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

April 2, 2004

NMFS Tracking No.:
2000/01449

L. John Iani,
Regional Administrator
U.S. Environmental Protection Agency, Region 10
1200 Sixth Avenue
Seattle, WA 98101

Re: Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for Potlatch Pulp and Paper Mill, Lewiston, Idaho, National Pollution Discharge Elimination System (NPDES) Permit No.: ID-000116-3 for the discharge of effluents into the Snake River, Nez Perce County, Idaho and Asotin County, WA (1 Project)

Dear Mr. Iani:

Enclosed is a document containing a biological opinion (Opinion) prepared by NOAA's National Marine Fisheries Service (NOAA Fisheries) pursuant to section 7 of the Endangered Species Act (ESA) on the effects of the proposed National Pollution Discharge Elimination System (NPDES) permit for the Potlatch Pulp and Paper Mill located in Lewiston, Idaho. In this Opinion, NOAA Fisheries concludes that the proposed action is not likely to jeopardize the continued existence of ESA-listed Snake River sockeye salmon, Snake River fall and spring/summer chinook salmon, Snake River steelhead, and all currently designated ESA critical habitat. As required by section 7 of the ESA, NOAA Fisheries includes in the Incidental Take Statement a description of reasonable and prudent measures with nondiscretionary terms and conditions that NOAA Fisheries believes are necessary to avoid or minimize the impact of incidental take associated with this action.

This document contains a consultation on essential fish habitat (EFH) pursuant to section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) and its implementing regulations (50 CFR Part 600). NOAA Fisheries concludes that the proposed action may adversely affect designated EFH for chinook salmon. As required by section 305(b)(4)(A) of the MSA, included are conservation recommendations that NOAA Fisheries believes will avoid, minimize, mitigate, or otherwise offset adverse effects on EFH

resulting from the proposed action. As described in the enclosed consultation, 305(b)(4)(B) of the MSA requires that a Federal action agency must provide a detailed response in writing within 30 days of receiving an EFH conservation recommendation.

If you have any questions regarding this letter, please contact David Mabe at (208)378-5696 or Ed Murrell at (208)378-5707 of my staff in the Idaho State Habitat Office.

Sincerely,

for Michael R Crouse

D. Robert Lohn
Regional Administrator

Enclosure

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Endangered Species Act Section 7 Consultation Biological Opinion
and
Magnuson-Stevens Fishery Conservation and Management Act
Essential Fish Habitat Consultation

Potlatch Pulp and Paper Mill, Lewiston, Idaho
National Pollution Discharge Elimination System (NPDES)
Permit No.: ID-000116-3 for the discharge of effluents into the
Snake River, Nez Perce County, Idaho and Asotin County, Washington

Agency: United States Environmental Protection Agency

Consultation Conducted By: National Marine Fisheries Service (NOAA Fisheries)
Northwest Region

Date Issued: April 2, 2004

Issued by: For Michael R Crouse
D. Robert Lohn
Regional Administrator

Refer to: 2000/01449

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ABBREVIATIONS AND ACRONYMS

11-KT	11-Ketotestosterone
AA	Abietic Acid
ACR	Acute to Chronic Ratio
AhR	Aryl Hydrocarbon Receptor
Al	Aluminum
AOX	Adsorbable Organic Halides
AQUIRE	Aquatic Information and Retrieval
ASB	Aeration and Stabilization Basin
AST	Activated Sludge Treatment
ASTM	American Society for Testing Materials
ATP	Adenosine Triphosphate
ATPase	Adenosine Triphosphatase
AWQC	Ambient Water Quality Criteria
BA	Biological Assessment
BAFs	Bioaccumulation Factors
BAT	Best Available Technology
BCF	Bioconcentration Factor
BCT	Best Conventional Pollutant Control Technology
BE	Biological Evaluation, considered synonymous to BA by NOAA
BKME	Bleached Kraft Mill Effluent
BOD	Biochemical Oxygen Demand
BOR	Bureau of Reclaimation
BPT	Best Practicable Control Technology Currently Available
BSAF	Biota-Sediment-Accumulation Factor
CCC	Criteria Continuous Concentration
CFR	Code of Federal Regulations
CM	Conservation Measures
CMC	Critical Maximum Concentration
COD	Chemical Oxygen Demand
COE	Corps of Engineers
CORMIX	Cornell Mixing Zone Expert System
CPC	Chlorinated Phenolic Compounds
CPOC	Chlorinated Phenolic Organic Compound
CWA	Clean Water Act
CWRM 2	Clearwater River Mile 2
DCDHAA	12,14-dichlorodehydroabietic acid
DHAA	Dehydroabietic Acid
DO	Dissolved Oxygen
E2	Concentrations of Estradiol
EFH	Essential Fish Habitat
ENAs	Erythrocytic Nuclear Abnormalities
EPA	Environmental Protection Agency
EROD	7-ethoxyresorufin-o-deethylase
ESA	Endangered Species Act
ESS	Experimental Stream Studies
ESU	Evolutionarily Significant Units
FAV	Final Acute Toxicity Values
FCRPS	Federal Columbia River Power System
FL	Fork-Length
ft	Feet

GSI	Gonadosomatic Index
Hg	Mercury
HMZ	Hydrodynamic Mixing Zone
HSDB	Hazardous Substances Data Base
HSI	Hepatosomatic Index
IDEQ	Idaho Department of Environmental Quality
IDFG	Idaho Department of Fish and Game
ILT	Incipient Lethal Temperature
lambda	Median Population Growth Rate
lb/day	Pounds per day
LC50	50% Lethal Concentration
LCR	Lower Clearwater River
LD50	50% Lethal Dose
LGD	Lower Granite Dam
LGR	Lower Granite Reservoir
LOEC	Lowest Observed Effect Concentration
MATC	Maximum Acceptable Toxican Concentration
MCDHAA	14-monochlorodehydroabietic acid
MFO	Mixed Function Oxygenase
MSA	Magnuson-Stevens Fishery Conservation and Management Act
MW	Megawatts
NAS	National Academy of Sciences
NCASI	National Council of the Paper Industry for Air and Stream Improvement
ND	Non-detectable
NEC	no effect concentrations
NOAA Fisheries	NOAA's National Marine Fisheries Service
NOEC	No Observed Effect Concentration
NOEL	No-Observable Effect Levels
NPDES	National Pollution Discharge Elimination System
OD	Oxygen Delignification
Opinion	Biological Opinion
Pb	Lead
PCBs	Polychlorinated Biphenyls
PCD	Partition-controlled delivery
PCDD	Polychlorinated Dibenzodioxins
PCDD/F	Polychlorinated Dibenzodioxins/Furans
PCDFs	Polychlorinated Dibenzofurans
PCP	Pentachlorophenol
PFMC	Pacific Fisheries Management Council
Potlatch Mill	The Potlatch Corporation – Lewiston Complex
ppq	Parts Per Quadrillion
ppt	Parts Per Trillion
Rkm	River Kilometer
RM	River Mile
RMZ	Regulatory Mixing Zone
Se	Selenium
SRM 110	Snake River Mile 110
SRM 120	Snake River Mile 120
SRM 131	Snake River Mile 131
SRM 137	Snake River Mile 137
SRM 139	Snake River Mile 139

SRM 144	Snake River Mile 144
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TEC	Toxicity Equivalent Concentration
TeCG	Tetrachloroguaiacol
TEF	Toxic Equivalent Factor
TEQ	Toxic Equivalent
TMDL	Total Maximum Daily Loads
TOC	Total Organic Carbon
TON	Total Organic Nitrogen
TRS	Total Reduced Sulfur
TSS	Total Suspended Solids
TU	Toxic Unit
UDP	Uridine Diphosphate
UILT	Upper Incipient Lethal Temperature
USFWS	U.S. Fish and Wildlife Service
USGS	U. S. Geological Survey
UUILT	Ultimate Upper Incipient Lethal Temperature
VTG	Vitellogenin
WET	Whole Effluent Toxicity
WSF	Water Soluble Fraction
wt	Weight
ZID	Zone of Initial Dilution

1. INTRODUCTION

1.1 Background

The U.S. Environmental Protection Agency (EPA), Region 10, National Pollution Discharge Elimination System (NPDES) Permits Unit has requested Endangered Species Act (ESA) section 7 consultation with National Marine Fisheries Service (NMFS) which has changed its name to NOAA Fisheries in 2001. The consultation is for the proposed reissuance of Potlatch Corporation's NPDES permit to discharge effluent into the Snake and Clearwater Rivers near Lewiston, Nez Perce County, Idaho. The ESA consultation on the EPA's proposed NPDES permit (June 19, 2003) was initiated on December 12, 2003.

The ESA of 1973 (16 USC 1531-1544), as amended, establishes a national program for the conservation of threatened and endangered species of fish, wildlife, and plants, and the habitat on which they depend. Section 7(a)(2) of the ESA requires Federal agencies to consult with U.S. Fish and Wildlife Service (USFWS) and NOAA's National Marine Fisheries Service (NOAA Fisheries), as appropriate, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or to adversely modify or destroy their designated critical habitats. This biological opinion (Opinion) completes consultation pursuant to section 7(a)(2) of the ESA and implementing regulations found at 50 C.F.R. 402.

The water and sediment quality resulting from the issuance of the proposed NPDES permit has the potential to affect four Evolutionarily Significant Units (ESUs) of salmonids listed under the ESA and that are under the regulatory purview of NOAA Fisheries. They include the Snake River spring/summer and fall chinook salmon ESUs (two distinct population segments of *Oncorhynchus tshawtscha* listed as 'threatened' pursuant to the ESA on April 22, 1992; 50 C.F.R. § 223.102), Snake River sockeye salmon ESU (*Oncorhynchus nerka*, listed as 'endangered' on November 20, 1991; 50 C.F.R. § 224.101(a)), and Snake River steelhead ESU (*Oncorhynchus mykiss*, listed as 'threatened' on August 18, 1997; 50 C.F.R. § 223.102).

Critical habitat was designated for Snake River fall chinook, spring/summer chinook, and sockeye salmon ESUs on December 28, 1993 (50 C.F.R. § 226.205). Critical habitat encompasses historically accessible reaches of all rivers and tributaries within each ESU's range, excluding specifically above Hells Canyon Dam, Dworshak Dam, and Napias Falls on Napias Creek.

Proposed actions are defined by NOAA Fisheries regulations (50 C.F.R. 402.02) 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 the EPA regulates and permits pollution discharges to waters of the United States pursuant to the Clean Water Act (CWA) of 1972, a Federal nexus exists for interagency consultation under ESA section 7(a)(2). The proposed discharge permit action would affect sediment or water quality beginning at the lower Snake and Clearwater Rivers confluence and continuing to the mouth of the Snake River. This area is occupied by Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River sockeye salmon, and Snake River steelhead. All river reaches that would be affected are designated as critical habitat for Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River sockeye salmon.

The objective of this Opinion is to determine whether the proposed Potlatch Mill NPDES Permit will jeopardize the continued existence of the Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, or Snake River steelhead

trout; or result in the destruction or adverse modification of critical habitat for Snake River sockeye, spring/summer chinook, and fall chinook salmon.

The analysis also fulfills the essential fish habitat (EFH) requirements under the MSA. The MSA, as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), established procedures designed to identify, conserve, and enhance EFH for those species regulated under a Federal fisheries management plan. Federal agencies must consult with NOAA Fisheries on all actions, or proposed actions, authorized, funded, or undertaken by the agency, that may adversely affect EFH (§305(b)(2)).

1.1.1 Consultation History

On October 30, 1985, EPA issued Potlatch an NPDES permit which was subsequently modified on June 30, 1988. The EPA issued Potlatch a new NPDES permit with an effective date of April 7, 1992. The 1992 permit regulated significantly more parameters than the 1988 permit. This permit was challenged by the Idaho Conservation League, the Dioxin/Organochlorine Center, and the Nez Perce Tribe. Because the status of the permit challenge was undecided and request was made for evidentiary hearing, the permit did not become effective by operation of EPA's regulations, and the 1985 permit remained in effect (40 C.F.R. 124.15(b)(2)).

These permit challenges were collectively withdrawn by February 1997, and the permit subsequently became effective on March 16, 1997. Because the expiration date of the permit had not changed, the permit expired shortly after becoming effective, on April 7, 1997. Potlatch submitted a timely NPDES permit application for renewal in 1996. The EPA was required by law to administratively extend the expired NPDES permit until such time as it reissues Potlatch a new permit (5 U.S.C. 558(c)).

The EPA initiated consultation with NOAA Fisheries on the Potlatch Mill NPDES permit in a letter dated March 6, 1997. A biological assessment (BA) for the proposed permit action was submitted November 1, 2000 (EPA 2000). On February 18, 2003, NOAA Fisheries provided EPA with a discussion draft biological opinion that had preliminary conclusions of "jeopardy" and "adverse modification."

After submittal of the draft biological opinion by NOAA Fisheries, the parties consulted during the summer and fall 2003 through numerous technical meetings and calls, addressing effects and concerns identified in the draft opinion. The 2000 draft NPDES permit was revised by EPA based upon comments from the public and significant new information since the close of the previous comment period, which included issuance of regional water quality standard guidance for temperature (EPA 2003a). The EPA determined the changes were substantive enough to re-open the public comment period on June 19, 2003. On July 27, 2003, NOAA Fisheries and the U.S. Fish and Wildlife Service jointly notified EPA that the revised permit constitutes a new Federal action and the BA should be withdrawn.

During the summer and fall of 2003, NOAA Fisheries met numerous times with USFWS, EPA, and Potlatch representatives in Boise, Idaho and Seattle, Washington to discuss the various scientific analyses needed to address the concerns expressed in NOAA Fisheries' February 18, 2003, draft Opinion. NOAA Fisheries received a new biological evaluation (BE) for the amended proposed permit action on December 12, 2003 (EPA 2003). On March 2, 2004, EPA amended the action and supplemented the BE with three important conservation measures. A detailed chronology of consultation events related to this project is included in Appendix 1.

The EPA thoroughly reevaluated the proposed Potlatch NPDES permit in their 2003-04 BE, which also included their latest proposed permit revisions (EPA 2003). Additionally, Potlatch Corporation volunteered to provide additional conservation measures that would result in lesser amounts of contaminants and heat discharging from its outfall into the receiving water (Appendix 3, Letter from Sue Somers, Potlatch Corporation to Robert Robichaud, EPA and David Mabe, NOAA Fisheries of December 19, 2003). Examples of significant analyses that are presented in EPA's 2003 BE and that were either not previously included or were improved from EPA's (2000) BA include the following:

- The 2003 EPA's BE provided information concerning the important mixing scenarios that occur at the confluence of the Clearwater River with the Snake River.

- The EPA (2003) made extensive analyses drawing heavily on recent research by the Pacific Northwest National Laboratory on the hydrodynamics of the confluence zone, which provided improved understanding of effluent dispersion and pollutant transport in the Snake River. Since the river's confluence is near the effluent outfall site, this new information was applied in the 2003 modeling efforts and resulted in more accurate information than that presented in EPA's 2000 BA.

- The 2003 BE modeling effort analyzed permit allowed concentrations during seasonal conditions when sensitive life stages of listed salmonids would likely be present.

- The 2003 BE more appropriately assessed the potential for adverse effects of the proposed action by analyzing for sublethal and behavioral effects on ESA-listed salmonids. This resulted in EPA using lower concentrations of some contaminants as benchmarks for concern.

- The additive toxicities of multiple contaminants were assessed by using a toxicity unit approach in the 2003 BE.

- The 2003 BE recognizes that other sources and loads of pollutants exist in the area and that those contributions limit the background assimilative ability of receiving waters within certain reaches or areas containing important salmonid habitat.

- The concern that particulates in the effluent and the organic contaminates that sorb onto them can affect sediment quality, fish forage items (aquatic invertebrates), and could thereby result in long-term bioaccumulative health problems for salmonids is recognized and evaluted in the 2003 BE.

- An extensive monitoring plan designed to determine the effects of the effluent on salmonids and their forage items has been jointly developed and is addressed by EPA's amendments to the 2003 BE.

In sum, the proposed action has changed in material respects, the 2003-04 EPA BE is a major improvement over previous versions, and significant new information and guidance has emerged since NOAA Fisheries completed its draft biological opinion in February 2003. In addition to new information in the BE, NOAA Fisheries analyzed new effluent chemicals of concern and better refined its data concerning listed species' use of the Lower Granite Reservoir (LGR). Consequently, the analysis in this final biological opinion differs considerably from that in the February 2003 draft opinion. Specific differences are discussed fully in section 2.3, Analysis of Effects.

1.2 Description of Action

1.2.1 Facility Background

The Potlatch Corporation – Lewiston Complex (Potlatch Mill) is owned and operated by Potlatch Corporation, which has its headquarters in Spokane, Washington. The facility site was developed as a saw mill since 1926. 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. The panel operation was closed and dismantled in the 1980's. Growth continued during the 1990's until, today, the Potlatch Mill has the capability to produce 500,000 tons per day of paperboard and tissue and 160 million feet per year of lumber.

The Potlatch facility is 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 (Figures 1-2).

1.2.2 Industrial Process

The Potlatch Mill manufactures wood products and bleached grades of paperboard, tissue and market pulp by the kraft (sulfate) process. In 1950, a bleached kraft pulp mill and paperboard machine was built at the site. The pulp and paperboard operations were expanded several times in the past fifty years. The first of three tissue machines was build in the mid-1960's and the two additional machines were added in the last 25 years. In the 1990's the mill began the conversion of its bleaching process from chlorine to chlorine dioxide. The mill met the standards for EPA's Cluster Rule in 2001.

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 lignins and resins, or because of the inefficient washing of the spent cooking liquor from the pulp. In order 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.

1.2.2.1 Wastewater Characterization

Potlatch monitors certain effluent parameters to comply with monitoring requirements specified in the permit under which the Mill currently discharges. Table 1 (from Table III-2 of the BE) summarizes the range and average concentrations of parameters monitored in effluent during 2002.

Table 1. Summary of Outfall 001 Discharge Composition (2002)

Parameter	Units	Range of Concentrations [1]	Average Concentration [2,3]
Flow	mgd	0 - 38.9	34.17
pH	s.u.	6.6 - 8.4	7.39
Color	c.u.	Note 13	1,800
Temperature	Deg F	67.7 - 93.7	79.92
Biochemical Oxygen Demand (BOD)	mg/L	19 - 86	33.37
Chemical Oxygen Demand (COD)	mg/L	320 - 831	522.92 [4]
Total Organic Carbon (TOC)	mg/L	Note 13	190
Total Suspended Solids (TSS)	mg/L	14.2 - 142.8	64.37 [5]
Turbidity	NTU	Note 13	77.1
Fecal Coliform Bacteria	MPN/100 mL	Note 13	50
a-Terpineol	µg/L	11 - 11	11
1,2,3,4,6,7,8,9-OCDD	µg/L	**3.25E-06 - 3.25E-06**	ND
1,2,3,4,6,7,8,9-OCDF	µg/L	**2.25E-06 - 2.25E-06**	ND
1,2,3,4,6,7,8-HpCDD	µg/L	**1.55E-06 - 1.55E-06**	ND
1,2,3,4,6,7,8-HpCDF	µg/L	**7.50E-07 - 7.50E-07**	ND
1,2,3,4,7,8,9-HpCDF	µg/L	**1.20E-06 - 1.20E-06**	ND
1,2,3,4,7,8-HxCDD	µg/L	**2.45E-06 - 2.45E-06**	ND
1,2,3,4,7,8-HxCDF	µg/L	**5.00E-07 - 5.00E-07**	ND
1,2,3,6,7,8-HxCDD	µg/L	**8.50E-07 - 8.50E-07**	ND
1,2,3,6,7,8-HxCDF	µg/L	**5.00E-07 - 5.00E-07**	ND
1,2,3,7,8,9-HxCDD	µg/L	**8.00E-07 - 8.00E-07**	ND
1,2,3,7,8,9-HxCDF	µg/L	**8.00E-07 - 8.00E-07**	ND
1,2,3,7,8-PeCDD	µg/L	**7.00E-07 - 7.00E-07**	ND
1,2,3,7,8-PeCDF	µg/L	**5.00E-07 - 5.00E-07**	ND
12-Chlorodehydroabietic Acid	µg/L	**20 - 20**	ND
14-Chlorodehydroabietic Acid	µg/L	**20 - 20**	ND
2,3,4,6,7,8-HxCDF	µg/L	**5.00E-07 - 5.00E-07**	ND
2,3,4,6-Tetrachlorophenol	µg/L	**1.25 - 1.25**	ND
2,3,4,7,8-PeCDF	µg/L	**3.50E-07 - 3.50E-07**	ND
2,3,7,8-TCDD	µg/L	**5.00E-07 - 4.30E-06**	ND
2,3,7,8-TCDF	µg/L	4.00E-07 - 3.60E-06	1.71875E-06 [6]
2,4,5-Trichlorophenol	µg/L	**1.25 - 1.25**	ND
2,4,6-Trichlorophenol	µg/L	**1.25 - 1.25**	ND
2-Methyl-2-cyclopenten-1-one	µg/L	76 - 76	76
3,4,5-Trichlorocatechol	µg/L	**2.5 - 2.5**	ND

Table 1. Summary of Outfall 001 Discharge Composition (2002)

Parameter	Units	Range of Concentrations [1]	Average Concentration [2,3]
3,4,5-Trichloroguaiacol	µg/L	**1.25 - 20**	ND
3,4,6-Trichlorocatechol	µg/L	**2.5 - 2.5**	ND
3,4,6-Trichloroguaiacol	µg/L	**1.25 - 1.25**	ND
3-Methyl-2-cyclopenten-1-one	µg/L	15 - 15	15
4,5,6-Trichloroguaiacol	µg/L	**1.25 - 1.25**	ND
9,10-Dichlorostearic Acid	µg/L	**20 - 20**	ND
Abietic Acid	µg/L	**20 - 20**	ND
Aluminum, Dissolved	µg/L	185 - 216	200.5
Aluminum, Total	µg/L	282 - 314	298 [7]
Ammonia as Nitrogen	µg/L	**25 - 25**	ND [8]
Arsenic, Dissolved	µg/L	**0.5** - 1	0.75
Arsenic, Total	µg/L	1 - 1	1 [9]
Chloroform	µg/L	1.4 - 2.5	1.95 [10]
Chromium, Hexavalent, Dissolved	µg/L	**5** - 64	34.5
Copper, Dissolved	µg/L	**5 - 5**	ND
Copper, Total	µg/L	**5 - 5**	ND
Dehydroabietic Acid	µg/L	**20 - 20**	ND
Dichlorodehydroabietic Acid	µg/L	**20 - 20**	ND
Halides, Adsorbable Organic (AOX)	µg/L	2980 - 37800	13875
Isopimaric Acid	µg/L	**20 - 20**	ND
Lead, Dissolved	µg/L	**1 - 1**	ND
Lead, Total	µg/L	**1 - 1**	ND [11]
Linoleic Acid	µg/L	**20 - 20**	ND
Magnesium	µg/L	Note 13	3740
Manganese	µg/L	Note 13	511
Mercury, Dissolved	µg/L	**0.1 - 0.1**	ND
Mercury, Total	µg/L	**0.1 - 0.1**	ND
Oleic Acid/Linolenic Acid	µg/L	**20 - 20**	ND
Pentachlorophenol (PCP)	µg/L	**2.5 - 2.5**	ND
Phosphorus	mg/L	Note 13	10
Pimaric Acid	µg/L	**20 - 20**	ND
Selenium, Dissolved	µg/L	**50 - 50**	ND
Selenium, Total	µg/L	**50 - 50**	ND
Sulfate	mg/L	Note 13	280
Surfactants	mg/L	Note 13	0.49

Table 1. Summary of Outfall 001 Discharge Composition (2002)

Parameter	Units	Range of Concentrations [1]	Average Concentration [2,3]
Tetrachlorocatechol	µg/L	**2.5 - 2.5**	ND
Tetrachloroguaiacol	µg/L	**2.5 - 20**	ND
Titanium	µg/L	Note 13	12
Total Dimethylcyclopenten-1-one	µg/L	**28 - 28**	ND
Total HpCDD	µg/L	**2.00E-06 - 2.00E-06**	ND
Total HpCDF	µg/L	**9.00E-07 - 9.00E-07**	ND
Total HxCDD	µg/L	**5.10E-06 - 5.10E-06**	ND
Total HxCDF	µg/L	**6.50E-07 - 6.50E-07**	ND
Total Organic Nitrogen (TON)	mg/L	Note 13	22.9
Total PeCDD	µg/L	**2.85E-06 - 2.85E-06**	ND
Total PeCDF	µg/L	3.30E-06 - 3.30E-06	0.0000033
Total TCDD	µg/L	**8.00E-07 - 4.30E-06**	ND
Total TCDF	µg/L	**1.50E-06 - 8.70E-06**	3.3E-06
Trichlorosyringol	µg/L	**1.25 - 1.25**	ND
Whole Effluent Toxicity (WET)	TU_c	Note 13	10
Zinc, Dissolved	µg/L	**5 - 5**	ND
Zinc, Total	µg/L	14 - 15	14.5 [12]

Notes:
1. Numbers in bold are equal to 1/2 Detection Limits.
2. ND: Not Detected.
3. In calculating the average concentration, one half 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.
4. The maximum reported value was 1650 mg/L
5. The maximum reported value was 206 mg/L.
6. The maximum reported value was 82 pg/L.
7. This is the maximum value.
8. The maximum reported value was 860 µg/L.
9. The maximum reported value was 9 µg/L
10. The maximum reported value was 33 µg/L.
11. The maximum reported value was 8 µg/L.
12. The maximum reported value was 99 µg/L.
13. No data.

1.2.3 Description of Discharge Points (Outfall 001 and ASB Pond)

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 [RM] 140). In addition to Outfall 001, the facility also discharges seepage into the groundwater from the Aerated Stabilization Basin (ASB) pond to the Clearwater River.

1.2.3.1 Outfall 001 Effluent

Effluent is released through Outfall 001 from a diffuser pipe 400 feet long sloping down the east-bank of the old Snake River channel at a water depth of about 26-32 feet, from the Lower Granite Reservoir (LGR) low pool elevation of 733 feet. The Snake River channel is 800-900 feet in width at this point. The diffuser extends across about 40% of the Snake River channel width at a 38 degree northward angle from perpendicular to flow. The diffuser consists of 80 individual ports on 5-foot centers, 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.

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 eastward and upward from each port at velocity of about 18 ft/s that is higher than the river velocity (0.8-3 ft/s). 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 et al. 1996). The effluent and entrained water completely mix within the jet, and thus the effluent is rapidly diluted.

The site of the mixing induced by the diffuser is at the discharge ports, near the bottom of the river. Model results with the Cornell Mixing Zone Expert System (CORMIX) model show roughly sevenfold dilution of the effluent by bottom water within 12 feet of the diffuser. Over 13-fold dilution occurs within 55 feet. Active mixing continues for 308 feet downstream of the diffuser, by which time the effluent is diluted more than 40 times with both deep and shallow water. The amount of dilution and distance over which it occurs changes depending upon ambient river and effluent discharge conditions, but the above-described general dynamics of rapid and voluminous mixing with deep water within a short distance of the diffuser prevail under all conditions (EPA 2003).

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

Before discharge through Outfall 001, the water is stored and treated in the 102-acre 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. The BE describes the ASB pond, estimates of its seepage and water quality. A summary of this information is presented below.

The ASB occupies the eastern portion of an old log pond that was drained just before 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 (LGD)by the U.S. Army Corps of Engineers (COE). 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 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. Groundwater quality testing by the Potlatch Corporation indicated elevated sodium and chloride concentrations that were attributed to ASB seepage.

The Hydraulic Connection Study completed by Cascade Earth Sciences in 1999 reported estimates of seepage rates from the ASB that ranged from a low of 0.021 mgd to a maximum of 5.98 mgd. The estimated seepage rate based on a water balance method that used annual recorded and estimated inflows and outflows to the ASB was approximately 3.3 mgd for 1998 data. The Hydraulic Connection Study also estimated groundwater flux between the ASB and the Clearwater River. Estimates of groundwater flux based on Darcy's Law were completed using a range of input for hydraulic conductivity, hydraulic gradient, and aquifer thickness due to variability and uncertainty in the estimates of each. The groundwater flux from the ASB to the Clearwater River based on the best available data was estimated to be 0.85 mgd. The quantity of groundwater flux from the ASB that discharges to the groundwater sink to the west was estimated to be 0.27 mgd for a combined total of 1.12 mgd.

During the Hydraulic Connection study, loading rates to the Clearwater River were estimated under the assumption that the groundwater conditions observed in the groundwater monitoring wells surrounding the ASB along the northern and eastern boundaries are representative of any discharge. Loading rate calculations were completed for ammonia-N, chloride, total suspended solids (TSS), and arsenic and values were estimated at 22 lbs/day, 1,559 lbs/day, 783 lbs/day, and 0.047 lbs/day, respectively from the ASB to the Clearwater River. Loading rates for biochemical oxygen demand (BOD), aluminum (Al), lead (Pb), mercury (Hg), and selenium (Se) were not completed due to the lack of detectable concentrations observed during the study period.

Potlatch (2000) submitted a report to EPA that concluded that, for the inorganic constituents considered (dissolved arsenic, mercury, selenium, lead and ammonia); the ambient water quality of the Clearwater River is not impacted by the seepage from the ASB. It was also concluded that ASB seepage has no measurable effect on the ambient river temperature.

1.2.4 Permit Limits

The EPA BE provides a good explanation of how NPDES permit limits are established. This pertinent information is restated here.

The 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 C.F.R. 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 CWA requires technology-based controls on effluents. This section of the CWA requires that, by March 31, 1989, all permits contain effluent limitations which: (1) control toxic pollutants and nonconventional pollutants through the use of "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 CWA.

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 (C.F.R.) at 40 C.F.R. Part 430. The Cluster Rule established revised subcategories for the pulp and paper industry. As a result of 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 C.F.R. 129. Within these guidelines, Subpart A (Barking), Subpart K (Sawmills and Planing Mills), and Subpart L (Finishing) apply to the discharge.

Section 304(h) of the 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, and 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 C.F.R. Part 136). All methods specified in the permit are published in 40 C.F.R. Part 136. All of these 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 C.F.R.

The receiving water body's ability to dilute and assimilate effluent is also factored into the development of effluent limitations (40 C.F.R. 122.44(d)(1)(ii)). Available dilution increases with distance downstream of the discharge point, and loads requiring assimilation decrease through time and distance downstream as receiving waters work to oxidize or degrade the load. The availability of an area for dilution and assimilation into receiving waters is termed a mixing zone. Under the Idaho water quality standards policy, mixing zones may be authorized for discharges to meet water quality standards. Mixing zones are areas or volumes of receiving water where wastewater mixes and assimilates into receiving water and where water quality standards may be exceeded following the state policy. Additional discussion of the mixing zones is provided in EPA (2003, at section VII.A and Appendix D). 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; and
- 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 C.F.R. section122.45(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 C.F.R. 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.

1.2.4.1 Current and Proposed Permit Differences

A comparison of the current and proposed effluent concentration limits for Outfall 001 are provided in EPA (2003, Table III-3). The metals limits in the existing permit were removed for the proposed final permit because EPA determined that there was not reasonable potential for these parameters to violate water quality standards (EPA 2003, Appendix B). The proposed effluent limits for adsorbale organic halides (AOX), 2,3,7,8- tetrachlorodibenzo-p-dioxin (TCDD), chloroform, and summer BOD and temperature are more stringent than the effluent limits in the current permit, and the low-end pH range is higher (i.e., more stringent) in the proposed permit than in current permit. However, loads or limitations of TSS, COD, effluent dissolved oxygen (DO), either were not provided or their limits were not reduced. The proposed final permit also includes narrative criteria that would prohibit the discharge of any floating solids, visible foam in other than trace amounts, or oily wastes that produce a sheen on the surface of the receiving water, and does not authorize the mill to use chemical agents containing trichlorophenol, pentachlorophenol, or zinc.

The proposed permit incorporates more stringent controls of chlorinated organic compounds than the current permit by establishing internal limitations on the fiber lines (i.e., the bleaching lines that are the source of chlorinated organic pollutants) to ensure that the mill's operations are meeting current technology standards for control of chlorinated organics. There are two bleaching lines at this facility: the chip line and the sawdust line. These internal limitations are provided in the BE (EPA 2003, Table III-4).

The permit includes compliance schedules for BOD_5 and temperature. The compliance schedule requires the final effluent limitation for temperature (July through September) is achieved within two years of the permit's effective date and BOD_5 (June through November) final effluent limitation is achieved within five years of the permit's effective date. Until the effluent limits are achieved, the permittee must comply with interim limits that are based upon their previous permit limit. The interim limits are provided in the BE (EPA 2003, Table III-5). A compliance schedule was not warranted for other parameters, since the proposed effluent limits are either promulgated in the ELGs or currently achievable by the Potlatch Mill.

1.2.4.2 EPA-Proposed Conservation Measures

In letter dated March 2, 2004, EPA amended the proposed action to include additional conservation measures (CMs) as part of its proposed action. The letter is included in this opinion as Appendix 4. The CMs are designed to mitigate potential adverse effects to ESA-listed species identified by the BE and to reduce risks associated with critical uncertainties. The temperature reduction and TSS reduction CMs are described in more detail in a December 19, 2003, letter from Potlatch Corporation which is included in this opinion as Appendix 3. EPA's BE amendments of March 2, 2004, also included a detailed monitoring and assessment plan, which is included in this opinion as Appendix 5.

The selected conservation measures are summarized as follows (the numbering of the conservation measures is from EPA's letter)

a. Conservation Measure 9: Potlatch Temperature Reduction Measures

Potlatch and EPA will ensure that effluent temperature meets EPA's final permit limits according to the proposed compliance schedule.

Potlatch will take the following steps to reduce temperature and increase DO: (1) install and begin operation of three heat recovery projects by April 1, 2004, that is expected to reduce effluent temperature; (2) install a pump and diffuser system in the outfall delivery pipe to dilute effluent as much as feasible with ambient yet variably cooler and more oxygenated Clearwater River water (at rates estimated to average of 6-7 mgd that would usually be less than 10 mgd for a variable dilution of 3-33% [average of 19%]); and (3) to initiate or increase operations and maintenance activities in the ABS pond to maximize its ability to shed heat with surface-foam blankets and peripheral sprinklers and improved surface aerators to maximize evaporation rates.

b. Conservation Measure 10: Potlatch Total Suspended Solids (TSS) Reduction Measures

Potlatch and EPA will ensure that the annual load of effluent TSS loads is reduced over 2002 levels by 25%. This reduction will be determined by comparing a 12-month rolling average of the Mill's TSS discharge to 2002 discharge levels. Potlatch will take the following steps to reduce effluent TSS: (1) maximize dewatering capture and mechanical removal of TSS in the primary clarifier; (2) enhance polymer flocculation and capture of TSS before entering the ASB pond; and (3) adjust nutrients in the ASB pond to maximize biological treatment and reduce TSS in the treated effluent. Because the efficacy of these efforts are not easily quantified, Potlatch will continue working on TSS reduction through the life of the permit to obtain reductions of at least 25%. The EPA and Potlatch believe that reductions in TSS would provide valuable reductions in attached loads of BOD, COD and color, chlorinated compounds, dioxins and furans, AOX and whole effluent toxicity (WET) to the receiving water.

c. Conservation Measure 12: Monitoring and Assessment Plan

The EPA provided a monitoring plan that will better define the exposure and effects of effluents from the Potlatch Pulp and Paper Mill to ESA listed salmonids and their habitats (Appendix 5). EPA's monitoring and assessment plan provided a listing of the types of studies EPA believed were necessary to further characterize the effects that Potlatch's NPDES discharge permit will likely have on endangered and listed species and the environment by conducting studies that address the uncertainties described in EPA's BA.

The monitoring and assessment plan included 4 elements that EPA considered necessary to achieve the goal of protection of human health and the environment. These were:

1. Characterization of the chemical and physical components of the action area: Potlatch effluent from its discharge point near the confluence of the Snake and Clearwater Rivers to the downstream end of Lower Granite Reservoir,

2. A determination if contaminants from the Potlatch effluent are bioaccumulated by endangered fish species, or fish species consumed by humans,

3. Development or modification of contaminant benchmarks for the protection of salmonids, their prey, habitat, and wildlife, and

4. Characterization of the potential effects to both ESA listed and other biota in the action area.

Each of these four elements is related to characterizing the action area, reducing uncertainties in the BO, and determining through *in situ* studies if ESA listed species, their prey and habitat are or will be harmed.

The monitoring and assessment plan was not intended to be a detailed work plan that described all sampling locations, numbers of samples, etc. It is the responsibility of Potlatch to prepare a work plan that provides the technical details of implementing the tasks described in this monitoring plan, and a schedule for performing the work. This more detailed work plan is to be submitted to EPA within 180 days after reissuance of the NPDES permit.

2. ENDANGERED SPECIES ACT

2.1 Critical Habitat and Biological Information

A detailed description of habitat use by ESA species within the action area follows the general descriptions of species life histories.

2.1.1 Essential Features of Critical Habitat

Effects of the proposed action are possible to essential features of Snake River salmon habitat. Essential habitat consists of four components: (1) spawning and juvenile rearing areas, (2) juvenile migration corridors, (3) areas for growth and development to adulthood, and (4) adult migration corridors.

The essential features of the rearing areas of the designated critical habitat for Snake River fall chinook salmon are: (1) water quality, (2) water quantity, (3) water temperature, (4) instream cover/shelter, (5) food for juvenile salmon, (6) riparian vegetation, and (7) living space. Fall chinook salmon migration and rearing habitat is found within the action area.

Essential features of the juvenile migration corridors for Snake River sockeye salmon, Snake River fall chinook salmon, Snake River spring/summer chinook salmon, and Snake River steelhead consist of adequate: (1) substrate, (2) water quality, (3) water quantity, (4) water temperature, (5) water velocity, (6) cover/ shelter, (7) food, (8) riparian vegetation, (9) space, and (10) safe passage conditions. Juveniles of all species migrate through and rear to varying extent in the action area.

Essential features of the adult migration corridors for Snake River sockeye salmon, Snake River fall chinook, and Snake River spring/summer chinook salmon include adequate: (1) substrate, (2) water quality, (3) water quantity, (4) water temperature, (5) water velocity, (6) cover/ shelter, (7) riparian vegetation, (8) space, and (9) safe passage conditions. Adults of all species migrate through and may hold temporarily in the action area.

Possible effects to any of those features within the action area must be identified and addressed for the proposed action. All actions or groups of actions associated with this consultation need to be considered both individually and collectively to determine if they may destroy or adversely modify any of the above features of critical habitat.

2.1.2 Action Area

The action area is defined as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action" (50 C.F.R. 402.02).

Direct and indirect effects of the proposed discharge could occur beginning at the mill site where effluent enters surface and ground waters and ending where significant amounts of contaminated water or solids are transported downstream. Thus, the action area is comprised of the surface waters and sediments in the lower Clearwater and Snake Rivers. The Snake River action area is bounded on the upper end by Outfall 001 (SRM139 at the confluence with the Clearwater River). The farthest downstream effect identified in the Snake River is a modeled decrease in DO near its confluence with the Columbia River (SRM 0). Therefore, the action area extends from SRM 139 to SRM 0. The Clearwater River action area is bounded on the upper end by the top of the mill site, because of seepage from the ASB, from Clearwater RM 3 to the confluence with the Snake River (RM 0). Potential downstream effects of specific and combined constituents within the effluent vary with diminishing concentration and interaction by distance.

The lower Snake River, which is a tributary to the Columbia River, has four locks/dams in the State of Washington: Ice Harbor, Lower Monumental, Little Goose, and Lower Granite dams. All are run-of-he 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 (PNNL 2002).

The LGD creates the pool called LGR that is the dominant habitat feature of the action area. The dam is on the Snake River at RM 107 near Almota, Washington. The LGR extends 39.3 miles upstream in the Snake and a further 4.6 miles into the lower Clearwater River. The LGR is considered to end near Asotin in the Snake River arm just upstream from Outfall 001 and adjacent to the Potlatch facility in the Clearwater River arm. The uppermost portion of LGR is more riverine in nature, while the lower portions more resemble a reservoir. Typical water particle retention time of LGR is 7 to 10 days.

Upstream of LGR, both the North Fork of the Clearwater and Snake Rivers are regulated by dams. The Dworshak Dam (1957) is 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.

Almost all salmon and steelhead spawning within the Snake River Basin occurs upstream of LGR. Rearing habitats for Snake River salmonids include upstream tributaries, and the main stem Snake and Clearwater Rivers, including LGR. All Snake River anadromous salmonids must navigate LGR twice (as juveniles leaving and adults returning). Even those juvenile migrants that are loaded onto barges or trucks are not collected until LGD. Therefore, all migrating salmon must swim past the confluence and through LGR as juveniles and adults.

Upstream migrating adult salmonids use a fish ladder to pass over the LGD. 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.

Table 2. 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 Appendix B of COE 2002)

The reservoir area exhibits a typical longitudinal impoundment gradient composed of three reach types. The uppermost portion of the LGR 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 LGRdue to greater water velocity, cooler water inflow from the Clearwater, and relatively shallow water depths. A mid-reservoir reach represents the largest section of the 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 in Appendix B of COE 2002). Table 2 summarizes physical characteristics of LGR.

Approximately 10% of the LGR is shallow water habitat (Bennett et al. 1993). The vast majority of this habitat, however, exists within the upper reaches of LGR upstream of RM 130 (Cook et al. 2003; Figure 2). These areas are often associated with higher velocities, gentle gradients and sandy substrates. Shallow areas are located along shorelines and some mid-channel shelf areas. These habitats primarily occur within the lower Clearwater River channel, in the area of the confluence, downstream of the confluence along both shorelines, and further downstream (to about RM 130) along inside bends, sloping points, and in-channel islands (Figure 2). Backwater areas, with very low water velocity, slightly warmer water, and fine substrate, are limited within LGR. These areas are favored by resident warm water species (e.g., centrarchids) for spawning and rearing. Aquatic macrophyte production in the LGR is minor due to lack of shallow areas and backwaters.

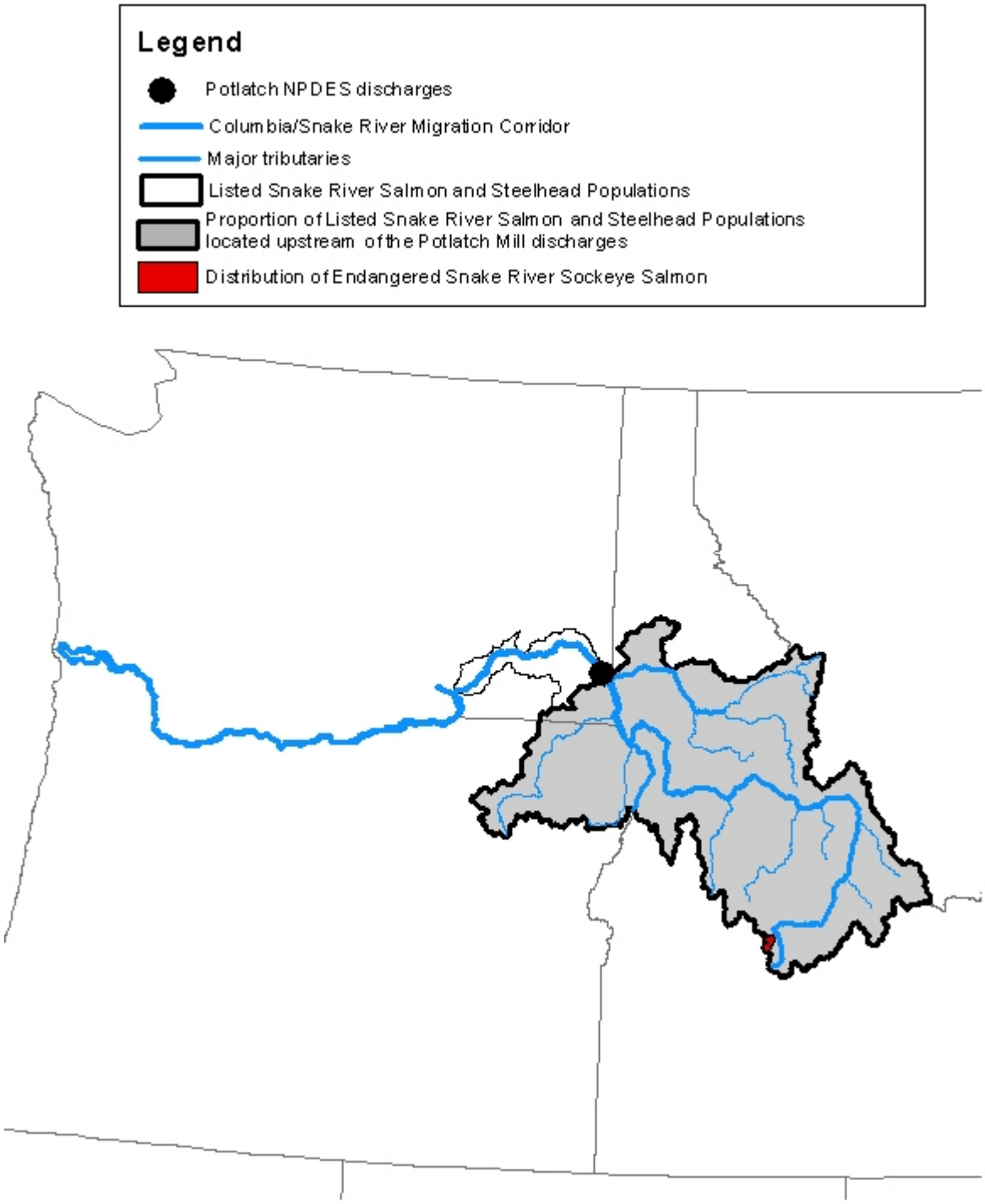

Figure 1. Nearly all of listed Snake River salmon and steelhead trout populations must pass the Potlatch Mill discharges located at the confluence of the Clearwater and Snake rivers.

