ND	ND	ND	ND	ND	ND	ND	ND
Nov. 2006	µg/L	**B-sitosterol**	0.396	0.453	0.408	0.402	0.7	0.513	0.766	0.949
	µg/L	**Campesterol**	0.024	0.056	0.028	0.042	0.072	0.049	0.064	0.0805
	µg/L	**Stigmastanol**	ND	ND	ND	ND	ND	0.026	0.049	0.0695
	µg/L	**Stigmasterol**	0.032	0.047	0.038	0.042	0.059	0.044	0.059	0.0575
	µg/L	**Retene**	ND	ND	ND	ND	ND	ND	ND	ND
Mar. 2006	µg/L	**B-sitosterol**	0.735	1.91	1.66	1.66	1.73	2	2.02	2.485
	µg/L	**Campesterol**	0.04	0.094	0.103	0.103	0.113	0.121	0.111	0.1025
	µg/L	**Stigmastanol**	ND	ND	0.034	0.034	0.041	0.047	0.056	ND

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	µg/L	**Stigmasterol**	0.053	0.075	0.085	0.085	0.091	0.118	0.089	0.089
	µg/L	**Retene**	ND	ND	ND	ND	0.001	0.002	0.001	0.001
Jun. 2006	µg/L	**B-sitosterol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Campesterol**	0.026	0.062	0.08	0.08	0.063	0.05	0.073	0.0685
	µg/L	**Stigmastanol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Stigmasterol**	0.099	0.129	0.177	0.177	0.126	0.105	0.105	0.1445
	µg/L	**Retene**	ND	0.003	ND	ND	ND	ND	0.114	0.002
Sept. 2006	µg/L	**B-sitosterol**	0.2	0.19	0.165	0.165	0.26	0.39	0.49	0.57
	µg/L	**Campesterol**								
	µg/L	**Stigmastanol**								
	µg/L	**Stigmasterol**								
	µg/L	**Retene**	0.0069	ND	ND	ND	ND	ND	ND	ND
Nov. 2006	mg/L	**Doc**	2.1	2.2	2.2	2.2	2.1	2.2	2.4	2.5
	mg/L	**TOC**	2	2.2	2.3	2.3	2.2	2.2	2.5	2.5
	mg/L	**TSS**	ND	6	ND	9	7	5	8	6.5
Mar. 2006	mg/L	**Doc**	2.6	2.8	2.8	2.8	2.7	2.9	2.8	3.05
	mg/L	**TOC**	2.6	2.9	2.9	2.9	3.1	3.1	3	2.9
	mg/L	**TSS**	ND	6	6	6	8	10	9	3.75
Jun. 2006	mg/L	**Doc**	2.4	2.4	2.5	2.5	2.5	2.6	2.5	2.5
	mg/L	**TOC**	2.9	2.7	2.5	2.5	2.6	2.8	2.7	2.65
	mg/L	**TSS**	ND	25	30	30	22	22	35	10.5
Sept. 2006	mg/L	**Doc**	1.8	2.7	2.8	2.8	2.8	2.6	2.6	2.5
	mg/L	**TOC**	1.8	2.6	2.5	2.5	2.5	2.7	2.7	2.6
	mg/L	**TSS**	ND	6	ND	ND	ND	2.5	2.5	6
Nov. 2006	µg/L	**2,3,4,6-Tetrachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,5-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,6-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichlorocatechol**	ND	ND	ND	ND		ND	ND	ND

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	µg/L	**Pentachlorophenol**	ND	ND	ND	ND		ND	ND	ND
	µg/L	**3,4,6-Trichlorocatechol**	ND	ND	0.001	ND		ND	ND	ND
	µg/L	**Tetrachlorocatechol**	0.001	ND	ND	ND		ND	ND	ND
	µg/L	**3,4,5-Trichloroguaiacol**	ND	ND	ND	ND		ND	ND	ND
	µg/L	**3,4,6-Trichloroguaiacol**	ND	ND	ND	ND		ND	ND	ND
	µg/L	**4,5,6-Trichloroguaiacol**	ND	ND	ND	ND		ND	ND	ND
	µg/L	**Tetrachloroguaiacol**	ND	ND	ND	ND		ND	ND	ND
	µg/L	**Trichlorosyringol**	ND	ND	ND	ND		ND	ND	ND
Mar. 2006	µg/L	**2,3,4,6-Tetrachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,5-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,6-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pentachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,6-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Tetrachlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,6-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**4,5,6-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Tetrachloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Trichlorosyringol**	ND	ND	ND	ND	ND	ND	ND	ND
Jun. 2006	µg/L	**2,3,4,6-Tetrachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,5-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,6-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pentachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,6-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Tetrachlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	µg/L	**3,4,6-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**4,5,6-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Tetrachloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Trichlorosyringol**	ND	ND	ND	ND	ND	ND	ND	ND
Sept. 2006	µg/L	**2,3,4,6-Tetrachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,5-Trichlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**2,4,6-Trichlorophenol**	ND	ND	ND	ND	ND	ND	0.00057	ND
	µg/L	**3,4,5-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Pentachlorophenol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,6-Trichlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Tetrachlorocatechol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,5-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**3,4,6-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**4,5,6-Trichloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Tetrachloroguaiacol**	ND	ND	ND	ND	ND	ND	ND	ND
	µg/L	**Trichlorosyringol**	ND	ND	ND	ND	ND	ND	ND	ND
Nov. 2006	pg/L	**2,3,7,8-TCDD**	ND	ND	ND	ND	ND	ND	ND	ND
	pg/L	**1,2,3,7,8-PeCDD**	ND	ND	0.003	ND	ND	ND	ND	ND
	pg/L	**1,2,3,4,7,8-HxCDD**	0.003	ND	0.004	0.005	0.005	0.004	0.004	0.0045
	pg/L	**1,2,3,6,7,8-HxCDD**	0.005	0.007	0.016	0.014	0.013	0.009	0.009	0.0105
	pg/L	**1,2,3,7,8,9-HxCDD**	0.005	0.006	0.009	0.011	0.011	0.007	0.008	0.0095
	pg/L	**1,2,3,4,6,7,8-HpCDD**	0.113	0.136	0.345	0.278	0.21	0.19	0.164	0.215
	pg/L	**OCDD**	0.942	1.08	1.97	1.78	1.03	1.3	1.23	1.74
Mar. 2006	pg/L	**2,3,7,8-TCDD**	ND	ND	ND	ND	ND	ND	ND	ND
	pg/L	**1,2,3,7,8-PeCDD**	ND	ND	ND	ND	0.003	0.003	ND	ND
	pg/L	**1,2,3,4,7,8-HxCDD**	0.006	ND	ND	ND	0.006	0.007	ND	ND
	pg/L	**1,2,3,6,7,8-HxCDD**	0.01	0.01	ND	ND	0.014	0.023	0.017	0.0125
	pg/L	**1,2,3,7,8,9-HxCDD**	0.008	ND	ND	ND	0.012	0.016	ND	0.0095

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	pg/L	**1,2,3,4,6,7,8-HpCDD**	0.137	0.187	0.138	0.138	0.252	0.49	0.276	0.202
	pg/L	**OCDD**	0.973	1.05	0.968	0.968	2.06	2.71	1.73	1.195
Jun. 2006	pg/L	**2,3,7,8-TCDD**	ND	ND	ND	ND	ND	ND	ND	ND
	pg/L	**1,2,3,7,8-PeCDD**	0.003	0.004	0.004	0.004	0.005	0.003	0.005	ND
	pg/L	**1,2,3,4,7,8-HxCDD**	0.005	0.007	0.007	0.007	0.011	0.006	0.006	ND
	pg/L	**1,2,3,6,7,8-HxCDD**	0.01	0.017	0.023	0.023	0.025	0.016	0.02	0.015
	pg/L	**1,2,3,7,8,9-HxCDD**	0.008	0.013	ND	ND	0.025	0.011	0.014	ND
	pg/L	**1,2,3,4,6,7,8-HpCDD**	0.261	0.347	0.463	0.463	0.49	0.275	0.417	0.317
	pg/L	**OCDD**	2.2	2.63	4.09	4.09	2.48	1.95	3.53	2.4385
Sept. 2006	pg/L	**2,3,7,8-TCDD**	ND	ND	ND	ND	ND	ND	ND	ND
	pg/L	**1,2,3,7,8-PeCDD**								
	pg/L	**1,2,3,4,7,8-HxCDD**								
	pg/L	**1,2,3,6,7,8-HxCDD**								
	pg/L	**1,2,3,7,8,9-HxCDD**								
	pg/L	**1,2,3,4,6,7,8-HpCDD**								
	pg/L	**OCDD**								
Nov. 2006	pg/L	**2,3,7,8-TCDF**	ND	0.006	ND	0.008	0.007	ND	ND	ND
	pg/L	**1,2,3,7,8-PeCDF**	ND	ND	Nd	0.003	0.003	ND	ND	ND
	pg/L	**2,3,4,7,8-PeCDF**	ND	0.003	ND	0.004	0.004	0.003	ND	ND
	pg/L	**1,2,3,4,7,8-HxCDF**	ND	0.004	0.003	0.011	0.01	0.006	0.004	0.006
	pg/L	**1,2,3,6,7,8-HxCDF**	ND	0.003	0.003	0.004	0.004	0.003	0.003	0.004
	pg/L	**1,2,3,7,8,9-HxCDF**	ND	ND	ND	ND	ND	ND	ND	ND
	pg/L	**2,3,4,6,7,8-HxCDF**	ND	ND	ND	0.003	ND	ND	ND	ND
	pg/L	**1,2,3,4,6,7,8-HpCDF**	0.018	0.026	0.032	0.041	0.033	0.028	0.031	0.0385
	pg/L	**1,2,3,4,7,8,9-HpCDF**	ND	0.004	0.003	0.004	0.003	0.003	0.003	0.004
	pg/L	**OCDF**	0.035	0.048	0.069	0.075	0.059	0.051	0.06	0.077
Mar. 2006	pg/L	**2,3,7,8-TCDF**	ND	0.006	0.008	0.008	ND	0.007	0.009	ND
	pg/L	**1,2,3,7,8-PeCDF**	ND	ND	ND	ND	0.004	0.003	0.003	ND
	pg/L	**2,3,4,7,8-PeCDF**	ND	0.003	ND	ND	0.005	0.004	0.003	0.003