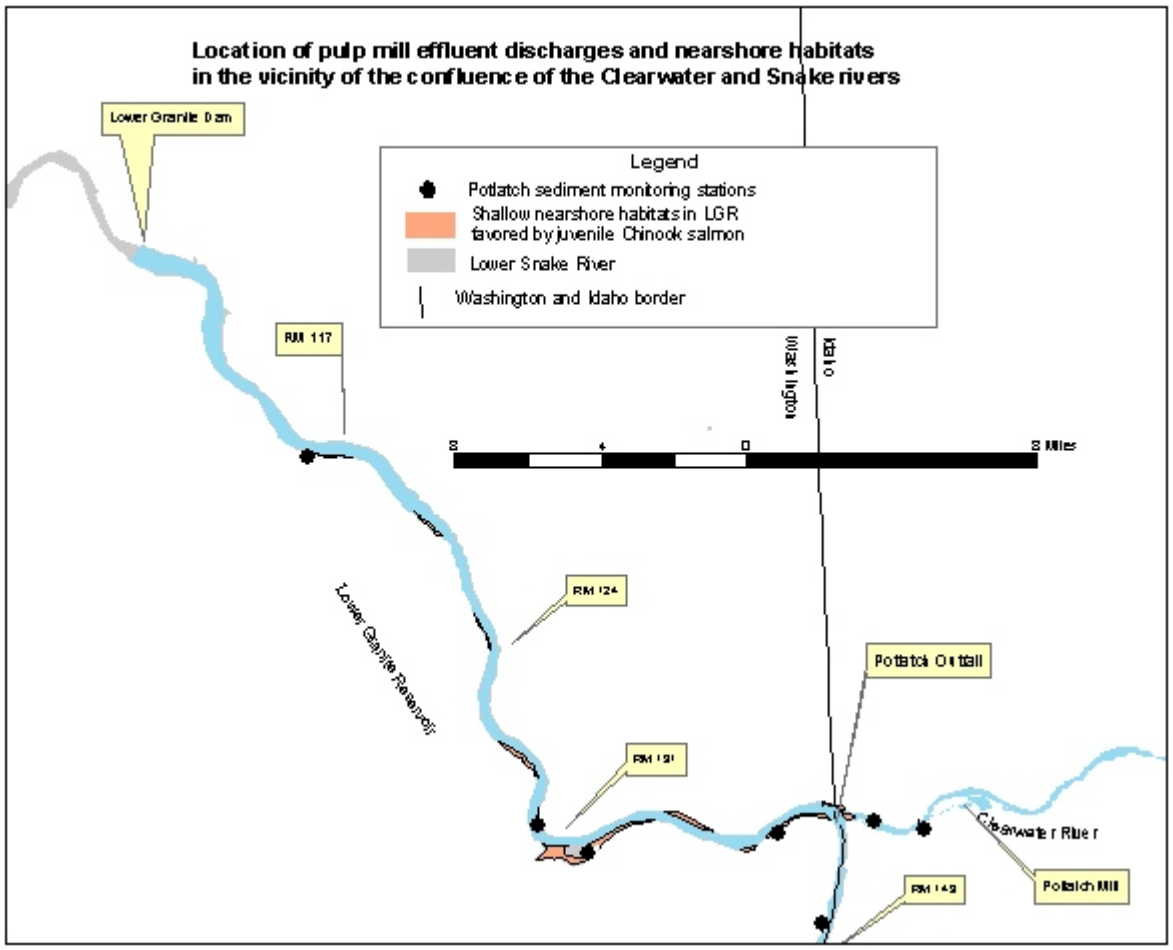

Figure 2. Locations of shallow, nearshore areas in LGR (drawn from PNNL 2002).

2.1.2.1 Physical Description of Receiving Water

a. River Morphology and Discharge

The lower Snake River below the confluence with the Clearwater River has a mean annual discharge of 49,800 cfs with mean peak discharge of 169,257 cfs (1959-1998) and average seasonal monthly low flow of around 20 kcfs in September. The Clearwater River is the largest tributary 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.

The construction and operation of Dworshak Reservoir on the lower North Fork Clearwater River has altered the baseline thermograph of the lower Snake River since about 1973. Before 1984 cooler flows from Dworshak Dam were limited with discharge less than 4 kcfs; however, since 1985 summer releases during average to wet flow years range between 6 to 20 kcfs. The highest summer flow releases were made since 1993, and have actively been used to mitigate warming of LGR and the lower Snake River. The frequency and duration of releases depend upon operational plans and storage volume of Dworshak Reservoir. During dry years, or at other times when limited storage is available, summer flows are not as well augmented with Dworshak Dam releases. Secondly, Dworshak storage and operations are not dependably sufficient to sustain flow augmentation releases throughout the entire spring, summer, and fall of any given year when temperatures may become harmful to salmon. For example, during 2001, Dworshak Dam releases of about 10 kcfs occurred only during July and August; whereas, June and September releases were much less (about 2 kcfs).

Because Outfall 001 is actually within the Snake River channel upstream of its confluence with the Clearwater River and because the two river plumes do not mix immediately, it is important to understand and describe conditions encountered by salmonids only within the Snake River plume. Inflows from the Salmon, and Grande Ronde Rivers are the primary contributors of flow to the Snake River above the confluence with the Clearwater River. The discharge of these tributaries are not regulated by dams. The Salmon River contributes an average 30% to 55% of the May to August flows of the lower Snake River within LGR upstream of the confluence with the Clearwater River.

The hydrograph for the lower Snake River is heavily regulated and base flows during summer and fall are consistently low among runoff years. During and following dry water years spring and winter flows can also remain at low levels for extended periods as upstream storage reservoirs are re-filled. (Figure 3)

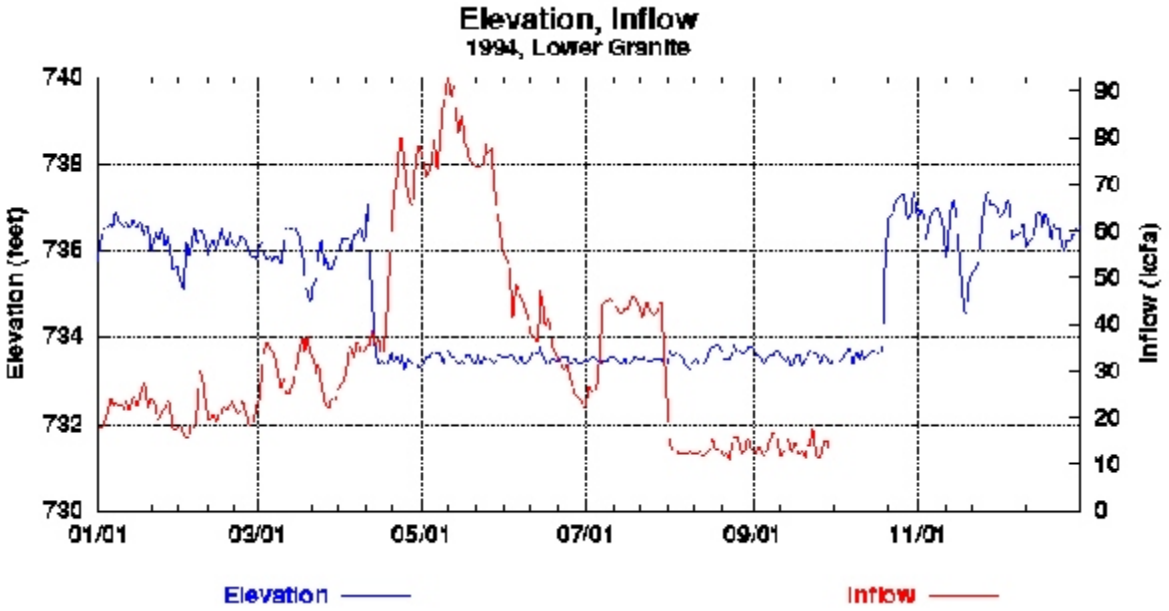

Figure 3. LGR inflow and pool elevation. Due to dam operations, the pool elevation and flow do not rise and fall together .

The 1Q10 and 7Q10 flows were used in the BE for critical analysis of reasonable worst-case acute and chronic effects. Comparing monthly and seasonal mean flows with the 1Q10 and 7Q10, shows that generally low flow conditions of the Snake River may endure for 4-9 months. Appendix 2 includes detailed discharge statistics for the Snake and Clearwater Rivers above and below the confluence. For example, the July 1Q10 and 7Q10 flows of 11,114 and 12,461 are only 26-17% less than the flow of 15,000 cfs that was maintained on average for 4-9 months during the low flows of 1992-1993, 1994-1995, and 2001-2002. Thus, when 1Q10 and 7Q10 flows occur they are usually inside 4-9 month periods of monthly flows averaging in the 14-15 kcfs range, which would be statistically predicted to return (or be repeated) every third year, representing about the 70% exceedence flow. From this exercise it can be seen that chronic exposure flows persisting for 7 days and returning every 10 years on average (12,461 cfs as the 7Q10) are not that much worse or different from the 30 day to 120 day or longer exposures to 14-15 kcfs flows that return every 5-10 years on average (the 30Q10 or 30Q5).

Thus, fish exposures to 1Q10 flows are entirely reasonable for estimating acute effects. Chronic exposure of early and late arriving juveniles and returning adults exposed to 7Q10 flows could result in sublethal or bioaccumulative concentrations and loads. The relative size of the mixing zone plume (by volume, area, and habitat footprint) to river cross-section is greater during periods with the lowest discharges, slowest velocities, and warmest temperatures. It is important to analyze effects during these critical periods because background water quality conditions are key habitat factors supporting salmonid survival. Extended periods of exposure or avoidance by even small proportions (or tails) of populations can rapidly extirpate sub-populations. Ultimately such a situation could reduce minimum population viability and increase the likelihood of extinction to at-risk-species.

b. Snake-Clearwater River Confluence Mixing

The Snake-Clearwater confluence is the lowest and largest in the Snake subbasin, at Snake River-mile (RM) 139 within the pool and near the head of LGR. River confluences are critically important habitat for salmonids. The dynamic mixing of rivers creates and maintains a diversity of habitats necessary for survival during various life stages. Substrate, water quality and quantity, and hydraulic diversity are higher in reaches receiving tributary inflows. Fish can move between differing river conditions and biological communities, and seek out optimal conditions (refugia) as runoff from different watersheds or basins mix.

The dynamics of mixing between the Snake River and the inflowing Clearwater River near the Potlatch 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.

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 can be approximated by examining only the momentum balance between the two rivers. Cook et al. (2003) described four "modes" of mixing dynamics.

There are two main modes of the rivers mixing as identified by Cook et al. (2003). The most common mode (as defined by Cook et al. as mode 1) is an unstratified scenario in which the rivers do not mix but rather flow side by side. This scenario is illustrated in Figure 4 (Figure IV-4 in the BE), from 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 vertical 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. As long as the rivers have comparable discharge flows, the rivers do not intermix, but instead flow side by side.

In the vertically stratified mode (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 more dense than the Snake River water under these circumstances and subducts under the Snake River. This subduction is referred to as vertical temperature stratification.

The occurrence of vertically stratified temperature conditions is shown by field data collected by Potlatch 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).

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 in excess of 10˚C were observed between the epilimnetic and hypolimnetic layers.

Figure 5 (Figure IV-3 in the BE) illustrates an example of vertical thermal stratification when the Clearwater River was approximately 10˚C cooler causing it to abruptly plunge beneath the Snake River. 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).

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 variably intermix.

Figure 4. Illustration of horizontal stratification at the confluence of the Snake and Clearwater Rivers. Infrared satellite image taken at midnight, April 4, 2002, at confluence zone. The red arrow indicates the direction of flow for the Clearwater River, while the blue arrow indicates the direction of flow for the Snake River. (Cook et al. 2003).

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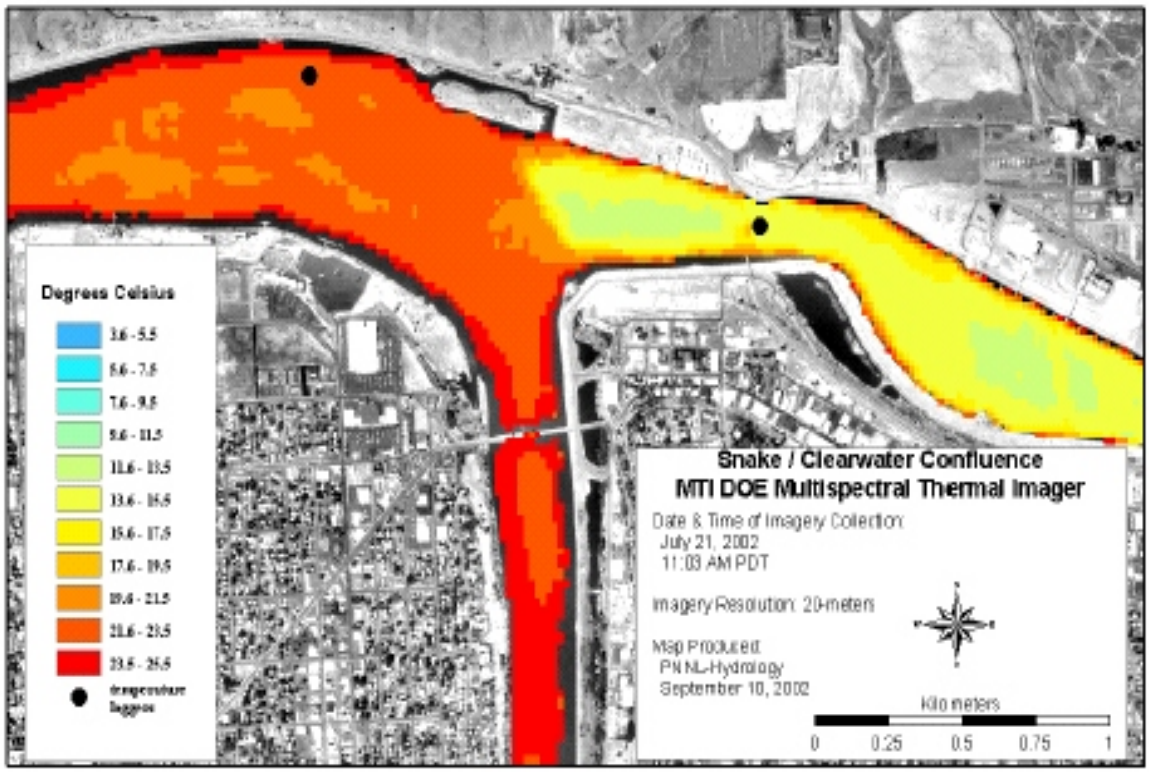


Figure 5. Illustration of vertical stratification at the confluence of the Snake and Clearwater Rivers. MTI satellite derived surface water temperatures on July 21, 2002 at 11 a.m. (Cook et al. 2003).

2.1.2.2 Quality of Receiving Water

Information on water quality parameters of LGR is substantial in general, yet site-specific data is often lacking around the confluence and mixing zone. Most data analyzed in this consultation were summarized by the COE (1999), which were primarily gathered and reported by Bennett et al. (1986, 1993) and Falter (2000, 2001a, 2001b). Additional information on water quality of LGR is reported annually by Potlatch (1997-2002). A summary of the water column data representing the environmental baseline is presented in the BE (EPA 2003, Tables IV-9 and 10). From these tables the water quality data upstream of the diffuser (Outfall 001) on the Snake River and the ASB on the Clearwater River represents the environmental baseline upstream of the action. The water quality data downstream of the confluence represents the environmental baseline within the action area.

The Potlatch data is from sampling and measurements made at the following seven targeted sites within the LGR: Clearwater River Mile 2 (CWRM 2), Snake River Mile 144 (SRM 144), Snake

River Mile 139 (SRM 139), Snake River Mile 137 (SRM 137), Snake River Mile 131 (SRM 131), Snake River Mile 120 (SRM 120), and Snake River Mile 110 (SRM 110). Water column sampling was conducted at each sample site weekly for 14 weeks between mid July and mid October. Potlatch collected both field-measured water quality data and water samples for water quality analyses at each sample site. The number of points sampled along transects of the river channel at each sample site varied depending on the location of the sample site. The CWRM 2, SRM 144, SRM 120, and SRM 110 had one sample point at mid channel, SRM 131 had three sample points, and SRM 137 and SRM 139 had five sample points. In the BE the sample sites are presented in Figure IV-7. All field measurements were taken at 1 foot below the surface and at 10-foot depth intervals thereafter to the bottom at each sample point for every site except at SRM 110 where measurements were only taken to a 90-foot depth. For water quality analysis samples, an equal number of grab water samples were collected and composited from five foot and mid depth of the water column at each sample point.

a. Temperature

Snake River summer water temperatures above LGR typically reach a maximum of 23ºC and normally exceed a base temperature of 20ºC for approximately 60 days each year (COE 1999). Warm inflows delivered from upstream are the primary cause of warm temperatures in LGR. These typically occur during the warmest climatic years with lowest river flows, generally during July and August. High water temperatures (20ºC to 25ºC) have been recorded in reservoirs upstream from Hells Canyon Dam. The Grande Ronde River regularly attains very warm temperatures during summer in its lower main stem reaches. The lower Salmon River is relatively cool with spring-summer heating being a factor during low water years. Clearwater River water temperatures above the LGR typically reach a maximum of 21ºC and normally exceed a base temperature of 20ºC approximately five days each year.

Compared to the Snake River flowing into it, average water temperatures within the reservoir system warm slower by about 1 week and cool slower by about two weeks (COE 1999). It should also be noted that scrollcase temperature readings at dams are taken 15 to 100 feet deep and thus misrepresent surface temperatures during summer by roughly 2ºC to 7ºC (Karr et al. 1998). Flow augmentation with cold water from Dworshak Reservoir on the North Fork Clearwater River is effective in reducing water temperatures in LGR and the lower Snake River, which have benefitted migrating and rearing juvenile salmon and upstream migrating adult salmon. Although Clearwater River average water column temperature decreases with flow augmentation from Dworshak Reservoir, temperature reductions within LGR are limited because the two river plumes do not mix well.

b. Dissolved Oxygen (DO)

The DO concentrations at various locations in the lower Snake River have been reported by the COE (1999), and U. S. Geological Survey (USGS) measurements at gauging stations, Potlatch (1999), and others. Generally, DO levels in the Snake River at the head of LGR may be quite low in the late summer/fall low-flow period, because DO is primarily reduced by high water temperatures. Therefore, DO levels in the period from July through September are the lowest of the year.

During the period 1968 to 1986, DO levels during August at Hells Canyon Dam ranged from 3.7 milligrams/liter [mg/L] in 1986 to 7.8 mg/L in 1982, and follow a significantly decreasing

trend through time (Falter 2001a). The USGS data since 1975 indicate low minimum DO levels of 2.3 mg/L and 4.8 mg/L have been recorded below LGD (RM 106.5) and in the Clearwater River (RM 11.6), respectively (COE 1999). Water quality data from upstream and downstream locations are pertinent to characterize longer term conditions, especially when temperature data in the LGR are limited

In the summer, increased water temperatures and subsequent decreased solubility of oxygen result in lower DO concentrations (COE 1999). Increased water temperatures accelerate the decomposition of organic substances, resulting in higher oxygen consumption. Changes in river flow have the potential to result in changes in DO concentrations, particularly in river systems with fixed continuous discharges of oxygen-demanding substances, such as the Potlatch 001 Outfall. Periods of low flow increase the magnitude of DO sags downstream. Periods of high flow offer more dilution and thus reduce the magnitude of DO sags experienced downstream of outfall discharges.

c. Total Suspended Solids (TSS) and Nutrients

Larger, mostly inorganic, particles transported by the Snake and Clearwater Rivers settle out in the upper parts of LGR. Fine suspended sediment and organic material is transported further downstream into LGR and other downstream reservoirs before settling out on the bottom. Highest concentrations of TSS usually occur during peak flows. Outside high flow periods, background TSS concentrations in LGR usually range between 10-20 mg/L (COE 1999).

Generally, high concentrations of total nitrogen (inorganic and organic) in the middle-Snake River above LGR are diluted by low concentrations in the Clearwater River. So after the two rivers mix the LGR contains moderately high levels. Nutrient concentrations generally decline moving downstream through the lower Snake reservoirs, yet remain higher than those of the Columbia River (COE 1999). Total nitrogen levels increase considerably during fall in the lower Snake River with concentrations reaching 0.8 mg/L to 1.1 mg/L in October. This late season increase may be due to a reduction in plant uptake associated with aquatic plant and algae dying back or going dormant and agricultural harvesting in the watershed. Early fall rains after prolonged dry periods also increase nutrient concentrations, although a corresponding increase in TSS levels was not detected (COE 1999). Nitrate levels also follow this seasonal pattern, with highest levels in spring and fall. Late-season increases in nitrate levels in the lower Snake River may nearly double concentrations in the Columbia River in McNary pool (COE 1999).

Phosphorus is generally expressed in terms of total phosphorus and orthophosphorus (COE 1999). Orthophosphorus represents the inorganic soluble fraction of the total phosphorus in water and is generally considered to be more readily available for biological uptake than is total phophorus. Total phophorus consists of both the soluble fraction and that portion adsorbed to sediments or tied up with biological materials in the water column. Phosphorus readily attaches to and travels with, adsorbed or biological quantities and usually represents the largest portion of total phophorus. In low oxygen conditions, the adsorption bond between phosphorus and the sediment particle becomes unstable and often results in a release of the adsorbed phosphorus into the water column. In contrast to nitrogen, phosphorus is usually the limiting nutrient for plant growth in freshwater systems (COE 1999).

Limnological conditions in the lower Snake River impoundments have generally been considered to be in the upper mesotrophic category (COE 1999; Falter 2001b). Based on a review of 1997 data, the average phosphorus levels throughout the lower Snake River appear to be in the

0.03 mg/L to 0.04 mg/L range during mid-summer and slightly higher to near the 0.06 mg/L to 0.07 mg/L range during June and fall months. This would suggest that the average phosphorus levels during summer in LGR would allow eutrophic conditions (COE 1999).

2.1.2.3 Receiving Water Substrate Description and Sediment Quality

The lower Snake River reservoir system accumulates approximately 3 to 4 million cubic yards of sediment per year, primarily within the LGR and the Palouse and Tucannon Rivers deltas (COE 2002, Appendix C). 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 2000). 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 are bioavailable to benthic organisms and can bioaccumulate in aquatic organisms via the food chain. Sediment quality is discussed further in sections on individual contaminants.

Substrate size in LGR 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 COE 2002, Appendix B). Clay content in LGR generally increased with distance downstream from RM 139 and the organic content is reduced to <5%. A substrate study of five shallow areas in the LGR found a high degree of embeddedness for substrates <150mm diameter (Bennett et al. 1988, as cited in COE 2002). Organic content ranged from 5.2-8.8%. In a study by the COE (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 75 m from the shoreline contained <1% fines.

2.1.2.4 Fish Assemblages within LGR

Potential predators and competitors of anadromous salmonids within 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 COE 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 channel 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 small steelhead parr because of their limited mouth gape. A small proportion of larger-sized smallmouth bass has the potential to consume limited numbers of yearling chinook and sockeye salmon, but the magnitude of this predation is probably low. Also, most yearling chinook migrate earlier in the year (March-May) when water termperatures are lower and the predators are less active. Channel catfish and northern pikeminnow grow to larger sizes and often forage in pelagic and near-surface portions of the channel eating crayfish and small fishes, so predation upon subyearling and yearling salmonids can be significant; however, these

predatory fishes are not as abundant within LGR as in other lower Snake River reservoirs. Crappie and yellow perch are relatively minor predators on juvenile salmonids in the reservoir and their small body size restricts consumption to fry or small juvenile chinook salmon and steelhead.

Juvenile resident species may compete with juvenile salmonids for limited forage and rearing habitats within LGR and the lower Snake River. The magnitude and dynamics of this competition are not well quantified, yet are probably most intense during series of low water years when reservoir temperatures are consistently warmer and flow velocities reduced to enable increased survival and abundance of juvenile resident fishes.

2.1.2.5 Invertebrate Assemblages within LGR

Salmonid forage within LGR is comprised primarily of benthic invertebrates, zooplankton, drifting macroinvertebrates (mostly from upstream habitats), and larval fishes. Four taxonomic groups accounted for 96% of prey items found within subyearling chinook salmon stomachs in LGR and the next pool downstream, Little Goose Reservoir (Curet 1993). Cladocera, primarily Daphnia, were the most numerous food items (46%) ingested, followed by Diptera (primarily Chironomidae and Simulidae), Ephemeroptera (primarily Baetidae), and Homoptera (primarily Aphididae). Larval fish accounted for 21% of prey items by weight and were most important later within the spring-summer seasons, likely due to larger size of subyearlings and possibly reduced availability of other forage. Seasonal trends in relative importance to differing ages (sizes) of subyearling chinook within LGR are described in more detail and discussed.

Curet (1993) also noted that some evidence exists suggesting the benthic community in LGR may not sustain all the salmonid emigrants during spring. Higher velocities within the upper portion of LGR likely increase the production and availability of drift relative to that within the lower reservoir and improve the relative value of upper reservoir habitats for foraging by salmonids. Reduced velocity and volume of discharge may also decrease the amount of drifting food items transported from upstream habitats, which is probably exacerbated by increased temperatures and lower concentrations of DO associated with low flow conditions during late-spring through fall in the lower Snake River.

Muir and Coley (1996) characterized and compared the stomach contents of yearling chinook salmon as they passed LGD with those sampled from fish passing McNary and Bonneville dams on the Columbia River. Muir and Coley (1996) results from the spring migration period differed somewhat from results of Curet (1993). After three years sampling Muir and Coley (1996) found that in one year (1987) yearling chinook smolts in LGR ate mostly amphipods and dipterans, yet in 1989 and 1991 chironomids were the primary prey items. Various terrestrial insects were also eaten, but were less important than chironomids. In contrast to Curet's (1993) findings, no fish, Cladocerans, or mayflies were found in stomach contents of yearling chinook in LGR in the three periods sampled.

One of the most significant findings of Muir and Coley (1996) was that high proportions of yearling chinook at LGR had empty stomachs (26-38%). This is in sharp contrast to the low (3-5%) empty stomachs observed at McNary and Bonneville dams. Muir and Coley (1996) considered the most likely cause of the high numbers of empty stomachs to be limited prey availability in LGR. They noted that LGR contains very fine sediments with high organic content. This sedimentation has led to a benthic community in LGR that is low in diversity and dominated by oligochaetes and chironomids. Reservoirs downstream from LGD in the Columbia

River generally have coarser sediments and greater benthic diversity. Muir and Coley (1996) suggested that starvation in LGR could result in smolts of poor quality that are unable to resist disease or predation, reduced survival could result from affected development, swimming performance, making them more susceptible to disease or by impairing fish immune response.

Other macroinvertebrate surveys have reported depauperate macroinvertebrate assemblages in LGR, with extraordinarily low diversity and in some cases low abundances. Cushing (1993) sampled macroinvertebrates in a few locations in LGR and Little Goose Reservoirs following the 1992 reservoir drawdown testing. In Little Goose, the amphipods *Gammarus* and *Corophium* (important salmonid prey) were very abundant, but were completely absent in the shallow water site in LGR sampled (Silcott Island, RM 131) and were rare in deep water sites. Oligochaetes and chironomids made up >99% of samples. At Wawawai Landing (near LGD) several mud samples were collected just above and below the water's surface and no living organisms were found (Cushing 1993).

Molluscs (primarily snails) may be important salmonid food items if available and other prey are limited. Snails were important food items for rainbow trout in long-term experimental stream studies conducted with Clearwater River water and Potlatch effluents (NCASI 1989). Frest and Johannes (1992) surveyed molluscs in LGR during the 1992 drawdown. They reported that the mollusc fauna in LGR is remarkably depauperate with only the introduced *Corbicula* surviving in any numbers. Even mollusc taxa that are widespread, eurytopic [i.e. those adaptable to a wide range of habitat conditions] and that they had commonly collected in great abundance from reservoir settings throughout the western and central U.S. and were known to historically occur in the lower Snake River were not found during their survey. Their absence in LGR was "very unusual and requires extraordinary explanation." Selected quotations from Frest's and Johannes's (1992) survey report follow:

> "The complete absence of any live freshwater snails from Clarkston to Nisqually John Landing is especially remarkable: no other comparable situation comes to mind."
>
> "The prevalence of mud substrate undoubtably limited some forms.... Still some of the eurytopic forms can thrive elsewhere in superficially similar environments. Their absence suggests that some water quality factor is seriously affecting major parts of the biota, to the point that the normal ecosystem has been distorted beyond recognition."
>
> "If [the relatively free-flowing portion of the Snake River] is being affected by such pollutants as lumber mill wastes, to cite just one obvious example, it is possible that frequent drawdowns would exacerbate pollutant effects on relatively sessile organisms, regardless of how helpful it may be to the more mobile organisms."

Benthic macroinvertebrates were surveyed annually from 1997-2001 in conjunction with ambient sediment chemistry monitoring in relation to Potlatch's discharges. Benthic organisms were collected at the locations sampled for sediment chemistry because they are often key indicators of water and sediment quality. Samples were collected at two reservoir higher-velocity areas, four reservoir lower velocity areas, and a COE levee pond. At each location with sufficient depth, samples were collected from different depth strata (A, 5-20 ft, B, 20-35ft, C, 35-50 ft, and D, >50ft). The sampling targeted soft bottomed depositional areas upstream and downstream of Potlatch's outfall site, with downstream sites at increasing distance from Potlatch's effluent discharge (CH2M Hill 1997, 1998, 1999, 2000, 2001).

Benthic invertebrate data from 2001 from selected stations are shown in Table 3. Patterns from the other years' data appear generally similar, but for simplicity only the most recent data are shown here. Because juvenile salmonids in the Snake River reservoirs preferentially use nearshore waters over moderately sloped bottoms, only data from the 5-20 ft depth strata are shown. Only sites from lower-velocity stations are compared since velocity is an important factor that structures benthic communities. The data are summarized into metrics according to whether the benthic organisms are important salmonid forage organisms based on findings of Muir and Coley (1996), Curet (1993), and Karchesky and Bennett (1999).

Table 3. Benthic community metrics at selected sites from Potlatch 2001 ambient sediment monitoring

Metrics	RM-142 (S1 Reference)	RM-137 (S-5)	RM-131 (slow water at Silcott Island, S3)	RM-116 (S2)
Total taxa richness	4	6	8	7
Salmonid prey taxa				
% Chironomids	16.90%	7.39%	21.34%	15.72%
% Amphipods	0	0	0	0
% Mayflies	0	0	0	0
Density of prey taxa	1033	1937	689	1421
Non-salmonid prey items				
% Oligochaetes	83.1%	91.6%	58.7%	74.8%
% Molluscs	0	0	0.00%	0.95
% Nematodes	0	0	4.00%	4.3
% Other	0.00%	0.98%	10.66%	2.8
Total invertebrate density	6114	26,221	3228	9042
% Inverts that are Salmonid prey organisms	16.90%	7.39%	21.34%	15.72%
% Inverts that are not salmonid prey organisms	83.10%	92.61%	73.33%	82.86%

Results of the Potlatch benthic community monitoring show taxa that are preferred salmonid prey make up a relatively small percentage (1 - 21%) of the benthic community at the sites sampled. A possible decline in the percentage of prey taxa collected was observed between the sites sampled upstream and downstream of Potlatch's effluent (RM 142 and RM 137, Outfall 001 is at RM 139). The percentage of salmonid prey organisms was lower at this site than at the other lower-velocity sites, and overall abundance of invertebrates was higher. This is consistent with changes to benthic communities expected from pulp mill influence (discussed in the analysis of effects of BKME in section 2.3.3.2). However, simple association does not necessarily equate to causation because many factors can influence benthic community structure. This situation warrants improvements in water and sediment quality where feasible and further research addressing cause and effect relationships.

In summary, the benthic invertebrate community is a key component of the salmonid prey base and habitat function in LGR. The community has unusually low diversity and abundance, and juvenile salmonids in LGR have poor feeding success relative to juvenile salmonids in other Snake River or Columbia River reservoirs. There is reasonable potential that pollutants from Outfall 001 contribute to this adverse condition and therefore a careful monitoring study to ascertain that information is warranted.

2.1.3 Status of Listed Species and their Use of the Action Area

2.1.3.1 Snake River Sockeye Salmon

Snake River sockeye salmon primarily occur in Redfish Lake in the upper Salmon River near Stanley, Idaho. Historically sockeye salmon occurred in four more upper Salmon River lakes, Petit, Alturas, Stanley, and Yellowbelly Lakes, and in lakes upstream of Hells Canyon Dam. Sockeye salmon returning to Redfish Lake travel a greater distance from the ocean (1450 kilometer [km]) and spawn at a higher elevation (2000 meter [m]) than any other known population of sockeye salmon. In Idaho, sockeye salmon travel along the Salmon and Snake Rivers during their out migration to the ocean and return migration to their natal lakes.

From 1991 to 1999 a total of 23 wild adult anadromous sockeye salmon returned to Redfish Lake. These wild fish were incorporated into the NOAA Fisheries /Idaho Department of Fish and Game (IDFG) captive broodstock program that begun in 1992. Releases from the NOAA Fisheries and IDFG captive broodstock programs generated seven returning adults in 1999, 257 adults in 2000, 26 adults in 2001, and 23 adults in 2002. Since 1991 the smolt-to-adult return data for Redfish Lake wild and hatchery sockeye averaged .0010. This means that on average one out of every 1000 smolts returns as an adult. The average Redfish Lake adult returns as a proportion of sockeye salmon adults passing over LGD average 54%. Therefore on average, almost half the adult sockeye salmon that pass over LGD never return to the upper Salmon River weirs and are assumed to be mortalities.

The captive broodstock program has focused on Redfish Lake, as the largest of the Stanley Basin sockeye lakes and the last to maintain anadromous runs. However since 1996, various numbers of juvenile sockeye salmon have been released into Alturas and Petit Lakes. NOAA Fisheries has released an interim recovery target for Snake River sockeye salmon of 1,000 anadromous adults returning to Redfish Lake, and 500 anadromous adults returning to one other Stanley Basin lake. Because of the low numbers of returning adults to Redfish Lake a CRI or QAR- type analysis of the status of this ESU was not performed for the Federal Columbia River Power System (FCRPS) Biological Opinion (NMFS 2000). NOAA Fisheries considers the status of this ESU to be dire under any criteria. Clearly the risk of extinction is very high ((McClure et al. 2000, 2003).

a. Specific Use of Habitat in the Action Area

The use and occurance of Snake River sockeye salmon in the lower Snake River and LGR are documented through observations and tag detections at dams. Some observations at dams may include kokanee salmon (a resident or hatchery form of sockeye salmon that are not listed under the ESA).

Downstream migration of yearling sockeye smolts begins in the upper Salmon River in late April and lasts through May (Bjornn et al. 1968). Median travel time of PIT-tagged smolts to the dam of first observation is 14 days. Low flows and warm temperatures in the Salmon and Snake Rivers may reduce migration rates 2-3 times both in river reaches upstream of LGR and within LGR proper. This trait delays run arrival times at LGR and the duration of time spent within LGR. Because they must travel almost 700 km downstream in variable flows to reach LGR, sockeye smolt runs can be considerably protracted and delayed. Hence, the primary outmigration of sockeye smolts passes the Clearwater River confluence and LGR from mid May through late June, with significant run tails sometimes extending into July and August (COE 2002, in EPA 2003; Figure 8).

Sockeye smolts are typically far along into the smolting process by the time they reach LGR and may be very buoyant in preparation for entering saltwater and traveling rapidly downstream (about one week duration in LGR; Table 4). Near-surface, mid-channel portions of the greater channel where velocities are highest are the most likely habitats used within LGR. If temperatures are warm, however, relatively more emigrants are likely to use deeper habitats in an attempt to avoid the warmest surface waters. Snake River sockeye smolts rear in cold, nearly sterile (oligotrophic) lakes, and thus are relatively small in size with comparatively limited energy reserves, such that delay or stress during migration can readily result in increased mortality. Because counts at LGD indicate spring emigrations are commonly protracted into the summer months, it is likely that increased temperatures, reduced velocities from reduced discharge, reduced concentrations of DO, and other parameters of late-spring and summer water quality are key factors of juvenile survival. Survival of sockeye smolts is reduced during low flow and warm temperature conditions, especially during late-spring and summer portions of runs.

Upstream migration of adults along the lower Snake corridor occurs mainly from June through August (Figure 7). Most adults likely negotiate LGR within 1-3 days (Table 4). They tend to swim and possibly hold in mid-depth habitats off of shorelines on either side of the channel. Important habitats are those that provide cooler water or eddies for resting such as gravel bars or the Clearwater River confluence. Most adult migrants likely pass the confluence area fairly quickly with limited resting or staging and continue migrating upstream. Migrations can be slowed or stopped when water temperatures warm and DO levels sag during summer.

Adult sockeye salmon migrations are well documented to be delayed or curtailed during summer months within the lower Snake River when increased temperatures, reduced velocities from reduced discharge, reduced concentrations of DO, and other parameters of degraded water quality are likely key contributors to reduced travel rates (Hodgson and Quinn 2002). Sockeye salmon populations whose migration routes exceed 19ºC average temperature display two distinct patterns of avoidance, migrating either before or after peak temperatures, which presumably incurs costs to reproductive potential. Snake River sockeye salmon spawn in high elevations with cold thermal regimes, so incubation temperatures require relatively early spawning. Delays longer than about 75 days probably do not occur based upon timing needed to successfully spawn and incubate eggs and because energy expenditures during long migrations and holding periods in freshwater are too great (Hodgson and Quinn 2002). Snake River sockeye salmon are migrating progressively earlier in apparent response to recent summer warming of the lower Columbia and Snake Rivers (Quinn et al. 1997).

2.1.3.2 Snake River Spring/Summer Chinook Salmon

The present range of spawning and rearing habitat for naturally spawned Snake River spring/summer chinook salmon is primarily limited to the Salmon, Grande Ronde, Imnaha, and Tucannon Rivers. Most adult Snake River spring/summer chinook salmon enter individual subbasins from May through September. Juvenile Snake River spring/summer chinook salmon emerge from spawning gravels from February through June (Peery and Bjornn 1991). Typically, after rearing in their nursery streams for 1 or 2 years, smolts begin migrating seaward in April and May (Bugert et al. 1990, Cannamela 1992). Snake River spring and summer chinook exhibit a diversity of shared life history traits. Both are stream-type fish, with juveniles that may initiate seaward migration as early as fry or migrate swiftly to sea as yearling smolts. Depending primarily on location within the basin (and not on run type), adults tend to return after either 2 or 3 years in the ocean. Both spawn and rear in small, high-elevation streams (Chapman et al. 1991), although where the two forms coexist, spring-run chinook spawn earlier and at higher elevations than summer-run chinook.

There is a long history of human efforts to enhance production of chinook salmon in the Snake River basin through supplementation and stock transfers. The evidence is mixed as to whether these efforts have altered the genetic makeup of indigenous populations. Adult straying rates appear to be very low.

For the Snake River spring/summer chinook salmon ESU as a whole, NOAA Fisheries estimates that the median population growth rate (lambda) over the base period ranges from 0.96 to 0.80, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to the effectiveness of fish of wild origin. NOAA Fisheries has also estimated median population growth rates and the risk of absolute extinction for the seven spring/summer chinook salmon index stocks in 2000 using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end assuming that hatchery fish spawning in the wild have not reproduced (i.e., hatchery effectiveness = 0), the risk of absolute extinction within 100 years for the wild component ranges from zero for Johnson Creek to 0.78 for the Imnaha River (NMFS 2000). At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years ranges from zero for Johnson Creek to 1.00 for the wild component in the Imnaha River (NMFS 2000).

a. Specific Use of Habitat in the Action Area

(1) Juveniles. Large numbers of subyearling (fry, parr, and pre-smolt life stages) spring/summer chinook move downstream from natal tributaries during summer, fall, and winter to rear and overwinter at lower elevations within the Snake River Basin, including the lower Snake, Clearwater, and Salmon Rivers. These fish may continue seaward migrations the following spring or rear until enough size and growth thresholds are met to enable smoltification. Immediately following emergence through mid-winter of the first year reduced habitat availability primarily caused by low flows cause increased use of larger rivers and LGR for rearing. During late-spring when flows decline and air temperatures warm, water temperatures increase, migration rates slow and survivals drop until temperature extremes drastically delay or curtail migrations.

Subyearling spring/summer chinook salmon rear in and migrate through LGR at least during spring and summer (Connor et al. 2001). The upper 10 miles of LGR that contain the majority of

shallow water habitats within the reservoir. Shallow water habitats are documented to be important sub-yearling feeding and resting habitats (Curet 1993; Connor et al. 2001). Mid-depth habitat is important at and downstream of the diffuser (and provides feeding and holding habitats) for older parr, yearlings, and residualized smolts, as documented for steelhead and chinook in Bennett et al. (1986, 1988, 1993). The Snake River at low flows is only 20-40 feet deep at the head of LGR. Rearing chinook salmon could be attracted to mostly reside in habitat less than 20 feet deep. In relatively clear waters flowing at low-velocity, juvenile salmonids would be expected to better use mid-channel habitats for feeding.

Spring/summer chinook yearling smolts typically travel rapidly downstream through LGR (within 1-2 weeks; Table 4) utilizing near-surface, mid-channel portions of the thalweg where velocities are highest. If temperatures are warm or the water column clear, relatively more emigrants are likely to use deeper water habitats. Various reports by Bennett et al. (1986, 1993) found juvenile chinook (> 75 mm length) utilizing a full range of habitats and depths during spring and early summer in LGR. Residualized juveniles were also present; these fish likely rear for extended periods (months) and would typically use mid-depth and deeper habitats. The upper portion of LGR includes the most valuable habitat for these fish.

Many stocks of Snake River spring/summer chinook smolts must migrate 300-400 miles from natal tributaries to reach LGR. So runs are often protracted into summer season and those rearing in higher elevations may be quite small and arrive later than average. These smolts likely have comparatively limited energy reserves, such that delay or stress during migration can readily result in increased mortality. Because counts at LGD indicate spring emigrations are commonly protracted into the summer months, it is likely that increased temperatures, reduced velocities from reduced discharge, reduced concentrations of DO, and other parameters of degraded water quality are key contributors to reduced travel rates and survival.

The most critical period for many Snake River spring/summer chinook is during late June through late August when significant proportions of wild yearlings and subyearlings arrive in LGR during poor water quality conditions. Seasonally poor water quality can compress stocks into central portions of "normal" runs by altering overall run-timing and reducing the breadth of population response to subsequent life stage timing windows, like peak spring flows and ocean conditions that are themselves variable over time. This breadth of variability is very important to preserving the viability of the populations and preserving the ESU as a whole (NMFS 2000).

For example, from PIT-tag analysis, outmigration of wild subyearling and yearling spring-summer chinook smolts from the Pahsimeroi River continues from mid-May through June. The median travel time required for these smolts to reach LGD is roughly 25 days. Many of these fish are migrating through the lower Snake River during July and into early August. Mean daily water temperatures in LGRand vicinity of the Snake-Clearwater confluence exceed 18-22ºC during this season for many days of many years. Thus, while June migrants may represent 30% or more of the spring production from the Pahsimeroi River, in the lower river these fish are often treated as an insignificant number when compared to large releases of hatchery fish or peaks in wild fish runs across all tributaries.

(2)Adults. Adult spring/summer chinook salmon migrate upstream through LGR from May to August and likely negotiate the reservoir within 1-3 days (Figures 7 and Table 4). They tend to swim and possibly hold in mid-depth habitats off of shorelines on either side of the channel. Important habitats are those that provide cooler water or eddies for resting such as gravel bars or the Clearwater River confluence. Adult chinook salmon likely pass the Snake-Clearwater River

confluence area fairly quickly when heading upstream because it offers limited resting habitat. Adult upstream migrations can be delayed for significant periods of time when high water tempertures and low DO levels occur during summer.

2.1.3.3 Snake River Fall Chinook Salmon

The distribution of fall chinook is more limited than that of the spring/summer chinook ESU, and includes the lower reaches and tributaries of the Snake River, Clearwater River, Salmon River, Tucannon River, and Grande Ronde River (NMFS 1995). Fish from all of these rivers are found at one time or another in the lower Snake River. Only those fall chinook that spawn below lower Snake River dams or in the Tucannon River do not migrate through LGR in Idaho waters.

The upper reaches of the mainstem Snake River were the primary areas used by fall chinook salmon, with only limited spawning activity reported downstream from river kilometer (Rkm) 439. Only limited spawning activity was reported downstream from Rkm 439, about 1 km upstream of Oxbow Dam. Since then, irrigation and hydrosystem projects on the mainstem Snake River have blocked access to or inundated much of this habitat causing the fish to seek out less preferable spawning grounds wherever they are available. Natural fall chinook salmon spawning now occurs primarily in the Snake River below Hells Canyon Dam and the lower reaches of the Clearwater, Grand Ronde, Salmon, and Tucannon Rivers.

Adult Snake River fall chinook salmon enter the Columbia River in July and migrate into the Snake River from August through October. Fall chinook salmon generally spawn from October through November, and fry emerge from March through April. Downstream migration generally begins within several weeks of emergence. Juveniles rear in backwaters and shallow water areas of major rivers and reservoirs through mid-summer before migrating to the ocean. Thus, they typically exhibit an ocean-type juvenile history. Once in the ocean, they spend 1 to 4 years (though usually, 3 years) before beginning their spawning migration. Fall returns in the Snake River system are typically dominated by 4-year-old fish. For detailed information on Snake River fall chinook salmon, NMFS (1991) and June 27, 1991, 56 FR 29542.

Some Snake River chinook historically migrated over 1,500 km from the ocean. Although the Snake River population is now restricted to habitat in the lower river, genes associated with the lengthier migration may still reside in the population. Because longer freshwater migrations in chinook salmon tend to be associated with more extensive oceanic migrations (Healey 1983), maintaining populations occupying habitat that is well inland may be important in continuing diversity in the marine ecosystem as well.

For the Snake River fall chinook salmon ESU as a whole, NOAA Fisheries estimates that the median population growth rate (lambda) over the base period ranges from 0.94 to 0.86, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin. NOAA Fisheries has also estimated the risk of absolute extinction for the aggregate Snake River chinook salmon population, using the same range of assumptions about the relative effectiveness of hatchery fish. At low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e. hatchery effectiveness), the risk of absolute extinction within 100 years is 0.40. At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish (hatchery effectiveness = 100%), the risk of absolute extinction within 100 years is 1.00 (McClure et al. 2000, 2003).

a. Specific Use of Habitat in the Action Area

(1) Juveniles. Subyearling fall chinook salmon rear, smoltify, and migrate through LGR from April through August (Figure 7); Connor et al. 2002, 2003a; Smith et al. 2003). Warm temperatures and other factors can cause fall chinook juveniles to residualize in LGR. These fish typically rear until the following spring before continuing their migration to the ocean as yearlings (Connor et al., in review). Subyearling fall chinook salmon primarily rear and feed in the upper 10 miles of LGR that contain the majority of shallow water habitats within the reservoir. Shallow water habitats are important sub-yearling feeding and resting habitats (Curet 1993; Connor et al. 2001). Mid-depth habitat is also important to older subyearlings for as long as a month before migrating through LGD (Curet 1993). Residualized chinook salmon smolts were also found in mid-depth and deep water habitat in Bennett et al. (1986, 1988, 1991, 1993). The Snake River at low flows is only 20-40 feet deep at the head of LGR, so rearing fish could be attracted to reside in habitat throughout the area and the confluence itself. In relatively clear waters flowing at low-velocity, juvenile fall chinook salmon would be expected to best use mid-channel and near substrate habitats for resting and feeding.

Peak out-migration of juveniles at LGD typically occurs from May through July. Dam counts and research on PIT-tagged fall chinook indicates significant numbers of subyearling migrants continue moving through LGR during July and August (Smith et al. 2003). Spikes in detections of tagged fall chinook juveniles occur before and after August at LGD and other lower Snake River dams, suggesting mid-summer river conditions curtailed migrations. These are likely caused by the formation of elevated water temperatures and patches of low DO that essentially cause a stress barrier to the continued growth, development, movement and downriver passage of the juvenile fish. Some types of habitat may become more limiting in this situation, such as surface layers and shoreline areas or safe passage ways through dams. Fall chinook salmon likely comprise major proportions (50-70%) of subyearling chinook salmon in the lower Snake River (Connor et al. 2001). These fish typically rear about 40 days within LGR before migrating through LGD (Table 4).

From late-May through early September is a critical period for Snake River fall chinook subyearlings entering LGR from upstream tributaries as water quality declines throughout the summer season. Warm temperatures and low flow conditions can increase residence time by 20-40% or more and cause residualization that may extend the duration of stay until the following spring.

(2) Adults. Adult fall chinook salmon migrate upstream through LGR from May to November and likely negotiate the reservoir within 1-3 days (Figures 6 and Table 4). They tend to swim and possibly hold in mid-depth habitats off of shorelines on either side of the main channel. Important habitats are those that provide cooler water such as the Clearwater River confluence. Adult fall chinook salmon likely pass the Snake-Clearwater River confluence area fairly quickly when heading upstream because it offers little resting habitat.