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	pg/L	**1,2,3,4,7,8-HxCDF**	0.003	0.003	ND	ND	0.008	0.009	0.004	0.005
	pg/L	**1,2,3,6,7,8-HxCDF**	ND	0.003	ND	ND	0.005	0.005	0.003	0.003
	pg/L	**1,2,3,7,8,9-HxCDF**	ND	ND	ND	ND	ND	ND	ND	ND
	pg/L	**2,3,4,6,7,8-HxCDF**	ND	ND	ND	ND	ND	0.003	ND	ND
	pg/L	**1,2,3,4,6,7,8-HpCDF**	0.016	0.025	ND	ND	0.06	0.051	0.035	0.026
	pg/L	**1,2,3,4,7,8,9-HpCDF**	ND	ND	ND	ND	0.005	0.005	ND	ND
	pg/L	**OCDF**	0.029	0.045	0.023	0.023	0.105	0.073	0.072	0.0505
Nov. 2006	pg/L	**2,3,7,8-TCDF**	ND	0.007	0.01	0.01	ND	ND	0.005	ND
	pg/L	**1,2,3,7,8-PeCDF**	ND	ND	ND	ND	ND	ND	0.004	ND
	pg/L	**2,3,4,7,8-PeCDF**	ND	ND	ND	ND	ND	ND	0.004	ND
	pg/L	**1,2,3,4,7,8-HxCDF**	ND	0.008	0.01	0.01	0.009	0.008	0.009	ND
	pg/L	**1,2,3,6,7,8-HxCDF**	ND	0.006	0.007	0.007	0.008	ND	0.007	ND
	pg/L	**1,2,3,7,8,9-HxCDF**	ND	ND	ND	ND	0.003	ND	ND	ND
	pg/L	**2,3,4,6,7,8-HxCDF**	ND	ND	ND	ND	ND	ND	0.005	ND
	pg/L	**1,2,3,4,6,7,8-HpCDF**	0.036	0.057	0.08	0.08	0.059	0.061	0.078	0.064
	pg/L	**1,2,3,4,7,8,9-HpCDF**	0.005	0.006	0.007	0.007	0.009	0.007	0.007	0.005
	pg/L	**OCDF**	0.071	0.128	0.175	0.175	0.089	0.122	0.167	0.1535
Sept. 2006	pg/L	**2,3,7,8-TCDF**	ND	0.0292	0.0252	0.0252	0.0282	0.0272	0.0224	0.01391
	pg/L	**1,2,3,7,8-PeCDF**								
	pg/L	**2,3,4,7,8-PeCDF**								
	pg/L	**1,2,3,4,7,8-HxCDF**								
	pg/L	**1,2,3,6,7,8-HxCDF**								
	pg/L	**1,2,3,7,8,9-HxCDF**								
	pg/L	**2,3,4,6,7,8-HxCDF**								
	pg/L	**1,2,3,4,6,7,8-HpCDF**								
	pg/L	**1,2,3,4,7,8,9-HpCDF**								
	pg/L	**OCDF**								
Nov. 2006	pg/L	**Total TCDD**	ND	0.01	0.042	0.019	0.02	0.017	0.018	0.019
	pg/L	**Total PeCDD**	0.003	0.006	0.022	0.042	0.049	0.012	ND	0.0085

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	pg/L	**Total HxCDD**	0.042	0.051	0.098	0.128	0.118	0.072	0.069	0.0855
	pg/L	**Total HpCDD**	0.225	0.266	0.696	0.544	0.417	0.385	0.329	0.4555
	pg/L	**Total TCDF**	0.029	0.056	0.051	0.085	0.075	0.056	0.059	0.0585
	pg/L	**Total PeCDF**	0.011	0.021	0.027	0.041	0.038	0.026	0.023	0.026
	pg/L	**Total HxCDF**	0.017	0.04	0.048	0.083	0.07	0.051	0.044	0.056
	pg/L	**Total HpCDF**	0.036	0.055	0.088	0.089	0.07	0.062	0.071	0.09
Mar. 2006	pg/L	**Total TCDD**	0.008	0.02	0.016	0.016	0.034	0.035	0.032	0.028
	pg/L	**Total PeCDD**	ND	0.047	0.014	0.014	0.044	0.039	0.038	0.0565
	pg/L	**Total HxCDD**	0.062	0.063	0.033	0.033	0.123	0.167	0.112	0.081
	pg/L	**Total HpCDD**	0.283	0.354	0.264	0.264	0.52	0.883	0.539	0.399
	pg/L	**Total TCDF**	0.031	0.04	0.053	0.053	0.074	0.079	0.065	0.056
	pg/L	**Total PeCDF**	0.012	0.026	0.015	0.015	0.052	0.045	0.037	0.0275
	pg/L	**Total HxCDF**	0.023	0.036	0.02	0.02	0.088	0.077	0.045	0.0405
	pg/L	**Total HpCDF**	0.034	0.05	0.015	0.015	0.131	0.118	0.077	0.0565
Jun. 2006	pg/L	**Total TCDD**	0.01	0.019	0.027	0.027	0.031	0.013	0.013	0.016
	pg/L	**Total PeCDD**	0.007	0.016	0.03	0.03	0.021	0.006	0.006	0.0075
	pg/L	**Total HxCDD**	0.063	0.102	0.126	0.126	0.19	0.101	0.101	0.1005
	pg/L	**Total HpCDD**	0.509	0.645	0.89	0.89	0.98	0.536	0.536	0.629
	pg/L	**Total TCDF**	0.039	0.087	0.102	0.102	0.13	0.079	0.079	0.047
	pg/L	**Total PeCDF**	0.018	0.049	0.062	0.062	0.052	0.039	0.039	0.027
	pg/L	**Total HxCDF**	0.04	0.081	0.119	0.119	0.094	0.081	0.081	0.074
	pg/L	**Total HpCDF**	0.087	0.146	0.202	0.202	0.135	0.136	0.136	0.159
Sept. 2006	pg/L	**Total TCDD**	[redacted]							
	pg/L	**Total PeCDD**								
	pg/L	**Total HxCDD**								
	pg/L	**Total HpCDD**								
	pg/L	**Total TCDF**								
	pg/L	**Total PeCDF**								
	pg/L	**Total HxCDF**								

Month	Unit	Type	CR-REF	SR-REF	LPG-14	LGP-13	LGP-11	LGP-09	LGP-06	LGP-01
	pg/L	**Total HpCDF**								

Appendix D: Cormix Modeling

Appendix E: Fish Abundance Summary Data

The source for all fish count data is the Columbia River Data Access in Real Time (DART) database.

Table E- 1: Adult fall Chinook counts at Lower Granite Dam

Month	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10-Year Average	Average % annual run
JAN	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
FEB	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
MAR	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
APR	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
MAY	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
JUN	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
JUL	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
AUG	902	963	1526	1647	1653	3239	1929	2825	2520	4132	2133.6	6.4
SEPT	5791	7051	14051	11494	30226	17789	28614	45673	48518	45040	25424.7	75.8
OCT	1716	2472	1388	2336	10124	5078	4242	8029	9350	10795	5553	16.6
NOV	71	144	183	106	503	420	620	726	930	626	432.9	1.3
DEC	3	3	2	3	1	9	8	0	4	12	4.5	0.0
Total	8483	10633	17150	15586	42507	26535	35413	57253	61322	60605	33548.7	

Table E- 2: Adult wild sockeye counts at Lower Granite Dam

Month	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10-Year Average	Average % annual run
JAN	0	0	0	0	0	0	0	0	0	0	0	0.0
FEB	0	0	0	0	0	0	0	0	0	0	0	0.0
MAR	0	0	0	0	0	0	0	0	0	0	0	0.0
APR	0	0	0	0	0	0	0	0	0	0	0	0.0
MAY	0	0	0	0	0	0	0	0	0	10	1	0.1
JUN	1	8	41	94	80	3	15	62	71	65	44	4.2
JUL	13	45	810	1090	1943	1444	410	611	2503	305	917.4	88.6
AUG	0	0	41	31	147	53	32	64	173	37	57.8	5.6

SEPT	1	0	9	2	31	1	13	17	30	15	11.9	1.1
OCT	1	-1	8	2	0	1	0	3	9	8	3.1	0.3
NOV	1	0	0	0	0	0	0	0	0	0	0.1	0.0
DEC	0	0	0	0	0	0	0	0	0	0	0	0.0
Total	17	52	909	1219	2201	1502	470	757	2786	440	1035.3	

Table E- 3: Adult spring/summer Chinook at Lower Granite Dam

Month	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10-Year Average	Average % annual run
JAN	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0
FEB	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0
MAR	0	1	0	0	1	0	0	1	0	6	0.9	0
APR	4	524	993	94	9314	68	36	246	1131	13720	2613	3.4
MAY	13896	13237	31092	31170	70791	42672	51144	27087	64973	80038	42610	55.6
JUN	11716	12792	32461	27886	33232	37083	22842	11841	19224	18704	22778.1	29.7
JUL	3537	3197	7690	4580	8951	14996	4782	3591	7872	6058	6525.4	8.5
AUG	902	963	1526	1647	1653	3239	1929	2825	2520	4132	2133.6	2.8
SEPT	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0
OCT	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0
NOV	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0
DEC	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	0
Total	30055	30714	73762	65377	123942	98058	80733	45591	95720	122658	76661	

Table E- 4: Jack Chinook counts at Lower Granite Dam

Month	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10-Year Average	Average % annual run
JAN	0	0	0	0	0	0	0	0	0	0	0	0.0
FEB	0	0	0	0	0	0	0	0	0	0	0	0.0
MAR	0	0	0	0	0	0	0	0	0	0	0	0.0

APR	0	7	26	1	118	3	0	21	18	147	34.1	0.1
MAY	429	6279	5903	21444	4994	9908	2405	15269	9134	5537	8130.2	22.0
JUN	761	4202	8193	17679	3556	19754	1901	8382	7150	4679	7625.7	20.6
JUL	390	1724	1823	7824	2859	8506	825	3339	4322	1899	3351.1	9.1
AUG	151	237	293	1224	425	598	424	993	497	685	552.7	1.5
SEPT	2839	5669	6519	32938	8580	12302	16727	16240	12677	6681	12117.2	32.7
OCT	3616	3956	3337	7294	3901	6933	4774	5287	6290	4150	4953.8	13.4
NOV	169	73	152	313	165	137	173	376	556	348	246.2	0.7
DEC	1	-1	0	-1	0	2	3	0	1	2	0.7	0.0
Total	8356	22146	26246	88716	24598	58143	27232	49907	40645	24128	37011.7	

Table E- 5: Wild steelhead counts at Lower Granite Dam

Month	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10-Year Average	Average % annual run
JAN	0	0	0	0	0	0	0	0	0	0	0	0.0
FEB	0	0	0	0	0	0	0	0	0	0	0	0.0
MAR	0	0	0	0	0	0	0	0	0	0	0	0.0
APR	0	7	26	1	118	3	0	21	18	147	34.1	0.1
MAY	429	6279	5903	21444	4994	9908	2405	15269	9134	5537	8130.2	22.0
JUN	761	4202	8193	17679	3556	19754	1901	8382	7150	4679	7625.7	20.6
JUL	390	1724	1823	7824	2859	8506	825	3339	4322	1899	3351.1	9.1
AUG	151	237	293	1224	425	598	424	993	497	685	552.7	1.5
SEPT	2839	5669	6519	32938	8580	12302	16727	16240	12677	6681	12117.2	32.7
OCT	3616	3956	3337	7294	3901	6933	4774	5287	6290	4150	4953.8	13.4
NOV	169	73	152	313	165	137	173	376	556	348	246.2	0.7
DEC	1	-1	0	-1	0	2	3	0	1	2	0.7	0.0
Total	8356	22146	26246	88716	24598	58143	27232	49907	40645	24128	37011.7	

Table E- 6: Steelhead counts at Lower Granite Dam

Month	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10-Year Average	Average % annual run

JAN	0	0	0	0	0	0	0	0	0	0	0	0.0
FEB	0	0	0	0	0	0	0	0	0	0	0	0.0
MAR	3467	6894	3812	4313	5934	5017	4451	4461	5419	7118	5088.6	3.0
APR	3924	3477	3412	5918	3986	6443	3863	2674	1665	1798	3716	2.2
MAY	200	211	547	550	542	830	613	304	381	264	444.2	0.3
JUN	87	271	107	486	490	73	108	107	325	113	216.7	0.1
JUL	691	1504	4301	3806	8939	3309	1202	1044	4511	814	3012.1	1.8
AUG	2063	3792	13596	8978	16337	21163	2374	4016	5448	3323	8109	4.7
SEPT	50958	58983	83413	154804	98690	84732	46049	39492	66495	59018	74263.4	43.3
OCT	68063	71535	56851	129599	63325	58065	40984	49717	66635	58255	66302.9	38.6
NOV	14958	8269	8322	14574	7620	3355	8927	5597	10137	8150	8990.9	5.2
DEC	1580	3181	1120	669	1022	661	2104	498	3072	899	1480.6	0.9
Total	145991	158117	175481	323697	206885	183648	110675	107910	164088	139752	171624.4	

Appendix F: Priority and Nonconventional Pollutants Analyzed for in Clearwater Treated Effluents

Pollutant[1]	# of Samples	# of Non-detect	Symbol[2]	Maximum	Units
CDDs/CDFs					
1,2,3,7,8-pentachlorodibenzofuran (PeCDF	2	2	ND	0.007J	pg/L
1,2,3,4,7,8-hexachlorodibenzofuran (HxCDF	1	1	ND	0.35U	pg/L
1,2,3,6,7,8-hexachlorodibenzofuran	2	2	ND	0.35U	pg/L
1,2,3,7,8,9-hexachlorodibenzofuran	2	2	ND	0.83U	pg/L
2,3,4,6,7,8-hexachlorodibenzofuran	2	2	ND	0.35	pg/L
1,2,3,4,6,7,8-heptachlorodibenzofuran	2	2	ND	0.83U	pg/L
2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD)	3	3	ND	0.333	pg/L
1,2,3,4,7,8,9-heptachlorodibenzofuran	3	3	ND	0.03	pg/L
Volatile Organics					
acrolein	1	1	ND		µg/L
acrylonitrile*	1	1	ND	500.0[4]	µg/L
benzene*	1	1	ND		µg/L
bromomethane	1	1	ND	500.0	µg/L
chlorobenzene*	1	`	ND		µg/L
chloroethane*	1	1	ND	500.0	µg/L
chloromethane	1	1	ND	500.0	µg/L
dibromochloromethane	1	1	ND	100.0	µg/L
ethyl cyanide	1	1	ND	100.0	µg/L
ethylbenzene*	1	1	ND	100.0	µg/L
tetrachloromethane	1	1	ND	100.0	µg/L
tetrachloroethylene	1	1	ND		µg/L
toluene	1	1	ND		µg/L
trans-1,2-dichloroethene	1	1	ND	100.0	µg/L
tribromomethane	1	1	ND	100.0	µg/L
trichloroethene	5	3	ND	1.3	µg/L
trichlorofluoromethane	1	1	ND		µg/L
vinyl chloride*	1	1	ND	100.0[4]	µg/L
1,1-dichloroethane*	1	1	ND	100.0	µg/L
1,1,1-trichloroethane*	1	1	ND	100.0	µg/L
1,1,2,2-tetrachloroethane*	1	1	ND	100.0[4]	µg/L
1,2-dichloroethane*	1	1	ND	100.0[4]	µg/L
1,2-trans-dichloroethylene	1	1	ND		µg/L
1,2-dichloropropane*	1	1	ND		µg/L
1,3-dichloropropylene	1	1	ND		µg/L
2-chloroethyl vinyl ether*	1	1	ND	100.0	µg/L
Chlorinated Phenolics					
2,4-dichlorophenol*	1	1	ND		µg/L
2,3,4,6-tetrachlorophenol	4	4		0.04 U	µg/L
2,4,5-trichlorophenol	4	4		0.02 U	µg/L
2,4,6-trichlorophenol	5	5		1.27 J	µg/L
3,4,5-trichlorocatechol	4	4		0.22 U	µg/L
Pentachlorophenol	5	5		0.05 U	µg/L

Pollutant[1]	# of Samples	# of Non-detect	Symbol[2]	Maximum	Units
3,4,6-trichlorocatechol	4	4		0.11 U	µg/L
tetrachlorocatechol	4			0.35 U	µg/L
3,4,5-trichloroguaiacol	4	3		0.51 J	µg/L
3,4,6-trichloroguaiacol	4	4		0.27	µg/L
4,5,6-trichloroguaiacol	4			0.60 U	µg/L
tetrachloroguaiacol	4	4		0.04 U	µg/L
trichlorosyringol	4	4		0.13 U	µg/L
Metals[3]					
aluminum	1	0		368000	µg/L
antimony*	1	0		0.1	µg/L
arsenic*	1	0	0	1.6	µg/L
barium	1	0		263	µg/L
beryllium*	1	1	ND		µg/L
boron	1	0		26	µg/L
cadmium*	1	1	ND		µg/L
chromium	1	0		11.8	
cobalt	1	0	ND	1	µg/L
copper*	1	0	ND	2.5	µg/L
iron	1			342	µg/L
lead*	1	0	ND	0.62	µg/L
magnesium	1	0		4290	µg/L
manganese	1	0		296	µg/L
molybdenum	1	0		3.1	µg/L
mercury	1	1	ND		
nickel*	1	0		3.6	µg/L
Pesticides/Herbicides					
alpha-chlordane	1	1	ND	2.5	µg/L
aldrin	1	1	ND		
alpha-BHC	1	1	ND		
beta-BHC*	1	1	ND	0.4[4]	µg/L
gamma-BHC	1	1	ND		
delta-BHC*	1	1	ND	0.3	µg/L
dieldrin	1	1	ND		
Endosulfan I (alpha-Endosulfan)*	1	1	ND	0.5[4]	µg/L
Endosulfan II (beta-Endosulfan)*	1	1	ND	0.3[4]	µg/L
Endosulfan sulfate*	1	1	ND	0.5	µg/L
endrin*	1	1	ND	0.3[4]	µg/L
endrin aldehyde*	1	1	ND	0.5	µg/L
heptachlor*	1	1	ND	0.2[4]	µg/L
heptachlor epoxide*	1	1	ND	0.2[4]	µg/L
p,p'-TDE (4,4-DDD)*	1	1	ND	0.5[4]	µg/L
p,p'-DDX (4,4-DDE)*	1	1	ND	0.5[4]	µg/L
p,p'-DDT (4,4-DDT)*	1	1	ND	0.4[4]	µg/L
PCB 1016 (Arochlor 1016)*	1	1	ND	2.0[4]	µg/L
PCB 1221 (Arochlor 1221)*	1	1	ND	2.0[4]	µg/L
PCB 1232 (Arochlor 1232)*	1	1	ND	2.0[4]	µg/L
PCB 1242 (Arochlor 1242)*	1	1	ND	2.0[4]	µg/L
Semi-volatiles					
acenaphthene*	1	1	ND	10.0	µg/L
acenaphthylene*	1	1	ND	10.0	µg/L