2.1.3.4 Snake River Steelhead

NOAA Fisheries has identified 15 ESUs for West Coast steelhead from Washington, California, Oregon, and Idaho. Of these ESUs, the Snake River steelhead were designated as threatened. The primary rivers within the Snake River Basin supporting Snake River steelhead include the Clearwater River, Salmon River, Tuccannon River, Imnaha River, Grande Ronde River, and

Asotin River. Snake River steelhead enters freshwater in a sexually immature condition and requires several months in freshwater to mature and spawn. There are two run types for steelhead in Idaho, "A-run" and "B-run." Since 1963, steelhead crossing Bonneville Dam from April 1 through August 25 have been classified by IDFG as A-run, and those crossing after August 25 as B-run. Run-timing separation is not observed at LGD, where the two runs are distinguished instead by ocean age and body size: A-run steelhead are considered to be mostly age-1-ocean, while B-run steelhead are mostly age-2-ocean. The later entry into freshwater by B-run steelhead results in a larger average size than A-run fish. The Salmon River drainage contains primarily A-run steelhead, except for the South Fork 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).

Steelhead spawn in cool, clear streams with suitable gravel size, depth, and current velocity. Intermittent streams may also be used for spawning (Everest 1973). Steelhead enter streams and arrive at spawning grounds weeks or even months before they spawn and are vulnerable to disturbance and predation. Cover, in the form of overhanging vegetation, undercut banks, submerged vegetation, submerged objects such as logs and rocks, floating debris, deep water, turbulence, and turbidity (Giger 1973), is required to reduce disturbance and predation of spawning steelhead. Summer steelhead usually spawn further upstream than winter steelhead (Withler 1966, Behnke 1992).

Depending on water temperature, steelhead eggs may incubate for 1.5 to 4 months (August 9, 1996, 61 FR 41542) before hatching. Summer rearing takes place primarily in the faster parts of pools, although young of the year are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Productive steelhead 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 mainstem rivers (Nickelson et al. 1992).

Juveniles rear in freshwater from 1 to 4 years, then migrate to the ocean as smolts. Winter steelhead populations generally smolt after 2 years in freshwater (Busby et al. 1996). Steelhead typically reside in marine waters for 2 or 3 years before returning to their natal stream to spawn at 4 or 5 years of age. Populations in Oregon and California have higher frequencies of age-1-ocean steelhead than populations to the north, but age-2-ocean steelhead generally remain dominant (Busby et al. 1996). Age structure appears to be similar to other west coast steelhead, dominated by 4 year old spawners (Busby et al. 1996).

Hatchery fish are widespread and stray to spawn naturally throughout the region. In the 1990s, an average of 86% of adult steelhead passing LGD were of hatchery origin. Hatchery fish dominate some stocks, but do not contribute to others.

For the Snake River steelhead ESU as a whole, NOAA Fisheries estimates that the median population growth rate lambda over the base period ranges from 0.91 to 0.70, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared to that of fish of wild origin. NOAA Fisheries has estimated the risk of absolute extinction for the A- and B-runs, using the same range of assumptions about the relative effectiveness of hatchery fish. At the low end, assuming that hatchery fish spawning in the wild have not reproduced (i.e. hatchery effectiveness = 0), the risk of absolute extinction within 100 years is .01 for A-run steelhead and

0.93 for B-run steelhead. At the high end, assuming that the hatchery fish spawning in the wild have been as productive as wild-origin fish, the risk of absolute extinction within 100 years is 1.00 for both runs (McClure et al. 2000, 2003).

a. Specific Use of Habitat in the Action Area

(1) Juveniles. Fry emergence after spawning occurs from June through September. Juvenile steelhead rear in their natal streams and larger rivers throughout the Snake River Basin, for one to three years, but normally two, before migrating to the ocean as smolts. Large numbers of subyearling (mostly parr) life stages of steelhead move downstream from natal tributaries during fall and winter to rear and overwinter at lower elevations, including the LGR, lower Snake, Clearwater, and Salmon Rivers. These fish may continue seaward migrations the following spring or rear until enough size and growth thresholds are met to enable smoltification. Low flows may cause increased use of low elevation rivers and LGR for rearing.

Although use is not well documented, subyearling steelhead likely rear and feed in the upper 10 miles of LGR. The duration of rearing time in LGR likely varies from a few weeks to months. Most stay over summer and/or winter until spring migration occurs during elevated flows (Figures 6 and 7). Mid-depth habitat is important for larger juveniles at and downstream of Outfall 001 (Bennett et al. 1986, 1988, 1993). The Snake River at low flows is only 20-40 feet deep at the head of LGR. So larger juvenile steelhead could reside in habitat throughout the area.

Steelhead smolts migrate through LGR during spring and early summer (Figure 7). Peak outmigration usually occurs just before and during each year's peak runoff. Smolts average 175 mm in length and have an average stay in LGR of five days. Low flows increase residence time in LGR by 2-4 times (Figure 6). Steelhead smolts are typically more sensitive to warm temperatures than are chinook salmon. During warm springs or early-summers with low flows steelhead smolts have a relatively greater risk of suffering delay, residualism, and mortality due to increased river temperatures and associated water quality conditions. A critical period for many Snake River steelhead is during late May through early August when significant proportions of wild yearling steelhead smolts continue to enter LGR from upstream headwater tributaries (some as far as 300-400 miles distant). During summer the temperature of the lower Snake River and LGR increases significantly.

(2) Adults. Adult steelhead typically arrive in LGR during the summer and fall and either continue migrating upstream (within 1-3 days) to overwinter before spawning the next spring or overwinter within LGR in the main stem Snake and Clearwater Rivers (Bjornn et al.2000, 2003; Figure 7). Substantial proportions of migrants enter LGR from July to September migrations can be slowed or stopped and holding time in stressful conditions increased when water temperatures warm and DO levels sag during summer. Large numbers of adult steelhead use the Clearwater River confluence area in LGR to overwinter (3-5 months; Table 4). Steelhead are usually found holding in the lower Clearwater River from the confluence upstream a few miles. During late winter these adult steelhead resume migration to upstream tributaries to spawn in April and May. Adult steelhead tend to swim and possibly hold in mid-depth habitats off of shorelines on either side of the channel. Important habitats are those that provide cooler water or eddies for resting such as gravel bars, depressions, channel bends, and the Clearwater River confluence.

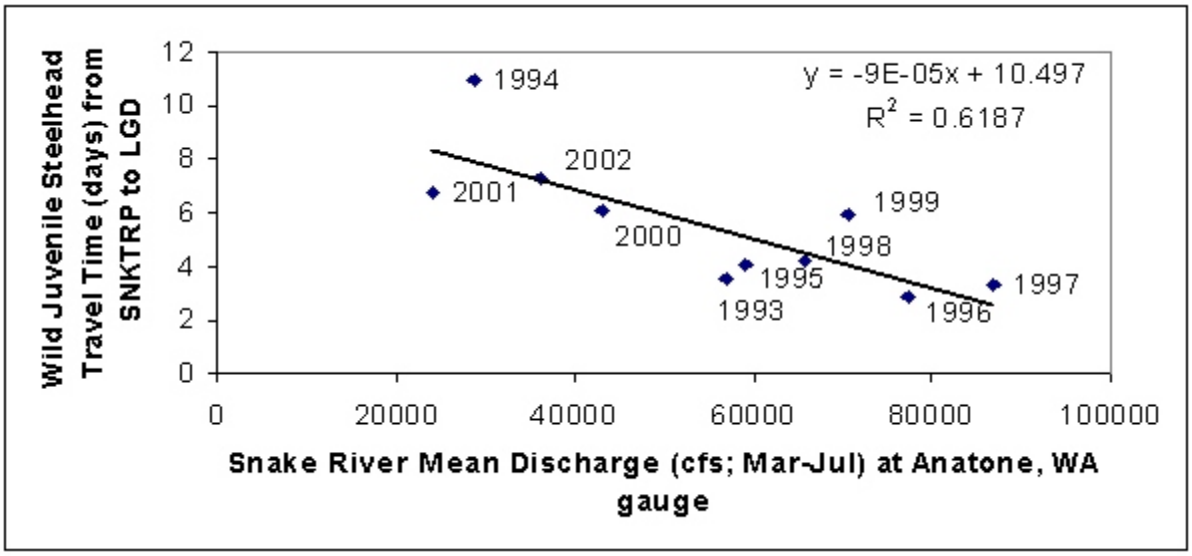


Figure 6. Time spent in LGR by steelhead smolts in relation to discharge.

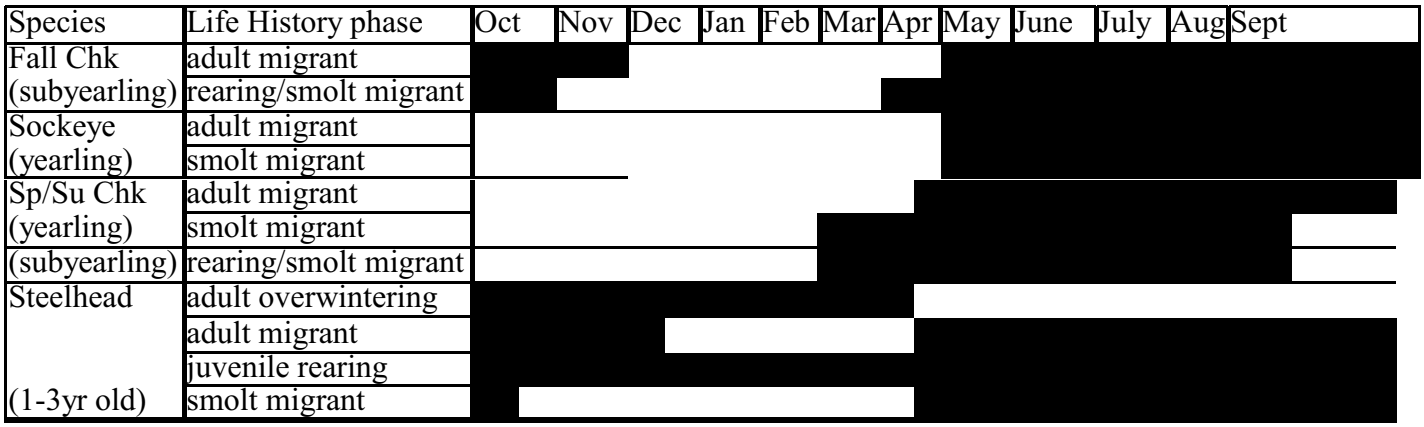


Figure 7. Presence and periodicity in LGR by life stage of ESA-listed salmonids.

Table 4. Summary of time spent in LGR by life stage for ESA-listed salmonids.

Species	Life History Phase	Estimated Duration in Lower Granite Reservoir (days)	
		Median	90th
Fall Chinook	adult migrant	3	7
(subyearling)	rearing/smolt migrant	43	60
Sockeye	adult migrant	3	7
(yearling)	smolt migrant	8-10	15-30
Sp/Su Chinook	adult migrant	3	7
(yearlings)	smolt migration	8-17	15-35
(subyearling)	rearing/smolt migrant	43	60
Steelhead	adult overwintering	100	180
	adult migrant	3	7
(0-3 yr old)	rearing/residualized	150	200
(1-3yr old)	smolt migrant	4-6	6-9

2.2 Evaluating Proposed Actions

The standards for determining jeopardy are set forth in section 7(a)(2) of the ESA as defined by 50 C.F.R. Part 402 (the Federal consultation regulations). NOAA Fisheries must determine whether the proposed action is likely to jeopardize the listed species or likely to destroy or adversely modify critical habitat. This analysis involves initial steps of: (1) defining the biological requirements and current status of the listed species; and (2) evaluating the relevance of the environmental baseline to the species' current status.

Subsequently, NOAA Fisheries evaluates whether the action is likely to jeopardize the listed species by determining whether the action, taken together with any cumulative effects and added to the environmental baseline, can be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of the affected species or to destroy or adversely modify their designated critical habitat. If so, NOAA Fisheries must identify any reasonable and prudent alternatives (RPAs) to the proposed action that would avoid jeopardy or adverse modification of critical habitat and meet the other regulatory requirements for RPAs. (See 50 C.F.R. § 402.02)

For the proposed action the jeopardy analysis assumes that aquatic organisms, including listed salmon and steelhead, would be exposed to chemical concentrations equal to the permit limits (including the Conservation Measures). Given this assumption, NOAA Fisheries analyzed whether the proposed action would be expected to reduce appreciably the likelihood of the listed species' survival and recovery, considering existing conditions and the following effects of the proposed action: (1) expected mortality of fish attributable to direct exposure to effluent in the water or sediments; (2) expected mortality of fish attributable to indirect exposure such as, through ingestion of prey or other materials; (3) harm that might result from sublethal effects from exposure, such as behavior changes or changes to the prey base; and (4) any harm or mortality expected from cumulative effects. NOAA Fisheries' critical habitat analysis considers the extent to which the proposed action impairs the function of essential elements necessary for juvenile and adult migration, spawning, and rearing of listed salmon and steelhead under the existing environmental baseline.

2.2.1 Biological Requirements

The first step in the method NOAA Fisheries uses for applying the ESA standards of § 7 (a)(2) to listed salmon and steelhead is to define the species' biological requirements that are most relevant to the particular consultation. The biological requirements are defined in this Opinion as the habitat characteristics necessary to support successful adult and juvenile migration, spawning, and rearing. Those necessary habitat characteristics include an appropriate range of channel substrate sizes, and adequate water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions (Busby et al. 1996; Spence et al. 1996; 62 FR 43937, August 18, 1997; 65 FR 7764, February 16, 2000). Spawning and egg incubation require clean gravels and an ample supply of cool, well-oxygenated water. Juvenile rearing requires a complex physical environment with ample pools, shade, cover, water quality, and food production. Successful juvenile and adult migration require ample stream flow and velocity, in-channel cover, low water temperatures, and unobstructed passage. All of the biological requirements of listed salmon and steelhead could potentially be impaired by effects of exposure to contaminated water, prey, or sediment. Therefore, all biological requirements related to freshwater life stages, and any latent effects in adults from exposure in early life history stages or in offspring from adult exposure, are considered in this Opinion.

2.2.2 Environmental Baseline

The environmental baseline is a description of the effects of past and on-going human and natural factors leading to the current status of the species or its habitat and ecosystem within the action area. The action area is defined as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action" (50 C.F.R. 402.02).

Direct and indirect effects of the proposed discharge could occur beginning at outfall 001 in the Snake River and adjacent to the mill site in the Clearwater River and ending where there are no longer effects known to occur or likely to occur using best available information and models. One of EPA's models indicated potential for reduction in DO from this action near the Snake River confluence with the Columbia River. Thus, the Snake River action area is bounded on the upper end by Outfall 001 (SRM139 at the confluence with the Clearwater River) and on the downstream end by the Snake River confluence with the Columbia River (SRM 0). The Clearwater River action area is bounded on the upper end by the top of the mill site, because of seepage from the ASB, from Clearwater River RM 3 to the confluence with the Snake River (RM 0). Potential downstream effects of specific and combined constituents within the effluent vary with diminishing concentration and interaction by distance.

The current status of ESUs of listed salmon and steelhead is described in Section 2.1.3 above. Available information suggests that essential biological requirements of listed Snake River salmon and steelhead are generally not being met, as indicated by the small population size of wild fish and the 10-year average return of hatchery fish. Each ESU affected by the proposed action has a currently declining population of wild fish. There are documented temperature and DO problems for ESA salmonids that have significantly reduced their survival in the lower Snake River action area and contributed to their decline (NMFS 2000).

A recent description of the environmental baseline is found in the Opinion for FCRPS (NMFS 2000). Under the existing environmental baseline, continued mortality is expected from operation of the hydroelectric dams in the Columbia and Snake Rivers, as well as from upstream

flow depletions and resulting temperature alterations (NMFS 1997), degraded water quality, other habitat alterations in rivers and estuaries, and mortality from fishing. Consequently, because the environmental baseline as described in the FCRPS is inadequate for water quality (especially flow and temperature; NMFS 1997, 2000), incremental improvements in freshwater habitat conditions are essential for the survival and recovery of listed Snake River salmon and steelhead.

2.2.2.1 Habitat Conditions for ESA Listed Snake River ESUs

Many factors have contributed to the decline of listed Snake River steelhead, Snake River sockeye salmon, Snake River spring/summer chinook salmon, and Snake River fall chinook salmon, including: grazing; agriculture development; timber harvest; mining; urban and rural development; mainstem Columbia and Snake River dams; ground and surface water withdrawals and storage; and historical commercial, recreational and native American harvest. Many of these factors are unaffected by the proposed discharge permit, but they have all contributed to low population sizes that increase the risk that inadequate water quality could appreciably reduce these listed species' continued survival and prospects for recovery.

Water quality in streams throughout the Columbia River basin has been degraded by human activities such as dams and diversion structures, water withdrawals, farming and grazing, road construction, timber harvest activities, mining activities, and urbanization. Over 2,500 streams and river segments and lakes do not meet Federally approved, state and Tribal water quality standards and are now listed as water-quality-limited under Section 303(d) of the CWA. Tributary water quality problems contribute to poor water quality where sediment and contaminants from the tributaries settle in mainstem reaches and the estuary.

Most of the water bodies in Oregon, Washington, and Idaho that are on the 303(d) list do not meet water quality standards for temperature or flow alteration. Temperature alterations affect salmonid metabolism, growth rate, and disease resistance, as well as the timing of adult migrations, fry emergence, and smoltification. Many factors can cause high stream temperatures, but they are primarily related to land and water-use practices rather than point-source discharges. Some common actions that result in high stream temperatures are the removal of trees or shrubs that directly shade streams, excessive water withdrawals for irrigation or other purposes, and warm irrigation return flows. Loss of wetlands and increases in groundwater withdrawals have contributed to lower base-stream flows, which in turn contribute to temperature increases. Channel widening and land uses that create shallower streams also cause temperature increases.

Columbia River basin anadromous salmonids, especially those above Bonneville Dam, have been dramatically affected by the development and operation of the FCRPS. Storage dams have eliminated spawning and rearing habitat and have altered the natural hydrograph of the Snake and Columbia Rivers, decreasing spring and summer flows and increasing fall and winter flows. Power operations cause flow levels and river elevations to fluctuate, affecting fish movement through reservoirs and riparian ecology, and stranding fish in shallow areas. The eight dams in the migration corridor of the Snake and Columbia Rivers alter juvenile and adult migrations. Juveniles, including smolts, experience a high level of mortality passing through the dams. The dams also have converted the once-swift river into a series of slow-moving reservoirs, slowing the journey to the ocean and creating habitat for predators. Water velocities and critical water temperatures throughout the migration corridor now depend far more on volume runoff than before development of the mainstem reservoirs.

There have been numerous changes in the operation and configuration of the FCRPS as a result of ESA consultations between the Action Agencies (Bonneville Power Association, COE, and Bureau of Reclaimation [BOR]) and NOAA Fisheries and USFWS. The changes may have improved survival for the listed fish migrating through the Snake and Columbia Rivers. Increased spill at all FCRPS dams allows some smolts to avoid both turbine intakes and bypass systems. Increased flow in the mainstem Snake and Columbia Rivers provides better in-river conditions for smolts. The transportation of smolts from the Snake River has also been improved by the addition of new barges and modification of existing barges. In addition to spill, flow, and transportation improvements, the COE implemented numerous other improvements to project operations and maintenance at all Columbia and Snake River dams. These improvements, such as operating turbines at peak efficiency, new extended-length screens at McNary, Little Goose, and LGD, and extended operation of bypass screens, are discussed in greater detail in the 1995 and 1999 FCRPS biological opinions (NMFS 1995 and 2000).

Although, remedial passage improvements may have accrued due to changes in FCRPS operations and configuration, including juvenile transportation around portions of the hydrosystem, periods of warm weather and low runoff continue to cause high rates of mortality among out-migrants in LGR, which is the uppermost reservoir in the FCRPS that fish must negotiate without the aid of transportation because juvenile collection facilities are further downstream at LGD.

The EPA, NOAA Fisheries, USFWS, and the Federal Action Agencies have been working to abate or offset temperature impacts associated with FCRPS operations (NMFS 2000). A primary measure for reducing summer-fall water temperatures of the lower Snake River and action area is cool water releases from Dworshak Dam on the North Fork Clearwater River. Given the significance of these cool water releases on the Snake River mainstem and the severe limitations of other substantive measures to alleviate high water temperatures in the Snake River mainstem, decisions regarding Dworshak Dam releases may be the most critical in the near-term attempt to moderate water temperature problems for migrating juvenile and adult salmon in the lower Snake River.

Background or ambient levels of pollutants in inflows carry cumulative loads from upstream areas in variable and generally unknown amounts. Municipal and industrial waste discharges occur in the greater Lewiston, Idaho-Clarkston, Washington area and are received from larger population centers in the upper Snake River basin. Major tributaries and drainages deliver higher background concentrations of metals, which are generally associated with mining areas that are common in portions of the Clearwater and Salmon Rivers and in tributaries throughout the upper Snake River. Contamination from pesticides is associated with upstream agricultural areas. These contaminates are primarily attached to organic matter that are delivered to depositional areas where sediments accumulate. The LGR also receives high loads of inorganic sediments from the Clearwater, Salmon, and Grande Ronde Rivers. The Clearwater River confluence with the Snake River is at the upstream entrance to LGR. In this reach inorganic and organic sediment deposition rates are highest because the reservoir slows the velocity of both rivers, thus allowing most particulates to settle out within the upper portions of LGR.

To maintain flood conveyance and navigation channels to ports in Lewiston and Clarkston area, the COE constructed riprap levees along the lower Snake and Clearwater Rivers and regularly dredges sediment from channels in the upper part of LGR. NOAA Fisheries' March 15, 2004, biological opinion entitled "Lower Snake and Clearwater Rivers 2004-2005 Dredging Snake River Fall Chinook, Snake River Spring/Summer Chinook, and Snake River Steelhead" was specifically analyzed and considered in this Opinion. In-water and upland methods are used to

dispose of dredged material. Levee ponds that collect storm runoff from the cities of Lewiston and Clarkston and potentially seepage from the mill site are discharged into the lower Clearwater River. These activities create multiple mixing zones of reduced water quality and could resuspend pollutants resting in or bound to sediments in the action area.

2.2.2.2 Water Quality Standards

The BE (EPA 2003) provides a good description of water quality standards used in the establishing the proposed permit limitations, a summary of that information is restated here.

Section 303(c) of the CWA 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 CWA. 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.

The EPA has established water quality standards regulations at 40 C.F.R. Part 131. Under section 510 of the CWA, 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 CWA 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, with the exception of waste transport and assimilation which is not an acceptable designated use. Section 303(a-c) of the CWA 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 CWA directs EPA to publish water quality criteria guidance to assist States in developing water quality standards. The 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 126 priority toxic pollutants (40 C.F.R. 131.36), 45 non-priority pollutants, and 23 organoleptic effects criteria (EPA 1986). The 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).

2.2.2.3 Status of Receiving Water Quality

The Snake River from its confluence with the Salmon River at RM 188 to its confluence with the Columbia River has been included on the 303(d) list (a list of impaired waters compiled under section 303(d) of the CWA) for Temperature by Idaho, Oregon and Washington. Additionally, Oregon and Washington have most of the Columbia River on their lists for temperature. The States of Idaho, Oregon and Washington and the EPA are working in coordination with the Columbia Basin Tribes to develop Total Maximum Daily Loads (TMDL) for Temperature. The preliminary draft TMDL states that the effects of point sources and tributaries (nonpoint sources) on cross sectional average water temperatures in the main stems are for the most part quite small. The point sources can cause temperature plumes in the near field but they do not result in measurable increases to the cross-sectional average temperature of the main stems. The cumulative impact of all the point sources is less than 0.14˚C when temperature criteria are exceeded in the river. This preliminary draft TMDL establishes a wasteload allocation for the Potlatch Mill of 298.76 megawatts (MW). The proposed permit limits will result in a range of thermal loading from 43 MW (August) to 176 MW (January) based on the highest temperatures recorded in the Snake River.

On February 25, 1991, EPA established a TMDL for 2,3,7,8- TCDD, a dioxin congener, for the Columbia River Basin, which includes the lower Snake River. The TMDL was developed because the state of Idaho had listed the Snake River, the state of Oregon had listed the Willamette and Columbia Rivers, and the state of Washington had listed the Columbia River under section 303(d) of the CWA as not meeting standards for dioxin. The TMDL for TCDD was set at 5.97 mg/day based on an ambient water concentration of 0.013 parts per quadrillion (ppq) at harmonic mean flow. More specifically, the TMDL set a goal for the Snake River watershed of 1.18 mg TCDD per day before its confluence with the Columbia River. The TMDL established a wasteload allocation for the Potlatch Mill of 0.39 mg/day, which was incorporated into the 1992 permit. The EPA formally consulted under the ESA with the USFWS on the TMDL for the endangered bald eagle. The USFWS (1994) reported in the resulting Opinion that a lower target concentration of 0.0013 ppq would be more protective for LCR bald eagles because the establishment of the TMDL for TCDD was likely to result in incidental take of bald eagles due to detrimental effects resulting from chronic toxicity such as reduced reproductive success, and other behavioral and physiological impairments that may act to reduce the eagle's ability to survive. However, the USFWS agreed to the 0.013 ppq TMDL level over the following five-year period, with the incorporation of conditions designed to help protect the eagles and the collection of additional data regarding dioxin bioaccumulation in the LCR (USFWS, 1994).

On July 2, 1993, EPA submitted a BA (EPA, 1993) to the NOAA Fisheries as part of an informal consultation under the ESA for Snake River sockeye salmon, Snake River spring/summer Chinook salmon and Snake River fall chinook salmon and proposed critical habitat for the three listed species. Based on the available information at the time, NOAA Fisheries determined that implementation of the proposed TMDL is not likely to adversely affect the three listed species based on the following conditions: the loading capacities cited in the TMDL would not be exceeded; predicted maximum levels of dioxin are not exceeded (i.e., a water concentration equal to 0.013 pg/L, salmonid tissue concentrations equal to 0.37 pg/g, and salmonid egg

concentrations equal to 0.53 pg/g). In addition, NOAA Fisheries urged EPA to conduct or support studies that will reduce uncertainties regarding impacts of dioxin (and dioxin-like compounds) on all life stages of Columbia River salmonids.

2.2.2.4 Mixing Zone

Mixing zones are areas of mixing downstream of outfalls that may exceed ambient receiving water quality standards under the CWA. The size of mixing zone and the concentration of effluent parameters, individually and in combination, within it are typically expected to create the most critical conditions from which to analyze effects on habitat and fish. Because ambient conditions of receiving waters and permit limitations on effluent parameters interact the area of mixing varies, but typically is designed to remain a small proportion of receiving water area or volume. The effects of loads of slowly degrading components to downstream reaches and sediments are not addressed in mixing zone analyses.

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. The BE (EPA 2003) refers to two types of mixing zones: the hydrodynamic mixing zone (HMZ) and the regulatory mixing zone (RMZ). Descriptions for each type of mixing zone are restated 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 five 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.

2.3 Analysis of Effects

This section evaluates the potential effects of the proposed action on listed salmon and steelhead and their critical habitat. The analysis identifies potential effects of each of the constituent limits that would be expected to occur if water concentrations were equal to the proposed effluent limit.

This permit also authorizes seepage from the ASB into the Clearwater River. The BE (EPA 2003) cites Cascade Earth Sciences (1999) estimating maximum seepage from the ASB to the groundwater at about 6 mgd, but another study estimated maximum rates at 9 mgd (Dames and Moore 1991) and little is known about the chemical concentrations or seasonality of this seepage. So it is possible that the seep area within the Clearwater River could contribute to significant loading of pulp and paper mill chemicals either in the water column or through bioaccumulative organic compounds. This area deserves meaningful monitoring to better describe its potential for adverse effects.

The presence of many organic chemical compounds associated with Potlatch's effluent have never undergone rigorous chemical analyses to detect their presence. Of those chemicals that are known, many have never been reported in bioassay analyses. For these chemicals it is impossible to describe their potential to adversely affect salmonids. So, the potential for organic compounds to affect salmonids cannot be fully evaluated at this time. Some organic compounds that are carried in the effluent, such as the chlorinated phenolic compounds (CPCs), retene, and dioxins, likely settle to the bottom and incorporated as sediments. These compounds also bioaccumulate, therefore monitoring and assessment of sediments, chemical, concentrations in benthic aquatic prey organisms, and chemical concentrations in suitable fish tissue will be required to fully determine potential impacts of Potlatch Mill effluent.

2.3.1 Issues That Are Common to All or Multiple Constituents

The following discussion presents information about the potential toxicity of the parameters with effluent limits in the draft permit to endangered and threatened species evaluated in this Opinion. The purpose of this section is to identify whether potential exposure to parameters evaluated in this Opinion have been observed to cause toxicity to aquatic species, and, if so, to identify the concentrations or amounts of parameters to which species were exposed when such toxicity was observed.

This section discusses important toxicological principles, including the concepts of dose-response, toxicity thresholds, development of toxicity-based benchmark concentrations, and the use of extrapolation of toxicity data across chemicals or species. A framework of toxicity thresholds is used to determine whether concentrations or amounts of permitted parameters downstream of the discharge could cause toxicity to listed species. Methods are identified that were used to collect aquatic toxicity information about the permitted parameters, including the sources of toxicity information and the approach used to determine whether data are suitable for use in this analysis. Also described is the approach to assign the aquatic toxicity data for each parameter into categories according to the type of toxicity it could cause, direct toxicity to the species of concern or indirect toxicity to the prey or habitat of the species of concern.

Analysis of individual constituent sections discuss the aquatic toxicity data collected for each permitted parameter and the parameter's potential to adversely affect listed species. The lowest concentration or amount of a parameter that was observed in toxicity studies to have caused direct toxicity to a listed species (or similar species) is identified and then adjusted to a no observable effect concentration (NOEC), when possible. Similarly, the lowest concentration or amount of a parameter that was observed in toxicity studies to have caused toxicity to the prey or habitat of a listed species (or similar species) was identified and converted to a NOEC. The

toxicity benchmarks used in this Opinion are the NOECs of a parameter determined from the toxicological response in toxicity studies. This Opinion developed NOECs from toxicity studies using a criteria explained in 2.3.1.8 below.

The presence of a chemical in the tissue of a species that results from the presence of compound in the water column is not a response in and of itself. The response is the biochemical, physiological, or pathological change that occurs as the result of systemic presence of a chemical or agent. Adverse effects associated with bioaccumulation of a chemical in tissues (that is, the change in biological function that may occur once a parameter has accumulated to a certain point in tissues) are addressed by reviewing toxicity studies that relate a toxicological response to accumulated tissue levels. More commonly, the accumulated tissue levels are not measured, and the toxicological response is related directly to measured concentrations in an environmental medium.

The typical way of experimentally defining a concentration-response relationship is to test chemicals at several water concentrations and note the concentration at which the response of interest is observed. As such, the curve representing the relationship is not perfectly known, but rather is interpolated between actual measurement points and sometimes extrapolated beyond the areas of measurement. A measurement frequently derived in aquatic toxicity testing is the $LC_{50,}$ or the concentration that was lethal to 50% of the animals tested. Also, a regression analysis producing an EC_p value is preferred for determining sublethal adverse effects. (e.g., EC_{10}; EC stands for the "effective concentration", effective meaning sublethal; p represents the proportion responding.) This is in contrast to the "no effect concentration" (NOEC) and "lowest observed effect concentration" (LOEC) for exposure concentrations, which is determined by ANOVA. These values (LOEC and NOEC) are often information-poor because they are dependent on the quantal nature of allocating exposure concentrations and sound experimental design with sufficient statistical power to avoid false negatives (i.e., accepting the null hypothesis of "no treatment effect" when in fact an effect exists, but it can't be detected with the current experimental design). If exposure concentrations are too far apart or few replicates are used in the experimental design, the LOEC value determined by ANOVA may severely overestimate the true threshold value. In contrast, the EC_p value is determined directly with the dose-response curve and is a good statistical representation of the response, especially when a low proportion (e.g., EC_{10}) of the population is considered. Unfortunately, many of the studies used in this analysis contained too few treatments to produce a regression equation linking exposure or concentrations with a biological effect. Many of these studies examined only two or three concentrations that differed by up to an order of magnitude, leading to large gaps between the NOEC and LOEC values. Because of this, LOEC values should be viewed with caution because they may be higher than the true LOEC value.

2.3.1.1 Methods for Evaluation

Toxicity information for each parameter is presented, such as the types of toxicity evaluated, the test species, the life stage tested (when known), the concentrations observed to cause responses, and the endpoints (LOEC, LC_{50}, EC_{25}) associated with various concentrations. Concentration benchmarks are selected separately for direct toxicity and indirect toxicity to listed species. Benchmarks are the no effects concentrations generated by LOECs or LC_{50}s, both of which were adjusted with safety factors. The approach of using the lowest reported concentration probably affords the greatest level of protection to the evaluation. Confidence in the selected benchmarks (and subsequent confidence in the conclusions of the evaluation that uses these benchmarks)

depends upon the quality of the available toxicity data. Ideally, to predict with the greatest accuracy whether an effect is adverse, one would use toxicity data from an experiment using the species of interest, and using the experimental design most easily extrapolated to the conditions of interest. However, such studies frequently do not exist when dealing with listed species, therefore, studies that examine surrogate species and situations were used in this evaluation.

2.3.1.2 Sources of Aquatic Toxicity Data

The draft permit specifies effluent limitations for 21 parameters, including 15 chemicals (12 CPCs, chloroform, 2,3,7,8-TCDD, 2,3,7,8-TCDF), AOX, BOD, TSS, pH, temperature, and WET. A thorough review of the scientific literature was conducted to identify as many sources of toxicity data for the 21 permitted 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. The toxicity data used by EPA to derive water quality criteria were also reviewed.

There are currently no promulgated AOX ambient water quality criteria (AWQC) for the protection of aquatic life. This is, in part, because the composition of the chemical mixture that constitutes AOX depends on the raw materials used and processes conducted at facilities, as well as conditions in the receiving water bodies. Considering that AOX is a varying mixture of chemicals, the potential toxicity of AOX is best indicated by the results of toxicity tests using whole effluent (which includes any toxicity caused by AOX). Therefore, AOX is not reviewed as a separate parameter. Instead, an extensive review of WET results is provided.

Toxicity data for the 12 CPC and chloroform were obtained from EPA's Aquatic Information and Retrieval (AQUIRE) database and additional literature not cited in AQUIRE. The AQUIRE database contains more than 172,000 records with toxicity data from studies using more than 6,700 chemicals and more than 3,700 freshwater and saltwater species. Nearly 13,000 publications are reported in AQUIRE. Searches were conducted up to June, 2003, to identify aquatic toxicity data for the 13 chemicals listed above.

2.3.1.3 Approach Used to Select Suitable Studies

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. For other parameters, no studies, or only a few toxicity studies have been published. In some studies, potential toxicity to listed species was examined, while other studies evaluated toxicity to non-salmonid aquatic species. In some studies, toxicity data were obtained from experiments using a single chemical in a controlled exposure setting (such as an aquarium). Other types of toxicity studies examined potential toxicity resulting from exposure settings that more closely mimic the actual exposure conditions encountered by aquatic species (such as an experimental stream).

In some cases, the exposure conditions in an experiment differed greatly from the setting in which the listed species evaluated in this Opinion live. Toxicity data from species or settings that are very different from the listed species and their natural setting are not as indicative of the potential toxicity that could occur to listed species in the Action Area as studies using a closely related species in a natural setting. Accordingly, toxicity studies with certain characteristics were

considered unsuitable for use in the Opinion and were eliminated from the collection of toxicity data for permitted parameters. The studies that were considered unsuitable had at least one of the following characteristics:

The study was deemed incomplete. The EPA's AQUIRE 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 are not reported. Data from "incomplete" studies were not used in the Opinion.

The study did not report the mean concentration at which toxicity was observed. In a small fraction of the studies reviewed, the range of tested concentrations was reported, but the mean concentration at which toxicity was observed was not provided. Because of the difficulty in determining the concentration at which the toxicity actually occurred, data from such studies were not used in the Opinion.

Occasionally toxicity data for saltwater species were used for some organic compounds. For organic compounds, the main controlling factor for bioaccumulation is organic carbon content, which is not substantially different between the freshwater and marine systems. Additionally, pH may be important for some of these chlorophenols (e.g. pentachlorophenol). In general, the pH in marine waters occurs in the range of 7.5 to 8.5, which is not different than that observed in the Snake River. For chlorophenols that have toxicity values dependent on pH, use of marine species in toxicity assessment will generally underestimate the toxicity that may occur for similar species in freshwater. For pentachlorophenol, the lower the pH, the more toxic this compound becomes and a lower pH is more likely to be observed in the Snake River compared to most marine systems.

2.3.1.4 No Aquatic Toxicity Data for a Parameter

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:

The AOX is a complex mixture of compounds that vary in content and potential toxicity from mill to mill, depending on the raw materials used and the bleaching and treatment processes employed. As a result, there are no studies in the literature that have evaluated the toxicity of AOX to aquatic species. Toxicity data for some of the likely chemical components of AOX (such as chloroform and COPC) are used in the Opinion to assess the potential effects of those parameters on listed species. However, other compounds comprising AOX have not been identified and, therefore, cannot be assessed.

While evaluation of the toxicity of AOX itself and all individual components of AOX is not possible, it is possible to evaluate the toxicity of whole effluent (which includes AOX). In addition to the many toxicity studies that have used other mills' bleached Kraft mill effluent (BKME), the potential toxicity of Potlatch's effluent has been studied in several different experimental designs. These experimental designs included experimental stream studies using rainbow trout and sensitive indicator species, quarterly WET tests using sensitive indicator species, and whole life-cycle studies using approved indicator species. Toxicity studies using

whole effluent provide somewhat useful information; however, these surrogate species are not ideal. Even though valuable information about potential synergistic or antagonistic toxicity of the components of Potlatch's effluent may be obtained, the species used and endpoints monitored are not the most relevant for listed salmonids. While NOAA Fisheries approves of the concept of testing whole effluent to assess additive and synergistic interactions of all compounds, the WET testing scheme needs to include relevant species and biological responses. The EPAs monitoring and assessment plan addresses possible modifications to the WET test procedure.

Very little or no aquatic toxicity studies were found for 3,4,6-trichloroguaiacol, 4,5,6-trichloroguaiacol, 3,4,6-trichlorocatechol, 3,4,5-trichlorocatechol, and trichlorosyringol, especially for direct toxicity to salmonids. In most cases 2,4,5 trichlorophenol was chosen as a surrogate chemical for the effects analysis. All of these CPCs likely exert the same sort of mode of action, which is an uncoupling effect on mitochondrial oxidative phosphorylation (Lundberg, et al. 1980). For those compounds without relevant toxicity information, 2,4,5 trichlorophenol was chosen because it exhibited a relatively large amount of toxicity data from several studies on various species, its structure contained three chlorine atoms, and its K_{ow} was similar to most CPCs without toxicity data. Based on the above information about bioaccumulation, choosing this compound was conservative. Pentachlorophenol was not chosen because of its very large K_{ow} value, which was 20 times higher than the next closest K_{ow} (tetrachloroguaiacol) (Table 8).

2.3.1.5 No Aquatic Toxicity Data for Listed Species

Time and resource constraints prohibit actual direct testing of potential toxicity of all chemicals to all listed species. Consequently, few toxicity studies have been conducted using listed species. However, toxicity studies using similar or other highly sensitive species (e.g., 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 surrogates 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. Sappington et al. (2001) 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 cold water fishes, and the fathead minnow (*Pimephales promelas*) has been considered a suitable surrogate for warmwater fishes (Sappington et al. 2001). Since the listed salmonid species evaluated in this Opinion are cold water species, the following paragraphs will examine the sensitivity of rainbow trout compared to the sensitivity of listed salmonid species.

Sappington et al. (2001) examined the results of static acute toxicity tests using five chemicals (permethrin, copper, pentachlorophenol, 4-nonylphenol, and carbaryl) and four species (rainbow trout and three endangered salmonids: Apache trout (*O. apache*), Lahonton cutthroat trout (*O. clarki henshawi*), and greenback cutthroat trout (*O. clarki stomias*)). Water quality and geochemistry (i.e., pH, DO) in the test systems were controlled. Alkalinity and hardness were

consistent with the nominal requirements of the American Society for Testing Materials (ASTM) and EPA methods used for the static acute toxicity tests. The average pH was slightly above the requirement (8.4 versus 8.0) of the methods.

Generally, the results of this study showed that LC_{50}s for rainbow trout were similar to or slightly higher than LC_{50}s for the listed or proposed *Oncorhynchus* species (Sappington et al. 2001). Only the rainbow trout LC_{50}s for pentachlorophenol and permethrin were statistically significantly different from (and higher than) the listed species LC_{50}s. It can be concluded from this study that rainbow trout LC_{50}s would likely be protective of most listed salmonids; however, a safety factor may be needed to estimate LC_{50}s for listed fish species. The data from this study indicates that a factor of 0.5 would be a conservative estimator. In other words, listed species may be up to two-fold more sensitive than rainbow trout. This study was based on a limited set of toxicants, the mortality response, and very few salmonid species. It is not known if expanding the study with more species and contaminants or examining other important biological responses would change the outcome.

2.3.1.6 Influence of Temperature, pH, and Other Water Quality Stresses on Fish Response to Toxicity

Variations in temperature, oxygen, and pH can affect fish in several ways. One mechanism is direct action by fluctuations in these parameters that occur near the physiological limits tolerated by the individual. Another is by the rate of change, which can be rapid in an effluent plume. Changes in temperature have been associated with several biological responses in salmonids, such as changes in physiological parameters (Adenosine Triphosphatase [ATPase] activity, behavioral changes (predator avoidance, equilibrium, swimming performance), changes in cortisol, glucose, cholesterol, and growth rate (Fagerlund et al. 1995 for a review of studies).

Several physical/chemical parameters, such as temperature, pH, oxygen content, and salinity have been shown to affect toxicity in aquatic organisms. The effects of these parameters on the toxicity response have been thoroughly reviewed by Sprague (1995). Several mechanisms are likely responsible for the increases and decreases observed in toxicity when these parameters are altered. For example, fish generally experience increased ventilation when temperature is increased. When temperatures are elevated, fish respire more and therefore can accumulate more of a toxicant across the gills. The pH can alter the speciation of many metals leading to increases or decreases in the most bioavailable forms. The pH may also modify the organism's physiology altering the rates of uptake and elimination.

Temperature changes can affect toxicity; however, the direction of change (more sensitive or less sensitive) depends on the chemical. For example, many pesticides are more toxic in warm water, including pentachlorophenol (Asano et al. 1969); however, other compounds such as phenol are more toxic in cold water than warm water (Brown et al. 1967). For an effluent plume, the direction of response may not matter. Fish that travel from colder water to the warm waters of a plume may exhibit a greater sensitivity to some toxicants, whereas, other fish that stay in the plume for a few days and accumulate toxicants, may show greater sensitivity once they travel back to cooler waters. One study examined pulp mill effluent directly and found that it became more toxic with an increase in temperature (8ºC to 19ºC), with the lethal threshold declining up to 50% (Loch and MacLeod 1974).

The effects on toxicity by pH are well known. For many metals, toxicity increases greatly when pH falls. Ionic copper is known to increase almost an order of magnitude for a 0.5 unit decrease

in pH (Stumm and Morgan 1981), which can cause large increases in toxicity. Several studies have examined the relationship between pH and metal toxicity (for a review Sorensen 1991). In contrast to this is the behavior of ammonia. When ionized (NH_4^+), the toxicity is low; however, when pH increases, ammonia becomes less ionized and much more toxic. An increase of one pH unit (at circumneutral pH) raises the amount of NH_3 about six fold. Ammonia also increases in toxicity with increasing temperature. Cyanide toxicity is also affected by pH. The HCN is more toxic than CN^-, so as pH increases, toxicity will decrease.

2.3.1.7 Combined effects of toxicants

Although guidelines and criteria are determined for single chemicals, in combination their effects may be additive, synergistic (where the net effect exceeds the sum of effects), or antagonistic. These interactions must be considered because contaminants in the environment do not usually occur alone, but as complex mixtures. It should also be noted that bioaccumulation is a continuous process. Concentrations of some toxicants below their criteria or guideline value are still bioaccumulated and may be causing subtle effects. For contaminants with the same mode of action (i.e., toxic effect) concentrations may be added together when assessing the combined effect of all constituents. Several studies have shown that adverse effects occur when several criteria are near or equal to acute AWQC because of additive or synergistic toxicity mechanisms (e.g., Alabaster and Lloyd 1982, Spehar and Fiandt 1986, Enserink et al. 1991, Sorenson 1991, Escher et al. 2001).

2.3.1.8 Data Analysis for Threatened and Endangered Species

Water and sediment quality guidelines are generally based on analyses of the responses exhibited by several species to a toxicant. Often the goal is to protect 95% of all species from effects that may impact population abundance (Stephan et al. 1985). Consequently, these guidelines are based on the biological responses (mortality, and alterations to growth and reproduction), which are generally recognized as responses that will likely impact population dynamics. In contrast to this, protection of listed species under the ESA must consider harm to individual fish because additional losses will impede the recovery of an already severely depleted population. Because the probability of extinction is higher for ESA listed species, protection of individuals is important to ensure overall population recovery to a more stable level. While the standard sublethal responses of growth and reproductive impairment may be reasonably translated into population effects, all other sublethal responses, such as altered hormone levels, increased enzyme activity, and disease susceptibility were considered in this analysis. This is primarily due to the potential effects that contaminants may have on the complex physiological processes that allow individual salmon to transition from a freshwater to marine mode of existence, resist disease, mature normally, and successfully complete their life-cycle.

Because threatened and endangered species are so vulnerable, it is appropriate to apply a conservative factor to convert effect levels (e.g., lowest observed effect concentrations; LOECs) to “a safe value” (e.g., no observed effect concentration; NOEC). A safety factor of 10 is appropriate and is supported by other such applications (Chapman et al. 1998, Duke and Taggart 2000). This factor was also used for the indirect toxicity benchmark because of the uncertainty of the data for prey and non-salmonid fish. It should be noted that these adjustment factors do not take into account the higher sensitivity (factor of 2) to toxicants by listed species compared to rainbow trout (Sappington et al. 2001). Selecting the lowest NOEC value from one particular study would not necessarily provide protection against adverse effects, therefore NOEC values

from various toxicity studies were not used. The justification for not selecting NOECs include, temporal response, endpoint protection, interspecies variability, and experimental variability. The NOEC in published studies is endpoint-specific, meaning that it is an NOEC for a given endpoint such as mortality, growth, or reproduction. To adequately protect against all adverse effects, it is better to use the LOEC value found in the literature and apply a safety factor. Additionally, because the LOEC has variability (e.g., 95% confidence interval), does not necessarily represent the steady state response, and may be lower for listed salmonids (see above), the safety factor of 10 is well justified. Finally, NOEC values are negative evidence. Assuming that no response will occur at a NOEC is not necessarily true. It is also better to use statistically significant response data and make adjustments for all the inherent uncertainty.

In some cases the lowest toxicity value was not chosen because the original paper could not be obtained and examined. In those cases, the next highest value from a paper that could be examined was chosen as the benchmark value. It should also be noted that in some studies the only data available are 24 hour, 48 hour, or 96 hour LC_{50}s. It is well known in the toxicity literature that steady-state mortality (incipient LC_{50}) can take many days to occur. Consequently, these short-term LC_{50}s may underestimate the true mortality response by several fold. This phenomenon can be observed in the paper by Buccafusco et al. (1981) showing both 24 hour and 96 hour LC_{50} values for most of the chlorophenols of interest. These results show substantial reductions in LC_{50} values from 24 hours to 96 hours for each compound indicating increased toxicity over time.