Pollutant[1]	# of Samples	# of Non-detect	Symbol[2]	Maximum	Units
anthracene*	1	1	ND	10.0	µg/L
benzidine*	1	1	ND	50.0[4]	µg/L
benzo(a)anthracene*	1	1	ND	10.0[4]	µg/L
benzo(a)pyrene*	1	1	ND	10.0[4]	µg/L
benzo(b)fluoranthene*	1	1	ND	10.0[4]	µg/L
benzo(k)fluoranthene	1	1	ND		µg/L
benzo(ghi)Perylene	1	1	ND		µg/L
bis(2-chloroisopropyl)ether*	1	1	ND	50.0	µg/L
bis(2-chloroethoxy)methane*	1	1	ND	10.0	µg/L
bis(2-chloroethyl)ether*	1	1	ND	10.0[4]	µg/L
Bis(2-ethylhexy)phthalate	1	1	ND		µg/L
butylbenzyl phthalate*	1	1	ND	10.0	µg/L
chrysene*	1	1	ND	10.0[4]	µg/L
di-n-butyl phthalate*	1	1	ND	10.0	µg/L
di-n-octyl phthalate*	1	1	ND	10.0	µg/L
dibenzo(a,h)anthracene	1	1	ND	20.0	µg/L
dichlorodifluoromethane (NR)	1	1	ND	10.0	µg/L
diethyl phthalate*	1	1	ND	10.0	µg/L
dimethyl phthalate*	1	1	ND	10.0	µg/L
fluoranthene*	1	1	ND	10.0	µg/L
fluorene*	1	1	ND	10.0	µg/L
hexachloro-1,3-butadiene	1	1	ND	10.0	µg/L
hexachlorobenzene*	1	1	ND	10.0[4]	µg/L
Hexachlorocyclopentadiene*	1	1	ND	10.0	µg/L
hexachloroethane*	1	1	ND	10.0[4]	µg/L
Indeno(1,2,3-CD)pyrene*	1	1	ND	20.0[4]	µg/L
Isophorone*	1	1	ND	10.0[4]	µg/L
methylene chloride	1	1	ND		µg/L
n-nitrosodi-n-propylamine*	1	1	ND	20.0	µg/L
n-nitrosodimethylamine*	1	1	ND	50.0[4]	µg/L
nitrobenzene*	1	1	ND	10.0	µg/L
naphthalene	1	1	ND		µg/L
p-chloro-m-cresol	1	1	ND		µg/L
phenanthrene*	1	1	ND	10.0	µg/L
phenol*	1	0	ND	0.097	µg/L
pyrene*	1	1	ND	10.0	µg/L
1,2-dichlorobenzene*	1	1	ND	10.0	µg/L
1,2-diphenylhydrazine*	1	1	ND	20.0[4]	µg/L
1,2,4-trichlorobenzene*	1	1	ND	10.0	µg/L
1,3-dichlorobenzene*	1	1	ND	10.0	µg/L
1,3-dinitrobenzene	1	1	ND	20.0	µg/L
1,4-dichlorobenzene*	1	1	ND	10.0	µg/L
2-chloronaphthalene*	1	1	ND	10.0	µg/L
2-chlorophenol*	1	1	ND	10.0	µg/L
2-nitrophenol*	1	1	ND	20.0	µg/L
2,4-dimethylphenol*	1	1	ND	10.0	µg/L
2,4-dinitrophenol*	1	1	ND	50.0	µg/L
2,4-dinitrotoluene*	1	1	ND	10.0[4]	µg/L
2,6-dinitrotoluene*	1	1	ND	10.0	µg/L
3,3'-dichlorobenzidine*	1	1	ND	50.0	µg/L

Pollutant[1]	# of Samples	# of Non-detect	Symbol[2]	Maximum	Units
4-bromophenyl phenyl ether*	1	1	ND	10.0	µg/L
4-chlorophenyl phenyl ether*	1	1	ND	10.0	µg/L
4-nitrophenol*	1	1	ND	50.0	µg/L
4,6-dinitro-o-cresol	1	1	ND		µg/L
Resin Acids					
12,14-dichlorodehydroabietic acid	2	2	ND		µg/L
dehydroabietic acid	3	2	ND	13.8	µg/L
isoprimaric acid	3	2	ND	20.9	µg/L
primic acid	3	1	ND	49.13	µg/L
12,14-chlorodehydroabietic acid	4	1	ND	1.38	µg/L
12-chlorodehydroabietic acid	1	1	ND		µg/L
14-chlorodehydroabietic acid	1	1	ND		µg/L
abietic acid	1	1	ND		µg/L
retene	4	1	ND	0.20	µg/L
9,10-dichlorostearic acid	1	1	ND		

Footnotes:
1. Priority pollutants are indicated by an asterisk (*).
2. Symbol of ND indicates that all results are non-detected; the maximum value is the greatest detection limit.
3. Detection limits are not available for metals analyzed semi quantitatively.
4. The maximum level is above the applicable water quality criteria for this pollutant parameter.
5. U: Nondetected-value is one half the reporting detection limit.
6: J: Estimated value

Appendix G: Dioxin Analysis

Resident Fish Tissue Sampling and Analysis Report, Potlatch NPDES Permit Renewal Compliance Monitoring

The average concentration of 2,3,7,8-TCDD, 2,3,7,8-TCDF and TEQ was calculated for each location where these species and tissue types were collected. Average concentrations were expressed on a dry-weight and lipid-normalized basis. Comparison of sampling stations using lipid-normalized concentrations is preferred because dioxin and furan concentrations are directly related to lipid content. Lipid normalization assures that any differences in concentration between sampling stations are representative of differences in the amount of dioxin available for fish to bioaccumulate and not caused by a difference in lipid content between stations. In order to capture the range of uncertainty introduced by assumptions about what concentration of dioxins and furans exist in non-detects, average concentrations for each station were calculated assuming non-detects are equal to the detection limit (ND=DL).

Biota to Sediment Accumulation Factors

BSAFs can be calculated on both a dry weight and an organic carbon and lipid normalized basis. Dry-weighted BSAFs are calculated by dividing the dry-weight concentration in fish by the dry-weight concentration in sediment. As noted above, dioxin and furan concentrations in sediment and fish are affected by organic carbon (OC) concentration and lipid content. To adjust for the effects of organic carbon and lipid, OC- and lipid-normalized BSAFs are calculated by dividing the lipid-normalized concentration calculated using the average concentration of 2,3,7,8-TCDD and 2,3,7,8-TCDFfor each species of fish and tissue type sampled from the LGR during the resident fish tissue sampling conducted in 2007 and the average sediment concentration of 2,3,7,8-TCDD and 2,3,7,8-TCDF collected in 2005. Table 4-9 presents sediment concentrations for the congeners and Table G- 1 and G-2 present fish tissue concentrations on a dry-weight and lipid-normalized basis, respectively. Both the resident fish and sediment sampling and analyses were conducted by Anchor Environmental, L.L.C.

OC-lipid-normalized site-specific BSAFs for 2,3,7,8-TCDD and 2,3,7,8-TCDF species and tissue combination are presented in Table 7-18.

Table G- 1: Summary of Dry Weight Dioxin Levels in Largescale Suckers and Smallmouth Bass

	2,3,7,8-TCDD (ng/kg)				2,3,7,8-TCDF (ng/kg)				TEQ (ng/kg)			
	RepA	Rep B	Rep C	Mean	Rep A	Rep B	Rep C	Mean	Rep A	Rep B	Rep C	Mean
Largescale Sucker – Whole Body												
CR-REF Clearwater River	0.022	0.033	0.031	0.029	0.325	0.067	0.180	0.191	0.164	0.172	0.182	0.173
SR-REF Snake River	0.050	0.042	0.022	0.038	0.341	0.444	0.494	0.426	0.185	0.204	0.226	0.205
LGP-01 Snake River	0.039	0.018	0.046	0.035	0.760	1.190	1.120	1.023	0.280	0.198	0.256	0.244
LGP-02 Snake River	0.020	0.034	0.028	0.027	1.000	0.878	0.513	0.797	0.214	0.212	0.230	0.219
LGP-03 Snake River	0.033	0.018	0.036	0.029	0.730	0.433	0.406	0.523	0.249	0.236	0.305	0.263
LGP-04 Snake River	0.034	0.022	0.017	0.024	0.268	0.468	1.000	0.579	0.228	0.233	0.308	0.256
LGP-05 Snake River	0.035	0.024	0.021	0.027	0.298	0.371	0.266	0.312	0.206	0.256	0.157	0.206
LGP-06 Snake River	0.028	0.020	0.040	0.029	0.280	0.481	0.641	0.467	0.224	0.227	0.272	0.241
Largescale Sucker – Fillet with Skin												
CR-REF Clearwater River	0.046	0.041	0.033	0.040	0.339	0.215	0.074	0.209	0.272	0.208	0.207	0.229
SR-REF Snake River	0.025	0.046	0.033	0.034	0.286	0.303	0.147	0.245	0.179	0.220	0.195	0.198
LGP-01 Snake River	0.061	0.041	0.029	0.043	0.195	0.099	0.252	0.182	0.195	0.203	0.191	0.196
LGP-02 Snake River	0.036	0.023	0.021	0.027	0.252	0.358	0.494	0.368	0.218	0.140	0.165	0.175
LGP-03 Snake River	0.027	0.022	0.024	0.024	0.065	0.079	0.237	0.127	0.164	0.132	0.180	0.159
LGP-04 Snake River	0.027	0.038	0.040	0.035	0.136	0.156	0.129	0.140	0.197	0.160	0.177	0.178
LGP-05 Snake River	0.025	0.036	0.026	0.029	0.238	0.091	0.131	0.153	0.216	0.150	0.155	0.174
LGP-06	0.023	0.032	0.023	0.026	0.288	0.143	0.373	0.268	0.142	0.153	0.161	0.152