Some of the available data examined only mortality, which is a severe response. To convert a lethal response into a sublethal response, a conversion factor was needed. A value of 10 is commonly used for translating acute mortality responses to sublethal or chronic responses (McCarty and Mackay 1993, Rand et al. 1995) and is often called the acute to chronic ratio (ACR). The LOEC is commonly the LOECs associated with a sublethal response to toxicant exposure. This conversion factor was applied to direct toxicity measures that were based on mortality as an endpoint. For those compounds where only mortality information was available, a factor of 100 was applied to convert from the LC_{50} to a NOEC. The same approach was used for the indirect toxicity benchmark to protect salmonid prey and other species that interact with salmonids.

2.3.1.9 Bioaccumulative contaminants

All of the organic pollutants of concern in this action bioaccumulate. In general, if a compound's log K_{ow} (Table 8) exceeds 3.0, it may accumulate and persist in sediment and become available for bioaccumulation. The higher the K_{ow}, the more likely it is to accumulate in sediment and biota. All organic pollutants of concern biomagnify to some extent in the food chain, although this is more of a serious concern for some contaminants than others. The EPA criteria development guidelines include a component designed to address the risks of elevated fish tissue residues of organic compounds to humans and avian and mammalian predators, but not the risk of that residue to fish (Stephan et al. 1985). In fact, this process drives nearly all of the numeric criteria established for organic contaminants. What is not considered in these evaluations, however, is whether these tissue residues would directly affect the health of the aquatic organisms themselves. Similar to metals, the consumption of aquatic invertebrates by fish is never formally considered in the development of the criteria for organic compounds. It is well-established that invertebrates may accumulate organic contaminants in aquatic systems, and that these contaminants are passed on to fish through the diet (Streit 1998). Consequently, if the

water quality criteria do not protect invertebrate prey species from organic residue accumulations, they may not protect listed salmon and steelhead juveniles from adverse effects associated with dietary exposure.

In particular, the use of ambient water concentrations alone leaves exposure pathways to several organic pollutants un-regulated. For example, CPCs are not highly water soluble, are persistent in both food and sediments, and are taken up by salmonids through their diet. A number of the organic compounds reviewed here have considerable potential to biomagnify in aquatic systems (Suedal et al. 1994). Proposed permit limits for such compounds do not consider food web transfer and bioaccumulation with respect to the target species. Consequently, they may greatly underestimate the toxicity of these chemicals in the environment. Ambient concentrations that are not associated with adverse effects in laboratory tests that only evaluate water-borne exposure may not be adequate when both water-borne and dietary exposures are taken into account. This is particularly important for the juvenile life stage of anadromous salmonids while they reside in rearing habitat, if such exposure later influences their downstream migration and subsequent ability to osmoregulate as they enter saltwater. Exposure to some of these compounds can also affect salmonid growth and disease resistance (Arkoosh et al. 1991, 1998; Glubokov 1990; Casillas et al.1995; Lundebye et al. 1999).

This is an especially significant concern for organic contaminants, such as dioxins, furans, and CPCs for which exposure is primarily via sediments and tissues of prey organisms. Indeed, environmental agencies in some other countries, including Canada, no longer recommend water quality guidelines for these substances, but regulate them through other media such as sediment, soil, or tissue (CCME 2001).

2.3.1.10 Dietary Exposure and Bioaccumulation

NOAA Fisheries believes that assessment of toxic effects from water column exposure to effluent contaminants is insufficient to adequately determined impacts to listed salmonids and their prey. Bioaccumulation of these compounds must be considered. The monitoring and assessment plan proposed in this BO addresses these deficiencies.

Because hydrophobic compounds are expected to show a similar or proportional affinity for the lipid of an organism as that for octanol, the degree of partitioning exhibited between water and octanol, as characterized by the partition coefficient K_{ow}, can be a useful means for evaluating and predicting bioaccumulation (Mackay 1982; Di Toro et al. 1991). For organic compounds that are not metabolized, the relationship between the bioconcentration factor (BCF) and K_{ow} is strong (Mackay 1982). The expected wet weight (wt.) BCF for a non-metabolized hydrophobic compound is a function of the lipid content of an organism and the value of K_{ow} for the compound. The standard equation for determining the expected BCF is:

$$BCF = 0.046 \times K_{ow}$$

which is derived from fish studies and is based on an average lipid content of 4.6% wet wt. McCarty (1986). This relationship is used in this Opinion for evaluating effects related to exposure and bioconcentration of the toxic organic pollutants. Compounds with a log K_{ow} above 3.5 are likely to bioaccumulate, especially if they are not metabolized.

Risk management via water concentration-based water quality criteria or benchmarks is not protective of listed salmonids against toxic pollutants that strongly bioaccumulate (e.g., mercury,

selenium, and some organic pollutants). This is because the true potential for toxic hazards to fish and wildlife through bioaccumulation is determined not only by an immediate water-borne exposure and direct toxicity effects, but also by the rate of mass loading into an aquatic ecosystem, the corresponding environmental partitioning of mass loads between the water column, sediments, and biota (food chain), and how the toxic pollutant is assimilated and acts on the organism. A water column concentration of a toxic pollutant may not reflect mass loading or be reflected in food chain bioaccumulation. Low concentrations of water-borne toxic pollutants can occur under very different scenarios: for example, when mass loading into the system is low, or alternatively through rapid biotic uptake and/or sediment deposition when there is elevated mass loading. Toxicity to fish and wildlife is ultimately determined by how much of the toxic pollutant is partitioned into the food chain. Therefore, water quality criteria or benchmarks are useful guides for risk management only to the extent that they protect aquatic food chains from extreme bioaccumulation.

Most of the organic contaminants of concern in this action have the potential to bioaccumulate. In general, if a compound's log K_{ow} (Table 8) exceeds 3.5, it may accumulate and persist in sediment and become available for bioaccumulation. The higher the K_{ow}, the more likely it is to accumulate in sediment and biota. Ambient concentrations that are not associated with adverse effects in laboratory tests that only evaluate water-borne exposure may not be adequate when both water-borne and dietary exposures are taken into account. This is particularly important for the juvenile life stage of anadromous salmonids while they reside in rearing habitat, if such exposure later influences their downstream migration and subsequent ability to osmoregulate as they enter saltwater. Exposure to some of these compounds can also affect salmonid growth and disease resistance (Arkoosh et al. 1991, 1998; Glubokov 1990; Casillas et al.1995; Lundebye et al. 1999). This is an especially significant concern for organic contaminants, such as dioxins, furans, and CPCs for which exposure is primarily via sediments and tissues of prey organisms. Indeed, environmental agencies in some other countries, including Canada, no longer recommend water quality guidelines for these substances, but regulate them through other media such as sediment, soil, or tissue (CCME 2001).

2.3.1.11 Benchmarks for Sediment and Tissue concentrations

Sediment and tissue benchmark concentrations were generated for many of the toxic compounds listed in this Opinion. These benchmark concentrations will be used for several of the monitoring studies to be conducted and any time sediment or tissue concentrations are available from monitoring and assessment activities. The values are listed in Table 9 and their derivation is described here.

Because chlorophenols generally act by a common mode of action, all sediment and tissue benchmarks are to be treated as summed values in the same fashion as the toxic unit approach. This concept is discussed and supported elsewhere in this BO. As stated in this BO, one recent study has demonstrated additivity for phenolic compounds and in some cases, the interactions were synergistic (Escher et al., 2001). Based on these data, additivity is a reasonable assumption for toxicity due to chlorophenol exposure. 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 toxicity equivalent quotient (TEQ) approach for tissue concentrations will be used for dioxins/furans, which is described elsewhere in this BO. The toxic unit approach will not be applied to the remaining compounds.

a. Sediment Benchmarks

Sediment benchmarks can be generated by two approaches. One approach entails predicting a sediment concentration by modeling the sediment-water partition coefficient normalized to organic carbon (=Koc). With this partition coefficient, the water benchmark can be equated to a sediment benchmark based on the expected partitioning between sediment and water. Another approach for determining the sediment benchmark is using biota-sediment-accumulation factor (BSAFs) and tissue concentrations known to cause adverse effects.

(1) Koc approach. The sediment benchmark concentrations in Table 9 were generated with benchmarks for water concentrations and predicted partition coefficients. First, a sediment-water partition coefficient (Koc) was determined. This partition coefficient is organic carbon normalized because organic carbon in sediment (TOC) this is the main controlling variable for the partitioning of organic compounds to water. These Koc values have been modeled by several researchers as a function of the octanol-water partition coefficient (K_{ow}) (Table 8) and the general equation (Di Toro et al. 1991) is:

$$K_{oc} = 0.983*K_{ow} + 0.00028$$

The Koc was then used to predict the sediment concentration (Sed_{oc}) that would be produced from the benchmark water concentration listed for each compound in Table 8. Sed_{oc} is the organic carbon normalized sediment concentration ([sediment]/f_{oc}).

$$K_{oc} = \frac{[sediment]/f_{oc}}{[water]}$$

where foc is the fraction of organic carbon.

$$Sed_{oc} = \text{Water benchmark} * K_{oc}$$

To convert to a bulk sediment concentration the Sed_{oc} is multiplied by the fraction of organic carbon found in the sediment (e.g., 0.01 for a 1% TOC content). For example, the water benchmark for 2,3,4 trichlorophenol is 3.4 ug/L and the K_{ow} for this compound is 5012. The Koc would equal 4,339, which would lead to a predicted sediment concentration of 14,753 ng/g oc. If the sediment organic carbon content was 1%, this would translate to a bulk sediment concentration of 147 ng/g. All values for the sediment benchmark are in Table 9.

To determine the toxic unit value for sediment concentrations, each measured sediment concentration will be divided by its respective sediment benchmark (Table 9) and all values will be summed. If the sum of toxic units (TU) exceeds 1.0, then adverse effects to biota may occur. The lowest value for each chlorophenol of the direct/indirect pair will be chosen. The equation is:

$$\text{Sediment benchmark TU} = \sum \frac{[\text{sediment}]_{\text{measured}}}{[\text{sediment}]_{\text{benchmark}}}$$

(2) BSAF approach. The BSAF approach can be used to generate sediment benchmarks when a tissue benchmark and BSAF value are known.

The BSAF is the lipid normalized tissue concentration divided by the organic carbon normalized sediment concentration.

$$\text{BSAF} = \left(\frac{[tissue]}{f_{lip}}\right) \Big/ \left(\frac{[sediment]}{f_{oc}}\right)$$

where [tissue], [water], and [sediment] are concentrations, f_{lip} is the fraction of lipid in tissue, and f_{oc} is the fraction of organic carbon in sediment. By convention, all values should be dry wts.

The BSAF is generally independent of chemical properties such as the partition coefficient. For example, for a given species, the same BSAF would be expected for all chlorophenols, regardless of their large variation in K_{ow}. The BSAF is usually expressed for steady state conditions; however, as with all bioaccumulation factors (BAF), time must be specified (i.e., 10-day BSAF). After a sufficient amount of time, steady state between tissue, sediment, and water partitioning will occur meaning the BSAF should not change. For most hydrophobic organic compounds, 28 days is usually sufficient to achieve a high percentage of steady state conditions (Lee et al. 1993), however some very hydrophobic compounds may take considerably longer.

The BSAF equation can be rearranged to express this relationship in terms of a sediment concentration. That equation is:

$$[sed_{oc}] = \frac{[tissue]}{\text{BSAF} * f_{lip}}$$

Using the tissue concentration associated with adverse effects in the numerator and BSAFs determined in the laboratory or field, a sediment concentration can be determined that is considered protective against adverse effects.

b. Tissue Benchmarks

Benchmark concentrations for tissue were generated by examining the literature. Several studies have exposed various species to toxicants relevant to this action and compared tissue concentrations with adverse biological effects.

The tissue benchmark for dioxin is addressed in Section V.C.4. The analysis for the CPC is found in Section V.C.45.

2.3.1.12 Monitoring Frequency and Exposure

An important consideration that is often not addressed in water quality monitoring is the issue of sampling frequency. In order to accurately compare water quality samples with regulatory criteria, samples need to be collected at least at the same frequency as the criteria (i.e., every hour for critical maximum concentration [CMC] and every four days for critical continuous concentration [CCC]). Otherwise, an exceedance could occur without detection. Samples, however, are seldom if ever taken at the specified frequency, and which allows the chance of undetected exceedances. This issue is a feature related to enacting and enforcing the proposed limits that has the potential to adversely affect listed salmonids.

2.3.2 Conventional Pollutants and Metals: Analysis of Individual Constituents

2.3.2.1 Temperature

Potlatch Mill discharges thermal effluent into the Snake River near its confluence with the Clearwater River at the head of LGR. Effluent exits through Outfall 001 and a single diffuser that is 122 m in length with 80 ports (72 active) at depth of 9.14 m. The proposed permit does not include a limitation on the mill's rate of discharge, which has been increasing over time, but is ultimately limited by a number of factors including capacity of the 48-inch diameter outfall delivery pipe. The mill's average discharge of thermal effluent was 56 cfs during 1997 to 1998. The 95th percentile discharge rate of 40 million gallons per day (mgd; or 66 cfs) was used by EPA for critical analyses and is also used herein to analyze effects, assuming that production of effluent will not continue to increase.

The permit includes a two-year compliance schedule for achieving the summer season daily maximum limit, to allow Potlatch time for design and construction. In the interim, the permit contains effluent temperature limits of 33°C during all months.

During consultation EPA modified the action and amended the BE to implement three heat recovery projects, augment discharge with Clearwater River water between May 15 and September 30, and implement operational and maintenance activities that will reduce the effluent temperature at the discharge port by approximately 3°C and at 3 m downstream of the diffuser by approximately 1°C. Potlatch has committed to completing heat recovery projects by April 1, 2004 and to report to relevant parties by June 30, 2004. Potlatch has additionally committed to completing the flow augmentation project by nine months after the new permit is issued, and to implementing ASB temperature management upon permit issuance. These measures will improve temperature quality of the effluent during and after the 2-year compliance schedule.

Temperature limitations in the proposed permit are as follows:

	Effluent Temperature Limitations	
Time Period	2-Year Compliance Daily Maximum	Daily Maximum
October - June	33	33
July	33	32
August - September	33	31

a. Temperature Effects

Temperature controls many physical and chemical processes in water, affecting concentrations of numerous constituents, such as: dissolved oxygen, pH, hardness and alkalinity, and the toxicity of some constituents such as, ammonia, organics, metals, cyanide, chlorine, and nitrogen. Temperature affects a number of biological interactions, which may affect or alter ecological regimes, including, competition and predation, metabolic function, disease, and prey forage. The National Academy of Sciences (NAS 1972a, in McCullough et al. 2001) recommendations for water temperature exposure for protection of aquatic life specify maximum acceptable temperatures for prolonged exposures (greater than one week), winter maximum temperatures, short-term exposure to extreme temperature, and suitable reproduction and development temperatures. Comprehensive literature reviews, issue papers, and guidance have recently been completed and most of the following information is from these sources (McCullough et al. 1999, 2001; EPA 2003a).

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

The upper incipient lethal temperature (UILT) is an exposure temperature, given a previous acclimation to a constant acclimation temperature, in which 50% of the fish will die within 7 days (McCullough et al. 2001). The duration for which a salmonid can tolerate an elevated temperature decreases with increasing temperature. As acclimation temperatures rise above 20ºC, the UILT essentially estimates the ultimate upper incipient lethal temperature (UUILT). The UUILT for salmonids is quite consistent ranging from about 23-26ºC among species (McCullough et al. 2001). Sockeye salmon are more sensitive than chinook with a 0.5-1ºC lower UUILT. Rainbow trout have a higher UUILT (about 1ºC) than chinook salmon, however, steelhead smolts appear to be much more sensitive than chinook smolts.

Although UUILT values for salmonids reported in the literature are up to 26ºC, fish in the field will not necessarily be acclimated to warm temperatures as they are in laboratory tests of UUILT. Given that wild fish are often more sensitive than domesticated laboratory stocks and are often acclimated to temperatures lower than laboratory fish, mortality in the field could be induced at

significantly lower temperatures. Therefore, McCullough et al. (2001) caution that UILT values in the field may be 1-2ºC lower than the UUILT values derived in the laboratory. Field data suggest maximum distribution of salmonids is limited to waters that have a temperature of about 2-3ºC less than the UUILT derived from laboratory experiments (McCullough et al. 2001).

The incipient lethal temperature (ILT) for a given fish species is determined by acclimating the fish to one temperature and then subjecting them instantaneously to another temperature. Acute thermal shock leading to death can be induced by rapid shifts in temperature (McCullough 1999). When salmonids are acclimated below 12ºC, substantial (50%) lethality (LT50) can be expected to occur almost instantly (1-60 s) at temperatures above 30-34ºC. For spring chinook salmon acclimated at 15ºC and subjected to water at 29.7ºC, survival time can be calculated to be 3.2 minutes (McCullough 1999).

Thermal shock can also cause sublethal effects and indirect mortality even with short duration exposures. Juvenile chinook salmon and rainbow trout acclimated to 15-16ºC and transferred to temperature baths in the range of 26-30ºC suffered significantly greater predation than controls (Coutant 1973). Coho salmon and steelhead trout acclimated to 10ºC and transferred to 20ºC water suffered physiological changes including hyperglycemia, hypocholestorolemia, increased blood hemoglobin, and decreased blood sugar regulatory precision (Wedemeyer 1973). Predation of sockeye salmon fry by juvenile coho salmon increased with a 10ºC increase in temperature (Sylvester 1972). Rainbow trout acclimated at 8ºC and transferred to 24ºC water remained at the surface for 1 to 2 minutes, expended large amounts of energy swimming haphazardly, then became lethargic at the surface, until after about one hour the fish could be handled with very little evasive response (Smith and Hubert 2003). In a fluctuating environment, multiple-day exposure to near lethal or lethal temperatures may create cumulative effects (McCullough et al. 2001). Multiple stresses can load or limit physiological systems, reduce growth rates, initiate disease, reduce osmoregulatory ability, and result in death when the physiological tolerance is overwhelmed (Budy et al. 2002; McCormick et al. 1998; Wedemeyer et al. 1990).

The smoltification process is tightly linked to temperature. At the time of smoltification, anadromous salmonids experience reduced ATPase levels at constant or acclimation temperatures greater than 11-13ºC that may result in delay or residualization (McCullough et al. 2001). It appears that temperatures of between 15ºC and 20ºC can block smoltification during migration of smolts to the ocean (McCullough et al. 2001). Temperatures above 18ºC may inhibit feeding in smolts.

At elevated temperatures (i.e., temperatures that cause changes in the smoltification process), several outcomes are possible. First, if smoltification cannot be completed then residualization occurs. Juvenile salmon usually wait for another year before out-migrating to the ocean. Remaining in freshwater for a second year subjects these juveniles to increased predation and sublethal effects associated with rearing in an area that reaches critically temperatures. Second, survival of smolts that reach the ocean depends heavily upon the degree of smoltification and general health to volitionally enter saltwater. If elevated temperatures have inhibited smoltification, reduced growth, or increased stress, then ocean survival can be decreased.

Adult salmonids are very sensitive to temperature and appear to have lethal tolerances 2-3ºC lower than the juvenile fish typically used in lethality testing (McCullough et al. 2001). The National Academy of Science (1972a, in McCullough 1999) recommended that a safety factor of -2ºC be added to the UILT. This safety factor was meant to minimize mortality but did not consider disease incidence or other sublethal effects, which were poorly understood at that time

(McCullough 1999). The UILT also does not consider other stressors that may be present, such as high turbidity, low dissolved oxygen, or chemical toxicants, that may cause fish to die at temperatures lower than the UILT (McCullough 1999).

Loss of equilibrium occurs at lower temperatures than lethality. In addition, as temperatures increase, incidence of disease increases. Holding adult chinook at 14ºC for 1.5 months resulted in no mortality but holding for a similar amount of time at 19ºC resulted in nearly total mortality from disease (McCullough 1999). Columbia River steelhead, acclimated at 19ºC, had a lethal threshold of 21ºC and have also been shown not to migrate until river temperatures fall below 21ºC (McCullough et al. 2001). Of note is that as temperature increases and causes adult salmon to hold in thermal refugia, the additional expenditure of energy while holding and not migrating may cause pre-spawning mortality through expenditure of limited energy reserves (McCullough et al. 2001). MacDonald (2000) noted that in a tributary to the Frazier River, 25% mortality of chinook salmon spawning runs occurred when mean summer temperatures were frequently above 20ºC and reached a high of 23ºC.

Migration behavior of adult chinook salmon can be changed by water temperature. Chinook and other salmon are assumed to delay migration until water temperature is conducive. Extensive delays can prevent adult salmon from reaching spawning grounds in time to spawn successfully.

Generally, spring and fall chinook salmon migrate upstream when temperatures are between 11ºC and 20ºC (McCullough et al. 2001). A migration threshold at a temperature of 21-22ºC is documented by numerous studies across major migratory salmonid species in the Columbia River (McCullough et al. 2001; McCullough 1999). On the Tucannon River in Washington, spring chinook salmon stopped migrating when water temperature reached 21ºC. Similarly, adult sockeye salmon migrations were also blocked at 21ºC according to one study. Another study reports that some passage occurred even when temperatures exceeded 22.2ºC. In the Okanogan River steelhead migration did not begin until temperatures fell below 22.8ºC, though optimal migration temperature was about 14.4ºC (McCullough 1999). Snake River sockeye salmon are reported to delay migration at temperatures greater than 21ºC (Quinn et al. 1997). At temperatures above 20-21ºC less than 50% of fall chinook entering the Snake River successfully passed LGD (Karr et al. 1998). Migrations during temperatures a few degrees lower resulted in almost twice as many fish reaching LGD.

Spawning success can change as a result of the temperature experienced by adult salmon before spawning. Pre-hatch mortality and developmental abnormalities were higher and alevin size was smaller in chinook salmon held at 17.5-19ºC for two weeks before spawning compared to chinook salmon held at 14-15.5ºC for the same period of time (McCullough 1999). As a result of these and similar observations, temperatures of 6-14ºC have been recommended for holding of chinook salmon broodstock (McCullough 1999). Pre-spawning mortality caused by elevated temperature and extended holding times are similar for sockeye salmon survival; 51% of adults survived in a diel temperature regime of 9.4-23.3ºC while 96% survived in a diel regime of 11.1-15.6ºC (McCullough 1999). Elevated holding temperature has also been shown to result in lower health and reproductive indices (McCullough et al. 2001).

Under stressful conditions mortality rates of gravid, and therefore sensitive, adult migrants are likely to increase at the population or run level. Furthermore, gametes inside body cavities of exposed adults are highly susceptible to thermal damage. Delayed pre-spawning mortality due to disease, stress, and exhaustion will increase in upstream areas (Wilkie et al. 1997). Also spawning vigor (penetration into low order streams, mate selection, redd construction, and other

such essential behaviors) may be severely impaired and result in lower rates of survival of progeny. Prespawning mortality rates of salmon often exceed 20-40% after passage through the lower Snake River (Wilkie et al. 1997).

b. Temperature Effects in LGR

More information on general effects of temperature on rearing and migrating juvenile salmonids and temperature effects on juvenile survival through LGR and the lower Snake River may be found in: Section 2.1.3 of this Opinion; McCullough et al. (1999, 2001); NOAA Fisheries (2003a, Table 2); EPA (2003a); Curet (1993); Smith et al. (2003); and Connor et al. (1998, 2003a, 2003b).

Within the four ESA-listed species addressed in this Opinion, fish react similarly to temperature, yet salmonid life stages react differently to the effects of temperature.

(1) Juveniles. The middle area of the channel containing the mixing zone is primarily mid-depth habitat. Subyearling salmonids prefer shallow, sandy substrate, and slow velocity water near faster current, yet apparently use pelagic habitats at times, such as about a month before continuing downstream migration or when shoreline or surface water temperatures exceed 17ºC (Curet 1993). Subyearling salmonids migrating from upstream areas must pass the diffuser. Their rates of travel are typically slow and although specific migration paths are not well documented, subyearlings are expected to prefer near-shore routes, which reduces their risk and incidence of exposure to the mixing zone by enabling a greater proportion to avoid or skirt the mixing zone while moving downstream. In LGR larger juvenile chinook and steelhead use deeper-water habitats for rearing similar to that occurring in the vicinity of the diffuser (Bennett et al. 1993). Salmon and steelhead smolts are usually yearlings more than 90 mm in length. These fish are typically traveling quickly (2-14 days) through LGR during their spring or early summer migration towards the ocean. Smolts typically use mid-channel habitats for migration and are thought to prefer the fastest velocity portions of the thalweg, swimming at depths averaging about 3 m.

Based upon the available literature summarized above, the temperature of 17ºC is a critical benchmark for rearing and migrating subyearling fall and spring/summer chinook salmon in the lower Snake River. As temperatures approach and exceed 17ºC subyearling chinook apparently vacate shallow water habitats in LGR (Curet 1993). This behavior likely results in cumulative stress, reduced feeding and growth, and increased rates of predation.

Within the first meter of discharge from the diffusers, the velocity of plume jets are about 6 m/s. The jets are angled 38 degrees from horizontal and 30 degrees from vertical from the direction of flow. Some smolts swimming downstream with the current at 2-3 m/s could encounter the broadside or topside of one of the 72-80 jets. This scenario is most likely when smolts migrate at night or in turbid peak-flows without the visual acuity to detect and avoid the jet plumes. Selected toxicity benchmarks for instantaneous temperature increases of 10ºC over ambient is likely to increase predation and cause physiologically sublethal stressors (Coutant 1973; Wedemeyer 1973). Although not well documented in combination, because compounds found in pulp mill wastes are somewhat unique, it is expected that a 5ºC increase over ambient in combination with other stressors (high velocity jet currents, low DO, and temporarily elevated concentrations of other pollutants) upon contact along the margins of a jet-stream or within a slower velocity plume is likely to at least disorient smolts and accumulate stress.

Migrating spring/summer chinook, fall chinook, sockeye, and steelhead yearling smolts are present in the action area from late-winter to mid-summer. These fish are characteristically swimming fast through the lower Snake River dams toward the ocean. Although thermal shock can occur during any season, the time required for warm effluent to reach equilibrium with background water temperature is very short during colder seasons and the size of the initial mixing zone includes only the first meter of the jet plumes, so the likelihood of exposure becomes very small by size and distance to exceed benchmarks.

Few yearling smolts migrate during summer. Warmer river and effluent temperatures during summer cause the mixing zone to expand to its maximum size. This expansion increases the likelihood of encounters with diffuser jets for those few juvenile salmonids migrating during the summer.

The diffuser jets potentially adversely affect juvenile salmonids by causing disorientation and a reduction in travel rate, or forced navigation around the mixing zone. Most juveniles would cross the diffuser, avoid the jets, continue downstream into the head of the passive mixing zone that is about 45 meters in length, and then swim through the entire length of mixing plume without changing course. Fish travel time to pass the mixing zone and affected confluence area would require 20 seconds for the fastest yearling smolts to maybe two minutes for the slowest or weakest swimming sub-yearling smolts. This route would result in a few seconds of increased exposure to a mix of toxicants within the mixing zone.

Critical condition modeling in the BE indicates that within 1-2 m of the diffusers the temperature of jets would not exceed 5ºC over ambient. Within the first meter of the zone of initial dilution (ZID) proposed temperature limits of 31-33ºC may exceed 10ºC over ambient, comprising less than 1% of the cross-section of the Snake River before it mixes with the Clearwater River. Rapid cooling occurs so that at about 3 m downstream from the diffuser where the individual jets combine into a single plume and passive mixing begins, the mixing zone includes not more than a 1ºC elevation from ambient and comprises about 5% of the Snake River channel cross-section. At 10 m and farther downstream the mixing zone plume is 3-6 m thick and may include about 15-25% of the Snake River channel at a temperature elevation of maybe 0.3-0.5ºC. During summer (July-September) when temperatures of the receiving waters exceed benchmarks for temperature, the effluent plume fills about 32% of the Snake River channel 45 m downstream from the diffusers. Here the plume is modeled to increase the temperature less than 0.3ºC. Given expected velocities of 0.3 m/s, water particle time to travel this length of the plume is about two minutes. The mixing zone would likely increase temperatures 0.1-0.3ºC in an area less than 5,500 m^2 (122 m wide by 45 m long) in the confluence area. Further downstream any temperature elevation caused by the mill effluent is slight or negligible.

(2) Adults. Based upon the available literature summarized above, selected toxicity benchmarks for various adverse effects to adult salmon and steelhead caused by temperature in LGR are: 21ºC for avoidance and lethality and 16-19ºC for sublethal effects of increased incidence of disease, loss of equilibrium, and loss of limited energy reserves.

Passage counts at LGD and Ice Harbor Dam indicate passage of a few chinook salmon, sockeye salmon, and steelhead at temperatures above 21ºC. Substantial proportions of these runs were adversely affected with higher rates of delay, stress, reduced fitness, and ultimately mortality.

In the vicinity of the Snake-Clearwater Rivers confluence water temperatures are above temperature benchmarks for adult salmonids for many days during summer each year. When

thermal barriers to adult migration form in the lower Snake River it is caused by background river temperatures. The mill contributes a measurable increase to water temperature within the jets and initial zone of mixing and variably slight to negligible increases to the surface waters of the Snake River within 45 m downstream. There are always passages around the mid-channel mixing zone for adult migrants by swimming upstream through the Clearwater River plume or up the western and southern side of the Snake River channel. The portion of the mixing zone near the diffusers with temperatures greater than 5ºC above ambient is only 1% of the river cross-section and the diffuser jets should be easily skirted by upstream swimming adults.

Most adult migrants likely pass the confluence area fairly quickly, with limited resting or staging, and continue migrating upstream. Significant proportions do so during July and August when Snake River temperatures exceed benchmarks.

Significant proportions of runs of adult summer steelhead also migrate into LGR and the confluence area during July and August and continue holding there until resuming upstream migration in February. Radiotelemetry studies show that many of these fish remain at or within a few miles of confluence for up to 2-5 months (Bjornn et al. 2000, 2003). It is believed that some fish would periodically hold in the turbulence zone between the two rivers. Because the mixing zone includes the confluence turbulence zone, some of these fish could remain in close proximity to the mixing zone where chronic exposures to effluent must be considered. During the winter cold periods, the warmer waters of the mixing zone may be attractive to adult steelhead and may cause them to hold longer within the mixing zone.

b. Summary Effects of Permit Limits (Temperature)

The proposed permit allows release of 33ºC water year-round for the next two years. After a two-year compliance schedule, the final effluent temperature maximum daily limitations are 33ºC during October through June, 32ºC during July, and 31ºC during August and September. Permit limitations allow continued release of effluent warmer than ambient temperatures when ambient receiving waters exceed temperature benchmarks for adverse effects to ESA species.

The mill's heat contribution is measurable but relatively negligible to downstream reaches. Adverse effects to salmon and steelhead are likely limited to the area within the first 45 m downstream of the outfall for avoidance and to chance encounters of smolts with the 70-80 diffuser jets within the first meter from the diffusers that could cause mild thermal shock to a small fraction of migrating smolts. For these fish, exposure increases the risk of predation mortality.

2.3.2.2 Total Suspended Solids (TSS)

The TSS include both organic and inorganic particulate matter in water and refer to the portion of total solids retained on a 2 m (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 (EPA 2003). The proposed TSS effluent limitations require a maximum daily effluent limitation of 94,400 lb/day and an average monthly effluent limitation of 50,600 lb/day. These limits are based upon the last five years production of unbleached kraft market pulp 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 (EPA 2003).

During consultation EPA modified the action and amended the BE to include a conservation measure (section 1.2.4.2) to permanently reduce TSS by 25% of the 2002 average. The measure would be implemented within 1-3 years of permit renewal.

a. Total Suspended Solids (TSS) Effects

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

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

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

Indirectly, the suspended solids affect other parameters such as temperature and DO. Because of the greater heat absorbency of the particulate matter, the surface water becomes warmer and this tends to stabilize the stratification (layering) in stream pools, embayments, and reservoirs. This, in turn, interferes with mixing, decreasing the dispersion of oxygen and nutrients to deeper layers and altering the vertical stratification of heat in the water column (Wilber and Clarke 2001).

High concentrations of TSS can block light from reaching submerged vegetation. As the amount of light passing through the water is reduced, photosynthesis slows down. Reduced rates of photosynthesis causes less DO to be released into the water by plants. If light is completely blocked from bottom dwelling plants, the plants will stop producing oxygen and will die. As the plants are decomposed, bacteria will use up even more oxygen from the water. Low DO can lead to fish kills. High TSS can also cause an increase in surface water temperature, because the

suspended particles absorb heat from sunlight. This can cause DO levels to fall even further (because warmer waters can hold less DO), causing additive harm to aquatic life (Mitchell and Stapp 1992).

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

b. Selected Toxicity Benchmarks: Total Suspended Solids (TSS)

Based upon a review of several sources, and by applying a safety factor of 10 to the lowest LOEC reviewed, EPA (2003) selected 10 mg/L as a TSS benchmark. NOAA Fisheries concurs with this benchmark.

c. Water Column Effects

The EPA (2003) modeled the dispersion of TSS from Potlatch's effluent into the Snake River under a variety of flows and stratification scenarios. NOAA Fisheries' evaluation relies upon the more conservative (for this purpose) scenario of the Snake River and Clearwater River being horizontally stratified, i.e. running side-by-side without mixing for long distances. The TSS dispersion was modeled by EPA (2003) using the 1Q10 flow (the lowest one day average flow occurring on a 10-year recurrence interval) was modeled for the months of July – December. Under these scenarios, TSS was predicted to exceed the benchmark for about 258 to 475 meters downstream of the outfall. This increase in TSS could reduce oxygen and light, increase adsorption of heat, reduce mixing and dispersion of the effluent plume in the mixing zone. The TSS increase could also irritate gill membranes with exposure or cause avoidance by salmon and steelhead of the mixing zone as it extends downstream over these distances.

d. Sediment Effects

The potential for the suspended solids in the Potlatch Mill's effluent to contribute to bedload sedimentation was also evaluated by EPA (2003).

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. The EPA 2003 (citing 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.

The EPA applied the trapping efficiency formula to LGR and predicted trapping efficiencies of 99%. However, it is not proven from field studies whether the TSS float, remain suspended, or settle before degrading, so the relative effects to site-specific habitats are difficult to measure.

Recent studies have shown that organic matter discharged from pulp mills flocculates and accumulates on the river bottom more rapidly than would be predicted by transport models if the matter remained as particulates, a phenomenon termed pulp mill effluent induced coagulation and flocculation (PMEICF). 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 (Young and Smith 2001; Joyce and Smith 2003; see also references cited by EPA 2003). Existing sediment transport models have failed to take into account the phenomenon of pulp mill effluent induced coagulation and flocculation. 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. Bacteria excrete polymeric substances, which may be significant in the floc formation. Only some bacterial species are considered "floc-formers" The physiological changes of the bacteria could influence the observed induced flocculation (EPA 2003).

The 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 food chain of juvenile salmonids.

The sediment data collected by Potlatch reviewed in the BE indicate that settling of the suspended solids in the effluent may be occurring between SRMs 137 and 131. The EPA (2003) noted that this would agree with the literature findings. Since the maximum effluent concentration of TSS is quite high (283 mg/L) compared with the benchmark of 10 mg/L, chronic effects on juveniles include reduced vision, possibly some stress, and avoidance of the area. The findings of several studies listed above indicate the discharge of TSS at the 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 upper LGR does include habitat needed to support juveniles (subyearlings, yearlings, and residualized smolts) and adults, and holding habitat for bull trout and steelhead.

Settling of suspended solids often occurs in low velocity waters, overlying a gently sloping bottom (low gradient), near the fringes of the channel. A number of these areas are located in the upper portions of the reservoir upstream from RM 120 (PNNL 2003). These habitat types correspond with the most heavily-used reservoir habitat areas by sub-yearling chinook salmon (Key et al. 1996). The settling of suspended solids from Potlatch's effluent would contribute to the LGR organic load, especially during low flow conditions.

Juvenile chinook salmon in LGR have been observed to have poor feeding success compared to riverine areas and reservoirs further downstream (Muir and Coley 1996). Poor feeding success could be a factor in poor survival of juvenile salmon (Muir and Coley 1996; Curet 1993). A high proportion (26 – 38%) of juvenile salmon collected after passing LGR had empty stomachs (Muir and Coley 1996). Few (3 – 5 %) juvenile salmon collected further downstream at McNary and Bonneville Dams had empty stomachs (Muir and Coley 1996).

Amphipods are taxa typical of impounded waters, but were scarce from LGR (Muir and Coley 1996; Cushing 1993). Reservoirs downstream from LGD in the Snake River generally have coarser sediments and greater benthic diversity (Muir and Coley 1996; Curet 1993), so habitats preferred by these invertebrates may be lacking in the mud and sand substrates of LGR. Yet, Frest and Johannes (1992) found the reach of Snake River downstream of Lewiston and Clarkston (and Outfall 001), was remarkably depauperate of molluscs and the native freshwater fauna has declined from 33 to 6 of the taxa present in upstream reaches. They postulate the absence of freshwater snails from Clarkston downstream to Nisqually John Landing in LGR is especially remarkable and requires extraordinary explanation.

It seems plausible to NOAA Fisheries that poor juvenile salmon feeding conditions are due in part to reservoir morphometry and river substrate, yet Potlatch's effluent also contributes. Studies are needed to specifically identify the environmental fate and biological effects of Potlatch's effluent solids. Adverse effects may occur directly on aquatic invertebrates, and indirectly via habitat modification. The TSS and associated pollutants from Potlatch might be a contributor to the poor rearing and feeding conditions for juvenile salmon in the upper portions of LGR by affecting critical habitat, specifically food resources in the habitat. TSS impacts on aquatic invertebrates may result from physical toxicity of the suspended solids themselves, or indirectly impact invertebrate populations by alteration of surficial sediment substrates after the suspended solids settle out of solution, resulting in changes in the species composition and abundances of benthic invertebrates.

2.3.2.3 Dissolved Oxygen (DO)

Three components of mill effluent can affect receiving water DO levels: (1) the actual concentration of DO by volume discharged into the mixing zone, (2) the load of COD material, and (3) the load of BOD material. These components cumulatively affect different areas within receiving waters based upon rate of assimilation and degradation, fate and transport.

The proposed permit contains no limitations on the concentration of DO or load of COD in the effluent. Limited reporting by Potlatch of voluntary self-monitoring data indicate COD concentrations within the effluent are significant with an average of 523 mg/l, including a maximum release of 1650 mg/l (BE Table III-2). These limits exceed generic environmental guidelines for COD from pulp and paper mills as established by some sources (World Bank 1998).

The BOD is a measure of the concentration of oxidizable organic material in the effluent. The BOD determination is an empirical test of BOD5 which is explained in further detail in the BE (EPA 2003). High BOD lowers the concentration of DO in water, and toxicity could occur as a result of insufficient concentrations of DO. Reduced DO concentrations mean less oxygen is available for respiration. Because a *decrease* in DO concentration can cause toxicity (in contrast to other parameters, for which an *increase* in concentration can cause toxicity), water quality criteria for DO are expressed as a *minimum allowable* concentration.

Although the permit limit is for BOD and is established as lbs/day, the BE evaluates the potential effects to listed salmonids, from DO concentrations (mg/L). The BE modeled BOD5 released from the mill and found it to have reasonable potential to impact water quality standards and adversely affect listed salmonids. The model showed that Snake River DO concentrations fell below water quality standards near its mouth approximately 100 miles downstream. NOAA Fisheries concurs with EPA (2003) that adverse effects to salmonids are likely from DO

reductions in the lower Snake River caused by loads of BOD in the mill effluent, and that reductions in BOD5 in the proposed permit are warranted. There is some uncertainty, however, regarding transport modeling of where these loads are most likely to degrade, which should be monitored and ground-truthed so that effects can be refined.

The proposed permit limitations for BOD are set by time period, which is more or less sensitive to regulated river discharge and water temperature. The proposed permit includes a 5-year compliance schedule for June-November limits on BOD. The interim BOD limitations in the permit are set by river discharge in a similar fashion to past permits (EPA 2003, Tables III-3 and III-5).

Time Period (BOD5)	Max Daily (lbs/day)	Monthly Average (lbs/day)
1999 Draft Permit		
River Flow: >22,000 cfs	53,800	28,100
>18,000 - 22,000 cfs	36,300	18,900
<18,000 cfs	24,600	12,800
Proposed Permit (after 5-year compliance)		
December - May	55,100	28,800
June - November	9,800	5,100

During consultation EPA modified the action and amended the BE to include conservation measures (section 1.2.4.2) to (1) reduce TSS by 25% of the 2002 average within 1-3 years of permit renewal and (2) increase effluent DO levels by a projected 5-20% dilution of ambient Clearwater River as an additional means to increase effluent DO concentrations from May 15 to September 30 each year before the end of the two-year compliance schedule.

The BE analyzed (modeled) effects of BOD in the lower Snake River, yet did not analyze the combined additional effect of COD on DO concentrations within the mixing zone, LGR, and lower Snake River.

a. Dissolved Oxygen (DO) Effects

A number of factors affect DO in receiving waters. The DO content of fresh water is about 14.6 mg/L for saturation at 0ºC and decreases gradually with increasing temperature to 9.1 mg/L at 20ºC and 7.5 mg/L at 30ºC (Rand and Petrocelli 1985). Other factors that tend to decrease DO in receiving waters include aquatic microbial, plant, and animal respiration. Factors that tend to increase DO include the equilibrium between atmospheric oxygen concentrations and the concentration of DO in water, wind mixing, and photosynthesis by aquatic algae and higher aquatic plants.

Spatial variability in DO includes longitudinal, vertical, and temporal components (COE 1999). Typical depletion of DO downstream of a source of oxygen demand is approximated by a specific first-order decay curve to a sag point, and then recovery based on reaeration. The shape

of the curve is dictated by the magnitude of the demand, the nature of the substances exerting the demand, the water temperature, hydraulic factors, stream geometry, the background DO concentration, and the reaeration potential of the reach downstream of the source (COE 1999).

Longitudinal variability in DO concentrations can also be related to the locations of sediment oxygen demand, stands of macrophytes, differences in reaeration rates related to channel morphometry, the presence of blooms of phytoplankton or the presence of large numbers of respiring organisms in localized areas. Vertical variability in DO levels typically occurs when water is isolated at depth through thermal or density stratification. This isolation removes the potential for reaeration of these waters while allowing for DO depletion through the settling into and decomposition of organic matter in the deep layers, which is additive to latent sediment oxygen demand. The DO concentrations vary over temporal time scales ranging from seasonal to hourly.

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

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

Reductions in DO can decrease swimming performance in both adult and juvenile fish, affecting the ability to migrate, forage and avoid predators (NMFS 1999c; Spence et al. 1996). Any reduction in DO below saturation at high water temperatures increases the risk of adverse effects to salmonids. Subyearling and smolt life stages are very sensitive to low DO. Dahlberg et al. (1968) found that a reduction in DO to 7.5 mg/L resulted in a five percent reduction in swimming speed. Dahlberg noted that swimming speed declined markedly below 7-8 mg/L DO. The ecological significance of increased stress and reduced swimming ability has only recently been increasingly verified and associated with latent declines in production and survival (Servizi and Martens 1991; Wilkie et al.1997; Wedemeyer et al. 1990; Budy et al. 2002). Complete migratory blockages of chinook salmon have occurred at DO concentrations of 4.3 mg/L or less (Materna 2001).

Sublethal effects that occur below 8 mg/L may control survival and success of juvenile salmonids in nature through reduced growth and size observed in juvenile salmonids at DO concentrations below saturation. Swimming speed in juvenile salmon declines markedly below DO concentrations of 7-8 mg/L (NMFS 1999c). Results of several growth experiments summarized for coho salmon (Warren et al. 1973, as cited in ODEQ 1995) show that growth rate appears closely related to DO concentrations below 6.0-6.5 mg/L. The ODEQ's issue paper further reports that concentrations from near 8 to 6.5 mg/L resulted in measurable reductions in swim

speed and maximum attainable growth and laboratory studies which have shown that blood is not fully saturated with oxygen at levels near 6.5 mg/L and changes in oxygen transfer efficiency occur.

Warm water levels, water temperatures work in synergy with DO concentrations to cause a range of adverse effects to salmonids. This range includes acute lethal toxicity, inability to complete essential foraging and predator avoidance behaviors, area avoidance, migration delays, increased stress, reductions in growth, and slower swimming speed. Low DO concentrations increase the acute toxicity of various toxicants such as metals (e.g., zinc) and ammonia (Rand and Petrocelli 1985; NMFS 1999c). Adverse effects of toxicants found in pulp and paper mills may be increased by low DO. Hicks and Dewitt (1971) show a 5-20% increase in toxicity as DO levels fall to 60% of saturation. Percent reductions in DO of this magnitude occur in LGR and the lower Snake River during summer. Also, TSS and toxicants may increase sensitivity to low concentrations of DO. For example, any toxicant which damages the gill epithelium can decrease the efficiency of oxygen uptake.

b. Selected Toxicity Benchmarks (DO)

Based on the information above, a water-column DO concentration equal to or greater than 8.0 mg/L is likely to meet the biological requirements of rearing, smoltifying, and migrating subyearling and yearling juveniles and migrating and staging adult salmon and steelhead in receiving waters.

c. Effects of low DO and Oxygen Demand of Effluent in LGR

The BOD and COD of the effluent have a combined effect of decreasing DO in the receiving water. Degradation rates and possibly transport of BOD and COD are different, yet overlap, to cause a range of effects. Upon discharge the diffuser jets force rapid mixing and the low concentration of DO in the effluent requires a draw upon receiving water to reach equilibrium. Then COD, which is expected to at least in part degrade rapidly, demands oxygen from receiving waters within the mixing zone and farther downstream in the plume. A BOD-caused reduction in DO of 0.5-1.2 mg/l may occur in LGR or within the lower Snake River 139 miles downstream of the diffuser in. The BOD degrades more slowly and also demands oxygen from receiving waters (EPA modeling indicated peak demand at Snake RM 40; EPA 2003, Figure VII.11).

The DO concentrations are lowest during summer when Snake River flows are low and ambient temperatures exceed 20ºC. Then DO is 6-7 mg/L and velocity less than 1 ft/s in the Snake River (COE 1999). During summer low flows Potlatch effluent is 0.5-1% of the Snake River volume. It consists of 32ºC water, 5 mg/L concentration of DO, an organic carbon load, and a chemical load. As a result of the Potlatch effluent, it is believed that DO levels of the Snake River will dip even lower. The width and breadth of these effects are not known. However, during summer low flows the 5 mg/L DO and 20ºC toxicity benchmarks described in the BE are not likely to be protective of salmonids in LGR.

The COE review of DO information for LGR and found that low levels occur in various habitats and seasons, especially near substrates and in surface layers of the water column during summer and fall seasons (COE 1999, Appendix C). Levels of DO, as averaged across the entire water column by EPA, are not sensitive to skewed distribution of reduced DO within portions of the channel. Furthermore, diel cycles, shorter-term fluctuations, and patchy distributions of DO are

common within reservoirs (COE 1999) and fish response is often closely linked to daily-minimum DO concentrations. The DO data from Potlatch monitoring studies document that DO sags may occur across the entire width and depth of the Snake River in the confluence area. The low DO ranges in the Potlatch monitoring data show all points sampled (i.e., across all transects at all depths) within the Snake River can be similar on the same days and that these conditions can persist for extended periods during summer (Potlatch 1997, 1998, 1999; Chapman and Hillman 2001).