Snake River												
Smallmouth Bass – Fillet with Skin												
CR-REF Clearwater River	0.020	0.030	0.036	0.029	0.088	0.066	0.049	0.068	0.176	0.144	0.196	0.172
SR-REF Snake River	0.033	0.021	0.042	0.032	0.081	0.073	0.077	0.077	0.195	0.148	0.180	0.174
LGP-01 Snake River	0.041	0.064	0.050	0.052	0.045	0.044	0.041	0.043	0.196	0.274	0.210	0.227
LGP-02 Snake River	0.031	0.024	0.053	0.036	0.050	0.069	0.069	0.062	0.148	0.135	0.264	0.182
LGP-03 Snake River	0.017	0.043	0.029	0.030	0.023	0.064	0.051	0.046	0.153	0.146	0.206	0.168
LGP-04 Snake River	0.038	0.088	0.031	0.052	0.042	0.034	0.058	0.045	0.169	0.269	0.221	0.220
LGP-05 Snake River	0.030	0.030	0.025	0.028	0.040	0.049	0.052	0.047	0.149	0.178	0.187	0.172
LGP-06 Snake River	0.026	0.034	0.034	0.031	0.068	0.071	0.068	0.069	0.203	0.159	0.158	0.173
Smallmouth Bass – Whole Body												
CR-REF Clearwater River	0.060	0.042	0.044	0.049	0.184	0.158	0.243	0.195	0.398	0.296	0.232	0.309
SR-REF Snake River	0.055	0.053	0.032	0.046	0.310	0.344	0.231	0.295	0.264	0.239	0.218	0.240
LGP-01 Snake River	0.052	0.076	0.092	0.073	0.215	0.241	0.198	0.218	0.247	0.382	0.376	0.335
LGP-02 Snake River	0.028	0.042	0.126	0.065	0.250	0.220	0.198	0.223	0.202	0.315	0.585	0.367
LGP-03 Snake River	0.022	0.019	0.016	0.019	0.149	0.255	0.175	0.193	0.277	0.162	0.145	0.195
LGP-04 Snake River	0.043	0.017	0.023	0.027	0.202	0.174	0.286	0.221	0.202	0.134	0.245	0.193
LGP-05 Snake River	0.052	0.030	0.018	0.033	0.360	0.312	0.339	0.337	0.226	0.174	0.170	0.190
LGP-06 Snake River	0.018	0.017	0.030	0.021	0.321	0.420	0.508	0.416	0.253	0.217	0.223	0.231

Table G- 2: Summary of Lipid-Normalized Dioxin Levels in Largescale Suckers and Smallmouth Bass

	2,3,7,8-TCDD (ng/kg)				2,3,7,8-TCDF (ng/kg)				TEQ (ng/kg)			
	RepA	Rep B	Rep C	Mean	Rep A	Rep B	Rep C	Mean	Rep A	Rep B	Rep C	Mean
Largescale Sucker – Whole Body												
CR-REF Clearwater River	0.483	1.500	1.800	1.261	7.065	3.032	10.588	6.895	3.562	7.817	10.697	7.359
SR-REF Snake River	1.459	1.005	0.519	0.994	10.029	10.571	11.762	10.788	5.450	4.858	5.390	5.233
LGP-01 Snake River	0.666	0.482	0.983	0.710	12.881	31.316	23.830	22.676	4.745	5.205	5.437	5.129
LGP-02 Snake River	0.345	0.673	0.428	0.482	17.241	17.216	7.892	14.116	3.693	4.149	3.542	3.795
LGP-03 Snake River	0.566	0.328	0.619	0.504	12.586	8.019	7.000	9.202	4.290	4.372	5.266	4.642
LGP-04 Snake River	1.425	0.562	0.289	0.759	11.167	12.000	17.544	13.570	9.496	5.963	5.403	6.954
LGP-05 Snake River	1.173	0.777	0.533	0.828	9.933	11.968	6.650	9.517	6.871	8.268	3.922	6.354
LGP-06 Snake River	0.789	0.442	0.608	0.613	7.778	10.689	9.712	9.393	6.228	5.039	4.117	5.128
Largescale Sucker – Fillet with Skin												
CR-REF Clearwater River	2.572	1.644	1.650	1.955	18.833	8.600	3.695	10.376	15.126	8.312	10.364	11.267
SR-REF Snake River	1.235	1.632	1.817	1.561	14.300	10.821	8.167	11.096	8.945	7.867	10.810	9.207
LGP-01 Snake River	3.788	3.375	1.718	2.960	12.188	8.300	14.824	11.770	12.179	16.903	11.249	13.444

LGP-02 Snake River	1.234	0.780	0.710	0.908	8.690	11.933	17.034	12.552	7.524	4.681	5.689	5.965
LGP-03 Snake River	1.571	1.137	0.790	1.166	3.829	4.163	7.900	5.298	9.674	6.955	5.992	7.540
LGP-04 Snake River	2.225	2.229	1.809	2.088	11.333	9.176	5.864	8.791	16.454	9.402	8.041	11.299
LGP-05 Snake River	1.000	3.033	2.758	2.264	9.520	7.592	13.789	10.300	8.649	12.460	16.346	12.485
LGP-06 Snake River	0.682	2.153	1.095	1.310	8.471	9.533	17.762	11.922	4.187	10.185	7.663	7.345
Smallmouth Bass – Fillet with Skin												
CR-REF Clearwater River	1.569	4.040	4.000	3.203	6.738	8.787	5.528	7.018	13.507	19.149	22.049	18.235
SR-REF Snake River	4.704	2.080	4.784	3.856	11.366	7.250	8.761	9.126	27.525	14.829	20.410	20.921
LGP-01 Snake River	7.121	10.443	7.677	8.413	7.707	7.246	6.292	7.082	33.851	44.938	32.326	37.038
LGP-02 Snake River	4.162	3.408	5.989	4.520	6.743	9.690	7.807	8.080	19.995	18.945	29.984	22.975
LGP-03 Snake River	2.361	4.696	3.043	3.366	3.208	6.967	5.372	5.183	21.318	15.892	21.886	19.698
LGP-04 Snake River	4.512	13.952	4.588	7.684	4.952	5.413	8.515	6.293	20.156	42.689	32.514	31.787
LGP-05 Snake River	3.683	2.107	2.236	2.675	4.854	3.521	4.718	4.364	18.197	12.745	17.033	15.992
LGP-06	2.663	3.744	3.843	3.417	6.888	7.833	7.685	7.469	20.708	17.614	17.795	18.706

Snake River												
Smallmouth Bass – Whole Body												
CR-REF Clearwater River	1.426	1.113	1.080	1.207	4.381	4.158	5.927	4.822	9.475	7.788	5.671	7.644
SR-REF Snake River	1.439	1.313	1.167	1.306	8.158	8.600	8.556	8.438	6.942	5.984	8.067	6.998
LGP-01 Snake River	1.622	2.915	2.863	2.467	6.719	9.269	6.188	7.392	7.705	14.686	11.753	11.381
LGP-02 Snake River	0.705	1.438	4.345	2.163	6.410	7.586	6.828	6.941	5.176	10.875	20.180	12.077
LGP-03 Snake River	0.694	0.588	0.543	0.608	4.656	7.969	5.833	6.153	8.642	5.072	4.845	6.186
LGP-04 Snake River	1.536	0.692	0.649	0.959	7.214	7.250	8.171	7.545	7.213	5.573	6.989	6.592
LGP-05 Snake River	1.305	0.833	0.520	0.886	9.000	8.667	9.686	9.117	5.659	4.825	4.861	5.115
LGP-06 Snake River	0.462	0.376	0.509	0.449	8.231	9.333	8.759	8.774	6.488	4.817	3.840	5.048

Appendix H: Influence of Chemicals on Species

Table H- 1: Overview of the Range of Effects from Chlorinated Phenols: Acute Values

Species	Method	Chemical	LC50/EC50 (µg/L)	Reference
Cladoceran *Daphnia magna*	S, U	4-chlorophenol	4,820	Kopperman et al., 1974
Cladoceran *Daphnia magna*	S, U	4-chlorophenol	4,060	USEPA, 1978b
Cladoceran *Daphnia magna*	S, U	2,4,5-trichlorophenol	2,660	USEPA, 1978b
Cladoceran *Daphnia magna*	S, U	2,4,5-trichlorophenol	780-3800	LeBlanc, 1980; LeBlanc et al., 1988, Spehar, 1986
Cladoceran *Daphnia magna*	S, U	2,4,6-trichlorophenol	6,040	USEPA, 1978b
Cladoceran *Daphnia magna*	S, U	2,4,6-Trichlorophenol	270 -15,000	Dence et al., 1980; Yoshioka et al., 1986; Kukkonen and Oikari, 1987; Holcombe et al., 1987; Virtanen et al., 1989
Cladoceran *Daphnia magna*	S, U	2,3,5,6-tetrachlorophenol	570	USEPA, 1978b
Cladoceran *Daphnia magna*	S, U	2,3,4,6-tetrachlorophenol	290	USEPA, 1978b
Cladoceran *Daphnia magna*	S, U	4-chloro-2-methyl phenol**	290	USEPA, 1978b
Cladoceran *Daphnia magna*	S, U	2,4-dichloro-6-methylphenol	430	USEPA, 1978b
Cladoceran *Daphnia magna*	24-hour	Tetrachloroguaiacol	4,960	Dence et al., 1980
Cladoceran *Daphnia sp.*	48-hours	2,3,4,6-Tetrachlorophenol	500-750	Liber et al., 1992
Cladoceran *Daphnia sp.*	S, U	2,3,4,6-Tetrachlorophenol	10-16,000	LeBlanc, 1980; Virtanen et al., 1989; Oikari et al., 1992; and Liber and Solomon, 1994
Cladoceran *Daphnia sp.*	S, U	2,3,4,6-Tetrachlorophenol	1,400 - 2,300	Shigeoka et al., 1988
Cladoceran *Daphnia sp.*	96-hours	Pentachlorophenol	320 -800	Adema and Vink, 1981; Ewell et al., 1986
Rotifer sp.	48-hours	2,3,4,6-Tetrachlorophenol	280-650	Liber et al., 1992
Copepod sp.	48-hours	2,3,4,6-Tetrachlorophenol	270 to 590	Liber et al., 1992
Fathead Minnow *Pimephales promelas* embryo	48-hours	Tetrachloroguaiacol	100-200	Woodland and Maly, 1997