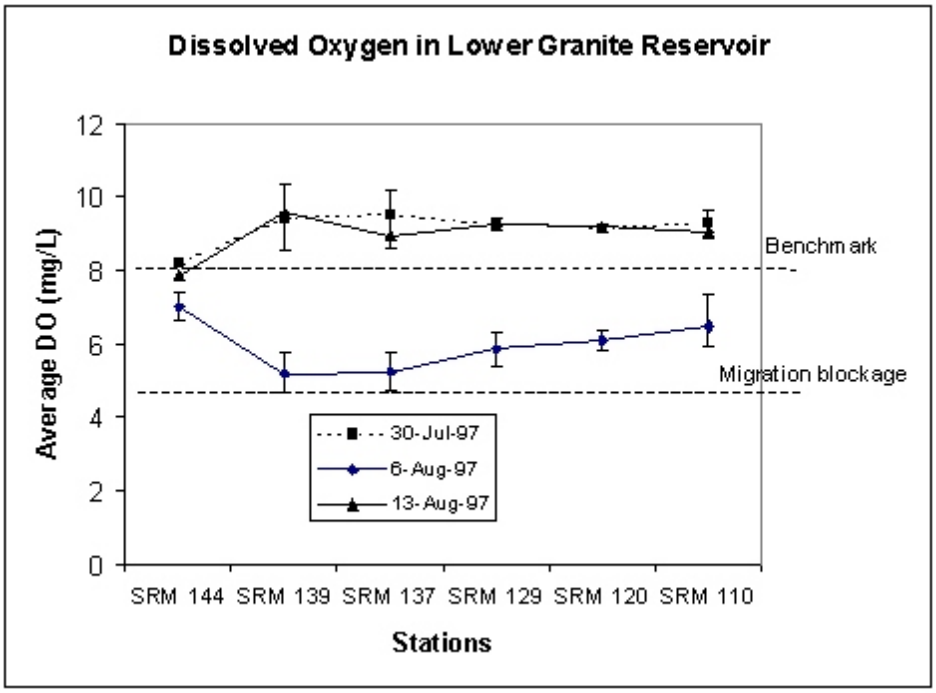


Figure 8. Selected DO longitudinal profiles in LGR. Error bars show data ranges. Data from Chapman and Hillman (2001).

On August 6, 1997, from river miles 139 to 129, DO concentrations from the surface to 6m of LGR ranged from 4.7 to 5.7 mg/L with corresponding temperatures ranging from 19 - 23°C. The DO concentrations averaged over the entire width and depth DO concentrations measured at the monitoring station near the head of the reservoir at river mile 144 on that day were less severe ranging from 6.6 to 7.4 mg/L at 22°C. This low DO episode likely took several days to set up and may have persisted for several days. By August 13, DO values had risen to be mostly at or above the 8 mg/L benchmark (Figure 8). During the August 13 sampling event, DO increased with depth, due to the subduction of the cold, oxygenated "Dworshak augmented" Clearwater River water (see Figure 5).

EPA (2003) estimated the reductions in ambient DO concentrations due to biological oxygen demand in Potlatch's effluents. Under the currently effective permit limits, if Potlatch discharged at their permitted BOD limitations, DO deficits ranging from 0.5 mg/L to 1.5 mg/L in

LGR and further downstream in the Snake River were predicted from June - November. Conditions may become especially serious during a spate of hot, calm weather that could result in the loss of many fish. EPA determined that reductions in these effluent limits were necessary to meet Washington's water quality standards for dissolved oxygen in LGR of 8 mg/L. After the final permit limits are attained, DO reductions in the Snake River due to permitted BOD limits were predicted to be less than 0.2 mg/L below background conditions in the summer. However, the State of Idaho authorized a 5-year compliance schedule to meet the final permit limits that will comply with Washington's DO standards (EPA 2003). In the interim, the current limits remain in effect.

The magnitude of decrease in DO to these habitats from the mill effluent is uncertain because the effects of COD were not modeled and the locations and quantities of TSS settlement are not well documented. Most BOD is thought to be attached to TSS while COD and nutrients are thought to remain as liquids in the effluent. It seems reasonable that many suspended solids (TSS) would settle within LGR as river velocity slows. The BOD and COD material that settles to the bottom of the reservoir demands oxygen from water in and above the sediments. The BOD or COD material remaining in suspension is likely to flow through LGD and be carried downstream and contribute to reduced DO levels as it degrades along the way. As modeled in the BE (EPA 2003), loads of BOD from the mill are estimated to exert peak oxygen demand around Snake RM 40. After compliance with proposed BOD limits, DO levels attributable to Potlatch should approach background levels in the lower Snake River. Increased dilution of the Columbia River reduces the chance of effects farther downstream. A 25% reduction in TSS included as a conservation measure of the permit (section 1.2.4.2) will help reduce the DO problem in LGR and the lower Snake River during and after the compliance period.

The actual DO deficits due to the BOD component of Potlatch's effluents are less than those permitted since Potlatch's actual discharges are less than the maximum permitted effluent limits. EPA modeled deficits predicted from their current and proposed effluent limits for the Potlatch Mill, but they did not predict deficits based on actual discharges. NOAA Fisheries made simple estimates of the deficits resulting from Potlatch's current BOD discharges by assuming that the BOD concentrations in the effluents are directly proportional to modeled in-reservoir deficits. EPA (2003, Figure V-11) modeled deficits based upon average interim (current) permit concentrations of about 86 mg/L and average final permit concentrations of about 15 mg/L. In 2002, the measured average BOD in Potlatch's effluent was 33 mg/L (Table 1). Interpolating EPA's modeled results to the 2002 average BOD concentrations gives a deficit estimate of about 0.2 to 0.6 mg/L. If it is further assumed that the 25% reduction in TSS from 2002 levels will result in a roughly proportional BOD reduction, the "post-conservation measure" reservoir DO deficit from Potlatch's BOD would be on the order of about 0.1 to 0.4 mg/L. This conservation measure is expected to be completed prior to the full attainment of the final BOD permit limits (within 3 years, Appendix 3).

d. Summary of Effluent Oxygen Demand

Low DO levels associated with high temperatures in LGR may occur for weeks during summer and fall and adversely affect juvenile, smolt, and adult salmon and steelhead. Because low DO levels could be the result of a combination of factors, including but not limited to Potlatch's effluent, in order to address DO deficits comprehensively and to allocate reductions fairly among sources, the lower Snake River should have a TMDL for DO.

2.3.2.4 Hydrogen Ion (pH)

The pH is a measure of the concentration (activity) of hydrogen, or hydronium, ions in water. Specifically, pH is the negative log of the hydrogen ion concentration. The pH of natural waters reflects the acid-base equilibrium achieved by various dissolved solids and gases, and is an important factor in the chemical and biological interactions found in waterbodies. On the pH scale of 0 to 14, waters of 0 to 7 are acidic, and waters from 7 to 14 are alkaline. Elevated hydrogen ion concentrations at low pH are directly toxic to fish, causing osmoregulatory problems (NMFS 1999c). Changes in pH also affect the solubility or toxicity of metals such as aluminum, manganese, zinc, copper, and cadmium in the water column and sediments, thereby affecting the exposure dose of metals to aquatic organisms (NMFS 1999c). Aluminum is the metal of greatest concern at low pH values. Un-ionized ammonia, which is directly toxic to aquatic organisms, is a problem at higher pH values. At a given temperature, the higher the pH, the greater the amount of un-ionized ammonia that will be present for a given amount of total ammonia (NMFS 1999c).

Rainwater without anthropogenic acids has a pH generally between 5.0 and 5.6. The buffering capacity of a waterbody is related to alkalinity, a trait that is determined by soil type and parent geology. Waters such as the Snake River with high alkalinity are able to neutralize or buffer a certain amount of acidic inputs. Discharge of water from reservoirs also affects alkalinity in downstream waters. Typically, reservoir water is stored up during spring runoff and has a low alkalinity. Alkalinities are lowest during periods of high surface runoff (winter and spring) and highest during periods when groundwater discharge dominates stream flow (summer and fall) (NMFS 1999c).

There is little species-specific information for pH effects on anadromous fish. Highly acid-sensitive fish species (e.g. fathead minnow and striped bass) may suffer decreased reproductive success. Below pH 6.0, reproductive success of lake trout declines in some waters, and lake and rainbow trout do not survive in the wild in aquatic habitats at pH 5.5 to 5.0 (NMFS 1999c). Davidson (1933, as cited in Heard 1991) reported a kill of pink salmon and other fish in an Alaska stream due to carbon dioxide asphyxiation where pH temporarily dropped to 5.6. Vulnerable life stages of chinook salmon are sensitive to pH below 6.5 and possibly at pH greater than 9.0 (NMFS 1999c).

Considering indirect effect to the salmonid food base, some insect larvae including those of the mayflies, stoneflies, and caddis flies are sensitive to low pH in the range of 5.5 to 6.0 (NMFS 1999c). In the pH range of 6.5 to 6.0, anticipated effects are a small decrease in species richness of phytoplankton, zooplankton, and benthic invertebrate communities resulting from the loss of a few highly acid-sensitive species (NMFS 1999c). No measurable change in community abundance or production is expected outside the mixing zone.

a. Effects of Effluent (pH)

Based on the information above, pH in the lower Clearwater and Snake Rivers outside the range of pH 6.5-9 is harmful to anadromous salmonids. Below 6.5 pH richness and diversity of food organisms decline. Therefore to protect against direct and indirect effects of pH, the benchmarks of 6.5 and 9.0 pH are justified. High alkalinity of the Snake River (pH 8.6) quickly buffers acidic effluent from the mill and the small volume of effluent dilutes rapidly after discharge. Concentrations harmful to listed salmonids are not documented downstream of the mixing zone.

Summary

1. Limits in the draft permit allow the discharge of effluent between pH 5.5 and pH 9.0 into receiving waters. Potlatch effluent is quickly mixed with the alkaline waters of the Snake River (8.6 pH). It is believed that within 10 feet of the diffuser, the effluent reaches a pH of 6.5 or greater.

2.3.2.5 Metals

At EPA's direction, Potlatch has monitored several metals and other inorganics in their effluents and in the receiving waters for several years. Based on their results, EPA concluded that there was no reasonable potential for Potlatch's effluents to exceed water quality criteria and thus no effluent limitations for metals were necessary. The EPA further concluded that discharges of metals were unlikely to adversely affect listed salmonids. Upon review of their data, NOAA Fisheries concurs with EPA's conclusions. Of the metals monitored in Potlatch's effluents, only hexavalent chromium and aluminum ever exceeded benchmarks in effluent. Maximum measured values for both metals exceeded benchmarks by factors of about 3X (Table 1). Since the minimum river to effluent dilution ratios are about 150X, and are considerably higher most of the time, metals from Potlatch's effluents would result in minimal risk to salmonids in the Snake River.

2.3.3 Organic Pollutants: Analysis of individual constituents

2.3.3.1 Adsorbable Organic Halides (AOX)

The AOX is a sum "parameter" measurement often used in waste water testing to indicate the overall level of the halogens, fluorine, chlorine, bromine and iodine. As the name implies, organic halide in water is determined by adsorption onto granular activated carbon, washing the adsorbed sample to remove inorganic halide, combustion of the sample and granular activated carbon to form the hydrogen halide, and coulometric titration. The result is reported as organic chloride (EPA Method 1650). The method is relatively simple and inexpensive, but results give no information on the toxicity or nature of the component organic chlorides. The AOX are a complex mixture of compounds that vary in content and potential toxicity from mill to mill, depending on the raw materials used and the bleaching and treatment processes employed. As a result, there are no studies in the literature that have evaluated the toxicity of AOX to aquatic species. Toxicity data for some of the likely chemical components of AOX (such as chloroform and CPCs) are used in the Opinion to assess the potential effects of those parameters on listed species. However, other compounds comprising AOX have not been identified and, therefore, cannot be assessed. The AOX is limited on about a 10 mg/L or part-per-million level whereas chemical components of AOX that likely contribute to toxicity (CPCs, chloroform, dioxins and furans) collectively account for only a few parts-per-billion of the effluent. Thus, >99% of the mass of organic chlorides allowed in an AOX discharge are unaccounted for on a mass basis. Therefore, analysis of AOX toxicity on the basis of individual compounds would be very difficult if not infeasible.

While evaluation of the toxicity of AOX itself and all individual components of AOX is not possible, it is possible to evaluate the toxicity of the whole effluent (including its AOX component). In addition to the many toxicity studies that have used other mills' BKME, the

potential toxicity of Potlatch's effluent has been studied in several different experimental designs. These experimental designs included experimental stream studies using rainbow trout and sensitive indicator species, quarterly WET tests using sensitive indicator species, and whole life-cycle studies using sensitive indicator species. Toxicity studies using whole effluent provide valuable information about potential synergistic or antagonistic toxicity of the components of Potlatch's effluent on sensitive indicator species. Because these studies examine the toxicity of the effluent that is the subject of this Opinion on species similar to the listed species and on sensitive species, the toxicity data from these studies are appropriate for assessing the toxicity of AOX. Toxicity data for whole effluent are used in lieu of selecting an AOX benchmark.

Average AOX concentrations in Potlatch effluents that were tested in the experimental stream studies ranged from 22,500 µg/L in studies with "pre-conversion" effluents resulting from conventional paper bleaching conducted in the 1980s to 8600 µg/L in studies conducted during the mid-1990's with "post conversion" effluents that reduced the use of elemental chlorine in the bleaching process by substituting a chlorine dioxide bleaching process and oxygen delignification for the elemental chlorine (NCASI 1997). The chlorine dioxide substitution had little influence on AOX concentrations, but the oxygen delignification reduced AOX concentrations substantially. At the target effluent volumes most frequently tested in the experimental stream studies (1.5% to 5% effluent), calculated AOX concentrations would have ranged from about 130 to 1100 µg/L during these tests. In the sections on BKME toxicity, a benchmark of 1.5% effluent was selected as the BKME concentration above which adverse effects to ESA listed salmonids were considered likely. Thus, with the "post oxygen delignification conversion" effluents tested in NCASI (1997), at benchmark concentrations of Potlatch effluents in river water, co-occurring AOX concentrations would have been about 130 µg/L. This calculation was made to relate AOX concentrations to the BKME benchmark, which is not the same as setting a benchmark for the toxicity of the AOX parameter.

2.3.3.2 Bleached Kraft Mill Effluent (BKME) or Whole Effluent Toxicity (WET)

The toxicity of BKME, also referred to as WET, has been studied extensively, especially in Canada and Europe (e.g., see reviews in Servos et al. 1996). This body of work clearly demonstrates that fish populations downstream of BKME discharges may be at risk for adverse health effects, including accumulation of chlorinated organic compounds, metabolic alterations, altered sexual maturation and reproduction, developmental abnormalities, changes in growth rates, and changes and in population structure (Servos et al. 1996). Changes in reproductive steroid hormones, chemical exposure, and chemical metabolism, as indicated by elevated activity of liver mixed-function oxygenases (MFOs) are particularly consistent effects at pulp mill sites (Munkittrick et al. 2002).

The chemicals responsible for the toxicity of BKME have not been fully identified, and the types and concentrations of potential toxicants may vary substantially from mill to mill, depending on the mill's processes and effluent treatment. Traditionally, bleached kraft mills have utilized chlorinated compounds in their bleaching processes, leading to discharge of dioxins and related compounds. Originally, it was believed that they were responsible for the toxic effects of BKME, such as MFO induction and reproductive and developmental abnormalities. More recently, alternative methods have been developed involving chlorine substitution that substantially decrease concentrations of dioxins and furans in mill effluents (EPA 1997; NCASI 1998, 1999). Biologically active compounds may also be removed by oxygen delignification or increased levels of wastewater treatment (e.g. secondary vs. primary treatment (Lippman and Schleisinger, 1979; Dence 1980).

Several studies have documented reductions in effluent toxicity following chlorine dioxide substitution, implementation of oxygen delignification, and increased wastewater treatment (EPA 1997; NCASI 1998, 1999). However, these process changes may not fully eliminate effects on aquatic biota, such as MFO induction or changes in endocrine function (Munkittrick et al. 1994, 1998. 2002). In addition to the dioxins and furans associated with chlorine bleaching, other bioactive compounds in effluents, such as plant sterols and other compounds derived from wood products (e.g., retene) that may not be removed by current treatment processes, are now thought contribute to these adverse health effects on aquatic life (Burnison et al. 1999; Fragoso et al. 1998). Since 1998, 114 field studies conducted on pulp mills in Canada as part of their Environmental Effects Monitoring program have revealed that despite mill modernization and installation of secondary effluent treatment, 90% of the mills still report effects such as alterations in growth, gonad size, and fecundity, MFO induction, and changes in plasma steroid levels in resident fish collected downstream of mill discharges (Munkittrick et al. 2002).

As outlined in the EPA's BE (EPA 2003) the Potlatch Mill currently uses oxygen delignification and has achieved substitution of chlorine dioxide for elemental chlorine in its bleaching process. Additionally, the Mill process water receives secondary biological treatment before being discharged. According to analyses documented in the BE, these process changes have resulted in decreased concentrations of some toxic materials in effluent (NCASI 1997).

Because the composition and toxicity of BKME is quite variable, in their BE EPA (EPA 2003) uses a WET approach to evaluate the toxicity of Potlach effluents. This involves the use of acute (96 hr) and chronic (usually 7-30 day) toxicity tests to measure the toxicity of wastewaters, and is designed to assess and protect against impacts upon water quality caused by the aggregate toxic effect of the discharge of multiple pollutants (Wang 1990). As part of the NPDES permitting process, Potlatch has been required to conduct quarterly chronic WET tests the water flea (*Ceriodaphnia dubia*) and fathead minnow (*Pimephales promelas*), and green alga (*Selanastrum capricornutum*). *Ceriodaphnia* and fathead minnow tests measure impacts on survival, growth, and reproduction. Similarly, the proposed final permit requires Potlatch to conduct quarterly chronic WET testing in the fourth year of the permit term, using water flea (*Ceriodaphnia dubia*), fathead minnow (*Pimephales promelas*), and green alga (*Selanastrum capricornutum*). The proposed final permit also requires bi-weekly toxicity testing if chronic toxicity from quarterly tests exceeds 55 TUc), where TU_c is calculated as the ratio of 100 to the NOEC. In the case of Potlatch's effluent, the NOEC is the highest percent effluent in a sample in which no effects were observed.

The proposed final permit does not establish numeric (i.e. concentration) effluent limitations for WET. There are no national criteria for WET, and 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." 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* (EPA 1997) to develop effluent limits protective of the narrative criteria. These limits are expressed not as concentrations, but as TU, where the TU are defined as the ratio of the exposure concentration to a benchmark concentration, EPA's recommended magnitudes for WET are 1 TU_c and 0.3 TU_a for the chronic and acute criteria, respectively. In accordance with this procedure, in the BE, the EPA established 1 TU_c for WET as a direct toxicity benchmark for listed salmonids and an indirect toxicity benchmark for prey species. The TU_a equals the ratio of undiluted effluent (100% effluent) to the NOEC for chronic effects, as determined through WET testing. Based on their review of the data on the toxicity of Potlatch effluent in both experimental stream and WET tests this level would correspond to concentrations of 0.6% effluent for effects on salmonid fry, 1.5%

effluent for effects on juvenile and adult salmonids, and 3% effluent for effects on the salmonid prey base.

a. BKME Toxicity Based on Studies of Effluents from other mills

In this section we review the current literature on the toxicity of pulp mill effluents to salmonids and their prey, with an emphasis on studies from mills using processes similar to those employed at Potlatch (e.g., chlorine substitution, oxygen delignification, and biological treatment of wastes). Because of the variability in mill conditions and processes, these studies may not accurately reflect of the toxicity of Potlatch effluents. However, they provide general information on the types of effects that might be expected as a result of BKME exposure, and the typical effluent concentration range at which such effects occur in similar mills.

b. Lethal Toxicity to Salmonids

Some data are available on acute toxicity of BKME to salmonids, mostly from older studies with effluents that had not undergone secondary treatment or chlorine substitution. Davis (1973) reports a 36% LC_{50} for underyearling juvenile salmon exposed to BKME in fresh water. Similarly, Whittle and Flood (1977) report 96-hr LC_{50s} ranging from 14-49% for juvenile rainbow trout (Salmo. gairdneri) exposed to BKME from a pulp mills in northern Ontario in static bioassays. Toxicity varied with testing methodology and effluent treatment systems at the mill where samples were collected. In continuous-flow bioassays, LC_{50s} were more consistent, ranging from 21.8 and 24.8%. Robinson et al. 1994 reports LC_{50} values ranging from 54% to 100% for rainbow trout exposed to effluents from Canadian mills using secondary treatment, and from 18-31% for mills using some form, of chlorine substitution, but only primary treatment.

With longer-term exposure, increased mortality may occur at lower BKME concentrations. Owens (1991) reported a no-observable effect levels (NOEL) of 4% for morality in juvenile chinook salmon exposed to BKME for several months. Lower values have been reported for other species exposed to BKME from plants using a chlorine dioxide bleaching process (a form a chlorine substitution). Mortality of adult longnose sucker and whitefish was significantly increased by exposure to concentrations as low as 0.5% BKME (Swanson et al. 1994), while a concentration of 0.8% BKME was associated with significantly increased mortality in fathead minnow (Robinson et al. 1994).

In summary, these studies suggest that acute toxicity is unlikely in salmonids exposed to BKME at 1.5% effluent concentrations. In longer term studies, increased mortality is reported at concentrations below 1.5% in other species, but not in salmonids.

c. Sublethal Toxicity to Salmonids

(1) MFO Induction. Probably the most widely document biochemical effect of BKME on fish is induction of xenobiotic metabolizing enzymes of the cytochrome P450 system, such as the mixed function oxygenase (MFO) enzyme 7-ethoxyresorufin-o-deethylase (EROD) (Hodson 1996). induction itself is not necessarily an adverse health effect, but is a good indicator of exposure to toxicants in BKME at concentrations high enough to elicit a biological response. Moreover,

MFO induction may be correlated with other, potentially harmful, changes physiological function, such as alterations in steroid hormone concentrations and reproductive abnormalities (Munkittrick et al. 2002).

The MFO enzyme system is involved in the metabolism of a variety of toxicants, including dioxins, dioxin-like polychlorinated biphenyls (PCBs), and PAHs, and can be induced not only by TCDD and other dioxin-like compounds in BKME, but by 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 eliminate mill effluent's ability to induce EROD activity (Martel et al. 1994; Munkittrick et al. 1992; Haley et al. 1995). In recent studies, several compounds have been identified that may be 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. 1996), 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 underway at the Potlatch Mill may not eliminate the ability of this effluent to induce MFO enzymes, although this process may reduce the potency of the effluent.

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. For chinook salmon, reported effect thresholds are in the 1-2% range. In one study, a concentration of 1.5% BKME was associated with a three-fold increase in EROD activity in fingerling Chinook, as well as increases in hepatic CYP1A messenger (mRNA) levels, in fingerling Chinook salmon exposed to treated BKME. (Campbell et al. 1996). Similarly, 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 in all but 2% effluent. Hewitt et al. 2003, reporting binding of ligands from pulp mill effluent to the rainbow trout aryl hydrocarbon receptor (AhR) at 1% effluent from a bleached sulfite/groundwood mill in New Brunswick, Canada; similar results were observed for other mills with processes more similar to Potlatch (Hewitt et al. 2000). Wood products derivatives, possibly jubaviones or pinosylvins, were thought to be the most likely active substances.

Even lower threshold concentrations are reported for rainbow trout (e.g., see Servos et al. 1996). Probably the lowest concentration of BKME associated with changes in EROD activity is 0.008%, using effluents from elemental chlorine free and totally chlorine free bleaching processes (Priha 1996). Other studies report MFO induction in at somewhat higher effluent concentrations. Lehtinin et al. 1990 exposed immature rainbow trout to six different BKME in a flow-through system for 7 weeks, at concentrations of 0.25% and 0.05% and measured MFO induction. For the effluent most similar to Potlatch, which was chlorine substituted and received biological treatment, no MFO induction was observed in fish exposed to 0.05% effluent, but fish exposed to 0.25% effluent showed MFO induction. Williams et al. 1996 report threshold concentration for MFO induction in 6 day exposures ranging from 0.57-9.1%, with a threshold of 1.12% for effluent from a plant with oxygen delignification, chlorine substitution, and secondary treatment, similar to the Potlatch Mill.

These data suggest that potential for MFO induction at effluent concentrations in the 0.6-1.5% range.

(2) Growth Changes. Changes in growth have been reported in salmonids exposed to BKME in several studies. There are a few reports of inhibited growth at higher effluent concentrations. For example, Whittle and Flood (1977) reported significantly reduced growth rates for juvenile rainbow trout exposed to 6% BKME under continuous-flow conditions for 18 days, and Webb and Brett (1972) reported reduced growth and food conversion efficiency in underyearling sockeye salmon exposed to 10% BKME for 56 days, with effluents from a mills using conventional bleaching processes (i.e., no chloride replacement of oxygen delignification). However, most studies report either growth enhancement or no growth effects at effluent concentrations in the 1-5% range, which is more similar to the likely concentration range of Potlatch effluents. For example, Hall et al. 1991 monitored the growth of juvenile rainbow trout in large-scale outdoor experimental streams to which biologically treated BKME was added. Results were reported for 9 to 10 month studies at effluent concentrations of 1.3-5.1% as well as for an extended 3.5-year study at 1.5%. Growth rates and average weight tended to be higher in fish from streams receiving effluent. McLeay and Brown (1974) also report increased growth rates and increased length, weight, and condition factor in juvenile coho salmon exposed for 200 days to neutralized, filtered BKME at concentrations of ~2.5% to 6% (.0.1 and 0.25 of the samples' lowest 96-h LC_{50} values of 25%). Servizi et al. 1993 found no effects on growth exposed 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, while Owen (1991) reported a No-Effect Level of 4% BKME for changes in growth of juvenile chinook salmon. Results with other fish species are comparable. For example, Borton et al. 1996 reports significantly increased growth in channel catfish and largemouth bass exposed to 8% chlorine-substituted BKME, compared to controls. Growth was similar to that in control streams at 4% BKME. Similar results (increased growth, condition, and liver size) were reported in wild slimy sculpin and white sucker exposed to effluents from pulp and paper mills in Canada (Galloway et al. 2003), although the effluent concentration was not reported for this study. Growth enhancement most likely occurs because the nitrogen and phosphorus present in BKME cause nutrient enrichment and may increased the available prey base (Owens 1991).

These data suggest that exposure to BKME at concentrations in the 0.6-1.5% range would not be likely to reduce growth of juvenile salmonids, and might even produce slight increases in growth. However, increased growth in BKME-exposed fish is not necessarily associated with improvement in other measurements of fitness. Hall et al. (1991) found increased mortality in BKME-exposed fish in comparison to controls, in spite of their enhanced growth rate. In a study with white sucker, Gagnon et al. 2000 found that BKME-exposed fish experienced increased growth, but showed reduced reproductive effort relative to body size, rather than increased reproductive effort typical of control fish under conditions of increased growth and nutrient enrichment.

(3) Reproductive Toxicity. The reproductive toxicity of BKME to feral fish has been studied extensively in Scandinavia and Canada (Sandstrom et al. 1996). Species examined include white sucker, lake, whitefish, longnose sucker, mountain whitefish, roach, perch, and sunfish. These studies show that depressed reproductive steroid levels, decreased gonad size and increased age to sexual maturation are commonly found in fish from sites in the vicinity of pulp mills. In studies with white sucker in Canada, these effects were reported in field studies where environmental BKME concentrations were in the 0.1-9% range (Munkittrick et al. 1994); effects

on steroid hormones and gonadal growth were observed at effluent concentrations as low as 0.8% for BKME and 1.3% for non-chlorinated effluents. Similarly, Soimasu et al. 1998 reported an LOEC of 3.5% for changes in hormone levels in whitefish exposed to BKME from a plant using chlorine dioxide bleaching. There is little comparable field data on BKME effects on steroid hormone levels and gonadal growth in salmonids. However, at 1% effluent, Hewitt et al. 2003 report binding of ligands in effluent from a bleached sulfite/groundwood mill in New Brunswick to rainbow trout hepatic estrogen and androgen receptor, as well as steroid binding proteins. While this mill uses a different process than Potlatch, these results are similar to those obtained from earlier studies with at bleached kraft mill more similar to Potlatch (Hewitt et al. 2000). The studies suggested the wood extractives or lignins were the most likely sources of the active agents.

There is some evidence of developmental abnormalities in salmonids exposed to BKME. Vuorinen and Vuorinen 1991 exposed rainbow trout to BKME in concentrations of 0, 0.5, 2 and 5% in a continuous-flow system over three different time frames: (1) from fertilization through embryo - alevin phases, (2) from sac fry-alevin phases, and, (3) as alevins only. The 5% BKME dose was lethal to sac fry and alevins, and produced significantly increased mortality in embryos. At 2% BKME, embryos were not affected, but sac fry and alevin mortality increased, with the greatest increases for those that had also been exposed as embryos. The 0.5% BKME dose led to increased mortality of alevins that had been exposed already as embryos and sac fry. However, this effluent may have been more toxic than Potlatch effluent, because it is not clear that is was chlorine-substituted or that it received biological treatment. In other studies with effluents from pulp mills with a variety of different processes, including conventional bleaching and chlorine dioxine subtitution, rainbow trout embryo survival in a short-term (7-day) test was unaffected (Kovacs and Megraw 1996).

Among other fish species, BKME concentrations associated with effects on egg hatchability and larval survival are quite variable (Kovacs and Megraw 1996). In bioassays testing zebrafish egg hatching and larval survival, LOEC for elemental chlorine free effluents ranged from 10-60% for hatching, and from 5-50% for larval survival (Ahtiainen et al. 1996). Brunsvik et al. 1991 (in Kovacs and Megraw 1996) report threshold values of 90-100% for elemental chlorine free effluents with oxygen delignification for zebrafish egg hatching, and threshold values of 45-90% for larval survival. Flink et al. 1994 (in Kovacs and Megraw 1996) reported threshold values of 35% for both egg hatching and larval survival for a elemental chlorine free effluent with oxygen delignification. In short-term (7-day) tests with fathead minnow larvae, reductions survival were seen at effluent concentrations of 16-24% 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 63-75% (Kovacs and Megraw 1996). Borton et al. 1991 tested effluents from mills with and without oxygen delignification and found even less toxicity (IC25 of 82-100%) for effluents from mills with oxygen delignification. However, effects may be more severe with chronic exposure. In whole life cycle tests with fathead minnow, Robinson et al. 1994 reported reduction in egg production and spawning at concentrations of 12% and 25%, while Kovacs et al. reported reductions in both of these parameters at concentrations of 2.5%. Kovacs et al. 1995 also reports changes in male/female sex ratio at effluent concentrations of 5%. Similarly, Larsson et al. 2000 observed male-biased sex ratios in eelpout embryos in the effluent gradient of a large Swedish pulp mill. More recent studies (Alsop et al. 2003) indicate that compounds in pulp mill effluent associated with wood furnish may bind to the retinoic acid receptor, which could potentially lead to developmental abnormalities in fish embryos, problems associated with vitamin deficiency in

adults. However, the level of binding activity appeared to be greater in effluents from mills using thermomechanical processing techniques than the type of processing used at the Potlatch Mill.

The data from studies with other fish species suggest that chronic exposure to BKME at concentrations in the 0.6-1.5% range could potentially affect steroid hormone levels and possibly other reproductive parameters in fish. Additionally, in some studies, effects on egg production, spawning, behavior, sex ratios, egg hatching, and larval survival have been reported at concentrations in the 0.6-1.5% range.

(4) Immune Toxicity. The immunotoxicty of BKME has not been examined in salmonid species, but one recent study shows evidence of immune dysfunction in non-salmonid fish exposed to BKME. 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. In order to study their capability to respond to foreign antigens they were immunized with bovine gamma-globulin before 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. These data suggest that immunotoxicity would be unlikely at effluent concentrations of 0.6-1.5%, but this is uncertain as threshold effluent concentrations for such effects have not been determined.

(5) Genetic Toxicity and Carcinogenicity. Some studies suggest that BKME may have mutagenic or genotoxic properties. Metcalfe et al. (1995) and Rao et al. (1995) tested extracts from BKME for mutagenicity using several different tests, including a Salmonella fluctuation assay, a bacterial genotoxicity assay, an in vivo hepatic micronucleus assay with trout, and an in vitro assay for DNA strand breaks with Chinese hamster lung cells and two mouse liver cell lines, and found evidence of genotoxicity. Less information is available on the concentration of BKME associated with mutagenic activity in live fish. In one study, genetic toxicity was observed by (Easton et al. 1997) in juvenile chinook salmon exposed to 4% BKME. Wilson et al. 2001 report increased levels of hepatic DNA adducts in juvenile chinook salmon exposed for 28 days to 8% and 16% treated effluent from an elemental chlorine free bleached kraft pulp mill.

However, there is little evidence that BKME is sufficiently mutagenic to have very much carcinogenic activity in fish. Metcalfe et al. 1995 tested the carcinogenicity of extracts from BKME in rainbow trout, and found that they did not induce hepatic neoplasms. They did, however, show some promotional activity, increasing the cancer risk in fish previously exposed to known carcinogens. In another study, Hall et al. 1992 used outdoor experimental streams to determine the effects of effluent on histopathology of rainbow trout. No neoplasms or preneoplastic lesions were observed after exposure for as 11 months at a BKME concentration of 5.1% or 42 months at a concentrations of 1.5%. Similar results have been observed in other species. Coullard and Hodson (1996) conducted a 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 preneoplastic or neoplastic lesions in fish near the pulp mill site. These studies suggest that BKME is not a potent mutagen or carcinogen in fish, although it could contribute to increased cancer risk if it is present in combination with more potent carcinogenic chemicals (e.g. high molecular weight PAHs).

There are relatively few reports of other histopathological conditions in salmonids or other fish exposed to BKME. In their study of trout exposed to BKME in experimental streams, Hall et al. 1992 found only parasite-related lesions similar to those found in fish from natural streams in

BKME-exposed fish at concentrations from 1.5-5%. Coullard and Hodson (1996) 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 was not reported in this study.

These data suggest that genetic damage, cancer, or precancerous lesions, and other histopatholgocial conditions are unlikely in salmonids exposed to BKME at concentrations in the 0.6-1.5% range.

(6) Hematological Toxicity. The literature on the hematological toxicity of BKME to salmonids is limited, but most studies suggest that hematological effects are unlikely at effluent concentrations associated with the Potlatch Mill. Hall et al. 1992 exposed rainbow trout to biologically treated 1.3-5.1% BKME for 10-11 months and to 1.5% biologically treated BKME for 42 months in outdoor experimental streams, but found no effects on hematocrit or leucocrit at either concentrations. McCleay and Brown (1974) report that juvenile coho salmon exposed for 200 days to neutralized, filtered BKME at concentrations of 2.5-6% (0.1 and 0.25 of the samples' lowest 96-h LC_{50} value of 25%) showed no changes in total blood cell, erythrocyte, leukocyte-thrombocyte and differential leukocyte counts. Numbers of circulating neutrophils were increased by the higher effluent concentration, but not by the lower effluent concentration. 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%. Hematological changes (e.g., changes in blood hemoglogin and hematocrits) have been observed in other fish species (bluegill, channel catfish, largemouth bass, whitefish) at BKME concentrations of 4-8% (Borton et al. 1996, Lappivaara 2001). These data suggest that hematological changes are unlikely in salmonids exposed to BKME at concentrations of 1.5% or less.

(7) Effects on Stress Response. There are some data to suggest that exposure to BKME may affect the hypothalamic-pituitary-adrenal axis and alter the stress response in fish, including salmonids. McCleay and Brown (1974) report that juvenile coho salmon exposed for 200 days to neutralized, filtered BKME at concentrations of 0.1 and 0.25 of the samples' 96-h LC_{50} values (2.5-6% for the lowest LC_{50} value of 25%) showed some physiological changes similar to those associated with chronic stress response. At the lower effluent concentration responses included changes in serum pyruvic acid:lactic acid ratios and elevated plasma glucose levels, and increases in the liver: muscle glycogen ratio.

More recent 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 six weeks to 4-8% 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 phosphylase 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.

These data suggest that effects on stress response are unlikely at BKME exposure levels of 1.5% and below, but data on salmonid species are too limited to make a clear determination.

d. Effects on Salmonid Prey

Although the number of toxicity studies that have been conducted on salmonid prey species is limited, there is some evidence to suggest that BKME has the potential to harm listed salmonids through indirect effects on their prey base. However, the results are not always simple to interpret because BKME contain both nutrients such as nitrogen and phosphorus that can stimulate productivity of the food webs that support salmon, and contaminants that can have an inhibitory effect on growth and reproductive processes.

e. Lethal Toxicity

Studies on the acute toxicity of BKME to salmonid invertebrate prey species are limited, with most data consisting of bioassays with daphnid species. For daphnids exposed to BKME, LC_{50} concentrations are variable, depending on the characteristics of the particular effluent. Robinson et al. 1994 reports LC_{50} values ranging from 85-100% for mills with some level of chlorine substitution and secondary treatment, and from 20-83% for mills with primary treatment only. Ahtiainen et al. 1996 reports a minimum LC_{50} of 37% was from a study in which Daphnia magna were exposed to effluent from an elemental chlorine free bleaching process. For Ceriodaphnia, O'Conner et al. 1994 reports IC25s (25% decline in survival and reproduction) ranging from 0.7 to 60%. For the effluents with oxygen delignification and chlorine substitution, probably the most similar to Potlatch, the values were in the 40-60% range.

Podemski (2000) assessed the chronic toxicity of BKME to two mayfly taxa (Ameletus subnotatus and Baetis tricaudatus), and found mortality at much lower BKME concentrations. Mortality rates as high as 50% were observed in first instar B. tricaudatus at concentrations as low as 1%, while mortality in older nymphs increased at 7% BKME. A subnotatus also exhibited increased mortality at 1%.

f. Sublethal Toxicity

(1) Growth Alterations. Exposure to BKME has been associated with growth stimulation of both mayflies and chironomids at concentrations as low as 1% (Dube and Culp 1996; Lowell et al. 1996). These results are very similar to those of more recent field and mesocosm studies conducted by Podemski and Culp (1996) Culp et al. (2000) and Podemski (2000) in the Athabasca River, Alberta, Canada. They found that BKME concentrations equivalent to levels in the Athabasca River (~1% 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 maybly species, and invertebrate abundance.

(2) Reproductive toxicity. Data on sublethal effects of BKME on reproduction of salmonid prey species are limited. 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 7 to 100%. Typical values for mills with oxygen delignification and some degree of chlorine substitution are in the 40-70% range, with one of the

lower values being 22% (Kovacs and Megraw 1996). Davis 1978 reports disruption of copulatory behavior in amphipods exposed to 10-15% BKME from a mill using conventional bleaching processes.

(3) Benthic community alterations. Some studies have found changes in benthic community composition in association with exposure to BKME. 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 4% 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.

A recent review of similar studies in Canada (Environment Canada 2003) came to the conclusion that exposure of benthic invertebrate communities to some low effluent concentrations result in a eutrophication response pattern (increases in abundance and taxa richness), likely due to elevated nutrient and organic matter content of pulp and paper mill effluent. Higher levels of effluent exposure lead to reductions in benthic invertebrate abundance and a potential for food reductions for fish preying on benthic invertebrates.

Overall, these data suggest that BKME at a concentrations of 3% has the potential to affect salmonid prey or benthic communities in areas where salmonids feed, but would be more likely to cause nutrient enrichment than toxic effects for most species. However, the available data are very limited.

g. Toxicity of Potlatch Effluent

As noted in the BE, over the past 20 years, many toxicity studies have been conducted using the whole effluent from the Potlatch Mill. These studies fall into four categories. The first category comprises a series of seven experimental stream studies conducted by the National Council of the Paper Industry for Air and Stream Improvement (NCASI) from 1980 to 1994. The second was a field study of adult steelhead migration in relation to Potlatch effluents. The third category included whole life cycle bioassays that exposed all life stages of fathead minnows to whole effluent. The fourth category encompassed 16-quarterly short-term "chronic" regulatory WET tests required by the NPDES permit under which the Mill currently operates.

(1) NCASI Experimental Stream Studies. The primary focus of the NCASI studies was to investigate the potential effect of different concentrations of whole effluent from the Potlatch Mill on the aquatic community living in experimental streams. Over the course of the 14 years that these studies were conducted, the Mill changed its bleaching process to.reduce the amount of chlorinated organic compounds in effluent. Five experimental stream studies were conducted in the 1980s, and are referred to as the pre-conversion studies (NCASI 1982, 1983, 1984, 1985, 1989). A study conducted in 1991, referred to as the post-conversion study, involved effluent

that had undergone chlorine substitution (NCASI 1993). The last study, conducted in 1993 and 1994, referred to as the post-conversion plus oxygen delignification (OD) study, involved effluent that had undergone both chlorine substitution and OD (NCASI 1997). Portions of the research were also published in book and journal articles (Haley et al. 1995, Hall and Haley 1995, Hall et al. 1991, 1992, 1996).

All of the studies monitored the health of rainbow trout, as well as other components of the aquatic community present in the streams during effluent exposure. In addition, in the four pre-conversion studies, early life stages of rainbow trout, coho salmon, chum salmon and chinook salmon were exposed to varying concentrations of effluent. One series of exposures involved burying eggs in the gravel of the experimental streams. The other series of experiments exposed various early life stages of rainbow trout and steelhead, chinook, coho and chum salmon to a range of effluent concentrations. The final 3.5 year pre-conversion study (NCASI 1989), was devoted to assessing full life cycle effects of effluent by allowing for the growth and sexual maturation of rainbow trout initially exposed as juveniles. Spawning was successfully completed in both control and effluent treatment streams and was carried through survival, egg hatching, and juvenile growth. The juvenile growth phase of that study with offspring from parents originally exposed as juveniles marked completion of the rainbow trout life cycle under conditions of continuous exposure to dilute (1.2%) effluent.

(a) Design and challenges interpreting the experimental stream studies:

The experimental streams were constructed to mimic natural stream functions, and were much larger than most mesocosm toxicity test chambers. Four stream channels, each about 110 m long, were constructed on a level grade with eleven alternating pools (1m deep by 3 m wide by 4 m long). In between the pools, riffles were constructed with pebble and cobble substrates. The streams were initially allowed to naturally colonize for 9-months before effluent additions. Each study examined primary productivity, benthic community structure, and fish survival and health. In each study, two streams were used as controls, and the other two for effluent additions. Depending on the study, effluent additions ranged from 1% to 10% on a effluent volume to Clearwater River volume basis (v/v). Each streams flow averaged about 0.6 cfs with complete water turnover in the streams taking about 90 minutes (NCASI 1989).

All the studies except NCASI (1997) were conducted before the "Dworshak augmentation" of the Clearwater River which results in unnaturally cold summertime temperatures in the dilution water (discussed in baseline and temperature sections of this report). Temperatures in the experimental stream studies ranged from 0.0 to 26˚ C in the pre-conversion studies, and 0.0 to 16˚ C in the post-conversion studies. The heated effluent tended to make the treatment streams about 0.4 to 0.7˚ C warmer than control streams in the 5% effluent additions. The potential effects of the somewhat warmer temperatures in the treatment streams were not evaluated in the studies. The DO tended to be slightly depressed in the treatment streams due to effluent effects. Total biological and COD of the effluent resulted in DO being depressed in the treatment streams by about 0.3 to 0.9 mg/L. Despite the depression, mean values were high in all streams, averaging between 9.5 and 13.3 with most values near saturation so DO differences were considered minimal (NCASI 1989, 1997). The streams had to be dredged annually because of sedimentation from the Clearwater River. The TSS from the effluent did not appear to contribute significantly to substrate accumulations but remained in suspension because of its fine to ultrafine particulate size and its primary composition of degraded bacterial cell

walls (T Hall, NCASI Anacortes, WA, personal communication with C Mebane, NOAA Fisheries, Boise).

The NCASI experimental stream study efforts were very ambitious in their use of large test facilities, long exposure duration, and measurements of endpoints that were relevant to stream ecosytems, including fish populations and health. A consequence of this design was a tradeoff between ecological realism and statistical rigor. Statistical rigor of observations was greatly limited by only having two control streams and one or two treatment streams. The large size and long exposures probably contributed to high variability among the control streams. Most statistical comparisons made throughout the study program can be questioned based on pseudoreplication issues, where the "replicates" are not statistically independent of one another, violating assumptions of stastical comparisons. Thus NOAA Fisheries' interpretation of the significance of observed effects reported in the experimental stream studies is based upon judgements of biological meaningfulness of apparent effects, rather than relying solely upon (suspect) conclusions of "statistical significance." Hurlbert (1984), Carpenter (1998), and Schindler (1998) provide good discussions of some of these issues in design and interpretation of poorly or replicated or unreplicated ecosystem studies.

Results of the Experimental Stream studies are summarized below, grouped by lethal effects to salmonids, sublethal effects to salmonids, and benthic community alterations.

(b) Lethal effects of Potlatch effluent to salmonids:

Results of tests of reduced survival due to effluent exposure for either the chlorine-substituted effluent or the effluent that underwent both chlorine substitution and oxygen delignification were variable, over exposure periods of about nine months. Before the studies with "final effluent" (i.e. chloride substitution and oxygen delignification, most similar to present effluents), reduced survival in juvenile rainbow trout was observed in exposures of 1.5% to 5.1% effluent. Two nine-month studies with 1.5% effluent were conducted; in the first survival was similar to controls but in the second survival was lower (Figure 9a). The investigators suggested the lower survival in the second test may have been due to unknown factors other than effluent since earlier tests with higher effluent concentrations had higher survival (NCASI 1989). In the study with "final effluent," (Figure 9b) there were no consistent reductions in survival in the effluent streams compared to the control streams. The tests with post-conversion effluents were unreplicated (NCASI 1997).

In addition to the experimental stream studies, effluent effects to different life stages of salmonids were tested in laboratory tests using a flow-through diluter system to replicate aquaria. The purpose of these tests was to compare traditional acute 96-hour toxicity tests and early-life stage test with results from the long-term, more natural (but less controllable) experimental stream studies. The laboratory tests included more concentrated effluents than the experimental streams, with exposures at 5.6, 10, 18, 32, 56 and 100 percent effluents (NCASI 1984). Selected results are presented in Table 5.

Larval survival studies were conducted with preconversion effluent using instream incubation boxes. While survival and hatching rates were quite variable in both control and treatment streams, results indicated that exposure of eyed rainbow trout, chum salmon, Chinook salmon, and coho salmon eggs during their incubation period effluent

concentrations in the 1-5% range had no effect on their survival to hatching or on the weight of larvae at the time of hatching (NCASI, 1984).

Dilution exposure studies were also conducted on rainbow trout, steelhead, chinook, coho, and chum salmon eggs with pre-conversion effluents (EPA 2003; NCASI 1982, 1984). Acute 96-hour NOEC concentrations (rainbow trout eggs only) ranged from 18-100%, with most results in the 32-56% range. In chronic experiments, NOEC concentrations for survival and growth of steelhead, rainbow trout, chinook, chum, and coho eggs were generally in the 5-10% range, depending on the length of exposure (~30-100 days). The lowest value obtained was a 5.6% LOEC for survival for rainbow trout early life stages with a 51 day exposure, and LOECs of 5.6% for growth for early life stages of chinook and chum salmon with a 106 day exposure.

Table 5. Selected lethality NOEC and LOECs from acute and early-life stage laboratory tests and long-term experimental stream studies. (NCASI 1984; NCASI 1997)

Species	Initial Life Stage	Exposure Time	NOEC (%)	LOEC (%)
Pre-conversion studies				
Rainbow Trout	Eyed egg through fry (a)	51 days	Control (0%)	5.6 %
Rainbow Trout	Fry	21 days	16.0 %	32%
Rainbow Trout	Fry	96 hours	56.0 %	100%
Rainbow Trout	Fry	96 hours	32.0 %	56 %
Chinook Salmon	Fry	96 hours	56.0 %	100%
Chinook Salmon	Fry	96 hours	10.0 %	18%
Rainbow trout	Juveniles	9 months	Control (0%)	1.2%
Rainbow trout	Juveniles	9 months	1.2%	2.2%
Post-conversion studies				
Rainbow trout	Juveniles	9 months	1.5%	5.1%

Figure 9a. Rainbow trout survival in control streams and streams treated with pre-conversion effluents (NCASI 1989).