Species	Method	Chemical	LC50/EC50 (µg/L)	Reference
Fathead Minnow *Pimephales promelas*	S, M	2,4,6-trichlorophenol	600	USEPA, 1972
Fathead Minnow *Pimephales promelas* (juvenile)	FT, M	2,4,6-trichlorophenol	9,040	Phipps et al., Manuscript
Fathead Minnow *Pimephales promelas*	S, M	4-chloro-3-methyl-phenol	30	USEPA, 1972
Rainbow trout *Salmo gairdneri*	48-hours	3-chlorophenol	10,000	Shumway & Palensky, 1973
Rainbow trout *Salmo gairdneri*	48-hours	2,4,5-trichlorophenol	1,000	Shumway & Palensky, 1973
Rainbow trout *Salmo gairdneri*	48-hours	2,4,5-trichlorophenol	260	Hattula et al., 1981
Rainbow trout *Salmo gairdneri*	48-hours	2,3,4,6-Tetrachlorophenol	334 to 506	Kennedy, 1990
Rainbow trout *Salmo gairdneri*	48-hours	Pentachlorophenol	18	Van Leeuwen et al. 1985
Rainbow trout *Salmo gairdneri*	96-hours	Pentachlorophenol	18-3,000	Bentley et al., 1975; Saarikoski and Viluksela, 1981; Hodson et al., 1984; Thurston et al., 1985
Rainbow trout *Salmo gairdneri*		Pentachlorophenol	NOEC: 11	Dominquez and Chapman, 1984
Rainbow trout *Salmo gairdneri*	48-hours	2,3,4,6-Tetrachlorophenol	334	Kennedy 1990
Rainbow trout *Salmo gairdneri*	48-hours	Tetrachloroguaiacol	LOEC: 100-200	Johansen at al., 1994
Rainbow trout *Salmo gairdneri*	48-hours	2,4,6-Trichlorophenol	730-3,304	Holcombe et al., 1987; Kennedy, 1990
Largemouth Bass *Micropterus salmoides*	S	Pentachlorophenol	0.2	Little et al. 1990
Atlantic sturgeon *Acipenseridae*	96-hours	Pentachlorophenol	<40	Dwyer et al. 2000
Chinook salmon *Oncorhynchus tshawytscha*	48-hours	Pentachlorophenol	31-68	Johnson and Finley, 1980 and Saarikoski and Viluksela, 1981
Brown Trout *Salmo trutta*	48-hours	2,4,5-trichlorophenol	900	Knott and Johnston, 1971
Brown Trout *Salmo trutta*	48-hours	2,3,4,6-Tetrachlorophenol	500	Hattula et al., 1981
Brown Trout *Salmo trutta*	48-hours	2,4,6-Trichlorophenol	730 -3,304	Holcombe et al., 1987; Kennedy, 1990

Species	Method	Chemical	LC50/EC50 (µg/L)	Reference
Brown Trout *Salmo trutta*	48-hours	2,4,6-Trichlorophenol	730	Huttula et al., 1981
Bleak *Alburnus alburnus*	96-hours	Tetrachloroguaiacol	110	Oikari, 1987
Silver Salmon *Oncorhynchus mykiss*	96-hour	Tetrachloroguaiacol	370	Johansen et al., 1994
Carp *Cyprinidae*	48-hours	Pentachlorophenol	9.5	Eisler, 1993
Lavae *Protozoa* *Tetrahymena pyriformis*	S	2,4,6-Trichlorophenol	3,990	Schultz and Riggin, 1985
Sea Urchin *Arbacia punctulata* Embyro		2,4,6-Trichlorophenol	39	Clowes, 1952
Sea Urchin *Arbacia punctulata* Embyro		2,4,6-Trichlorophenol	6.2	Clowes, 1951

Table H- 2: Overview of the Range of Effects from Chlorinated Phenols: Chronic Values

Table H-2. Overview of the Range of Effects from Chlorinated Phenols: Chronic Values				
Species	**Method**	**Chemical**	**IC50 (µg/L)**	**Reference**
Cladoceran *Daphnia magna*		2,4,5-trichlorophenol	550-1040	Hattori et al., 1984; Steinberg et al., 1992
Cladoceran *Sphaerium corneum*		2,4,5-trichlorophenol	LOEC: 42	Heinonen et al., 1997
Cladoceran *Daphnia magna*		Tetrachloroguaiacol	140-370	Virtanen et al., 1989; Oikari et al., 1992
Cladoceran *Daphnia magna*		3,4,5-Trichlorocatechol	339	Dence et al., 1980
Cladoceran *Daphnia magna*		3,4,5-Trichloroguaiacol	450-730	Virtanen et al., 1989; Oikari et al., 1992
Cladoceran *Daphnia magna*		4,5,6-trichloroguaiacol	580-22,000	Dence et al., 1980; Kukkonen and Oikari, 1987; Petersen and Petersen, 1988; Neilson et al., 1991
Cladoceran *Daphnia magna*		Tetrachlorocatechol	2,230	Dence et al., 1980
Cladoceran *Ceriodaphnia dubia*	7 day	Pentachlorophenol	MATC: 80	Masters et al., 1991
Cladoceran *Ceriodaphnia dubia*		2,4,6-Trichlorophenol	Behavor change 4200	Bitton et al., 1996

Cladoceran *Ceriodaphnia dubia*		Pentachlorophenol	161	Hedtke et al., 1986
Cladoceran *Ceriodaphnia dubia*		4,5,6-trichloroguaiacol	1,800	Dence et al., 1980; Kukkonen and Oikari, 1987; Petersen and Petersen, 1988; Neilson et al., 1990
Cladoceran *Daphnia sp.*	16 day	Pentachlorophenol	130	Hermens et al., 1984
Cladoceran *Daphnia sp.*		2,3,4,6-Tetrachlorophenol	NOEC: 10	LeBlanc, 1980
Cladoceran *Daphnia sp*		2,3,4,6-Tetrachlorophenol	MATC: 650-1200	Shigeoka et al., 1989
Cladoceran *Daphnia sp.*		2,4,6-Trichlorophenol	5,000	Schauerte et al., 1982
Rotifer *Brachionus calyciflorus*		Pentachlorophenol	1,410-16,000	Ferrando et al., 1992; Crisinel et al., 1994; Liber and Solomon, 1994
Rotifer *Brachionus rubens*		Pentachlorophenol	160	Halbach et al., 1983
Rotifer sp.	7 day	2,3,4,6-Tetrachlorophenol	NOEC: 110-200	Liber et al., 1992
Copepod sp.	7 day	2,3,4,6-Tetrachlorophenol	NOEC: 210-510	Liber et al., 1992
Fathead Minnow *Pimephales promelas*	ELS	2,4,6-trichlorophenol	530 – 970	USEPA, 1978b
Fathead Minnow *Pimephales promelas*		2,4,5-trichlorophenol	LOEC: 398-725	Norberg-King, 1989; Arthur and Dixon, 1994
Fathead Minnow *Pimephales promelas*		2,4,5-trichlorophenol	NOEC: 297-536	Norberg-King, 1989; Arthur and Dixon, 1994
Fathead Minnow *Pimephales promelas*		2,4,5-trichlorophenol	MATC: 344-623	Norberg-King, 1989; Arthur and Dixon, 1994
Fathead Minnow *Pimephales promelas*	8 day	Pentachlorophenol	95-8,000	Phipps et al., 1981
Fathead Minnow *Pimephales promelas*		Tetrachlorocatechol	1,270	Geiger et al., 1985
Fathead Minnow *Pimephales promelas* embryo		Tetrachloroguaiacol	LOEC: 100	Woodland and Maly, 1997
Rainbow trout *Salmo gairdneri*		2,4,6-Trichlorophenol	LOEC: 200	Castren and Oikari, 1987
Rainbow trout *Salmo gairdneri*		2,4,5-trichlorophenol	NOEC: 4.6	McKim et al., 1985
Rainbow trout *Salmo gairdneri*	7 or 28 day	2,4,5-trichlorophenol	NOEC: 211	Neville, 1995

Rainbow trout *Salmo gairdneri*	7 or 28 day	2,4,5-trichlorophenol	LOEC: 438	Neville. 1996
Rainbow trout *Salmo gairdneri*	7 or 28 day	2,4,5-trichlorophenol	LOEC: 34-125	Neville, 1995
Rainbow trout *Salmo gairdneri*	7 or 28 day	2,4,5-trichlorophenol	NOEC: 62.5	Neville, 1995
Rainbow trout *Salmo gairdneri*		Pentachlorophenol	5.67-14.46	Eisler, 1991
Rainbow trout *Salmo gairdneri*		Pentachlorophenol	0.035 - 1	Eisler, 1992
Rainbow trout *Salmo gairdneri*		Pentachlorophenol	5.7-14.5	Eisler, 2000
Silver Salmon *Oncorhynchus mykiss*		Tetrachloroguaiacol	320	Leach and Thakore, 1975, 1977
Silver Salmon *Oncorhynchus mykiss*		Tetrachloroguaiacol	LOEC: 20	Johansen et al., 1995
Silver Salmon *Oncorhynchus mykiss*		Tetrachloroguaiacol	LOEC: 100	Yang and Randall, 1996
Silver Salmon *Oncorhynchus mykiss*		3,4,5-Trichloroguaiacol	750	Leach and Thakore, 1975
Brown Trout *Salmo trutta*		Tetrachlorocatechol	1,100	Hattula et al., 1981
Chinook salmon *Oncorhynchus tshawytscha*		Pentachlorophenol	3.9	Iwama et al., 1986
Chinook salmon *Oncorhynchus tshawytscha*		Pentachlorophenol	22	Nagler et al., 1986
Largemouth Bass *Micropterus salmoides*	120 day	Pentachlorophenol	54	Johansen et al., 1985
Bluegill *Lepomis macrochirus*		2,4,6-Trichlorophenol	LOEC: 320	Buccafusco et al., 1981
Guppy *Poecilia reticulata*	7 day	Pentachlorophenol	40-1,442	Adema and Vink, 1981
American flagfish *Jordanella floridae*		2,4,6-Trichlorophenol	LOEC: 750	Smith et al., 1991
American flagfish *Jordanella floridae*		2,3,4,6-Tetrachlorophenol	LOEC: 1,035	Smith et al., 1991
pond snail *Lymnaea acuminata*		Pentachlorophenol	0.16-0.293	Dence et al., 1980; Kukkonen and Oikari, 1987; Petersen and Petersen, 1988; Neilson et al., 1992
Caddisfly *Hydropsyche siltalai*		4,5,6-trichloroguaiacol	50	Gupta and Rao, 1982
Dugesiidae		2,4,6-Trichlorophenol	850	Yoshioka et al., 1986

Dugesia japonica			
Protozoa *Tetrahymena pyriformis*	2,4,5-trichlorophenol	680	Yoshioka et al.,1985

Table H- 3: Summary of Temperature Considerations for Salmon and Trout Life Stages

Life Stage	Temperature Consideration	Temperature (˚C)	Reference
Spawning and Egg Incubation	Temperature range spawning is most frequently observed in the field	4-14 (daily avg)	USEPA, 2001a USEPA, 2001b
	Egg incubation studies		
	- Results in good survival	4-12 (constant)	USEPA, 2001b
	- Optimal range	6-10 (constant)	
	Reduced viability of gametes in holding adults	>13 (constant	USEPA, 2001b
Juvenile Rearing	Lethal temperature (1 week exposure)	23-26 (constant)	USEPA, 2001b
	Optimal growth		
	- unlimited food	13-20 (constant)	USEPA, 2001b
	- limited food	10-16 (constant)	
	Rearing preference temperature in lab and field studies	10-17 (constant) <18 (7DADM)	USEPA, 2001a Welsh et al. 2001
	Impairment to smoltification	12-15 (constant)	USEPA, 2001b
	Impairment to Steelhead smoltification	>12 (constant)	USEPA, 2001b
	Disease risk (lab studies)		
	- high	>18-20 (constant)	USEPA, 2001c
	- elevated	14-17 (constant)	
	- minimized	12-13 (constant)	
Adult Migration	Lethal temperature (1 week exposure)	21-22 (constant)	USEPA, 2001b
	Migration blockage and migration delay	21-22 (average)	USEPA, 2001a USEPA, 2001b
	Disease risk (lab studies)		
	- high	>18-20 (constant)	USEPA, 2001c
	- elevated	14-17 (constant)	
	- minimized	12-13 (constant)	
	Adult swimming performance		
	- reduced	>20 (constant)	USEPA, 2001b
	- optimal	15-19 (constant)	
	Overall reduction in migration fitness due to cumulative stresses	17-18 (prolonged exposures)	USEPA, 2001b

Table H- 4: Summary of Temperature Considerations for Bull Trout Life Stages

Life Stage	Temperature Consideration	Temperature (˚C)	Reference
Spawning and Egg Incubation	Spawning initiation	<9 (constant)	USEPA, 2001b
	Peak spawning temperature	<7 (constant)	USEPA, 2001b
	Optimal egg incubation temperature	2-6 (constant)	USEPA, 2001b
	Substantially reduced egg survival and sized	6-8 (constant)	USEPA, 2001b
Juvenile Rearing	Lethal temperature	22-23 (constant)	USEPA, 2001b

Life Stage	Temperature Consideration	Temperature (˚C)	Reference
	(1 week exposure) Optimal growth - unlimited food - limited food	 12-16 (constant) 8-12 (constant)	 USEPA, 2001b Bull trout peer review, 2002
	Highest probability to occur in the field	12-13 (daily maximum)	USEPA, 2001a USEPA, 2001b Dunham et al, 2001 Bull trout peer review, 2002
	Competition disadvantage	>12 (constant)	USEPA, 2001a Bull trout peer review, 2002

The following papers were part of the 2002 bull trout peer review:

Myrick, Christopher A. et al. 2002. Bull Trout Temperature Tresholds Peer Review Summary.

Bull Trout Peer Review Questions and EPA's "Straw" Proposal. 2002.

McCullough, D. and Spaulding, S. 2002. Multiple Lines of Evidence for Determining Upper Optimal Temperature Thresholds.

Idaho Department of Environmental Quality (IDEQ). 2002. Dissenting Opinion on Biological Threshold Numbers Proposed by Regional Temperature Criteria Development Technical Workgroup.

Washington Department of Ecology (WDOE). 2002. Evaluating Standards for Protection of Aquatic Life in Washington's Surface Water Quality Standards, Temperature Criteria, Draft Discussion Paper and Literature Summary. pp. 17-30.

Table H- 5: Dissolved Oxygen Concentrations (mg/L) Versus Quantitative Level of Effect

Salmonids			
Life Stage	**Toxicity Effect**	**Water Column DO (mg/L)**	**Intergravel DO (mg/L)**
Embryo and larval stages	No production impairment Slight production impairment Moderate production impairment Severe production impairment Limit to avoid acute mortality	11 9 8 7 6	8 6 5 4 3
Other life stages	No production impairment Slight production impairment Moderate production impairment Severe production impairment Limit to avoid acute mortality	8 6 5 4 3	
Invertebrates			
	No production impairment Some production Impairment Acute Mortality Limit	8 5 4	

Appendix I:Washington State Toxics Monitoring Program Results for the Snake River: 2004, 2005 and 2009

Site	Species Code	Sample ID	Date collected	Total PCB aroclors (µg/kg)	q	Total PCB congeners (µg/kg)	q	T-DDT (µg/kg)	q	Total PBDE (µg/kg)	q	Total Chlordane (µg/kg)	q	2378 TCDD TEQ (ng/kg)	q	2378 TCDD (ng/kg)	q	Mercury (mg/kg)	q	Lipid MEL (%)	q	Lipid CL (%)	q	Mean Total Length (mm)	Mean Weight (g)	Mean Age (years)
Snake R. blw Lower Monumental Dam	CC	05084283/4315	11/8/04	111	m	165		373	m	26	m	9.1		1.11		0.520	U	0.347	m	7.2	m	7.3		491	1162	11.5
Snake R. blw Lower Monumental Dam	CC	1001015-25	11/9/09	89				583.85		11.64		6.41		0.474		0.369		0.341		9.59		12.4		467.5	1065	11.8
Snake R. blw Lower Monumental Dam	CC	1001015-26	11/9/09	74				195.8						0.278		0.229		0.194		10.4		10.6		513.3	1372	9.7
Snake R. blw Lower Monumental Dam	CC	1001015-27	11/9/09	91				277.6						0.383		0.323		0.382		6.04		7.7		488.7	1303.7	14
Snake R. blw Lower Monumental Dam	NPM	1001015-48	11/9/09	51				72.59		4.532		0.938		0.127		0.045		0.552		2.69		3.1		371.8	457.6	6
Snake R. blw Lower Monumental Dam	PEA	1001015-45	11/9/09	10.6				32.073		1.393		2.73	U					0.183		1.06				288.3	186.8	6.8
Snake R. blw Lower Monumental Dam	PEA	1001015-46	11/9/09	21.4				32.3										0.157		1.1				289.7	186.7	6
Snake R. blw Lower Monumental Dam	PEA	1001015-47	11/9/09	18.1				33										0.285		1.08				308.3	225	9.3
Snake R. blw Lower Monumental Dam	SMB	1001015-43	11/9/09	5.6				9.76		0.61		2.683	U					0.15		0.45				215.7	127	2.7
Snake R. blw Lower Monumental Dam	SMB	1001015-44	11/9/09	8.2				15.76										0.0874		0.89				217.7	132.3	2.7
Snake R. ds Clarkston	BG	1001015-12	10/20/09	2				7.96		1.002		2.395	U					0.1		0.45				157.2	80.8	2
Snake R. ds Clarkston	BG	1001015-13	10/20/09	2.7	U			4.5										0.107		0.29				162.6	104.6	2.5
Snake R. ds Clarkston	BG	1001015-14	10/20/09	2.7	U			6.1										0.0917		0.58				159.6	88	2.4
Snake R. ds Clarkston	CCP	1001015-55	10/20/09	79				357.79		33.38		3.94		0.283		0.113		0.439		10.4		10.3		588.2	3085.4	11
Snake R. ds Clarkston	LMB	05084316	11/30/04	4.2	U			22		1.8		0.85	U					0.140		0.7				283	346	1.9
Snake R. ds Clarkston	LMB	1001015-71	10/20/09	4.2				18.894		3.21		2.465	U					0.194		0.45				352.8	738	3.5
Snake R. ds Clarkston	LMB	1001015-72	10/20/09	2.8	U			9.92										0.139		0.6				337.7	840.3	3.3
Snake R. ds Clarkston	LMB	1001015-73	10/20/09	2.7	U			4										0.106		0.39				291	415	2
Snake R. ds Clarkston	MWF	05084317	11/29/04	106		70		38		9.4		0.98	U	0.413		0.100	U	0.120		2.0		1.4		299	231	2.5
Snake R. ds Clarkston	PEA	05084318	11/30/04	26				86		12		0.47						0.296		1.9				273	155	4.3
Snake R. ds Clarkston	PMP	1001015-59	10/20/09	0.96	U			5.23		0.47		2.455	U					0.0697		0.6				146	68.5	2.4
Snake R. ds Clarkston	SMB	1001015-60	10/20/09	4.4				13.348		1.515		2.39	U					0.263		0.76				295.4	366.2	4
Snake R. ds Clarkston	SMB	1001015-61	10/20/09	3.1				9.74										0.319		0.99				313.8	395.2	5
Snake R. ds Clarkston	SMB	1001015-62	10/20/09	2.6	U			9.94										0.218		0.47				298.4	369.2	4
Snake R. ds of Lower Granite Dam, RM 103-105	CC	1001015-20	10/21/09	44				212.1		13.55		4.28		0.703		0.157		0.395		9.01		11.4		534.8	1859.3	13
Snake R. ds of Lower Granite Dam, RM 103-105	CC	1001015-21	10/21/09	51				130.2						0.146		0.087		0.508		9.9		11.2		515	1315.3	10.5
Snake R. ds of Lower Granite Dam, RM 103-105	CC	1001015-22	10/21/09	55				182.5						0.246		0.158		0.514		6.32		7.2		547.3	1771	12