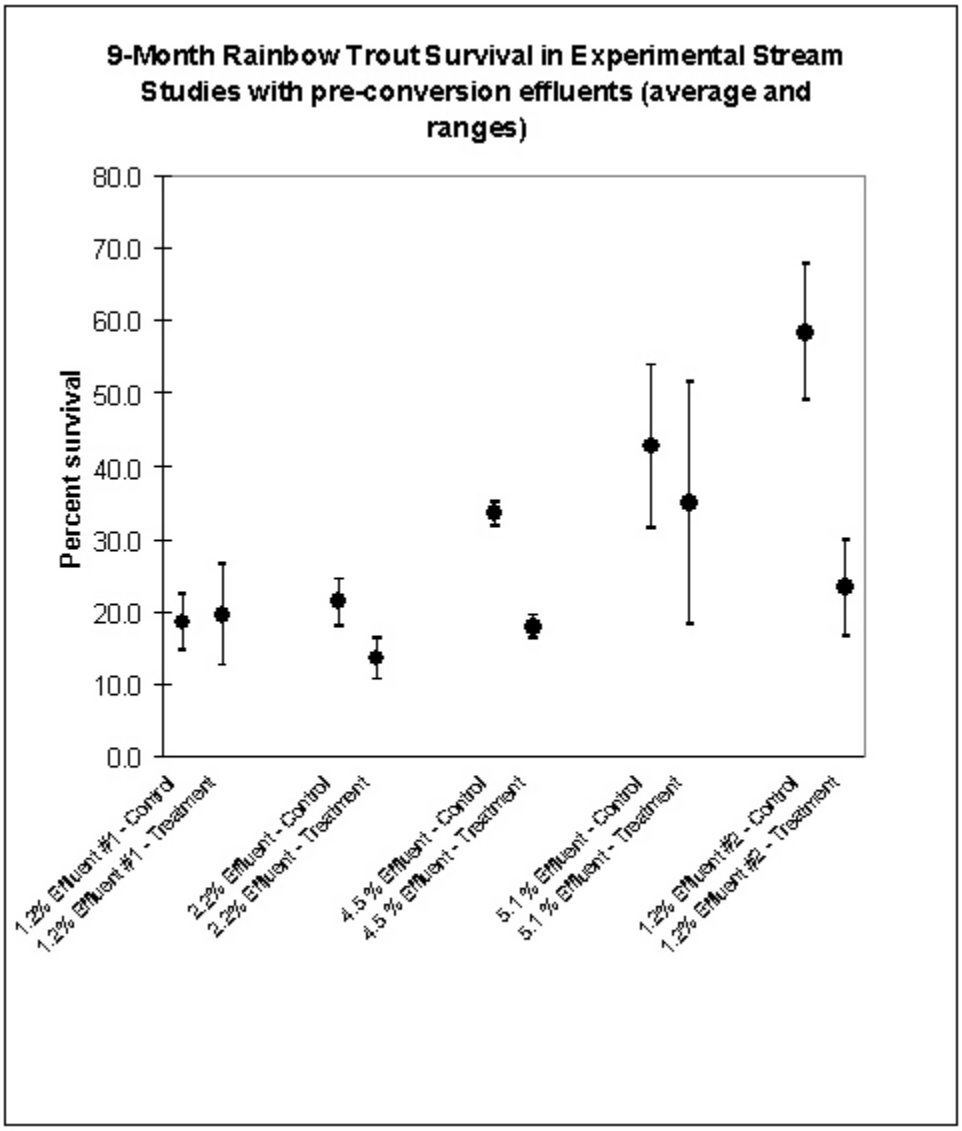
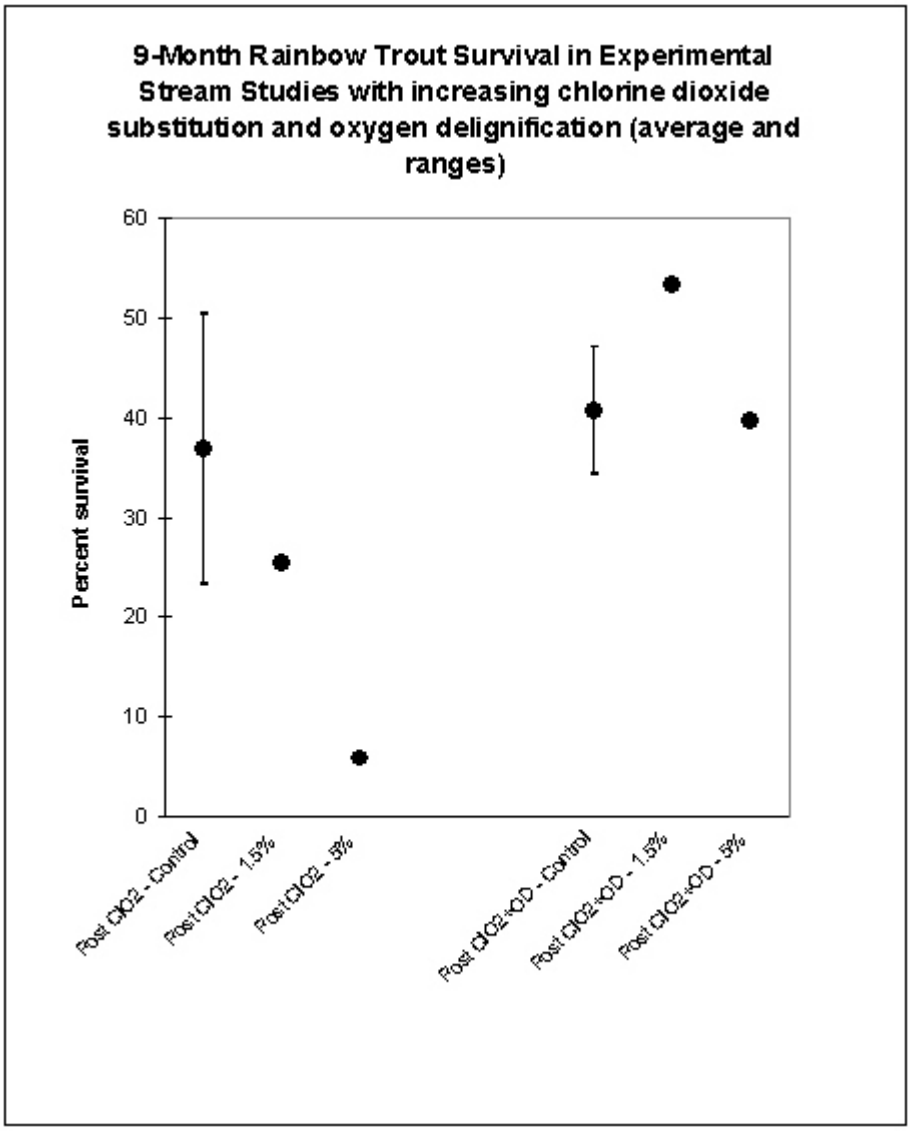


Figure 9b. Rainbow trout survival in control streams and streams treated with post-conversion effluents. (NCASI 1997).



(c) Sublethal effects of Potlatch effluent to salmonids:

A tendency toward increased growth was observed in rainbow trout from the effluent streams containing both 1.5% and 5.0% effluent. With chloride substitution only, trout growth was higher in the 1.5% effluent stream than in control streams, but the 1.5%effluent treatment stream had a production rate that was intermediate between the two control streams. In the 5.0% effluent stream, increases in fish growth, biomass, and condition were more marked than in the 1.5% effluent stream. When oxygen delignification was added, both the 1.5% and 5.0% effluent treatments had significantly higher growth and cumulative trout production than either of the control streams.

Gonadal growth and sex ratios were monitored in rainbow trout from effluent streams containing both 1.5% and 5.0% post-conversion and post-conversion plus DO effluent, and no significant effects were observed.

Rainbow trout exposed to Potlatch effluent at concentrations of 1.5-5% showed increases in MFO activity for both post-conversion and post-conversion plus oxygen delignification studies (Haley et al. 1995). In the 1.5% treatment, MFO activity increased 2-fold, and in the 5% treatment it increased 5-fold. The differences in the 1.5% treatment were not statistically significant, but this may be a reflection of the low power of test and lack of replication in the stream studies.

With chlorine substitution only, some pathological changes were observed in trout from the 5.0% effluent stream (fatty change in the liver and lateral line degeneration), but none in the 1.5% effluent stream.

Hematological parameters (e.g. hematocrit, leucocrit, plasma proteins) were monitored and no differences were found between treatment and control streams for either 1.5% or 5.0% effluent concentrations, for both effluents with chlorine substitution only, and effluents with chlorine substitution and oxygen delignification.

(d) Effects on benthic communities and salmonid foraging due to Potlatch's effluents:

In the stream studies with preconversion effluents, significant detrimental impacts were observed on the periphyton community and macroinvertebrate community structure. Periphyton populations in deeper pool depths was reduced in streams receiving effluents, probably due to reduced light penetrations from the increased turbidity of the effluents. This was observed in both pre-conversion studies and post-conversion studies (NCASI 1989, 1997).

Overall, macroinvertebrate densities and biomass were higher in effluent treated streams than control streams. Overall diversities (taxa richness) were only slightly lower in the effluent treated steams than controls, 34 vs. 38 taxa in the 5% effluent treatment, averaging 29 taxa in controls and 27 in effluent streams over five treatments. In contrast, consistent patterns in relative abundance of several taxa groups were observed (NCASI 1989).

Among the molluscs, clam populations increased in the effluent treated streams compared to the control streams. Snail populations in both control and effluent streams decreased greatly, apparently due in part to pressure on the snail population as a food source for progressively larger sized trout.

An overall trend toward the accumulation of organic sediments in the stream channels might have benefitted the development of clam populations while at the same time been detrimental to snails (NCASI 1989).

Chironomid and mayfly populations were consistently lower in the effluent treated streams than the control streams. Ostracod (seed shrimp), oligochaetes, and gammarid amphipod densities were consistently higher in effluent streams than controls. Differences were particularly pronounced with ostracods and oligochaetes. Caddisfly populations tended to be similar in control and effluent streams (NCASI 1989).

Trout feeding habits in the experimental streams were analyzed by sampling stomach contents . The dominant stomach contents were chironomids, amphipods, and snails (NCASI 1989). The trout were clearly opportunistic feeders, feeding on the dominant macroinvertebrate forms available, which undoubtably influenced why the survival and growth of trout was not depressed in the effluent-influenced streams, despite the depression of some taxa. The exception to that observation is the oligochaetes (tubificid worms). They were not reported in the stomach samples and thus apparently had no food value to the trout despite their increased abundance in effluent-affected streams.

Benthic community shifts apparently were less pronounced under the post-conversion "final effluent" exposures. Chironomid densities were lower in the 5% effluent exposures than the controls, but were much higher in the 1.5% effluent treatments. Amphipod densities showed no pronounced differences between control and effluent streams. Oligochaete densities still appeared to increase in either the 1.5% or 5% effluent streams (NCASI 1997). Unfortunately, the information reported on the benthic communities in the post-conversion studies (NCASI 1997) was much less detailed than that from the pre-conversion studies (NCASI 1989), which hampered comparisons.

(e) Field studies of adult steelhead migration in relation to BKME:

Falter and Ringe (1974) conducted a three-year field study from 1969-1971 to assess the relationship of Potlatch's BKME and pre-impoundment limnological conditions to adult steelhead trout behavior in the Snake River near Lewiston, Idaho. Steelhead were tagged with an ultrasonic transmitter and followed through a 25 km section of the then proposed LGR. Limnological parameters were measured in the Clearwater and Snake Rivers and then compared with fish behavior.

Falter and Ringe (1974) were unable to relate steelhead behavior to pollution inputs in the Lewiston area under free flowing conditions. The maximum concentrations of approximately 2.6% Potlatch Mill effluent occurred in August 1970, about 0.2 km below the confluence. Temperature increases in the river from the pulp mill effluent was undetectable beyond 200m from the outfall. No steelhead avoidance of these areas was noted, nor were there consistent differences in movements above and below the confluences. Steelhead seldom stayed in the mid-channel of the Snake River but frequently crossed from side to side, resting in low velocity areas just off the main

channel. A favored rest area was on the north shore about 2.5 km below the confluence despite the highest total volatile solids being measured there. After mid-September, fish commonly sat for extended periods 100 – 200 m below the Clearwater River mouth.

No significant correlations were observed between any chemical water quality parameter and steelhead behavior. However, as the temperature dropped below 15C fish movement slowed, fish generally stopped moving at night, and resting periods increased in length and number. Steelhead generally showed a preference to move in water with off-bottom current velocities of about 0.2 to 0.5 m/s and showed a definite pattern of crossover and resting points in the river (Falter and Ringe 1974).

1. These results suggest that if migrating adult steelhead were briefly exposed to modern pulp mill effluent under typical dilutions (<1% effluent), no migratory disruption would result.

2. Conversely, if the steelhead trout that overwinter in the immediate vicinity of the confluence fail to avoid areas of high effluent concentrations, they may be repeatedly exposed over a 4- 5 month overwintering period.

(f) Laboratory life cycle tests with fathead minnows:

The most recent NCASI research reviewed on the toxicity of Potlatch's effluents used the "final effluent" (70% chlorine dioxide substitution and oxygen delignification) (NCASI 2000). Of the effluents tested, these were most similar to present effluents (Potlatch presently uses full chlorine dioxide substitution, so present effluents should less toxic than those tested in NCASI (2000)). The study, extending over a 7-month period of time, included the exposure of fathead minnow (*Pimephales promelas*) eggs to 0, 3, 12, 24, 50, and 100% v/v effluent with a determination of growth, survival, spawning success and several fish health biomarkers. Measurements were also made of hatching success of eggs produced at each effluent concentration as well as growth and survival to 28-day post-hatch. Effluent used was also characterized with respect to chemical characteristics, including BOD, COD, AOX, color, CPC and phytosterols, and biological characteristics based on 7-day chronic *Ceriodaphnia dubia* and *P. promelas* freshwater bioassays and echinoderm sperm/egg and bivalve embryo/larval marine bioassays.

No dose-dependent effects were observed in growth or survival of fathead minnows or Ceriodaphnia at even 100% effluent. However, reproduction was reduced or prevented at high concentrations. One of the most sensitive test endpoint was the 48-hour marine mussel embryo-larval development. No similar test protocol is commonly used for freshwater mussels. Thus it is unknown whether the high sensitivity of the bivalve development test is due to effluents being somehow more toxic in marine environments, or whether if similar tests were conducted with native freshwater mussel species (which are in decline in LGR), such as the western floater *Anodonta kennerly,* the freshwater mussels would prove equally sensitive. Among biomarkers, gonad/somatic index in females and hormonal changes (estradiol) were the most sensitive endpoints (Table 6).

Table 6. Selected results of Potlatch's post conversion effluents on marine mussel (*Mytilus edulis*), *Ceriodaphnia dubia*, and fathead minnow (*Pimephales promelas*) in chronic testing (NCASI 2000)

Test and Endpoint		Average concentration causing 25% inhibition in endpoint (IC25)
Bivalve development		2.4
Ceriodaphnia, 7-day		
	Survival	>100
	Reproduction	45
Fathead minnow, 7-day		
	Survival	>100
	Growth	>100
Fathead minnow, 28-day		
	Survival	>100
	Growth	>100
Fathead minnow, 56-day		
	Survival	>100
	Growth	>100
Fathead minnow, life cycle		
	Egg production	
	Egg hatchability	>100
	Adult survival	>100
Biomarkers (fathead minnow life-cycle)		
	Gonad/somatic index: males	17
	Gonad/somatic index: females	5.3
	Testosterone production	64

Test and Endpoint		Average concentration causing 25% inhibition in endpoint (IC25)
	Estradiol production	1.6
	Liver/somatic index: males	20
	Liver enzyme changes (P4501A)	11
	Condition factors	>100
	Histopathology (lesions)	24

(g) Standard Regulatory Whole Effluent Toxicity (WET) tests.

As part of the NPDES permitting process, Potlatch has conducted quarterly chronic toxicity tests evaluating the effects of their effluent on the growth, reproduction, and survival Ceriodaphnia dubia and fathead minnows (Pimephales promelas). The most recent set of WET tests was conducted under Potlatch's current permit, between 1999 and 2002. These required tests consisted of quarterly chronic toxicity tests evaluating the effects of their effluent on the growth, reproduction, and survival 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. Results of these studies showed that with the one exception of the third quarterly 2002 test, no significant differences in survival, growth, or reproduction of *C. dubia* or *P. promelas* were observed at effluent concentrations of 10% or lower in WET tests conducted during 1999 through 2002. In the third quarterly 2002 *C. dubia* test, there was no effect on survival at 100% effluent concentration; however, the number of young (reproduction) were reduced at 10% effluent concentration, resulting in a NOEC of 3% effluent.

(h) Summary of toxicity of Potlatch Effluent:

In summary, toxicity tests with Potlatch effluent that has undergone chlorine substitution and oxygen delignification suggest that, at effluent concentrations of 1.5% and below, the risk of increased mortality and several other sublethal effects (reduced growth, pathological changes in tissues, haematological changes, reduction in gonadal growth) in juvenile and sub-adult salmonids is low. However, some sublethal changes would be likely to occur at concentrations of 1.5% and below, including MFO induction and increases in fish condition factor and production (growth). Additional impacts, such as alterations in steroid hormones or impacts on survival and development of early life stages, and behavioral changes, have not been adequately tested with post-conversion effluents.

Similarly, decreased availability of invertebrate prey is unlikely at effluent concentrations of 3% and below, based on *Ceriodaphnia* bioassays and experimental stream studies. However, nutrient enrichment effects, shifts in composition, and increases in invertebrate prey density and biomass could occur at these effluent concentrations.

The types of effects that have been reported to fish and invertebrates due to BKME exposures are summarized in Table 7.

h. Determination of Whole Effluent Toxicity (WET) and Effects

Because the composition and toxicity of BKME is quite variable, EPA uses a WET approach to evaluate the toxicity of Potlatch effluents. This involves the use of acute (96 hr) and chronic (usually 7-30 day) toxicity tests to measure the toxicity of wastewaters, and is designed to assess and protect against impacts upon water quality caused by the aggregate toxic effect of the discharge of multiple pollutants (Wang 1990). As part of the NPDES permitting process, Potlatch has been required to conduct quarterly chronic WET tests the water flea (*Ceriodaphnia dubia*) and fathead minnow (*Pimephales promelas*), and green alga (*Selanastrum capricornutum*). *Ceriodaphnia* and fathead minnow tests measure impacts on survival, growth, and reproduction. Similarly, the proposed final permit requires Potlatch to conduct quarterly chronic WET testing in the fourth year of the permit term, using water flea (*Ceriodaphnia dubia*) and fathead minnow (*Pimephales promelas*). The proposed final permit also requires bi-weekly toxicity testing if chronic toxicity from quarterly tests exceeds 55 chronic TU_c), where TU_c is calculated as the ratio of 100 to the NOEC. In the case of Potlatch's effluent, the NOEC is the highest percent effluent in a sample in which no effects were observed.

The proposed final permit does not establish numeric (i.e. concentration) effluent limitations for WET. There are no national criteria for WET, and 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." 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* (EPA, 1997) to develop effluent limits protective of the narrative criteria. The EPA's recommended magnitudes for WET are 1 TU_c and 0.3 TU_a for the chronic and acute criteria. The 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. LC_{50}). By EPA's (2003) definition in the BE, TU_c is equal to the ratio of undiluted effluent (100% effluent) to the IC25 (where 25% of the organisms are affected) determined through WET testing. The TU_a equals the ratio of undiluted effluent (100% effluent) to the LC_{50}, the concentration that results in 50% mortality as determined through WET testing.

In the BE, the EPA established 1 TU_c for WET as a direct toxicity benchmark for 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. The rationale for this approach is that WET is facility-specific so no data from literature can help to evaluate the level of protection provided by the permit limits. By definition, this would be the concentration at which 25% of the test organisms are affected; however, in their analysis in the BE, which is summarized below, EPA appears to be using actual NOEC values rather than the EC_{25} values to assess the current toxicity of Potlatch effluents (e.g. they propose a benchmark of 3% for effects on prey based on WET tests with *Ceriodaphnia*, which was lowest NOEC in the series of tests conducted by Potlatch from 1999-2002. However, the EC25 for this series of tests was 8.3%).

In their 2003 BE, EPA reviewed the data on the toxicity of Potlatch effluent in both experimental stream and WET tests to determine whether current discharges would be likely to adversely affect listed salmonids and their prey. The types of tests reviewed included rainbow trout in experimental stream studies, fathead minnow and daphnid life cycle tests, and fathead minnow and daphnids in WET tests. They identified the following NOEC/LOEC:

> An LOEC of 5.6%effluent for early life cycle stages, based on reduced growth of rainbow trout fry in dilution experiments with pre-conversion effluents. They converted the early life stage LOEC to a NOEC of 0.6%(166 TU_c), by applying a safety factor of 10 to the LOEC.
>
> An NOEC of 1.5% effluent (67 TU_c) for juvenile and adult life stages, based on results of the artificial stream studies
>
> An NOEC of 3% effluent (33 TU_c) for indirect toxicity effects based on impacts on *Cerodaphnia* reproduction in WET tests. The BE also lists a NOEC of 10% effluent (10 TU_c) for indirect effects on all life stages, however, no clear basis for it was found in our review of the BE document and it was not used in EPA's subsequent analyses of effects.

Because EPA established a water column toxicity benchmark of 1 TU_c for Potlatch effluent, these results mean that the toxicity of the whole effluent (166 TU_c for direct effects on salmonid fry, 67 TU_c for direct effects on juveniles and adult and 33 TU_c for indirect effects on the salmonid prey base) is greater than the water column toxicity benchmark (1 TU_c). Consequently, they then used a computer simulation model (CORMIX v4.2GT) to estimated the area where the concentration of the effluent plume and of the receiving waters would exceed the toxicity benchmark. In contrast to the approach used elsewhere in the BE of predicting effects based on the lowest bencharks, EPA elected to model the results based upon the higher of the two direct effects benchmark (1.5% effluent). No explanation for this inconsistency was presented and we thus presume it was an oversight. The model was run until the receiving water concentration was below the level safe for threatened and endangered species (1 TU_c). The model predicts that the water column concentration that is safe for juvenile and adult threatened and endangered salmonids from WET exposure of 67 TU_c is met within 24,426 meters (15.2 miles). The modeled effects would extend even further downstream on fry if the exposure level of 166 TU_c for this lifestage were used in the model. The EPA used the same simulation model to simulate toxicity to the salmonid prey base, using a WET exposure level of 33 TU. The model predicts that the water column concentration that protective against effects on the salmonid prey base (WET exposure of 33 TU_c) is met within 29.24 meters (96 feet). The EPA's modeling used scenarios based upon the 1Q10 flows, that is, the lowest daily average flow predicted in a 10-year recurrence interval. Since most effects observed with BKME testing developed over longer, chronic exposures, NOAA Fisheries is primarily basing our estimation of the extent of BKME by modeling longer-duration minimum flow scenarios (lowest weekly and monthly average flows observed in the last 10-years). These results are shown in later in section V.D.

On the basis of these analyses, EPA concluded that WET is likely to adversely affect listed salmonids up to 15.2 miles beyond the discharge, and to affect their prey base up to 96 feet beyond the discharge. Outside of these ranges, they conclude that discharges are not likely to salmon or their prey.

We agree with EPA's determination that Potlatch effluent is likely to adversely affect salmonids and their prey. However, based upon our review of the available information, NOAA Fisheries is selecting a 1.5% effluent concentration as the NOEC benchmark for both direct and indirect effects to salmonids. Our reasons for this conclusion included the following:

- The EPA's direct effects benchmark of 0.6% effluent was extrapolated from a egg-through-fry early life stage test of rainbow trout with pre-conversion effluent. This test ran for 51 days starting with eyed eggs through about 18 days after fry swim up. However, no salmon or steelhead spawning is believed to occur in LGR and NOAA Fisheries thinks it unlikely that steelhead trout fry that are only about 18-days post hatch to have traveled from their natal streams to LGR. Thus since the egg to early fry lifestages are unlikely to occur, this benchmark is not used. Clearly detrimental effects to older fry were only observed at higher concentrations (Table 5).

- While the 1.5% benchmark would protect adult and juvenile salmon from clearly detrimental effects of BKME, the data suggest that some sublethal effects could occur at concentrations below this level.

- There was a 2-fold increase in MFO induction in both pre- and post-conversion artificial stream tests with 1.5% Potlatch effluent. The fact that a difference of this magnitude was not statistically significant may indicate the low power of test in these studies because of the lack of replication and small sample size. While MFO induction in itself may not be a cause for concern, data on other health effects to salmonids that are often associated with EROD induction (alterations in plasma steroid hormones, reduced gonadal growth, and egg and larval viability) are limited for Potlatch effluents. (Also note that a reduction in estradiol was observed in fathead minnow in the WET tests at 1.6%.)

- Shifts in periphyton and benthic communities and food web stimulation were observed in artificial stream studies at 1.5% post-conversion effluent.

- Some potential BKME effects have not been tested with Potlatch effluent such as immunocompetence. Also, the toxicity of effluent to returning adult salmon has not been established. These animals are in poor physical condition and may be especially sensitive to toxicant effects. This could contribute to increased prespawn mortality, a phenomenon observed in a number of urban streams.

- The toxicity benchmarks for salmonids are very heavily based on toxicity tests from Potlatch's effluent, which consist of a series of artificial steam studies with pre-conversion effluents, as well as post-conversion effluents with chlorine substitution and chlorine substituted/oxygen delignification. While the information from these studies is valuable, it should be noted that this data set includes only one study with effluent undergoing current processes, and no studies on early life stages with post-conversion effluent. NOAA Fisheries is making a critical assumption that the post-conversion effluent is no more or less toxic than the pre-conversion toxicity. We think that is a reasonable assumption for two reasons, (1) several measured classes of pollutants declined from 38 - 90% between the pre-conversion and post conversion periods (including resin acids, CPC, and TSS), and (2) fewer effects were noted in the post-conversion study than in the pre-conversion studies (Haley et al. 1995, NCASI 1997).

- NOAA Fisheries' toxicity benchmark for effects on salmonid prey is based upon the experimental stream studies which showed some benthic community alteration in the most dilute treatements tested (1.5% effluent). It could be argued that any biological

effects in response to effluents are detrimental. However, since the benthic community effects were not obviously detrimental to salmonids in the experimental stream studies, we surmise that if similar effects were to occur in the receiving waters, any adverse effects on listed salmonids due to Potlatch's effluents at lower concentrations would likely be slight.

These areas of uncertainty point out the need for regular effluent testing, as well as environmental effects monitoring in salmonids and other resident species in the area likely to be impacted by the Potlatch effluent plume.

Table 7. Summary of likely effects on fish and invertebrates that may be caused by BKME at a range of concentrations in rivers.

Concentration (% effluent)	~TU (using 1.5% as 1 TU)	Likely effects
100%	67	Risk of acute mortality for all salmon life stages and prey
		Increased chronic mortality for all salmon life stages and prey
		MFO induction
		Growth reduction
		Reduce gonadal growth, altered steroid levels, reduced fecundity
		Developmental abnormalities and larval mortality
		Immunosuppression, reduced disease resistance
		DNA damage
		Reduced hematocrit and leucocrit, other hematological changes
		Physiological changes associated with chronic stress response; reduced capacity to respond to additional acute stressors
		Growth and reproductive impairment in prey
		Alterations in benthic community
50%	33	Risk of acute mortality for all salmon life stages and prey
		Increased chronic mortality for all salmon life stages and prey
		MFO induction
		Growth reduction

Concentration (% effluent)	~TU (using 1.5% as 1 TU)	Likely effects
		Reduce gonadal growth, altered steroid levels, reduced fecundity
		Developmental abnormalities and larval mortality
		Immunosuppression, reduced disease resistance
		DNA damage
		Reduced hematocrit and leucocrit, other hematological changes
		Physiological changes associated with chronic stress response; reduced capacity to respond to additional acute stressors
		Growth and reproductive impairment in prey
		Alterations in benthic community
25%	17	Immunosuppression, reduced disease resistance
		Reduce gonadal growth, altered steroid levels, reduced fecundity
		Physiological changes associated with chronic stress response; reduced capacity to respond to additional acute stressors
		Developmental abnormalities and larval mortality
		Reduced hematocrit and leucocrit, other hematological changes
		Reduced growth of juvenile and fry
		Growth and reproductive impairment in prey
10%	7	Threshold for increased lethality in acute (96-hour) exposures to salmonid fry

Concentration (% effluent)	~TU (using 1.5% as 1 TU)	Likely effects
		Developmental abnormalities and larval mortality
		Reduced growth of juvenile and fry
		Reduce gonadal growth, altered steroid levels, reduced fecundity
		DNA damage
		Physiological changes associated with chronic stress response; reduced capacity to respond to additional acute stressors
		Increased long-term mortality
		Changes in benthic community composition
		Possible reproductive impacts on prey, but not commonly observed
5%	3	MFO induction
		Increased growth or condition in juveniles and fry
		Increased productivity of benthic community
		Altered reproductive hormones levels in studies with juvenile trout; no gonad growth effects in artificial stream studies
		Pathological changes, e.g. fatty change in liver/lateral line degeneration (NCASI)
1.50%	1	MFO liver enzyme induction
		Altered reproductive hormones levels in life-cycle studies with fathead minnows Possible increased growth or condition in juveniles and fry

Concentration (% effluent)	~TU (using 1.5% as 1 TU)	Likely effects
1.50% (continued)	1 (continued)	Increased abundance of overall benthic invertebrate densities Possible shifts in benthic community composition, with increased dominance of oligochaetes and chironomids

2.3.3.3 Chloroform

The following discussion of chloroform effects is taken directly from EPA's (2003) BE of chloroform. NOAA Fisheries concurs with that analysis, and other than minor editing, has repeated EPA's (2003) analysis on chloroform verbatim here. See EPA (2003) for references from the chloroform section.

a. Introduction

Chloroform ($CHCl_3$), also known as trichloromethane or methyltrichloride, 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 x 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.

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 readily dissolves in water and dissipates through aeration. The volatilization half-life for chloroform is 36-hours; therefore, half the chloroform added to the system from the discharge would be volatilized within 16 miles (assuming a river flow rate of 0.2 m/s), which is half the distance to the LGD. 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 0.462 per day for chloroform.

b. Environmental Baseline

Chloroform has not been tested for in either the Snake or the Clearwater Rivers in the Action Area; therefore, the environmental baseline is unknown. Since there are other sources (such as sewage treatment plants and air deposition) above the Action Area that would contribute to the background concentration of chloroform in the receiving water, there is likely some chloroform in the background. The proposed final permit includes monitoring of chloroform in the receiving

water body to establish the levels of chloroform in the waterbody. In order to account for the uncertainty of the environmental baseline, this assessment assumes that the environmental baseline is 2.9 µg/L, which is one-half the water quality criterion set by the states of Idaho and Washington as protection of human health. Therefore, the environmental baseline is below levels that are considered safe for threatened and endangered salmonids. 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.

c. Water Quality Standard

The most stringent water quality standard in Idaho and Washington for chloroform is 5.7 ug/l as a long-term average for the protection of human health.

d. Effluent Limitation

Any chloroform found in bleach plant effluent that is not emitted to the air before 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 TSS in the effluent.

The proposed final permit requires 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 effluent due to the fiberline limitations and a fiberline (14.6 mgd[1]) to effluent (40 mgd) flow ratio of 0.365 would be 86.3 µg/L and 51.5 µg/L, respectively.

e. Toxicity Benchmarks

Salmonid Species

(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 under estimate the toxicity because concentrations in static tests are likely to diminish during the progress of the exposure as a result of loss from water to air.

Several studies have measured the concentration of chloroform required to cause mortality in aquatic organisms. The range of LC_{50s} (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).

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.

[1] This flow rate is the combined flow rate of both fiber lines.

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 LC_{50} value reported by Birge and Black (1979). The behavioral endpoint reported by Birge et al. (1993). 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 resulted in 12.4 µg/L as the direct water column toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead.

(2) Indirect Effects. Nonsalmonid fish have been found to have chloroform LC_{50s} 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 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).

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_{50s} 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 311,000µg/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 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. 27 µg/L is used as an indirect toxicity benchmark for prey species.

(3) Direct Effects. In order to evaluate the effects within the exposure volume, EPA (2003) has used a computer simulation model (CORMIX v4.2GT) to simulate the effluent chloroform plume in the Snake River. The model was run until the receiving water concentration was below the level safe for threatened and endangered species (12.4 µg /L).

The modeling predicted that concentrations would be diluted to meet the benchmarks within about 2 meters downstream of the diffuser (EPA 2003, Table VII-1). The model predicted that the water column concentration that is safe to threatened and endangered salmonids from direct exposure of chloroform (12.4 µg /L) is met within 1.95 meters (6.4 feet) when a background concentration of 2.9 µg/L is assumed. The modeling predicted that the velocity of the plume as a function of distance downstream drops below 5 ft/s. at approximately 5 meters downstream. Because the maximum swimming speed of salmon and steelhead is 5 ft/s or less, it is a safe assumption that the fish will not be exposed to chloroform concentrations within 5 feet of the diffuser because they will not be able to swim into the plume when the plume velocity is higher than their maximum swim speed. However, it is improbable that the fish would be exposed to chloroform within 10 feet of the diffuser because they would tend to avoid the currents created by the jet action of the plume.

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 two times lower than the toxicity benchmark (12.4 µg /L), it is not likely that threatened or endangered salmonids would be exposed to unsafe levels of chloroform.

Therefore, NOAA Fisheries concludes that the discharge of this compound at the maximum effluent concentration may directly affect, but is not likely to adversely affect Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead.

(4) Indirect Effects. As predicted by the model, the indirect toxicity benchmark will be met within the jet action of the plume; therefore, it is unlikely that prey species would be exposed to unsafe levels of chloroform.

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 this compound may indirectly affect, but is not likely to adversely affect Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead.

2.3.3.4 Dioxins and Furans

The polychlorinated dibenzo-*p*-dioxins (PCDDs) and the closely related polychlorinated dibenzofurans (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, incineration plants, domestic fires, motor vehicle exhausts, and disposal of municipal sewage treatment plant sludge. Thus, they are broadly dispersed in the atmosphere, soil, and water on a global scale. The high hydrophobicity

and lipophilicity of dioxins accounts for their very low solubility in water, though they do adsorb to organic material in the particulate and dissolved phases, and readily bioaccumulate. Also, PCDD/Fs are immobile once they become incorporated into sediment, where they are highly resistant to environmental and biologic degradation, and persist for decades (reviewed in Fletcher and McKay 1993; Clarke et al. 1996).

Of the 235 possible PCDD/F isomers, the ones possessing four to six chlorine atoms have the highest potential for adverse biological activity. The 17 most toxic isomers all include chlorine substitutions in the 2,3,7,8 positions -- with TCDD being the most toxic (Fletcher and McKay 1993; Feeley 1995).

To facilitate toxicological comparisons, a toxicity equivalent factor (TEF) was established for the 17 toxic PCDD/F isomers, wherein TCDD was assigned a TEF of 1.0. When the TEF is multiplied by the isomer concentration, a toxicity equivalent concentration (TEC) is obtained. When assessing total exposure to mixtures of dioxins (such as 2,3,7,8-TCDD and 2,3,7,8-TCDF) or other contaminants that have a similar mechanism of action through the AhR (such as PCBs), the TEC for each component in the mixture is summed to obtain a TEQ. In spite of some observed nonadditive interactions in mixtures, it has been suggested that an additive model still seems most plausible in light of studies with environmentally derived mixtures (Van den Berg et al. 1998).

Dioxins mediate their toxic effects through a highly specific, high affinity binding interaction with a cellular protein called the AhR. The AhR exists in a multiprotein cytosolic complex, which, upon binding with dioxin, undergoes a series of changes which ultimately leads to the activation of various genes, including CYP1A1 and CYP1A2 (reviewed in Grassman et al. 1998). In fish, adverse biological responses such as reduced survival, reduced growth, reproductive toxicity (i.e. reduced development of secondary sexual characteristics), developmental toxicity, immunotoxicity, behavioral effects (i.e. erratic or listless swimming), histopathological effects (i.e. liver necrosis), biochemical effects (i.e. presence and functionality of the AhR, Phase I and II enzyme induction), fin disease, hemorrhaging, edema, and epithelial lesions have been reported due to exposure to dioxins (Fletcher and McKay 1993; Walker and Peterson 1994a; Boening 1998).

a. Concentrations of dioxins and furans in Snake River/LGR environment:

No data have been located on waterborne concentrations of dioxins or furans in the Snake River/Lower Granite reservoir area. Dioxins and furans have been measured in sediments by the COE (2002) and in a program of annual monitoring by the Potlatch Corporation of several sites up and down gradient of the outfall (CH2M Hill 1997, 1998, 1999, 2000, 2001, 2002). Since the monitoring by Potlatch was intended to evaluate spatial and temporal trends in constituents of concern, those data are discussed here.

Similarly to the toxicological comparisons, these sediment data are illustrated using a TEQ approach for total toxic PCDD isomers, which is based on the specific fish TCDD toxic equivalency factor (TEF) for each individual isomer (Van den Berg et al 1998). This is done to facilitate convenient approximations or comparisons of relative total dioxin concentrations in sediments between the sites. There are two important caveats. One is that this comparison assumes that the total organic carbon content of the sediments are roughly similar at the different sites. Nonetheless, such extrapolations of fish tissue-based TEFs to estimate TEQs in different media such as water or sediment are commonly made (e.g. EPA 2000, AMEC 2003).

The second caveat is that, while these relative comparisons between otherwise similar sites may be convenient, it is not appropriate to compare them to the absolute values for sediment toxicological benchmarks. That is because the TEQ approach accounts for toxicity, but does not account for the fact that the different dioxin isomers may have different bioavailability from the sediments. Acknowledging this limitation, alternative approaches for illustrating dioxin contamination in sediments such as ignoring all chlorinated dioxins other than TCDD, or assuming equal toxicity are rejected. Figure 10 shows means and standard deviations of the seven dioxin isomers at several stations in LGR above and below the Potlatch outfall. These were calculated from the data summaries in EPA (2003 Tables 16 and 17) setting non-detected values equal to the detection limits and using "less than" TEFs as "equal to" TEFs. The data clearly illustrate a concentration gradient of dioxin congeners in sediment from the Snake River as the sampling sites migrate downstream from the Potlatch outfall, with large variation observed at RM 137, which is the closest station to the diffuser.

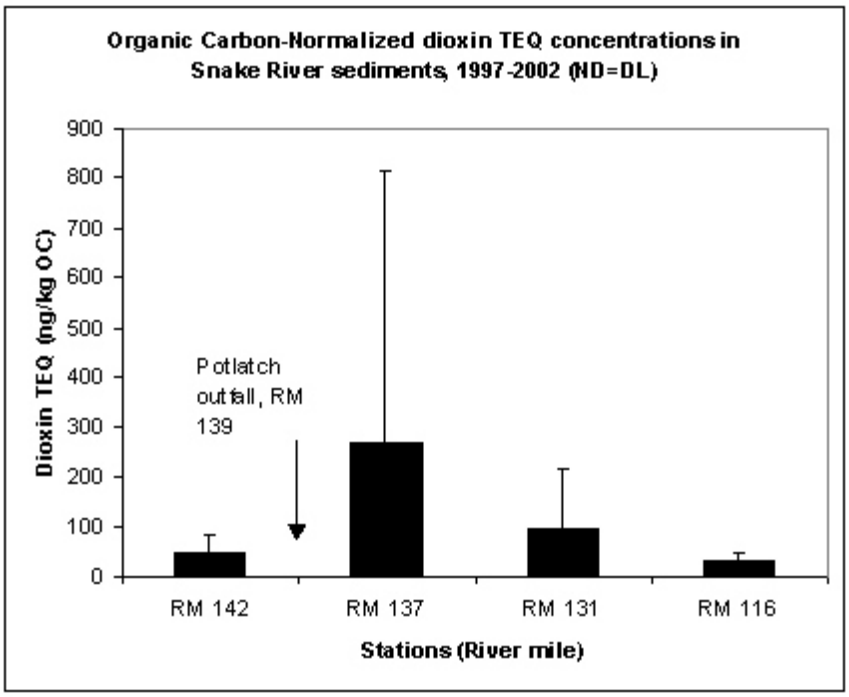


Figure 10. Mean and standard deviation (error bars) dioxin concentrations at selected sediment sampling sites above and below Potlatch's outfall.

b. Direct Lethal Effects of TCDD to Salmonids:

For larger fish species, the major route of exposure to dioxins and related compounds is through the diet, rather than bioconcentration via direct uptake of chemicals from water across the gill membrane. Laboratory toxicological studies conducted on salmonids encompass both routes of exposure. Also, environmental exposure is usually chronic, as is the toxicological mechanism of TCDD (Batterman et al. 1989; Fletcher and McKay 1993; Jones et al. 1993; Giesy et al. 2002). 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; Guiney et al. 1996). Presented below are the lowest tissue or water-born exposure concentrations of TCDD that have been reported to cause adverse effects in salmonids. Although toxicological information was not available for the specific salmonid species identified in the NPDES permit for the Potlatch Corporation, information on the various life stages of other salmonid species (such as coho salmon, lake herring, and rainbow, lake and brook trout) was available and is discussed below.

Information on the lethal (and sublethal) effects of TCDD on adult salmonids and their eggs includes a recent report by Giesy et al. (2002), in which female rainbow trout (*Onchorynchus mykiss*) were fed TCDD-spiked food every third day, with an average daily dietary dose of 1.8 ng TCDD/kg (w/w) food (equivalent to an actual daily intake of ~0.027 ng TCDD/kg body wt. per day). The Lowest Observed Effect Concentration (LOEC) for increased mortality after 224 days of exposure, as well as decreased survival of spawned eggs, was 1 ng TCDD/kg whole body wet wt. or 0.21 ng/kg in muscle in adults. However, high background TCDD equivalents in the feed make interpretation of results difficult, and therefore this study will not be used to evaluate tissue benchmarks.

Progressing to the next youngest life stage, Miller et al. (1973) exposed juvenile coho salmon (*Onchorhynchus kisutch*) for 24 hours to 0.056 ng TCDD/L in a water-born exposure study, which resulted in reduced survival over sixty days. Additional studies on juvenile salmonids report adverse effects at somewhat higher water concentrations. For example, increased mortality was observed within 10 days in juvenile rainbow trout exposed for 96 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 ug TCDD/kg (Hawkes and Norris 1977; Kleeman et al. 1988; van der Weiden et al. 1990).

For lake trout swim-up fry (*Salvelinus namaycush*), 50% mortality (LD_{50}) was noted by Walker et al. (1991) at a whole body tissue concentration of 300 ng TCDD/kg. Somewhat lower sensitivity was reported by Mehrle et al. (1988) for rainbow trout swim-up fry, in which water-born exposure to 0.038 ng TCDD/L for 28 days, followed by an additional 28 day depuration period, resulted in increased mortality with a corresponding whole body TCDD concentration of 740 ng/kg. Although tissue concentrations were not reported, additional studies with higher water concentrations have included a 96 hour exposure of rainbow trout yolk sac fry to 1 or 10 ng/L TCDD, causing reduced survival, generalized edemas, growth retardation, degeneration and necrosis of liver parenchymal cells, foreshortened maxillas, and opercular defects (Helder 1981; Eisler 1986).

Salmonid eggs are also quite sensitive to TCDD exposure. Helder (1981) exposed rainbow trout eggs for 96 hours to 1 ng/L TCDD, 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 1994b; 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 - 31 ng/L or whole body tissue concentrations of 55 - 270 ng/kg, after exposure times of 20 minutes to 48 hours.

In summary, TCDD tissue concentrations which cause increased mortality in salmonids are 40 ng/kg in eggs, or range from 300 ng/kg in sac fry to 740 ng/kg in swim-up fry. The lowest TCDD concentrations in water that caused increased mortality ranged from 0.038 ng/L for rainbow trout swim-up fry to 10 ng/L for lake trout eggs, with intermediate values for adults, juveniles, and sac fry. Therefore, based on the best available toxicological information, the concentrations for lethal effects in salmonids are 40 ng/kg in eggs, 300 ng/kg in sac fry, or 0.038 ng/L aqueous concentration.

Various studies on salmonids have focused on maternal transfer of TCDD to eggs. A EPA Interim Report (1993, and references therein) indicates that in lake trout, TCDD concentrations in eggs were approximately 50% of the TCDD whole body concentration in the maternal adult fish. Similar analyses in brook trout indicated that TCDD concentrations in eggs were approximately 39% of those in the maternal animal (Tietge et al. 1998). Using these percent values, one can estimate that a lethal concentration in eggs of 40 ng/kg would translate into a whole body maternal tissue concentration of 80 – 102 ng/kg.

c. Direct Lethal Effects of TCDF to Salmonids:

Toxicity information on TCDF is more limited than for TCDD. A water-born exposure study was reported by Mehrle et al. (1988) in which rainbow trout swim-up fry were exposed to TCDF in a flow-through experiment with measured concentrations of contaminant. They reported a significant increase in mortality after seven days of exposure to 3.93 ng/L TCDF. Increased mortality was still evident at 28 days of exposure, and at the end of a 28 day depuration period.

In addition, Walker and Peterson (1991) injected newly fertilized rainbow trout eggs with TCDF, and looked for signs of increased mortality from hatching to swim-up. Through this evaluation, they determined an LD_{50} of 8.086 ug/kg in egg tissue.

d. Direct Sublethal Effects of TCDD to Salmonids:

The toxicological literature on sublethal effects of TCDD on salmonids focuses on coho salmon, rainbow and brook trout, and includes information on all life stages. The range of LOECs for tissue or water is similar to that identified for lethal effects. Some of the relevant sublethal studies are highlighted below.

In brook trout, a body burden of 1,200 ng TCDD/kg resulted from dietary exposure for 182 days, and was associated with minor behavioral effects and a delay in onset of spawning by 13 days (Tietge et al. 1998).

Juvenile coho salmon exhibited decreases in food consumption and weight gain when exposed for 24-48 hours to ≈ 5.6-10.53 ng/L TCDD, followed by a 20 or 56 day post-exposure period (Miller et al. 1979). The reported whole body concentration after 114 days was ≈ 478 ngTCDD/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.

Studies on exposure of eggs to TCDD include the work of Johnson et al. (1998), who exposed adult brook trout via the diet, and noted that the spawned eggs, which had tissue concentrations of 41 ngTCDD/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/L TCDD for 96 hours, wherein the resulting fry exhibited significant growth retardation for 72 days.