Site	Species Code	Sample ID	Date collected	Total PCB aroclors (µg/kg)	q	Total PCB congeners (µg/kg)	q	T-DDT (µg/kg)	q	Total PBDE (µg/kg)	q	Total Chlordane (µg/kg)	q	2378 TCDD TEQ (ng/kg)	q	2378 TCDD (ng/kg)	q	Mercury (mg/kg)	q	Lipid MEL (%)	q	Lipid CL (%)	q	Mean Total Length (mm)	Mean Weight (g)	Mean Age (years)
Snake R. ds of Lower Granite Dam, RM 103-105	CCP	1001015-66	10/21/09	70				488.09		5.02		4.96		0.383		0.214		0.332		12		12.7		630.8	3866.8	15.3
Snake R. ds of Lower Granite Dam, RM 103-105	CCP	1001015-67	10/21/09	225				1002.3										0.251		15				645	4801.7	13.7
Snake R. ds of Lower Granite Dam, RM 103-105	MWF	1001015-23	10/21/09	8.7				16.893		3.3		2.405	U	0.076		0.03	UJ	0.0844		1.67		3		259.7	172.7	1.7
Snake R. ds of Lower Granite Dam, RM 103-105	MWF	1001015-24	10/21/09	15.2				36.1										0.106		1.67				280.3	195	2.5
Snake R. ds of Lower Granite Dam, RM 103-105	NPM	1001015-40	10/21/09	25				40.289		4.06		0.564		0.16		0.055		0.994		1.89		3.4		358.2	420.7	6.8
Snake R. ds of Lower Granite Dam, RM 103-105	NPM	1001015-41	10/21/09	19.8				50.1										0.795		2.95				354.2	401.9	5.8
Snake R. ds of Lower Granite Dam, RM 103-105	NPM	1001015-42	10/21/09	17.7				38.1										0.825		2.04				369.4	461.4	5.8
Snake R. nr Central Ferry	BG	1001015-54	10/28/09	1.5				3.57		0.767		2.45	U					0.0681		0.81				148	71.5	2.3
Snake R. nr Central Ferry	CC	05084311	12/1/04	148		65		389		14		9.9		1.12		0.370		0.283		13		11		565	1842	12.0
Snake R. nr Central Ferry	CC	1001015-09	10/28/09	53				206.18		20.88		3.916		0.254		0.067		0.414		4.78		6.5		547	1525	13.8
Snake R. nr Central Ferry	CC	1001015-10	10/28/09	94				637						0.597		0.336		0.427		5.14		5.5		568.8	1448	12.3
Snake R. nr Central Ferry	CC	1001015-11	10/28/09	31				211.3						0.462		0.17		0.522		6.88		7.1		563.5	1795.8	12.3
Snake R. nr Central Ferry	CCP	1001015-63	10/28/09	65				678.27		4.701		7.34		0.389		0.164		0.252		10.3		13.7		611	2933	11.3
Snake R. nr Central Ferry	CCP	1001015-64	10/28/09	48				219										0.302		10.1				696.3	4985.3	14
Snake R. nr Central Ferry	CCP	1001015-65	10/28/09	138				518										0.233		19.9				671.7	5022	17
Snake R. nr Central Ferry	LMB	05084312	12/1/04	11				9.3		0.47		1.0	U					0.092		0.7				295	399	2.1
Snake R. nr Central Ferry	PEA	05084313	12/1/04	10				29		2.1		0.91	U					0.264		2.2				284	186	5.1
Snake R. nr Central Ferry	PEA	1001015-52	10/28/09	5.7				13.7				2.5	U					0.281		1.08				297.8	195.5	6
Snake R. nr Central Ferry	PMP	1001015-53	10/28/09	0.97	U			2.17		8.86	U	2.41	U					0.0543		0.35				128.3	47.1	2

Site	Species Code	Sample ID	Date collected	Total PCB aroclors (µg/kg)	q	Total PCB congeners (µg/kg)	q	T-DDT (µg/kg)	q	Total PBDE (µg/kg)	q	Total Chlordane (µg/kg)	q	2378 TCDD TEQ (ng/kg)	q	2378 TCDD (ng/kg)	q	Mercury (mg/kg)	q	Lipid MEL (%)	q	Lipid CL (%)	q	Mean Total Length (mm)	Mean Weight (g)	Mean Age (years)
Snake R. nr Central Ferry	SMB	1001015-49	10/28/09	4.6				10		0.94		2.37	U					0.347		0.54				345.7	573.7	5.3
Snake R. nr Central Ferry	SMB	1001015-50	10/28/09	2.8	U			7.7										0.115		0.3				237	196.7	2.3
Snake R. nr Central Ferry	SMB	1001015-51	10/28/09	2.8	U			6.7										0.281		1.08				297.8	195.5	6
Snake R. nr Central Ferry	YP	05084314	12/1/04	5.0	U			5.9		6.2	UJ	1.0	U					0.196		0.5				258	232	3.3
Snake R. nr Lyons Ferry, RM 59-60	CC	1001015-01	10/26/09	46				148.46		22.52		5.56		0.635		0.165		0.233		8.48		11.1		553.5	1764.8	10
Snake R. nr Lyons Ferry, RM 59-60	CC	1001015-02	10/26/09	53				169.6						0.353		0.03	UJ	0.254		6.94		8.2		522.5	1455	9.5
Snake R. nr Lyons Ferry, RM 59-60	CCP	1001015-56	10/26/09	25				201.74		15.27		4.636		0.104		0.072		0.189		4.78		6		563.3	2600.3	7.3
Snake R. nr Lyons Ferry, RM 59-60	CCP	1001015-57	10/26/09	113				422										0.212		18.7				616.7	3210.3	9
Snake R. nr Lyons Ferry, RM 59-60	CCP	1001015-58	10/26/09	70				390.7										0.233		9.02				555.7	3046	6.7
Snake R. ups Ice Harbor Dam RM 11-12	CC	1001015-05	11/10/09	61				275.46		12.7		3.06		0.863		0.22		0.233		10.4		12.2		494.6	1250	9.4
Snake R. ups Ice Harbor Dam RM 11-12	CC	1001015-06	11/10/09	142				432.2						1.248		0.413		0.251		13.6		14.7		527.2	1505.4	10.6
Snake R. ups Ice Harbor Dam RM 11-12	CC	1001015-07	11/10/09	84				224.3						0.725		0.215		0.243		10.4		11		548.5	1697	12.3
Snake R. ups Ice Harbor Dam RM 11-12	CCP	06024751	11/14/05	115		65		146		30		5.1		0.417		0.100		0.180		5.4		1.7		675	4207	13.6
Snake R. ups Ice Harbor Dam RM 11-12	LMB	1001015-16	11/10/09	1.8				5.79		0.82		2.46	U					0.0441		0.35				238.7	208.7	1.3
Snake R. ups Ice Harbor Dam RM 11-12	NPM	1001015-15	11/10/09	24.2				53.868		3.32		0.567		0.006		0.041	UJ	0.271		2.54		3.8		383.2	608.4	4.2
Snake R. ups Ice Harbor Dam RM 11-12	PEA	05524731	11/14/05	43				22		2.5		0.98	U					0.190		1.8				286	4207	5.4
Snake R. ups Ice Harbor Dam RM 11-12	PEA	1001015-35	11/10/09	22.8				75.314		3.856		0.771						0.164		1.34				295.8	203.5	5.8
Snake R. ups Ice Harbor Dam RM 11-12	PEA	1001015-36	11/10/09	16.5				40.5										0.163		1.09				283.3	187.3	6.7
Snake R. ups Ice Harbor Dam RM 11-12	PEA	1001015-37	11/10/09	10				21.85										0.156		0.96				297.7	205.7	6.7
Snake R. ups Ice Harbor Dam RM 11-12	SMB	1001015-32	11/10/09	5				11.2		1.06		2.42	U					0.111		0.36				270.7	249.3	3.7
Snake R. ups Ice Harbor Dam RM 11-12	SMB	1001015-33	11/10/09	2.7	U			7										0.0845		0.2				205.3	99.7	2.3
Snake R. ups Ice Harbor Dam RM 11-12	SMB	1001015-34	11/10/09	2.6	U			5										0.0876		0.19				207	99.3	3

Site	Species Code	Sample ID	Date collected	Total PCB aroclors (µg/kg)	q	Total PCB congeners (µg/kg)	q	T-DDT (µg/kg)	q	Total PBDE (µg/kg)	q	Total Chlordane (µg/kg)	q	2378 TCDD TEQ (ng/kg)	q	2378 TCDD (ng/kg)	q	Mercury (mg/kg)	q	Lipid MEL (%)	q	Lipid CL (%)	q	Mean Total Length (mm)	Mean Weight (g)	Mean Age (years)
Snake R. ups Ice Harbor Dam RM 11-12	YP	05524730	11/14/05	4.9	U			6.7		0.60		0.99	U					0.045		0.6				204	94	1.2
Snake R. ups Ice Harbor Dam RM 11-12	YP	1001015-38	11/10/09	1.8				4.75		0.664		2.425	U					0.141		0.54				242.3	182.7	3.3
Snake R. ups Ice Harbor Dam RM 11-12	YP	1001015-39	11/10/09	2.7	U			4										0.0745		0.19				223.3	156.3	1.7

Data Qualifiers and Notes

U = The analyte was not detected at or above the reported value.

UJ = The analyte was not detected at or above the reported estimated result.

m = mean value from analyses of field duplicates where two results are available. Where analysis was not done on only one sample, that sample result is given. Where both values were non-detect, the highest value was used. Where one duplicate was qualified as a non-detect (U, UJ), the reported value was used in determining the mean value.

Species Codes: BC = Black crappie, BG = Bluegill, BNT = Brown trout, BUR = Burbot, CC = Channel catfish, CCP = Common carp, CHK = Chinook salmon, CTT = Cutthroat trout, GCP = Grass carp, KOK = Kokanee salmon, LMB = Largemouth bass, LWF = Lake whitefish, MWF = Mountain whitefish, NPM = Northern pikeminnow, PEA = Peamouth, PMP = Pumpkinseed RBT = Rainbow trout, SMB = Smallmouth bass, WAL = Walleye, YP = Yellow perch.