An overview of these studies reveals that the range of tissue LOECs for sublethal effects of TCDD to various salmonid life stages is 41 ng/kg in eggs to ≈ 478 ng/kg in tissue. Information on aqueous exposure levels indicates an LOEC of ≈ 5.6 ng/L.

e. Direct Sublethal Effects of TCDF to Salmonids:

A single study by Mehrle et al. (1988) indicated a significant reduction in growth of rainbow trout swim-up fry when exposed to water-born TCDF. In this study, fish were kept in a flow-through system with measured concentrations of contaminant for twenty eight days, followed by an additional depuration period of twenty eight days. The LOEC for growth was 0.9 ng/L.

f. Bioaccumulation and Bioconcentration of TCDD and TCDF in Salmonids:

Dietary exposure to TCDD is a potentially significant concern, since this compound has been shown to bioaccumulate in salmonids. Independent studies have indicated a whole body BAF of 0.5 for rainbow trout (Kleeman et al. 1986; Fletcher and McKay 1993; Yamada 1997). Additionally, Jones et al. (1993) estimated a field-derived whole body TCDD-equivalents BAF of ~7.7 for lake trout and other species in the Great Lakes that feed at the same trophic level. If these BAF values are applied, then salmonid prey harboring a whole body TCDD-equivalents level of 10.4 – 160 ng/kg could lead to TCDD accumulation in an adult salmonid of 80 ng/kg (discussed above as a whole body maternal tissue concentration which would be associated with an LOEC for eggs). These prey values are environmentally relevant, since salmonid prey fish from the Great Lakes have been shown to harbor body burdens of ~5-15 ng TCDD equivalents/kg (Jones et al. 1993). It is also worth noting that compromised fry that hatch from eggs that may have received significant levels of TCDD via maternal transfer may suffer additional adverse effects due to potential aqueous or dietary exposure at the fry or later stages.

Salmonids have also been shown to significantly bioconcentrate TCDD. The BCFs for rainbow trout exposed to 0.038-167 ng/L TCDD range from 4,450 to 39,000, whereas the lake trout BCF has been noted at 1,430 (reviewed in: Sherman et al. 1992; Fletcher and McKay 1993; Yamada 1997). The highest rainbow trout BCF of 39,000 was estimated by Mehrle et al. (1988) in a flow through experiment wherein swim-up fry were exposed for 28 days to 0.038 ng/L TCDD, with a corresponding whole body TCDD concentration of 980 ng/kg and increased mortality during a 28 day depuration phase. If the BCF of 39,000 is generally applied to salmonids, then a TCDD water concentration of 0.002 ng/L could result in a whole body TCDD level of 80 ng/kg. As discussed above, 80 ng/kg is a maternal tissue concentration that would be associated with an LOEC for eggs of 40 ng/kg. Also, 0.002 ng/L is below the analytical detection limit of 10 pg/L,

and is below the salmonid lethal or sublethal water-born LOEC of 0.038 ng/L. Furthermore, this scenario does not take into account possible additional exposure via the diet, which is likely to be more significant than water-born exposure (Batterman et al. 1989; Jones et al. 1993).

The compound 2,3,7,8-TCDF (TCDF) has also been found to bioaccumulate in salmonids, although to a lesser extent than TCDD. Mehrle et al. (1988) exposed rainbow trout swim-up fry to a measured concentration of 0.41 ng/L TCDF in a flow through experiment for 28 days. The resulting whole body residue of TCDF was 2.48 ug/kg, with a measured BCF value of 6,049.

g. Indirect Lethal Effects of TCDD to Salmonids Through the Prey Base:

There is only extremely limited information regarding adverse effects of TCDD on the health of invertebrates, with only two species discussed in the toxicological literature - *Daphnia magna* and *Physa* (snail). The only biological endpoints that have been studied are survival and reproduction, whereas effects on growth, development, behavior, or immune function have not been reported.

Adams et al. (1986, and citations within) did not observe any effects on survival of *D. magna* exposed for 48 hours to 0.2-1,030 ng/L TCDD, nor did effects occur during a one week post-exposure period. Also, Wu et al. (2001) reported that *D. magna* did not suffer acute effects due to a 14 day exposure to 100-1,000,000 ng/L TCDD. Likewise, adult *Physa* species exposed to 200 ng/L TCDD suffered no effects on survival -- nor did the eggs that they laid, which hatched and developed into juveniles during the 36 day exposure period (Miller et al. 1973). For sublethal effects, the only invertebrate LOEC that was forthcoming in the literature was reported by Wu et al. (2001), wherein *D. magna* exhibited decreased reproduction after the eighth day of exposure to 100 ng/L TCDD. Although this value is well above the lethal and sublethal LOECs discussed earlier for salmonids, it's utility in this evaluation is nonetheless limited by the lack of range of species and biological endpoints that it represents.

The threat of TCDD to salmonids through their invertebrate prey base lies more in the ability of invertebrates to bioaccumulate this compound, rather than through direct toxicological effects on invertebrates themselves. For example, *D. magna* exposed to 0.002 ng/L TCDD for 15 days developed a TCDD body concentration of ˜16 ng/kg, with a corresponding BCF that can be estimated at 8,000 (Wu et al. 2001). The exposure concentration of 0.002 ng/L is well below the salmonid lethal or sublethal LOEC of 0.038 ng/L. As discussed above, when salmonid BAF values are applied, it can be determined that salmonid prey tissue concentrations of 10.4 – 160 ng TCDD/kg can elicit a salmonid maternal tissue concentration that would result in significant transfer of TCDD to eggs, thus causing adverse effects in offspring. The *D. magna* body concentration of ˜16 ng/kg falls within this range. Therefore, the information in this study raises the concern that a TCDD water concentration of 0.002 ng/L may promote TCDD body burdens in salmonid prey that could lead to adverse effects in salmonids themselves. Also as noted above, 0.002 ng/L TCDD could lead to unsafe tissue levels in salmonids simply through direct bioconcentration from water.

Indeed, various field studies illustrate the fact that invertebrate body burdens of 10 ngTCDD/kg and above can be achieved in the environment. Brown et al. (1994) reported that soft shell clam, *Mya arenaria*, collected near Newark and Elizabeth, NJ had body burdens of 4.8-20 ng/kg, whereas freshwater mussels near Troy, New York also had 20 ng/kg body burdens (Hong et al. 1992), and the meat of crabs from Newark Bay and the New York Bight harbored 100 ng TCDD/kg (Rappe et al. 1991).

h. Selected LOECs and Benchmarks for Tissue and Water:

Though the sensitivity to TCDD varies with species and with life stage, the tissue LOECs for lethal and sublethal effects to salmonids at any life stage are 40 ng/kg in eggs, which translates into a maternal whole body concentration that is as low as 80 ng/kg. When a safety factor of ten is applied to generate an NOEC from the LOEC, the resulting tissue benchmark of 8 ng/kg is obtained. This is in close agreement with the tissue benchmark of 9 ng/kg that was put forth in EPA (EPA 2003). Therefore, NOAA Fisheries supports the tissue TCDD-equivalent benchmark of 9 ng/kg. [It is of note that the Washington State Department of Ecology uses a criterion of 0.07 ng/kg for fish tissue as a state water quality standard, which is based on the EPA human health criterion of 0.013 pg/L in water and a BCF of 5,000 (Johnson et al., 1991).]

Various scenarios of water-born exposure to TCDD can be evaluated with respect to direct or indirect adverse effects on salmonids. First, the information on the direct toxicity of water-born TCDD to various salmonid species and life stages indicates a lethal concentration of 0.038 ng/L. If a safety factor of one hundred is applied to obtain an NOEC, the resulting value equals 0.00038 ng/L (or 0.38 pg/L).

Second, bioconcentration via exposure to aqueous TCDD should also be considered. When a salmonid BCF of 39,000 (Mehrle et al. 1988) is applied, then a TCDD concentration of 0.23 pg/L could result in a whole body tissue benchmark concentration of 9 ng/kg.

Third, as discussed above, work by Wu et al. (2001) indicates that a water-born concentration as low as 0.002 ng/L (2 pg/L) TCDD can elicit salmonid prey body burdens that could promote, through dietary transfer to salmonids, a salmonid maternal tissue concentration that would result in significant transfer of TCDD to eggs, thus causing adverse effects in offspring. If a safety factor of ten is applied to the water-born concentration of 2 pg/L to obtain an NOEC, the resulting value is 0.2 pg/L.

Of these three scenarios, the second two deal with the lower aqueous TCDD benchmark concentrations of 0.2 - 0.23 pg/L. These values are relatively close to the water-born benchmark of 0.063 pg/L that was put forth in EPA (2003). NOAA Fisheries therefore agrees with a water-born benchmark of 0.063 pg/L TCDD.

For TCDF, there are also various approaches to calculating a water-born benchmark that are based directly on toxicity studies. First, Mehrle et al. (1988) reported a lethal concentration for salmonids of 3.93 ng/L. By applying a safety factor of 100, one can use this value to estimate a NOEC of 39.3 pg/L. Second, a sublethal LOEC of 0.9 ng/L was reported in the same study. When a safety factor of ten is applied, a NOEC of 90 pg/L is obtained.

The BCF value of 6,049 from Mehrle et al. (1988) can be used to calculate a water-born benchmark based on the EPA's tissue benchmark of 9 ng/kg TCDD-equivalents. Using the TEF value of 0.05 for TCDF (Van den Berg et al. 1998), one can determine a tissue benchmark that is specific for TCDF: 9 ng/kg multiplied by 20 to obtain a TCDF-specific tissue benchmark of 180 ng/kg. The BCF of 6,049 = tissue concentration (180 ng/kg)/ water concentration. In this scenario, a water concentration of 29.8 pg/L is obtained, which is the estimated NOEC for water that would be associated with the TCDF tissue benchmark.

The three estimates of a TCDF water benchmark are in close agreement (39.3 pg/L, 90 pg/L, and 29.8 pg/L), and NOAA Fisheries recommends the lowest value of 29.8 pg/L as the water

benchmark for TCDF. This value is approximately 150-fold higher than the EPA's recommended water-born benchmark of 0.2 pg/L for 2,3,7,8-TCDF [outlined in EPA 2003], which uses the relationship between BSAF, BAF, and K_{ow} to derive a water-born benchmark.

i. Sediment benchmarks

A sediment benchmark was generated with the simple partitioning approach for predicting sediment concentrations from water benchmarks that was presented later in Section V (Table 9). The predicted sediment concentration is 194 ng/Kg oc, which is somewhat lower than the value of 381 ng/Kg oc developed by the EPA in the BE; however based on the uncertainty of the assumptions that go into both approaches for generating sediment benchmarks, the values are not likely to be statistically different.

Based on a review of EPA's BE of the NPDES Permitted Discharge from the Potlatch Mill in Lewiston, Idaho (EPA 2003), which outlines the use of the relationship between BSAF, BAF, and K_{ow} to derive sediment benchmarks, NOAA Fisheries concurs with the sediment benchmarks of 381 ng/kg-oc for 2,3,7,8-TCDD and 955 ng/kg-oc for 2,3,7,8-TCDF. Dry wt. organic carbon normalized 2,3,7,8-TCDD and 2,3,7,8-TCDF sediment concentrations at various sampling sites in the action area between 1997 and 2002 did not exceed these sediment benchmark values. However, although sediment concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF were below the benchmark values in the year 2000, the overall combined values for all measured dioxins and furans at SRM 137 (Red Wolf Marina, closest to the diffuser) were significantly elevated (EPA 2003).

Also, the COE (2002) reported non-organic carbon-normalized sediment concentrations for 2,3,7,8-TCDD and 2,3,7,8-TCDF near Clearwater RM 3, SRM 139, and the Corps East Pond near Lower Granite Pool. The concentrations for 2,3,7,8-TCDD included values between 0.43 – 2.8 ng/kg, and those for 2,3,7,8-TCDF included values between 1.3 – 3.4 ng/kg. If a 1% organic carbon content is assumed for these sediment samples, then the ranges for organic carbon-normalized values would be approximately 43 – 280 ng/kg 2,3,7,8-TCDD and 130 – 340 ng/kg TCDF. These adjusted values are relatively close to the sediment benchmarks.

j. Summary:

The scientific body of information on the toxicological effects of TCDD on salmonids includes work on coho salmon, rainbow trout, lake trout, brook trout, and lake herring, and encompasses all of the major life stages.

Based on a review of the toxicological literature, NOAA Fisheries concurs with EPA's water-born benchmark of 0.063 pg/L for 2,3,7,8-TCDD toxicity to salmonids. However, NOAA Fisheries recommends a water-born benchmark for 2,3,7,8-TCDF of 29.8 pg/L, which is approximately 150-fold higher than the EPA recommendation for this parameter.

The analytical detection limit of 10 pg/L for water-borne 2,3,7,8-TCDD is too high to adequately determine whether the aqueous concentration in the mixing zone is below 0.063 pg/L. The hydrophobicity of 2,3,7,8-TCDD and 2,3,7,8-TCDF greatly limits their occurrence in the aqueous phase. Therefore, the analysis and consideration of the effects of TCDD and TCDF in tissue of fish and in sediment is warranted.

Based on a review of the toxicological literature, NOAA Fisheries concurs with EPA's fish tissue benchmark of 9 ng/kg 2,3,7,8-TCDD-equivalents for salmonid toxicity. In EPA (2003), fish tissue concentrations of chlorinated dioxins and furans in the Columbia River are summarized, based on the EPA report "Columbia River Fish Contaminant Survey" (EPA, 2002). However, more specific environmental monitoring will be required to assess the potential impact of the release of Potlatch effluent on salmonids in the vicinity of the diffuser.

NOAA Fisheries agrees with the EPA's sediment benchmarks of 381 ng/kg-oc for 2,3,7,8-TCDD and 955 ng/kg-oc for 2,3,7,8-TCDF. At this time, there is insufficient sediment data to make a determination of how the release of Potlatch Mill effluent may affect listed salmonids via this route of exposure. Therefore, additional environmental monitoring of sediment will be essential for such an evaluation.

2.3.3.5 Chlorinated Phenolic Compounds (CPCs)

a. Introduction

Chlorinated phenolic compounds (CPCs) are formed as byproducts of the bleaching process and are found in the effluents of pulp and paper mills. 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 guaiacols are mono-ortho-methoxy substituted phenols with additional chlorines as indicated in the name. Substituted syringols are di-ortho-methoxy substituted phenols with additional chlorines as indicated in the name.

The octanol-water partition coefficient is one of the most useful chemical physical properties for predicting its behavior. These values for CPC are summarized in Table 8. The K_{ow}s for CPC range from 671 for 4, 5, 6- trichloroguaiacol to 724,436 for pentachlorophenol. Given that chemicals with log K_{ow} greater than 5.0 are considered to have a high potential for bioaccumulation (Carey et al. 1998), 4,5,6- trichloroguaiacol (log K_{ow}=2.83) would have a lower potential to bioaccumulate than pentachlorophenol (log K_{ow}=5.86).

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 CPCs to fungi is decreased as the degree of dissociation increases.

With the exception of pentachlorophenol, EPA, WDOE, and Idaho Department of Environmental Quality have not derived water quality criteria for these CPCs. The EPA has derived freshwater acute and chronic AWQC for pentachlorophenol based on the pH of the receiving water.

(1) Environmental concentrations and toxic effects of CPCs The draft permit specifies a daily maximum effluent limitation for each CPC of either 2.5 µg/L or 5 µg/L (the analytical detection limit for the compound).

The CPCs were not detected in samples collected from the Snake and Clearwater Rivers upstream of the Mill therefore it was assumed that concentrations of these compounds in upstream water is likely very low. However, because of the relatively high detection limits, the

concentrations of these chlorophenols may be as high as the detection limit, which in many cases is 2.5 µg/L (EPA 2003, Table VII-12). Two CPCs (3,4,5 trichloroguaiacol and tetrachloroguaiacol) were included as analytes in the Potlatch ambient sediment monitoring program which collected sediment samples from about 17 sites annually from 1997-2002. These two CPCs were not detected in any sediment samples with a usual detection limit of 0.3 mg/kg (CH2M Hill 1997, 1998,1999,2000,2001,2002).

In a few cases, data were not available to generate direct or indirect toxicity benchmarks. In those few cases, the data for other chlorophenols were used. For five chemicals, the direct toxicity benchmark for 2,4,5 trichlorophenol was used because several studies had been conducted with this compound, its K_{ow} was similar to that of the other compounds, and the benchmark value was low and therefore conservative.

The direct toxicity benchmarks were surprisingly consistent across all compounds, except for pentachlorophenol, which was expected to be much lower due to its high bioaccumulation potential. The relatively high variability for the indirect toxicity benchmarks was likely caused by several factors compared to the direct toxicity benchmark, which only considered salmonids. These include the use of several widely different taxa that would be expected to exhibit large differences in their response to a given chemical and few data with limited species and toxic endpoints.

It should be kept in mind that many of these compounds are hydrophobic and 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. Assessing the concentrations of these permitted compounds downstream can only be accurately determined by direct measurement and not by prediction based on laboratory experiments to determine persistence in pure water systems. Because of the large variable in half life values, direct measurement of these compounds downstream at various intervals is strongly recommended.

(2) Environmental Baseline Very little information exists regarding the concentrations of chlorophenols in water, tissue, and sediment downstream and upstream of the pulp mill. A monitoring and assessment plan will be presented to assure that environmental concentrations are adequately characterized in the action area (LGR).

(3) Effluent Limitation Nearly all of the 28 CPCs 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 CPCs, 12 of the higher substituted chlorinated compounds are associated with the presence of 2,3,7,8-TCDD and 2,3,7,8-TCDF.

The NPDES permit limits the concentrations of CPCs in the fiberline, which is only part of the total effluent concentration. Because other waste streams flow into the treatment pond and the EPA has concluded that none of the permitted chemicals are found in these waste streams, the predicted effluent concentration will be lower. The effluent concentration is determined by multiplying the detection limit for a CPC by the dilution factor. This dilution factor is based on the ratio of the fiberline flow to the total effluent producing a value of 0.365. This assumes that

none of these permitted compounds are present in the waste streams from the other areas of the plant. The total waste stream that is not from the fiberline is 23.9 mgd and includes the pulp mill (7.6 mgd), paper machines (4.1 mgd), recovery boilers (4.0 mgd), consumer products (8 mgd), and wood products (0.2 mgd) sections of the plant (Figure III-1 in the BE).

For those chlorinated compounds with a detection limit of 2.5 µg/L, the effluent concentration is predicted to be 0.91 µg/L. These CPCs are 2,4,5 trichlorophenol, 2,4,6 trichlorophenol, 3,4,5 trichloroguaiacol, 3,4,6 trichloroguaiacol, 4,5,6 trichloroguaiacol, and trichlorosyringol.

For those chlorinated compounds with a detection limit of 5.0 µg/L, the effluent concentration is predicted to be 1.8 µg/L. These CPCs are 2,3,4,6 tetrachlorophenol, pentachlorophenol, 3,4,5 trichlorocatechol, 3,4,6 trichlorocatechol, tetrachlorocatechol, and tetrachloroguaiacol.

(4) Toxicity Benchmarks Because the toxicity of chlorinated phenols to various species from water exposure is compound-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 on the basis of individual chemical, when sufficient information becomes available (EPA, 1980). Criteria have only been developed for 2,4,5-trichlorophenol, 2,4,6-trichlorophenol, and pentachlorophenol. This applies to criteria developed for water and sediment exposure. For criteria or benchmarks based on tissue concentrations, a different approach is used (see section on Tissue and Sediment Benchmarks).

- The most stringent water quality criterion developed by EPA (EPA, 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 2,4,6-trichlorophenol is 2.1 µg/L as a long-term average, for the protection of human health. The most stringent water quality criterion developed by EPA (EPA, 1986) for 2,4,6-trichlorophenol is 2 µg/L for organoleptic effects.

- The most stringent water quality standard in Idaho and Washington for pentachlorophenol is 0.28 µg/L as a long-term average for the protection of human health.

For all other CPCs, 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.

b. 2,4,5-Trichlorophenol

<u>Salmonid Species</u>

Direct Effects

The range of LC_{50s} for rainbow trout and brown trout was 260 µg/L to 900 µg/L 2,4,5-trichlorophenol (McKim et al. 1985, 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 LOECs for growth inhibition ranging from 34 µg/L to 125 µg/L for exposure to 2,4,5- trichlorophenol.

McKim et al. (1985) reported a NOEC of 4.6 µg/L 2,4,5-trichlorophenol for respiratory effects (ventilation rate, ventilation volume, oxygen uptake efficiency) in rainbow trout; however, this was the only concentration tested. Neville (1995) reported a NOEC for physiological changes of 211 µg/L and a LOEC of 438 µg/L.

The lowest direct toxicity concentration (LOEC) for 2,4,5-trichlorophenol is 34 µg/L (Neville 1995). An safety factor of 10 for coverting the LOEC to NOEC was applied resulting in a value of 3.4 µg/L for this study. If the mortality values were used and the standard safety factor applied (10 for ACR and 10 for LOEC to NOEC), the benchmark concentration would be very close (2.6 ug/L) to the selected benchmark using the LOEC value from Neville (1995). The direct toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 2,4,5-trichlorophenol will be 3.4 µg/L.

Indirect Effects

Non-salmonid fish have been found to have LC_{50s} 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 LC_{50s} 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 LC_{50} 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, maximum acceptable toxican concentration (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 EC_{50}s 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 behavioral changes in valve closure in *Sphaerium corneum* (a clam) at concentrations of 42 µg/L.

For non-salmonid fish and prey species, the LOEC was 42 µg/L (Heinonen et al. 1997). This value was divided by an safety factor of 10 to convert the LOEC to a NOEC of 3.4 µg/L.

The indirect toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 2,4,5-trichlorophenol will be 4.2 µg/L.

Summary

1. 2,4,5-trichlorophenol, in combination with other chlorophenols, may exceed benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are very few studies that examined chronic effects of this compound on salmonids and their different life stages.

3. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

4. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

c. 2,4,6-Trichlorophenol

Salmonid Species

Direct Effects

The range of LC_{50s} (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 (LC_{50}) for 2,4,6-trichlorophenol is 730 µg/L in brown trout (Hattula et al., 1981). 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. The direct water-toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 2,4,6-trichlorophenol will be 7.3 µg/L.

Indirect Effects

Nonsalmonid fish have been found to have LC_{50s} 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 LC_{50s} 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 an 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 EC 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.

Castren and Oikari (1987) reported a reduction in UDP-glucuronosyltransferase (an enzyme) in rainbow trout (*Oncorhynchus mykiss*) at a concentration of 200 µg/L 2,4,6-trichlorophenol. This was the only concentration, other than the control, that was tested. It could be considered an LOEC; however, lower concentrations may also cause this effect.

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 invertebrate prey species, the lowest acute LC50 was 270 µg/L. A NOEC was established by application of a safety factor of 100 to obtain a chronic NOEC of 2.7 µg/L for non-salmonid fish and prey. The indirect water toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 2,4,6-trichlorophenol will be 2.7 µg/L.

Summary

1. 2,4,6-trichlorophenol, in combination with other chlorophenols, may exceed the NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are very few studies that examined chronic effects of this compound on salmonids and their different life stages.

3. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

4. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

d. 2,3,4,6-Tetrachlorophenol

Salmonid Species

Direct Effects

For salmonid *Oncorhynchus mykiss*, Kennedy (1990) reported a range of mean LC_{50s} (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 LC_{50} of 500 µg/L 2,3,4,6-tetrachlorophenol.

The lowest direct toxicity concentration (LC_{50}) for 2,3,4,6-tetrachlorophenol is 334 µg/L in rainbow trout (Kennedy, 1990). Because no LOEC could be found for direct toxicity, a NOEC was established by application of two safety factors (10 for the ACR and 10 for the LOEC to NOEC). These safety factors produces a water concentration benchmark of 3.3 µg/L. The direct water-toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 2,3,4,6-tetrachlorophenol will be 3.3 µg/L.

Indirect Effects

Nonsalmonid fish have been found to have mean LC_{50s} 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 LC_{50s} ranging from 100 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 an 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 MATCs in the water flea ranged from 650 to 1200 µg/L.

Liber et al. (1992) reported 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. For the copepods, the EC_{50} values ranged from 270 to 590 µg/L 2,3,4,6-tetrachlorophenol (Liber et al., 1992). In the rotifers, 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). The lowest value was 270 ug/L for a 50% reduction in abundance of a calanoid copepod. For many of these, the EC50 is a 50% reduction in abundance, which is a severe effect.

Based on the numerous studies showing severe adverse effects between 100 and 300 ppb, the mortality endpoint of 100 ug/L was chosen. Applying the uncertainly factors for the ACR and for converting from an LOEC to NOEC, the benchmark becomes 1.0 ug/L. The indirect water-toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 2,3,4,6-tetrachlorophenol will be 1.0 µg/L.

Summary

1. 2,3,4,6-tetrachlorophenol, in combination with other chlorophenols, may exceed the NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined chronic effects of this compound on salmonids and their different life stages.

3. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

4. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

e. Pentachlorophenol

The water quality criteria for pentachlorophenol established by the EPA is pH dependent. At a pH of 7.8, the freshwater criteria are 19 µg/L for acute exposures and 15 µg/L for chronic exposures (EPA 1998), which supercedes EPA (1986a). In general, the toxicity of PCP increases with decreasing pH. To determine the freshwater criteria as a function of pH the following equations are used:

1. $CMC - e^{(1.005*pH - 4.869)}$

2. $CCC = e^{(1.005*pH - 5.134)}$

The CMC is the Critical Maximum Concentration and is defined as the highest one-hour average concentrations that should not result in adverse effects to aquatic organisms. The CCC is the Criteria Continuous Concentration is defined as the highest four day average concentration that should not result in adverse effects.

Because the pH in Idaho's freshwater streams can be variable and occur at low values, this pH adjustment for the Water Quality Criteria is important for determining the protective concentrations for aquatic life. For example, based on these equations the CCC at pH 6 would be 2.45 µg/L. This is a 5.3 fold reduction from the CCC determined at pH 7.8.

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 PCP 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). The PCP 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 PCP exposure (Webb and Brett 1973, Hodson and Blunt 1981, Dominquez and Chapman 1984, Brown et al. 1985). This pesticide is also considered a probable human carcinogen

Salmonid Species

Direct Effects

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 (Eisler, 1989; EPA, 1980, 1986a). Warmwater species are generally less sensitive than coldwater species in acute lethal toxicity tests (EPA, 1986a). 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, suggesting that an additional margin of safety may be needed to protect listed salmonids when using rainbow trout test data in toxicity assessments (Sappington et al 2001).

Eisler (1989) reviewed the effects of pentachlorophenol on invertebrates' growth, survival, and reproduction at levels of 3-100 µg/L. Fish are affected at concentrations from 1-68 µg/L, while

birds are affected at dietary concentrations greater than 3,580 mg/kg. 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 LC_{50} was determined for carp larvae at 9.5 µg/L at a pH of 7.2 (Eisler, 1989).

The LC_{50s} for chinook salmon were reported to be 31 and 68 µg/L pentachlorophenol (Johnson and Finley, 1980 and Saarikoski and Viluksela, 1981). The LC_{50} 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 LC_{50} 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 LC_{50} 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). The

lowest reported endpoint in AQUIRE was an LC_{50} of 18 µg/L for early life stage rainbow trout (Van Leeuwen et al., 1985). 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. The direct water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to pentachlorophenol will be 0.18 µg/L.

Indirect Effects

The LC_{50s} 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 LC_{50s} ranging from 40 to 1,442 µg/L pentachlorophenol for the guppy (Poecilia reticulata) exposed for 7 days; and the lowest LC_{50} for largemouth bass was 54 µg/L for a 120- day exposure period (Johansen et al., 1985). A 96 hour LC_{50} 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 four species of the salmonidae family. This conclusion was based on acute exposure and LC_{50} values for five different compounds (copper, carbaryl, nonylphenol, pentachlorophenol, and permethrin). The 96 hour LC_{50} 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). LC_{50} 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; Gupta and Rao, 1982). Static, flow- through and renewal systems were used. The range of LC_{50} values were generally similar across the species. LC_{50} 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). For the three water flea genera, LC_{50} 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 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).

The LOEC for pentachlorophenol is 40 µg/L, which was reported as a LC_{50} for sturgeon (Dwyer et al. 2000). This value is similar to other values for fish demonstrating toxicity in the

40 to 100 µg/L range. Using a factor of 10 to convert from an acute lethal effect concentration to a LOEC and an additional factor of 10 to determine the no effect level (NOEC), the indirect toxicity benchmark becomes 0.4 µg/L for non-salmonid fish and prey. The indirect water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to pentachlorophenol is 0.4 µg/L.

Summary

1. Pentachlorophenol is very toxic and concentrations of this compound in the mixing zone far exceed the NOEC benchmark level.

2. There are very few studies that examined chronic effects of this compound on salmonids and their different life stages.

3. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

4. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

5. Concentrations of pentachlorophenol in the mixing zone may exceed the EPA Water Quality Criteria and Human Health Criteria.

f. 3,4,5-Trichlorocatechol

Salmonid Species

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. The direct water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,5-trichlorocatechol is 3.4 µg/L.

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 LC_{50} (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.

A NOEC was established using a factor of 10 to convert from an lethal concentration to a chronic LOEC and an additional factor of 10 to convert this LOEC to a no effect level (NOEC). This

value is consistent with that derived for 2,4,5 trichlorophenol, which would had been used if no data were available. The indirect water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,5-trichlorocatechol is 3.4 µg/L.

Summary

1. 3,4,5-trichlorocatechol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined lethal or sublethal effects of this compound on salmonids and their different life stages.

3. A surrogate compound (2,4,5 trichlorophenol) was used for the determination of direct effects because no relevant studies have been performed with this compound.

4. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

5. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

g. 3,4,6-trichlorocatechol

Salmonid Species

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. The direct water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,6-trichlorocatechol is 3.4 µg/L.

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).

Due to the lack of data for species other than algae, the indirect benchmark for 2,4,5 trichlorophenol was used for 3,4,6 trichlorocatechol. The indirect water-toxicity NOEC

benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,6-trichlorocatechol is 4.2 µg/L.

Summary

1. 3,4,6-trichlorocatechol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined lethal or sublethal effects of this compound on salmonids and their different life stages.

3. A surrogate compound (2.4,5 trichlorophenol) was used for the determination of direct and indirect effects because no relevant studies have been performed with this compound.

4. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

5. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

h. Tetrachlorocatechol

Salmonid Species

Direct Effects

A mean LC_{50} (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).

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. The direct water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to tetrachlorocatechol is 11 µg/L.

Indirect Effects

Geiger et al. (1985) reported a mean LC_{50} of 1,270 µg/L tetrachlorocatechol for the fathead minnow (Pimephales promelas). Mortality tests using Daphnia magna have resulted in a mean LC_{50} of 2,230 µg/L tetrachlorocatechol (Dence et al., 1980).

For prey species, the LOEC for tetrachlorocatechol is 1,270 µg/L, which is based on the LC_{50} value reported by Geiger et al. (1985). A factor of 10 was used to convert from an acute lethality value to a chronic sublethal effect concentration. An additional safety factor of 10 was used to convert to a no effect level producing an indirect toxicity benchmark is 1.3 µg/L for

non-salmonid fish and prey. The indirect water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to tetrachlorocatechol is 1.3 µg/L.

Summary

1. Tetrachlorocatechol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined chronic effects of this compound on salmonids and their different life stages.

3. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

4. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

i. 3,4,5-Trichloroguaiacol

Salmonid Species

Direct Effects

The mean LC_{50} (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).

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. The direct water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,5-trichloroguaiacol is 7.5 µg/L.

Indirect Effects

Mortality tests performed using Daphnia magna have resulted in mean LC_{50s} ranging from 450 to 730 µg/L 3,4,5-trichloroguaiacol (Virtanen et al., 1989; Oikari et al., 1992).

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 4.5 µg/L. The indirect water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,5-trichloroguaiacol is 4.5 µg/L.

Summary

1. 3,4,5-trichloroguaiacol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined chronic sublethal effects of this compound on salmonids and their different life stages.

3. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

4. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

j. 3,4,6-trichloroguaiacol

Salmonid Species

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. The direct water-toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,6-trichloroguaiacol will be 3.4 µg/L.

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. The indirect water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 3,4,6-trichloroguaiacol is 4.2 µg/L.

Summary

1. 3,4,6-trichloroguaiacol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined lethal or sublethal effects of this compound on salmonids and their different life stages.

3. A surrogate compound (2,4,5 trichlorophenol) was used for the determination of direct and indirect effects because no relevant studies have been performed with this compound.

4. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

5. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

k. 4,5,6-trichloroguaiacol

Salmonid Species

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. The direct water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 4,5,6-trichloroguaiacol is 3.4 µg/L.

Indirect Effects

Mortality tests using the invertebrates have resulted in LC_{50s} 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 LOEC for 4,5,6-trichloroguaiacol is 50 µg/L, which is based on the LC_{50} value reported by Peterson and Peterson (1988). Using a factor of 10 to convert from an acute to a chronic effect concentration and a safety factor of 10 to convert to a no effect level, the toxicity benchmark is 0.5 µg/L for non-salmonid fish and prey. The indirect water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to 4,5,6-trichloroguaiacol is 0.5 µg/L.

Summary

1. 4,5,6 trichloroguaiacol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined lethal or sublethal effects of this compound on salmonids and their different life stages.

3. A surrogate compound (2,4,5 trichlorophenol) was used for the determination of direct effects because no relevant studies have been performed with this compound.

4. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the Lower Granite Reach.

l. Tetrachloroguaiacol

Salmonid Species

Direct Effects

The LC_{50s} (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 LC_{50} 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 leuocrit 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 direct toxicity benchmark concentration for tetrachloroguaiacol is 2 ug/L, which based on the reported increased leucocrit and decreased cortisol in juvenile rainbow trout (Johansen et al. 1994). The LOEC for these responses was 20 ug/L, which was adjusted by a factor of 10 to convert this value to a no effect level. This benchmark value is supported by several studies listed above demonstrating LC_{50s} in the 320 to 370 ppb range. If these LC_{50} values were used as the direct toxicity benchmark, and adjusted with the safety factors, benchmark concentrations would range from 2.0 ug/L to 3.7 ug/L.

The direct water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to tetrachloroguaiacol is 2.0 µg/L.

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 LC_{50} for this compound was between 100 µg/L and 200 µg/L.

Static 48-hour mortality tests using Daphnia magna have resulted in mean LC_{50s} 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 LC_{50} 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 LC_{50} 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 non-salmonid fish and prey species, the LOEC for tetrachloroguaiacol is 100 µg/L, which is based on several LC_{50} values. Using a factor of 10 to convert from a lethal effect to a chronic effect and an additional safety factor of 10 to convert the chronic effect value to a no effect level, the toxicity NOEC benchmark becomes 1.0 µg/L for non-salmonid fish and prey. The indirect water-toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to tetrachlorocatechol is 1.0 µg/L.

Summary

1. Tetrachloroguaiacol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are only a few studies that examined chronic effects of this compound on salmonids and their different life stages.

3. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

4. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

m. Trichlorosyringol

<u>Salmonid Species</u>

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. The direct water-toxicity benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to trichlorosyringol will be 3.4 µg/L.

Indirect Effects

In an acute, 96-hour static laboratory test, Kuivasniemi et al. (1985) reported an EC50 of 0.0033 mM (ca. 850 µg/L) based on population-level effects in green algae (Selenastrum capricornutum).

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.

The indirect water-toxicity NOEC benchmark for Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead exposed to trichlorosyringol is 4.2 µg/L.

Summary

1. Trichlorosyringol, in combination with other chlorophenols, may exceed NOEC benchmark concentrations in the mixing zone and downstream in the LGR.

2. There are no studies that examined lethal or sublethal effects of this compound on salmonids and their different life stages.

3. A surrogate compound (2,4,5 trichlorophenol) was used for the determination of direct and indirect effects because no relevant studies have been performed with this compound.

4. Analytical detection limits are too high for an adequate evaluation of ambient concentrations upstream, in the mixing zone, and downstream in the LGR.

5. Based on the K_{ow}, this compound may accumulate and persist in the sediment and the food web.

Effects Analysis

As individual compounds, many of the chlorophenols in the effluent occur at concentrations just below the NOEC benchmark concentrations listed in Table 9. The analysis of effects in water due to chlorophenol exposure is accomplished with the toxic unit approach described below.

Because the main concern for near-field effects from the effluent is to listed species, the toxic unit approach for water exposure will use the direct NOEC benchmark. The indirect NOEC water benchmarks, in addition to tissue and sediment NOEC benchmarks, will be used in the monitoring and assessment plan for determination of effects to all species. There is only one tissue and one sediment NOEC benchmark, which will be used for all cases where such data are available.

One of the primarily modes of action of chlorophenols is inhibition of oxidative phosphorylation, which causes a decrease in the production of 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 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).

An adequate toxicological assessment of the chlorophenols, catechols, and guaiacols must consider their additive effects. Because these compounds likely act by a common mode of action (inhibition of oxidative phosphorylation), 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 interactions were 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 these compounds, additivity is a reasonable assumption. Hence a simple toxic-unit approach would be valuable in protecting listed species and their prey from multiple toxicants that are at, or close to toxic levels.

For each toxicant found in a body of water, its potential for toxicity can be determined by the equation:

$$\text{Toxic Unit (TU)} = \sum \frac{[\text{water}]}{[\text{benchmark}]}$$

where [water] is the measured water concentration and [benchmark] is the lowest observed or no observed effect concentration for that toxicant. If TU values occur above 1.0, the potential for toxic effects increases. The TU value of 1.0 can also be considered the combined benchmark value. If the TU calculation is below 1.0, then the combined water concentrations for all compounds would be considered unlikely to cause toxicity.

It should be kept in mind that the benchmark concentrations used for this analysis are NOECs, which means a TU value of 1.0 is still a no effect value.

Table 9 shows the results of the TU calculations for chlorophenols and dioxin. The sum of TU in the effluent when it enters the Snake River is 14.1.

NOAA Fisheries has concluded that the discharge of chlorophenols at the maximum effluent concentration is likely to have chronic adversely effects on Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River steelhead.

Table 8. Summary of Physical/Chemical Parameters of selected chlorinated organic compounds

Parameter	Mol Weight (g/mol)	K_{ow}	log K_{ow}	Water Solubility (mg/L)		H (atm-m3/mol)	Vapor Pressure (atm)	Volatilization Half Lives (hr)	Min of Half Lives in Surface Water (hr)	Max of Half Lives in Surface Water (hr)	Geometric Mean of Half Lives in Surface Water (hr)
2,4,5-Trichlorophenol	197.5	5248	3.7	1.07E+03		1.30E-06	2.78E-05	768	0.5	336	13
2,4,6-Trichlorophenol	197.5	5754	3.8	8.00E+02		2.40E-06	1.53E-05	480	2	96	14
2,3,4,6-Tetrachlorophenol	231.9	26303	4.4	1.49E+02			2.76E-06	1032	1	336	18
Pentachlorophenol	266.4	724436	5.9	1.40E+01		7.50E-08	1.97E-07	3120	1	110.4	11
3,4,5-Trichlorocatechol	213.5	5623	3.7	3.88E+01	(a)			NA	NA	NA	NA
3,4,6-Trichlorocatechol	213.5	4467	3.7	4.87E+01	(a)			NA	NA	NA	NA
Tetrachlorocatechol	213.5	16982	4.2	1.30E+01	(a)			NA	NA	NA	NA
3,4,5-Trichloroguaiacol	227.5	14125	4.1	1.66E+01	(a)			NA	NA	NA	NA
3,4,6-Trichloroguaiacol	227.5	8710	3.9	2.68E+01	(a)			NA	NA	NA	NA
4,5,6-Trichloroguaiacol	227.5	671	2.8	3.37E+02	(a)			NA	NA	NA	NA
Tetrachloroguaiacol	261.9	40738	4.6	2.60E+01		1.37E-05	1.36E-06	NA	NA	NA	NA
Trichlorosyringol	257.5	15849	4.2	1.68E+01	(a)			NA	NA	NA	NA
2,3,7,8-TCDD	322	4365158	6.6	1.80E-04		2.77E-05	1.59E-10	46 d - 50 y	10074	14191	11957
2,3,7,8-TCDF	306	3388442	6.5	1.40E-03		1.48E-05	4.18E-10	15.5 d - 38 y	NA	NA	NA
AOX		NA						NA	NA	NA	NA
Chloroform	119.4	83	1.9	7.90E+03		3.73E-03		36	NA	NA	NA

(a) Formula from Lyman et al., (1990): log S = -0.9874 * log K_{ow} for halobenzenes

Sources: NCASI (1992), CRC Handbook of Chemistry and Physics (75th Edn) (1995) , Merck Index (10th Edn), Mackay et al. (1991) Texas NRCC(2000) HSDB (2000), Howard et al. (1991).

Table 9. Benchmarks for pulp mill chemicals.

	Benchmarks					
	Water					
	Direct	Indirect	Tissue	Sediment	Effluent baseline	Toxic units (water)
Units	µg/L	µg/L	mg/kg	µg/Kg-oc	µg/L	Unitless
		27				
Chloroform	12.4				710	57
	3.4	4.2		11,281		0.27
2,4,5 trichlorophenol			0.3		0.91	
2.4.6 trichlorophenol	7.3	2.7	0.3	39,722	0.91	0.12
2,3,4,6 tetrachlorophenol	3.3	1.0	0.3	69,823	1.8	0.55
pentachlorophenol	0.18	0.02	0.3	113,567	1.8	10
3,4,5 trichlorocatechol	3.4	3.4	0.3	11,281	1.8	0.53
3,4,6 tetrachlorocatechol	3.4	4.2	0.3	11,281	1.8	0.53
tetrachlorocatechol	11.0	1.3	0.3	148,004	1.8	0.16
3,4,5 trichloroguaiacol	7.5	4.5	0.3	80,471	1.8	0.24
3,4,6 trichloroguaiacol	3.4	4.2	0.3	17,739	0.91	0.27
4,5,6 trichloroguaiacol	3.4	0.5	0.3	1,471	0.91	0.27
tetrachloroguaiacol	2.0	1.0	0.3	66,545	1.8	0.90
trichlorosyringol	3.4	4.2	0.3	34,983	0.91	0.27
Sum of toxic units for chlorophenols						14.1
phytosterols	1.0					
resin acids	1.0					
retene	0.5			987,250		
Dioxins/Furans	pg/l		ng/kg	ng/kg-oc	pg/l	
	0.06			381	3.65	
2,3,7,8 TCDD	3		9 (TEQ)			58
2,3,7,8 TCDF	29.8		9 (TEQ)	955	11.6	0.4

Units in µg/L, µg/Kg, µg/Kg-oc, except for 2,3,7,8 TCDD, 2,3,7,8 TCDF. oc = organic carbon. TEQ = toxicity equivalency factors for dioxins and furans. Fish tissue benchmarks are 0.3 mg/Kg for direct effects and 1.6 mg/Kg wet weight for indirect effects. Sum of toxic units must be used for chlorophenols. Sum of toxic units for chlorophenols = 14.1 for exposure to effluent. Value for phytosterols is based on beta-sitosterol.

n. Benchmarks for Sediment and Tissue concentrations

Based on recent studies, the BSAF for 2,4,5 trichlorophenol (TCP) for Chironomus riparius exposed to sediment was found to be in the range of 1 - 10 (Penttinen et al. 1996, Ristola et al. 1999). One of these studies (Penttinen et al. 1996) also reports the 2,4,5 TCP BSAF for Lumbriculus variegatus, a common oligochaete worm, to be in the 50 to 100 range. Using these BSAF data in conjunction with a tissue benchmark, a sediment concentration was generated for salmonids and other species (indirect). A BSAF of 10 was selected for the indirect benchmark based the studies by Penttinen et al. (1996) and Ristola et al. (1999). The indirect tissue benchmark (see below) of 1.6 mg/Kg and a BSAFs of 10 produces a sediment benchmark of 16,000 ng/g organic dry wt. If the organic carbon content of sediment is 3%, then the benchmark sediment concentration would be 480 ng/Kg.

For salmon a much lower BSAF is expected because of the reduced interaction with sediment and possible metabolism. If a BSAF of 0.5 is selected and entered into the equation with the tissue benchmark of 0.3 mg/Kg (see below), the sediment benchmark would equal 60,000 ng/g organic carbon. If the sediment content was 3%, the bulk sediment benchmark would become 1,800 ng/g dry wt. As with the other benchmarks, the toxic unit equation will be applied to all measured chlorophenols. For this approach, the denominator of the toxic unit equation ($[Sed_{oc}]$benchmark) would be the lowest value determined above (16,000 ng/g oc).

This approach is valid only if reliable BSAF values are obtained. As part of the monitoring program, such values will likely be generated and used to refine these estimates for the sediment benchmarks. The sediment benchmarks generated with K_{oc} values will be used until such site specific bioaccumulation data are generated. If additional data on BSAFs indicate that the sediment benchmark should be lower, then the BSAF approach will be revisited.

<u>Tissue</u> <u>Benchmarks</u>

The following analysis is for the CPCs.

Most of the data has been generated for pentachlorophenol and a few other CPC. The most common endpoint was mortality. Tables V-6, V-7, V-8 show the available data. Mortality, expressed as the LR50 or tissue concentration associated with 50% mortality, was relatively consistent among species. The exception is *Oncorhynchus mykiss*, which will be used as a surrogate for listed salmonids. The mean (sd) LR50 for all PCP exposures (except *Oncorhynchus*) was 118 (67) mg/Kg wet wt. The mean LR50 for other chlorophenols was 147 (87) mg/Kg wet wt. As expected, the LR50s for the different CPCs were not significantly different. The overall mean (sd) LR50 for all CPCs is 123 (72) mg/Kg wet wt.

There are a few studies that examined sublethal effects and almost all of these are for PCP (Table 10). The overall mean (sd) lowest observed effects tissue (residue) (LOER) is 16.3 (17.6) mg/Kg wet wt. The ratio between the LR50 and LOER is 7.5 times, which is a value that supports our previous safety factor of 10 for converting lethality values to those that would be protective for sublethal responses.

The direct tissue benchmark for salmonids will be 0.3 mg/Kg wet wt., which is considered the no observable effects residue (NOER). This is based on the value for *Oncorhynus mykiss* in Table 10 and two safety factors as described elsewhere in this BO. The first safety factor is 10 for protecting salmonids from sublethal effects when only lethality data are available. The second safety factor (=10) will be applied to convert the LOER value to one that would give a reasonable assurance of no effects (NOER).

The indirect tissue benchmark will be 1.6 mg/Kg wet wt. This value was derived by dividing the LOER value for all sublethal effects by 10 to generate a NOER value.

A modeling exercise that examines the tissue benchmark based on bioaccumulation was as performed. Using the standard bioaccumulation equation (shown below) for species with a similar lipid content as that found salmonids, a series of tissue benchmarks was generated for each chlorophenol with the water benchmark and log K_{ow} for that compound.

Log BCF = 0.85*Log K_{ow} - 0.70, where BCF = $\frac{[\text{tissue}]}{[\text{water}]}$, tissue the unknown value and water is the benchmark value.

The mean (sd) of all tissue benchmarks using this approach was 2.7 (2.2) mg/Kg, which was not substantially different compared to the indirect benchmark. There are many more assumptions inherent with a tissue benchmark generated by the BCF approach compared to one that uses actual tissue concentrations. The BCF approach is indirect and if bioaccumulation is actually higher than that predicted with the equations the tissue benchmark increases. Therefore, NOAA Fisheries will use the values determined from actual measured values that are presented above. Interestlingly, the tissue benchmarks (1.6 and 2.7 mg/Kg wet wt.) determined by two approaches are generally similar, indicating that the bioaccumulation predictions for these chlorophenols may be relativley accurate.

Each measured tissue concentration for all chlorophenol will be assessed with the toxic unit approach with the equation below. This will be done for both the direct (salmonid) and indirct (other species) assessment. For the direct benchmark, each chlorophenol concentration will be divided by the NOER benchmark of 0.3 mg/Kg wet wt. and all values will be summed. If the sum of TU exceeds 1.0, then adverse effects will be assumed. For the indirect tissue benchmark, the NOER value will be 1.6 mg/Kg wet wt.

The equation is:

$$TU = \sum \frac{[\text{tissue}]_{\text{measured}}}{[\text{tissue}]_{\text{benchmark}}}$$

where the tissue benchmark is the NOER value.

Because the LR50 for dichlorophenol was not substantially different compared to all other chlorophenol values, all chlorophenolic compounds should be measured in tissue. Even though only a few chlorophenolic compounds are addressed specifically in this BO, the effluent from this pulp mill contains additional chlorophenols. Even though many of these compounds will have low bioaccumulation values, tissue concentrations will likely result because of the expected high concentrations. These additional chlorophenols should be considered in the toxic unit approach because they will contribute to the additive toxicity as demonstrated by Escher et al. (2001).

Table 10. Critical Body Residues for Pentachlorophenol. Lethal values (LR50s)

Species	LR50 (mg/Kg)	sd	n	Ref
Lumbriculus variegatus	81	46	3	1
Chironomus riparius	49	13	2	1
Carassius auratus	90	6	4	2
Oncorhynchus mykiss	29	8	5	3, 4
Salmo trutta	200		1	5
Eisenia fetida	48	53	3	6
Eisenia eugeniae	168			6
Lumbricus terrestris	192			6

Whole body values. LR50 is the concentration in mg/kg associated with the 50% mortality response. N is number of replicates for each, which are usually different exposure times and conditions from the same study. Mean (sd) for all values (except Oncorhynchus) is 118 (67) mg/Kg. References are 1. Kukkonen (2002), 2. Kishino & Kobayashi (1995), 3. Van den Huevel et al. (1991), 4. Hodson et al. (1984), 5. Hattula et al. (1981), 6. Fitzgerald et al. (1996). All concentrations as wet wt.

Table 11. Critical Body Residues for Pentachlorophenol.
Sublethal values (LOERs).

Species	LOER (mg/Kg)	sd	n	response	ref
Mercenaria mercenaria	0.5			physiological	1
Micropterus salmoides	10.2			physiological	2
Micropterus salmoides	9.6			growth	2
Mytlius edulis	2.34			physiological	3
Anodonta anatina	3.1			behavior	4
Micropterus salmonides	10.8			behavior	2
Nereis virens	28			biochemical	5
Glycera dibranchiata	4.1	1.8	2	cellular	6
Pimephales promelas	23.6	1.1	2	growth	7
Pimephales promelas	56.4	8.9	2	morphology	7

The LOER is the concentrations as mg/kg and is associated with the lowest observed effects residue (LOER). Growth and mortality are whole body concentrations. N is number of replicates for each, which are usually different exposure times and conditions from the same study. The overall mean (sd) for all endpoints is 16.3 (17.6) mg/Kg. The mean (sd) for growth is 16.6 (9.9) mg/Kg. References are 1. Anderson et al. (1981), 2. Mathers et al. (1985), 3. Wang et al. (92), 4. Makela and Oikari (1990), 5. Carr and Neff (1981), 6. Anderson et al. (1984), and 7. Spehar et al. (1985). All concentratons as wet wt.

Table 12. Critical Body Residues for several chlorophenols.
Lethal values (LR50s).

Chlorophenol	Species	LR50 (mg/Kg)	sd	n
2,3,4,6 TeCP	Lumbriculus variegatus	216	7	2
2,4,5 TCP	Chironomus riparius	22		
2,4,5 TCP	Lumbriculus variegatus	116	13	2
2,4,6 TCP	Lumbriculus variegatus	145	21	2
2,6 DCP	Lumbriculus variegatus	240	55	2

Whole body values. LR50 is the concentration in mg/kg associated with the 50% mortality response. N is number of replicates for each, which are different exposure times from the same study. Mean (sd) with out DCP is 147 (87) mg/Kg. All values from Kukkonen (2002). All concentrations as wet wt.

2.3.3.6 Retene

Retene (7-isopropyl-1-methylphenanthrene) is a polycyclic aromatic hydrocarbon that is formed from anaerobic bacterial metabolism of resin acids (e.g. abietic or dehydroabietic acid) that are found in wood extracts and many pulp mill effluents (Wakeham et al. 1980; Hodson et al. 1997; Kovacs 1986). It is most commonly found in sediments (Judd et al. 1995; Tavendale et al. 1995), but has also been found in unbleached kraft pulp and paper mill effluent and may form from its precursors in anoxic pockets within aeration treatment ponds (Zender et al. 1994). It is potentially a concern for the Potlatch Mill because it is not eliminated by process changes designed to reduce outputs of chlorinated compounds in effluents.

Retene has a K_{OW} of 6.4 (Basu et al. 2001), so it is relatively insoluble in water, but highly persistent in sediments. Reported background concentrations range from 0.01-0.8 ug/g dry wt. in uncontaminated lakes of forest watersheds to up to 3,500 ug/g dry wt in sediments downstream of pulp mills (Wakeham et al. 1980; Leppaenen and Oikari 1999).

Retene can be taken up by aquatic organisms, and may bioaccumulate in benthic invertebrates (Nikklae et al. 2001; Moradai et al. 2001) so there is a risk of dietary exposure for salmonids and other fish. Information on BSAFs for retene is limited, however, and estimates are quite variable. In an exposure with spiked sediments, Nikklae et al. 2001 estimated BSAFs for the oligochaete worm, Lumbriculus, ranging from 0.1 to 2.5, based on initial retene concentrations in the experiment, and from 0.8 to 13.5, based on final retene concentrations.

Like other PAHs, retene is metabolized by fish, and breakdown products are excreted as metabolites in bile (Fragoso et al. 1998, 1999). Based on laboratory exposure in rainbow trout, retene metabolites can be detected in the bile within 6 hours of initial exposure. Levels decline in about 48 hours after transfer of fish to clean water. Retene exposure has been assessed in the field by measuring bile metabolites in a few studies (Lepannenen and Oikari 1999; Oikari et al. 2002), and it appears to be a useful screening technique. Moreover, some of these data suggest that the risk of retene exposure is greatest in benthic fish. Leppanenen and Oikari (1999) determined concentrations of retene metablites in the bile of roach (Rutilus rutilus) and perch (Perca fluviatilis) caught 1 to 2 km downstream of pulp and paper mills in Finland, they found significant retene exposure in roach, which primarily feed on benthic organisms, but low levels of retene in bile of perch, which feed primarily in the water column.

Because of its rapid metabolism, retene does not bioaccumulate in fish tissue; even at lethal exposure levels, concentrations are near detection limits (Billiard et al. 1999). Consequently, the risks of bioaccumulation or bioconcentration in wildlife or humans that eat the fish is minimal.

a. Direct Lethal Effects

Studies on the acute toxicity of retene to salmonids are limited. Hawkins et al. (2002) reported no mortality in juvenile rainbow trout exposed to retene at a concentration of 100 ug/L for 96 hours.

Several studies have estimated chronic lethal concentrations for early life stages of salmonids. Billiard et al. 1999 reports an LOEC of 100 ug/L for mortality in rainbow trout eggs and fry exposed to waterborne retene in a 42 day exposure. Similarly, Hawkins et al. 2002 observed 20% mortality in rainbow trout eggs and fry exposed to a retene concentration of 100 ug/L for

22 days. Retene may be more toxic to larvae in the presence of UV radiation. Hakkinen et al. 2003 found that retene was acutely lethal to larval whitefish at a concentration of (13.3 ug/L) in the presence of UV radiation, but retene alone induced no mortality at concentrations as high as 100 ug/L. Vehniainen et al. 2003 report a larval mortality LC_{50} of 41 ug/L for vendace (Coregonus albula) and an LC_{50} of 15-16 ug/L for whitefish for retene in the presence of UV-B radiation. These doses were not lethal in the absence of UV light. However, comparable data are not available for salmonid larvae.

b. Direct Sublethal Effects

(1) MFO Induction. Like many other PAHs, retene binds to the Ah receptor in salmonids and other fish, and induces the mixed function oxidase enzymes (e.g. CYP1A) that metabolize these toxicants (Fragoso et al. 1998, 1999). Although CYP1A induction itself is not necessarily an adverse health effect, it may be correlated with other, potentially harmful, changes in physiological function, such as alterations in steroid hormone concentrations and reproductive abnormalities (Munkittrick et al. 1998).

Reported threshold concentrations of retene necessary for CYP1A induction in salmonids range from 10 to 32 ug/L. Fragoso et al. 1998 found that MFO induction [as measured by EROD (7-ethoxyresorufin-o-deethylase) induction] in juvenile rainbow trout occurred at retene concentrations as low as 10 ug/L. Billiard et al. 1999 report a threshold for MFO induction of ~ 32 ug/L for larval rainbow trout.

(2) Growth Changes. Billiard et al. 1999 report reductions in growth (weight) of larval rainbow trout exposed to retene in the water column for 42 days. Trout growth at swim up was decreased, as shown by an overall decrease in biomass with increasing retene concentration. Weights of fish in higher treatments were significantly different from those of control fish for both whole fish (threshold, 180–320 ug/L) and fish without yolk sacs (threshold, 100–180 ug/L).

(3) Reproductive and Developmental Toxicity. The effects of retene on the reproduction physiology of salmonids or other fish have not been extensively studied. Retene does not appear to have any significant estrogenic activity, as Sherry et al. 1999 found that it did not induce vitellogenin production (an indicator of estrogen exposure) in brown trout. Evanson and Van der Kraak (2001) found that in vitro exposure to retene increased gonadotropin-stimulated testosterone production in goldfish testis tissue, but these effects were not seen with goldfish or rainbow trout exposed to retene in vivo, so it is hard to know the significance of this observation in wild fish. Other PAHs that interact with the Ah receptor have been associated with anti-estrogenic activity, reduced gonadal development, and altered plasma estradiol concentrations (Fairbrother et al. 1999), so retene may have similar effects, but this has not yet been established.

There is substantial evidence that retene can produce developmental abnormalities in early life stages of salmonids and other fish. Typically, it induces a syndrome that closely resembles blue-sac disease caused by dioxin and related dioxin-like compounds, as well as other high molecular weight PAHs that interact with the Ah receptor (Fairbrother et al. 1999). Blue-sac disease is characterized by pericardial and yolksac edema, craniofacial malformations, hemorrhaging, decreased growth, and increased mortality (Billiard et al. 1999).

Billiard et al. (1999) found that chronic (42 day) exposure of rainbow trout (Oncorhynchus mykiss) to retene from the eyed egg stage to hatch and from hatch to the onset of swim up behavior caused increases in blue sac disease posthatch at concentrations as low as 32 ug/L. Fin erosion and opercular sloughing were also seen in 100% of retene-exposed swim up larvae. Similar symptoms were observed in larval zebrafish (Dania rerio), but at concentratons of 320 ug/L and higher. Some newer research suggests that the true threshold effect levels for blue sac disease may be even lower. Kiparissis et al. 2003 tested the toxicity threshold for retene in a partition-controlled delivery (PCD) method that maintains the concentrations of chemicals in test solutions at or below solubility limits for extended exposure times. Concentrations are maintained by equilibrium partitioning of test chemicals from a series of poly(dimethylsiloxane) films loaded with a range of concentrations of each chemical. This method is thought to be more accurate at predicting toxicity for lipophilic compounds such as retene than traditional water column exposure assays. When Japanese medaka (Oryzias latipes) were exposed to retene in the PCD assay, the median effective concentration (EC50) for blue sac disease was 10 ug/L. In contrast, the nominal EC50 values for blue sac disease that Kiparissis et al. 2003 obtained in semi-static 24-h and static assays were 150 ug/L and 2500 ug/L.

As noted earlier, larval mortality is reported at concentrations in the 100 ug/L range under routine exposure conditions for salmonids (Billard et al. 1999; Hawkins et al. 2002), but could occur a concentrations an order of magnitude lower in the presence of UV light, based on studies with other fish species (Hakkinen et al. 2003; Vehniainen et al. 2003).

(4) Genetic Toxicity, Carcinogenicity, and other Histopathological Changes. There is some evidence that retene may be genotoxic. Gravato and Santos (2002) examined genotoxic responses, measured as erythrocytic micronuclei and nuclear abnormalities in sea bass (Dicentrarchus labrax) exposed for 6 h to 0.0125 muM (~3 ug) concentrations of retene, and found increases the frequency of nuclear abnormalities. However, in other tests of mutagenicity and carcinogenicity (e.g., the AMES test, structure-activity modeling), retene showed no activity (Muller et al. 1985, Richard and Woo 1990).

While there is little evidence that retene is carcinogenic, under certain conditions it may be hepatotoxic. Hakkinen et al. 2003 report that larval whitefish treated with retene in combination with UV light had necrotic lesions in liver at concentrations as low as 10 ug/L. These same fish also exhibited skin damage and increases in the number of mucous cells in the skin at the same exposure concentration. Without UV radiation, these symptoms did not appear at 100 ug/L, the highest concentration tested.

(5) Behavioral Effects. Hakkinen et al. 2003 report that larval whitefish treated with retene at concentrations as low as 10 ug/L in combination with UV light exhibited signs of behavioral irritation and hypoxia. Without UV radiation, these symptoms did not appear at 100 ug/L, the highest concentration tested. In the BE (EPA 2003), EPA calculated the likely threshold level for narcotic effects of retene, based on its general model for narcotic effects of PAHs. Using this methodology, they estimated an approximate effect concentrations of 0.56 ug/L, which is substantially lower than threshold concentrations for other effects documented in the literature.

(6) Toxicity of retene in sediments. Sediments are likely a major source of retene, so it is useful to know the sediment retene concentrations that associated with threshold retene concentrations in the water column for the biological effects described above. This can be calculated from the following equation:

$SQC_{oc} = K_{oc}FCV$

Where SQC_{oc} = the sediment contaminant concentration in mg/g organic carbon, K_{oc} = the partitioning coefficient for sediment organic carbon, and FCV = the water concentration, expressed in mg/L (Di Toro et al. 1991). K_{oc} can be calculated from the octanol/water partitioning coefficient, K_{ow}, using the formula:

$Log_{10}K_{oc} = 0.00028 + 0.983 \log_{10}K_{ow}$

For retene, log K_{ow} = 6.4 and log K_{oc} = 6.29.

Estimated effects for several endpoints are shown in Table 13.

Table 13. Estimated sediment retene concentrations associated with water column toxicity benchmarks, based on equilibrium partitioning.

Effect	Retene water concentration (ug/L)	Sediment concentration (mg/g OC)	Sediment concentration (ug/g dry wt; 1% TOC)	Reference
MFO induction	10	19.5	195	Fragoso et al. 1998
Reduced larval growth	100-180	195-351	1950-3150	Billard et al. 1999
Blue sac disease	10-32	19.5-62.4	195-624	Billard et al. 1999; Kiparris et al. 2003
Behavioral irritation	10	19.5	195	Hakkinen et al. 2003
Narcosis	0.56	1.1	11	EPA 2003

These estimates correspond well with the sediment retene concentrations that have been associated with MFO induction in field and laboratory studies. Oikari et al. 2002 measured CYP1A activity in fingerling rainbow trout exposed to retene-spiked or naturally contaminated sediments collected downstream of a pulp mill for four days, and observed significant increases in CYP1A activity at sediment retene concentrations in the 500-2300 ug/g dry wt range. The approximate NOEC for CYP1A activity in this study was 100-500 ug/g dry wt, depending on how long the sediment had been stored. It should be noted that the mill near the sediment collection site for this study had eliminated elemental chlorine bleaching in 1992 and used only 1% to 2% elemental chlorine bleaching between 1985 and 1992 (Oikari et al. 2002), so the dioxin content of effluent had been minimal for some time, although some residues of dioxins and furans from earlier discharges could have been present in the naturally contaminated sediments. Data on their concentrations was not available, however. Data on sediment concentrations associated with other biological effects from field data or sediment exposures are not available.

It should be kept in mind that if fish are exposed to retene-contaminated sediments in the natural environment, they will be exposed not only to retene in pore water or retene transferred to the water column, but also through other routes, especially the diet. This means that sediment concentrations estimated through equilibrium partitioning may be higher than sediment concentrations that would be associated with biological effects in the field.

c. Indirect Effects on Salmonid Prey Base

Although the number of toxicity studies that have been conducted on salmonid prey species is limited, there is some evidence to suggest that retene has the potential to harm listed salmonids through indirect effects on their prey base.

Lethal Toxicity

Some data are available on the toxicity of retene to daphnid species. Huovinen et al. 2001 report no acute toxicity in Daphnia magna exposed to retene at concentrations as high as 320 ug/L. However, in combination with UV exposure, toxicity (immobilization) was observed at this concentration, with an estimated EC_{50} of 360 ug/L.

Sublethal Toxicity

Growth Alterations**.** Nikkilae et al. 2001 assessed growth in oligochaete worms (Lumbriculus variegatus Mueller) exposed to retene in sediments at concentrations of 50 and 200 ug/g dry wt. for 28 days. No effects on growth were seen at either concentration.

Reproductive and developmental toxicity. Nikkilae et al. 2001 assessed reproduction in oligochaete worms (Lumbriculus variegatus Mueller) exposed to retene in sediments at concentrations of 50 and 200 ug/g dry wt. at three sediment organic carbon levels (1.1%, 3%, and 23%) for 28 days. No effects on reproduction were seen at any concentration.

Hamalainen et al. (1997, 2000; cited in Oikari et al. 2002) report that midge in sediments contaminated with retene showed morphological deformities, but data on sediment retene concentrations were not available. Since the study was done in a lake contaminated with pulp mill effluent, concentrations were likely within the 500-3000 ug/g range documented for sediments in the vicinity of pulp mills.

Macroinvertebrate communities. In NCASI (1996) initiated an aquatic food chain study at its experimental streams facility in New Bern, NC that evaluated the environmental disposition of 12 different neutral organic compounds, including retene, in surface water, sediment, vegetation, macroinvertebrates and fish. The design employed two control streams (Neuse River water) and two dosed streams (36% effluent, 64% Neuse River water). In these streams, retene was present in surface water and sediment at the low ppb (ng/g) range. No effects were observed on macroinvertebrate populations.

d. Toxicity Benchmarks

Results of the studies reviewed above show the following toxicity benchmarks for retene:

- Narcotic effects at concentrations as low as 0.56 ug/L, based on EPA modeling studies (or an estimated sediment toxicity threshold of 11 ug/g sediment at TOC of 1%, based on equilibrium partitioning).

- Larval toxicity of ~10 ug/L in the presence of UV radiation (in whitefish; comparable data are not available for other salmonids) (or an estimated sediment toxicity threshold of 195 ug/g sediment at a TOC of 1%.

- CYP1A induction in salmonids at concentrations as low as 10 ug/L (195 ug/g sediment at TOC of 1%); field studies indicate a sediment NOEC of 100 ug/g dry wt (Oikari et al. 2002).

- Blue-sac disease at 32 ug/L in trout; but EC50s as low as 10 ug/L in medaka using a PCD-controlled delivery system rather than a traditional waterborne exposure. In the traditional system, EC50 was 150 ug/L or more.

e. Analysis of toxicity of retene in Potlatch effluent

The current Idaho and Washington water quality standards do not specifically address retene, and proposed final permit does not limit retene in Potlatch effluents. In EPA's Opinion (EPA 2003) the implementation of best management practices (BMPs) and proper operation and maintenance of the ASB system virtually eliminates retene in the discharge, so exposure to salmonid species would be minimal. However, to our knowledge, no supporting data on the levels of these compounds in Potlatch effluent are available to confirm that this is true. Moreover, retene has not been tested for in either the Snake or the Clearwater Rivers in the Action Area, so the environmental baseline is unknown.

Some data on retene concentrations and effects are available from mill with processes similar to those used at Potlatch. A pilot study, described in the BE (EPA 2003) was conducted by NCASI in 2000, on sixteen separate mill wastewater facilities to evaluate how the treatment plant influent and effluent concentrations of neutral organic compounds (primarily diterpene compounds) covaried with concommitant changes in wastewater water quality parameters. Retene was included in the analytical suite employed in the study and was measured in a composite samples that consisted of four separate grab samples over a period of three months. Concentrations of retene observed in mill effluents and wastewaters were reportedly very low, below levels associated with blue-sac disease (~32 ug/L; Billard et al. 1999) but actual data were not available. Some data are also available on sediment concentrations in receiving water near the Buckeye Foley Mill, a dissolving sulphite mill whose effluent concentrations of retene would be comparable or higher than those from the Potlatch Mill (EPA 2003). Sediment retene concentrations found in the Fenholloway River below the Buckeye outfall were 0.45 ug/g dry wt or lower.

The retene concentrations in the effluent and sediment mentioned above are below levels associated with adverse effects in salmonids and their prey, based on the current literature. If retene concentrations in Potlatch effluent and local sediments are in a similar range, then impacts

on listed salmonids would likely be minimal. However, without data on retene concentrations in effluent and sediments in the Potlatch Mill receiving environment, this cannot be confirmed. Moreover, retene concentrations have not been measured in either the Snake or the Clearwater Rivers in the Action Area, so the environmental baseline is unknown. This information is critical for an accurate evaluation of the effects of retene on listed salmon, because even if current discharges are low, it is very likely that some levels of retene is present in sediments from historical discharges.

2.3.3.7 Resin Acids

a. Resin Acid Toxicity

Resin acids are natural plant products that are present in wood extracts and in cooking liquors and other liquid wastes from pulp mills (Pesonen and Anderson 1992; Hodson et al. 1997; Kovacs 1986). They have also been identified in foam from aerated lagoons treating kraft pulp mill effluents (Servizi et al. 1976). The most common are abietane monocarboxylic acids, including abietic acid (AA) and dehydroabietic acid (Mutton 1962, Leach and Thakore 1976). In mills where chlorine is used as a pulp bleaching agent, chlorinated resin acid derivatives are also generated (Leach and Thakore 1973, 1975; Holmbom and Lehtinen 1980; Oikari et al. 1980). Resin acids and other wood derived compounds are thought to account for a significant proportion of the toxicity of pulp mill effluents, especially those that are elemental chlorine free (Lehtinen 1996).

In addition to posing a hazard because of their own toxicity, resin acids can undergo biological hydrogenation/dehydrogenation and/or decarboxylation to produce retene (Bright et al. 1999), a PAH compound that has dioxin-like activity. Retene is more resistant to biodegradation than resin acids, and can be persistent in sediment contaminated with pulp mill effluents. Soaps of resin acids (isopimaric, abietric, and dehydroabietric acids) have also been identified as causing biologically deleterious effects in kraft mill effluents. Studies in Canada indicate that these compounds are contained mainly in combined condensates rather than black liquor. Secondary treatment is effective in removing some resin acids from effluents (Soimasu et al., 1998; Bright et al., 1999; Mikkelson and Huuskonen, 2000) by partitioning them into the organic carbon phase of suspended organic matter (floc) during the biological treatment process. This floc is eventually removed from the water column in the sludge settling process.

Resin acids can accumulate in sediments adjacent to pulp mills to relatively high concentrations. For example Leppaenen and Oikari (1999) observed total resin acid concentrations as high as 1500 ug/g dry wt in a lake in Finland receiving pulp mill effluents. Twelve kilometers downstream from the point of BKME discharge, the concentration of resin acids was 139 mu g/g dry wt. (1,700 ug/g OC). Upstream of the BKME discharge, the background concentration of resin acids was below 70 mu g/g dry wt. Substantial concentrations of resin acids (1470 ug/g d.w.) were also detected in association with suspended sediment particles collected at the sediment sampling sites.

Resin acids are taken up by fish and have been measured in blood plasma and bile of rainbow trout and other fish exposed to both bleached and unbleached kraft mill effluents (Oikari et al. 1982, 1984, 1985; Oikari and Nittyla 1985). Measurement of resin acids in bile may be an effective way of monitoring exposure to these compounds in the field, and has been employed in a several studies. For example, Tavendale et al. 1996 measured concentrations of resin acids by

GC/MS in bile collected from goldfish (*Carassius auratus*) sampled at five sites on a river receiving the biologically treated discharge from a New Zealand bleached kraft mill, and found abietanic, 13-abietenic and seco-dehydroabietic acids at concentrations up to 507 ug /g bile in fish collected near the mill's discharge point. Similarly, Karels et al. 2001 observed elevated concentrations of resin acids in bile of perch (*Perca fluviatilis* L.) and roach (*Rutilus rutilus* L.) populations downstream of two pulp and paper mills in southern Lake Saimaa, Finland. The mills used elemental chlorine-free bleaching and activated sludge effluent treatment technologies. Bile resin acid concentrations in exposed perch and roach (260-320 mu g/ml) were 10 to 30 times higher compared with the references.

Resin acids can also accumulate to some extent in fish tissues, particularly in the brain (Kruzynski 1979; Oikari et al. 1982; Pritchard et al. 1991). Bioaccumulation is not as great in the gonads and skeletal muscle (Oikari et al. 1982). Nimi and Lee (1992) determined BCFs in rainbow trout for nine resin acids (abietic, dehydroabietic, chlorodehydroabietic, dichlorodehydroabietic, neoabietic, pimaric, isopimaric, sandaracopimaric, and palustric acids), and obtained values ranging from 25-130 in fish exposed for 20 days. Elimination rates were also monitored over a 10-d period, and the half-lives of these acids were estimated at < 4 d. Invertebrates can also absorb resin acids. Burggraaf et al. 1996 showed that freshwater mussels incubated in the effluent of a kraft pulp and paper mill rapidly accumulated resin acids in their tissues, with mean BCFs for individual resin acids varying from 110 to 330. Resin acids were depurated rapidly from mussel tissue, with biological half-lives of about three days.

b. Regulatory Framework

The current Idaho and Washington water quality standards do not specifically address resin acids, so no numerical criteria have been established. Alberta and Quebec have established chronic criteria for the protection of aquatic life from resin acids of 100 ug/L and 77 ug/L, respectively (AEP 1999; MDEQ 1996).

In the Potlatch NPDES permit, EPA has not established specific limits for resin acid discharges, because it believes that the adoption of the BMPs it has promulgated to control black liquor leaks and spills (see 40 C.F.R. section 430.03 would reduce the discharge of resin acids from pulp and paper mill effluent discharges (EPA 2003). Additionally, EPA states in the BE that the measurement of BOD, which is regulated in the permit, is an adequate surrogate for measurement of resin acids (EPA 2003). The BOD is an indicator of the total organic load that is being discharged to a receiving stream, from resin acids as well as other compounds including terpenes, fatty acids, phenols, formic acid, acetic acid, sacharinic acids and other small organic acids. Studies by TAPPI (1975) indicate that these compounds are successfully removed by biological treatment, and are reduced to the same extent as overall BOD removal efficiency. Thus, in EPA's Opinion, appropriate reductions of BOD in the wastewater through biological treatment would effectively control resin acids discharges and resin acid toxicity, and it is unnecessary to consider them as a separate pollutant parameter in the permit.

c. Direct Lethal Effects

The 96-h median lethal concentrations (LC_{50s}) reported in rainbow trout range from 0.7 to 1.5 mg/L for AA, from 0.8 to 1.79 mg/L for dehydroabietic acid, and from 0.6 to 1.2 mg/L for 12,14-dichlorodehydroabietic acid (DCDHAA) (Leach and Thakore 1976; Davis and Hoos1975;

Kennedy et al. 1995; Straus et al. 1997). Leach and Thakore 1978 report median lethal concentrations (96-h LC_{50}) of seven resin acids; dehydroabietic, abietic, isopimaric, palustric, pimaric, sandaracopimaric, and neoabietic for juvenile coho salmon (*Oncorhynchus kisutch*) ranging from 0.2-0.8 mg /L in bioassays with solution replacement every 4-8 h. The toxicity of resin acids may be increased if fish are exposed to compounds that induce EROD activity. Strauss et al. 1997 found that treatment of rainbow trout with piperonyl butoxide (PBO), which induces EROD activity, reduced the mean 96-h LC_{50} for dehydroabietic acid (DHAA) from 1.79 mg/L to 1.58 mg/L. Because EROD-inducing compounds (e.g., retene, dioxins) are often found in pulp mill effluents or in sediments associated with pulp mill effluent release, this could be an important concern.

With chronic exposure, resin acid may be lethal to salmonids at concentrations substantially below the 1-2 mg/L concentration typically reported for 96-hour LC_{50} tests. Johnsen et al. 1995 found that, in 8-week exposures, a resin acid concentration of 170 ug/L was lethal to rainbow trout.

d. Direct Sublethal Effects

Jaundice/Impaired Liver Function. Jaundice, which is caused by the buildup of bilirubin in plasma, is one of the most widely documented effects of resin acids in fish (Kruzynski 1979, Matsoff and Oikari 1987, Croce et al. 1995). It can occur with acute exposure if resin acid concentrations are high enough. Nikinmaa and Oikari 1982 observed increased bilirubin levels in rainbow trout exposed to a 2 mg/L concentration of resin acids for 24 hours. Mattsoff and Oikari 1987 report that at 0.4 mg/l or higher resin acid concentrations, rainbow trout developed jaundice in 2-4 days at water temperatures over 17 degree C. Resin acids may induce jaundice at lower concentrations with chronic exposure or in combination with other contaminants. Croce and Stagg (1997) observed significant elevations in the concentrations of bilirubin in the plasma and yellow pigmentation on the ventral surface and around the gill arches of Atlantic salmon exposed for nine days to a mixture of three resin acids (isopimaric, dehydroabietic, and AA) at concentrations ranging from 10-100 ug/L in combination with diesel fuel oil, water soluble fraction (WSF) at a concentration of 2.0 mg/L.

Raised bilirubin levels in fish exposed to resin acids have been attributed to impaired liver function (Nikinmaa and Oikari 1982; Oikari and Nakari 1982), stimulation of hemoglobin release from erythrocytes (Bushnell et al. 1985) and subsequent degradation of the hemoglobin to bilirubin (Pritchard et al. 1991), blocking or depression of hepatic UDP-glucuronyl transferase activity (Matsoff and Oikari 1987; Oikari et al. 1985), inhibition of the bile acid transport through inhibition of sodium/potassium pump activity (Rabergh et al. 1992), and insufficient secretion of conjugated bilirubin into the bile (Mattsoff and Oikari 1987). Resin acids can also cause changes in erythrocyte or liver cell pH (Bogdanova and Nikinmaa 1998; Niikinmaa et al. 1999) impair cell integrity and function. Nikinmaa observed a decrease in intracellular pH of rainbow trout (*Oncorhynchus mykiss*) hepatocytes at exposure concentrations of 1.5 mg/L for isopimaric and dehydroabietic acid. These phenomena were observed in fish exposed to resin acids *in vivo*, or in *in vitro* experiments with hepatocytes or erythrocytes exposed to resin acids.

(1) Changes in hematological parameters and other biochemical alterations. Exposure to resin acids at sublethal concentrations can affect a number of hematological and metabolic parameters, even if exposure is only short-term. Kennedy et al. 1995, 1996 exposed juvenile

rainbow trout for 24 h to sublethal concentrations of 14-monochlorodehydroabietic acid (MCDHAA) or DCDHAA. Hematocrit, plasma lactate, and liver protein were significantly affected by exposure to the doses of 0.7-0.8 mg/L. Nikinmaa and Oikari 1982 exposed rainbow trout to a 2 mg/L concentration of resin acids for 24 hours. The resin acids affected respiration by causing a decrease in arterial P_O^2, red cell volume and pH, and an increase in ATP/Hb ratio. Ion balance was also disturbed as shown by a decrease in plasma chloride concentration. The fish also showed increases in plasma lactate concentration and enzyme activity that are indicative of metabolic dysfunction.

Similar effects may occur at lower concentrations with chronic exposure. Tana (1998) examined the effects of dehydroabietic acid on hematological parameters rainbow trout and found effects on blood hematocrit, leucocrit, hemoglobin, mean cellular hemoglobin count, plasma glucose, lactate, liver glycogen at exposure concentrations in the 5-50 ug/L range following exposures of several weeks. Similarly, Johnsen et al. 1995 observed reductions in hematocrit and leucocrit in rainbow trout exposed to effluent containing 85 ug/L of resin acids for 8 weeks, and reductions in leucocrit in trout exposed to 30 ug/L of resin acid for the same period of time.

(2) Behavior and Nervous System Function. Acute exposure to high concentrations of resin acids can cause resin acid poisoning, whose symptoms include abnormal reactions to auditory, visual, and tactile stimuli, and similar evidence of impaired neurological function (Oikari et al. 1982). It has been shown the resin acids cause rapid release of neurotransmitters from rainbow trout brain synaptosomes at concentrations in the 3-30 mg/L range (Zheng and Nicholson 1996, 1998), with the chlorinated derivatives of resin acids being the most potent. Few behavioral alterations have been reported in fish exposed to resin acid concentrations of 1 mg/L or less. For example, Kennedy et al. 1995, 1996 exposed juvenile rainbow trout for 24 hours to concentrations of MCDHAA or DCDHAA in the 0.7-0.8 mg/L range, but found no effect on swimming performance.

Although these tests have not been conducted with salmonids, tests with other species suggest that fish do not have an avoidance response to resin acids at environmentally relevant concentrations. Wildish et al. 1977 found no tendency for herring (*Clupea harengus*) to avoid resin acids at concentrations of 1 mg/L. Similarly, Myllyvirta and Vuorinen (1989) showed no evidence of an avoidance response in vendace (*Coregonus albula* L.) exposed to wastewater containing resin acids.

(3) MFO activity. Resin acids to not seem to be very effective inducers of CYP1A activity. There are some reports of induced MFO activity in juvenile eel (*Anguilla anguilla*) and juvenile sea bass exposed to abietic and dihydroabietic acid concentrations as low as 30 ug/L (Pacheco and Santos 1997; Pacheco and Santos 1999). However, CYP1A induction is generally not reported in studies with salmonids, exposed to resin acids. Croce et al. 1995 found no induction of induce hepatic cytochrome P450 1A (measured as EROD activity) in Atlantic salmon parr (*Salmo salar*) injected intraperitoneally with dehydroabietic acid, isopimaric acid and AA) at total resin acid concentrations up to 120 mg/kg fish weight, while Ferguson et al. 1992 found no induction of EROD activity in rainbow trout injected with resin acids at concentrations as high as 370-500 mg/kg. In waterborne exposures, the same group of researchers saw no induction at 100 ug/L after 16 days of exposure (Ferguson et al. 1992). Similarly Hewitt et al. 1996 that BKME fractions enriched in resin acids were not effective EROD inducers in sea bass (*Dicentrarchus labrax* L).

(4) Genotoxic Effects. Evidence for mutagenicity or carcinogenicity of resin acids is not strong, but some studies (Pachebo and Santos 1997, 1999; Gravos and Santos 1992) indicate that resin acids can induce nuclear abnormalities in fish erythryocytes. Pachebo and Santos 1999 observed erythrocytic nuclear abnormalities (ENAs) in adult eel Anguilla anguilla L. injected intraperitoneally with AA or DHAA, at a dose of 14.7 umol/kg, or exposed to waterborne AA or DHAA above 0.1-0.3 uM (30-90 ug/L based on MW or 302 for AA). In juvenile eels ENA frequency was increased at a concentration of 2.7 uM (~800 ug/L). In another study they observed increased erythrocytic micronuclei and nuclear abnormalities after a 2 hour exposure to 0.9 uM (270 ug/L) AA (Pacheco and Santos 1997).

(5) Histopathological Changes. Johnsen et al. 1995 observed increased necrosis of liver cells in rainbow trout exposed for 8 weeks to thermomechanical pulping effluents with resin acid concentrations of 30 ug/L and above.

(6) Growth. Johnsen et al. 1995 found no effect of resin acid exposure on juvenile/adult rainbow trout growth following at 8 week exposure at concentration as high as 85 ug/L. However, earlier life stages may be more sensitive. Sepulveda et al. 2003 observed a decrease in the weight of largemouth bass fry from female bass exposed for 56 days to mill effluents containing resin acids at concentrations of ~18 ug/L and above. Elevated resin acid levels were also observed in bile of exposed adult females.

(7) Reproductive and Developmental Toxicity. Several field studies show correlations between resin acid exposure and altered reproductive parameters in fish residing near pulp mills. For example, Sepulveda et al. 2003 exposed largemouth bass (Micropterus salmoides) up to 56 days various concentrations of bleached and unbleached kraft mill effluent (0-80%) and resin acid concentrations ranging from 2-3 ug/L to about 50 ug/L. Elevated resin acid concentrations in bile were associated with lower concentrations of estradiol (E2) and vitellogenin (VTG) in females, lower concentrations of 11-ketotestosterone (11-KT) and increased E2 in males; increased hepatosomatic index (HSI) values in females, and decreased gonadosomatic index (GSI) values in both sexes. Increased HSI was first observed at effluent resin acid concentrations of ~18 ug/L, while effects on vitellogenin and steroid hormones were first observed at ~26 ug/L. A reduction in the number of fry produced, an increase in the frequency of fry abnormalities, and a decrease in fry weight were also observed at resin acid concentrations of ~18 ug/L and above.

Kukkonen et al. 1998 found that adult male crucian carp with elevated concentrations of resin acids and beta-sitosterol in bile had higher HSI, lower GSI, and increased plasma testosterone concentrations lower gonad somatic index. These fish also exhibited reduced haemoglobin and haemocrit levels, and changes in plasma thyroid hormone concentrations (higher T_3, but lower T_4). Karels and Oikari (2000) observed elevated plasma estradiol and testosterone concentrations in pre-spawn female and male perch and male roach with high resin acid concentrations in bile collected downstream of two pulp and paper mill sites in Finland. Soimasuo et al. (1998a, 1998b) also observed significant reductions in plasma concentrations of 17 beta-estradiol (37%) and testosterone (40%) in whitefish (*Coregonus lavaretus* L.) exposed for 30 days secondary treatment effluent from an elemental chlorine free pulp mill at concentrations of 3.5% and above, but resin acid levels in these effluents were quite low (~5 ug/L in 100% effluent) and fish

exposed to the 3.5% concentration showed no resin acid accumulation in bile, even though plasma hormone concentrations were altered. This suggests that compounds other than resin acids were responsible for these changes.

There are also some reports that pulp mill effluents containing resin acids may have estrogenic activity. Mellanen et al. 1999 and Soimasu et al. 1998 observed expression of the vitellogenin gene, a biomarker of exposure to estrogenic effluents, in juvenile whitefish (*Coregonus lavaretus* L.) caged in pulp mill effluents containing resin acids and phytosterols. However, in another study, Kukkonen 1998 found no vitellogenin production in the exposed male fish and some females with high concentrations of resin acids in bile showed decreased vitellogenin levels. Unfortunately, it is difficult to attribute these effects to resin acids because fish were exposed to whole effluents containing other compounds. Controlled exposure studies that would establish a clear cause and effect relationship between resin acid exposure and alterations in reproductive function are lacking.

(8) Immune Function. Short-term exposure to high concentrations of resin acids can reduce disease resistance in salmonids. Kennedy et al. 1995, 1996 exposed juvenile rainbow trout for 24 h to sublethal concentrations (0.7-0.8 mg/L) of MCDHAA or DCDHAA. Resistance to infection by the bacterium Aeromonas salmonicida was significantly reduced; cumulative percent mortalities in fish exposed to MCDHAA or DCDHAA reached 20 and 26%, respectively. The reduction in disease resistance was associated with 3 to 14-fold increase in plasma cortisol concentrations. However, little is available on the impact of chronic exposure to lower concentrations of resin acid on immune function or disease resistance. Johnsen et al. 1995 observed a decrease in leucocrit in rainbow trout exposed for 8 weeks to effluents containing 30 ug/L of resin acids. This could contribute to reduced disease resistance, but disease challenge studies to confirm this have not been conducted.

e. Indirect Effects

Information on the effects of resin acids on salmonid prey and other benthic organisms is very limited. Resin acids can be taken up be invertebrates (Burggraaf et al. 1996) so could be passed on to fish through diet, but the effects of resin concentration on salmonid prey and benthic communities themselves are less clear. Historical studies of benthic communities changes associated with effluent loadings from pulp mills suggest that these compounds may have a significant impact on some salmonid prey species. Merilaeinen et al. 2001 found that changes in profiles of resin acids (pimaric, isopimaric, sandaracopimaric, dehydroabietic and AA) in sediment cores from a Finland lake were associated with distinct changes in chironomid assemblages in benthos, which are an important food source for juvenile salmonids. However, sediment resin acid concentrations in this study area were quite high, up to 1100 ug/g dry wt (Leppaenen and Oikari 2001).

Dube and Culp (1996) studied growth responses of chironomids exposed to biologically treated BKME using artificial streams in the Thompson River, British Columbia, Canada. Effluent concentrations tested were 0.25, 0.5, 1.0, 5.0, and 10.0%. They found that chironomid growth (total biomass and individual weight) was significantly stimulated at low effluent concentrations (less than or equal to 1.0%). The effect was less marked at higher concentrations (5.0 and 10.0%), but was still greater than in the controls. Increases in growth were attributed to the effects of nutrient and organic enrichment from BKME, especially the high concentration of

phosphorus. However, because resin acid concentrations in the effluents were low (2-4 ug/L) it was not clear that they were responsible for inhibiting chironomid growth at the higher effluent concentrations. Similarly, Tana et al. 1994 examined the effects of effluents from various bleaching processes on aquatic invertebrates in mesocosm exposures, but at least in effluents from newer bleaching technologies such as those currently employed at Potlatch, was unable to correlate resin acid concentrations in effluents with toxic effects.

f. Studies with Potlatch Effluent

As noted in the BE, over the past 20 years, many toxicity studies have been conducted using the whole effluent from the Potlatch Mill, including a series of seven studies conducted by the (NCASI) from 1980 to 1994. Five experimental stream studies were conducted in the 1980s before Potlatch converted to chlorine substitution (pre-conversion studies); one study was conducted in 1991 on effluent that had undergone chlorine substitution; and a final study was conducted in 1993-1994 on effluent that had undergone both chlorine substitution and oxygen delignification. The studies conducted from 1991-1994 are described in NCASI 1997 and Haley et al. 1995. In these studies, fish were exposed to two concentrations of effluent, 1.5% and 5%.

Resin acid concentrations in effluents were monitored in a number of these studies. In the initial study conducted in 1980, the total resin acid concentration in 100% effluent was 1142 ug/L, and in subsequent pre-conversion studies, concentrations ranged from 105-204 ug/L. Resin acid concentrations were also monitored in the treatment streams monitored in one artificial stream experiment conducted with Potlatch pre-conversion effluent (NCASI 1989). In this study, resin acid concentrations ranged from 1.9-4.1 ug/L. Conversion to chlorine substitution and oxygen delignification did not seem to affect resin acid levels to any great degree. In the first post-conversion study, with chlorine substitution only, the resin acid concentration was 61 ug/L; in the study with chlorine substitution and oxygen delignification, it was 205 ug/L. If we assume that typical resin acid concentrations of 100% effluent would be up to about 200 ug/L, then in 5% effluent, the concentration would be ~10 ug/L, while in 1.5% effluent, it would be ~3 ug/L.

Given these assumptions, the results of the Experimental Stream studies indicate that:

- There was no clear evidence of reduced survival due to effluent exposure at resin acid concentrations of ~10 ug/L or below over a time period of about six months.

- Gonadal growth and sex ratios were not affected by exposure to effluent containing 10 ug/L resin acids and below.

- Some pathological changes were observed in trout from effluent stream containing 10 ug/L resin acids (fatty change in the liver and lateral line degeneration), but none in the effluent containing 3 ug/L resin acid.

- Hematological parameters (e.g. hematocrit, leucocrit, plasma proteins) were monitored and no differences were found between treatment and control streams for effluents containing 10 ug/L resin acids or below. In the one pre-conversion study in which resin acid concentrations in treatment streams was actually measured, no effects were seen in these parameters at resin acid concentrations of 4.1 ug/L (the highest concentration measured).

- Larval survival studies were conducted with preconversion effluent using instream incubation boxes. Exposure of eyed rainbow trout, chum salmon, Chinook salmon, and coho salmon eggs during their incubation period to effluents with resin acid concentrations of 10 ug/L or below had no effect on their survival to hatching or on the weight of larvae at the time of hatching (NCASI, 1984).

- In effluent dilution studies (EPA 2003; NCASI 1982, 1984), acute 96-hour NOEC concentrations for rainbow trout eggs ranged from 18-100%, with most results in the 32-56% range, which would correspond to resin acid concentrations of ~64-111 ug/L.

- In chronic effluent dilution experiments (EPA 2003; NCASI 1982, 1984), NOEC concentrations for survival and growth of steelhead, rainbow trout, chinook, chum, and coho eggs were generally in the 5-10% range, depending on the length of exposure (~30-100 days). The lowest value obtained was a 5.6% LOEC for survival for rainbow trout early life stages with a 51 day exposure, and LOECs of 5.6% for growth for early life stages of chinook and chum salmon with a 106 day exposure. The estimated resin acid concentration in 5.6% effluent would be ~11 ug/L.

In whole bioassay and whole life cycle tests which may be relevant to effects on salmonid prey:

- In WET tests conducted with Potlatch effluent from 1999-2002, the typical NOEC for impacts on survival, growth, or reproduction of *C. dubia* or *P. promelas* was 10% effluent. This would correspond to an estimated resin acid concentration in effluent of 20 ug/L, if resin acid concentrations are similar to those measured in post-conversion effluents used in artificial stream studies.

- In whole life-cycle studies on fathead minnows with Potlatch effluent (EPA 2000), the NOEC was 10% effluent, corresponding to an approximately resin acid concentration of 20 ug/L.

- In summary, toxicity tests with Potlatch effluent suggest that, in effluent with a resin acid concentrations of 10 ug/L and below, the risk of increased mortality and most sublethal effects (reduced growth, hematological changes, reduction in gonadal growth) in juvenile and sub-adult salmonids is low. Pathological lesions and effects on survival and development of early life stages could occur at this level, but would be unlikely at concentrations of 3 ug/L and below. Decreased survival, growth, and reproduction of invertebrate prey are unlikely at resin acid concentrations of 20 ug/L and below, although in one study effects on *Ceriodaphnia* reproduction was observed at an estimated resin acid concentration of 20 ug/L.

g. Toxicity Benchmarks

Results of the studies reviewed above show the following toxicity benchmarks for direct effects of resin acids on salmonids:

- 96-hour LC_{50} values for juvenile salmonids as low as 200 ug/L

- 76% mortality of rainbow trout at with chronic exposure (8 weeks) at concentrations of 170 ug/L

- Acute 96-hour NOEC concentrations for rainbow trout eggs in the 64-111 ug/L range, based on studies with preconversion Potlatch effluent

- A LOEC of 30 ug/L for liver necrosis in rainbow trout with chronic exposure (8 weeks)

- A chronic (9 day) LOEC of 10 ug/L for increased bilirubin levels in Atlantic salmon in mixtures with fuel oil, or 400 ug/L alone

- A chronic LOEC of 5 ug/L for alterations in hematological parameters (blood hematocrit, leucocrit, hemoglobin, mean cellular hemoglobin count, plasma glucose, lactate, liver glycogen) in rainbow trout: LOECs in the 20 ug/L range are more commonly reported.

- A chronic NOEC of ~4 ug/L for effects on hematocrit, leucocrit, and liver somatic index from studies with Potlatch pre-conversion effluent

- A chronic NOEC of 3 ug/L and LOEC of 10 ug/L for tissue pathology, based on based on resin acid concentrations measured in stream studies with Potlatch effluent

- A chronic LOEC of ~10 ug/L for effect on survival and growth of salmonid early life stages, based on resin acid concentrations measured in stream studies with Potlatch effluent

Benchmarks based on studies with other species include:

- ENA in adult eel Anguilla anguilla exposed to resin acids at concentrations above 30 ug/L

- Reduced fry production, reduced fry weight in largemouth bass with chronic maternal exposure (8 weeks) at resin acid concentrations as low as ~18 ug/L

For indirect effects on salmonid prey:

- There are insufficient data from controlled exposure studies with resin acids alone to conclusively establish a toxicity benchmark for their indirect effects on salmonid prey.

- A chronic NOEC of 20 ug/L is estimated for effects on growth and reproduction of prey based on resin acid concentrations measured in stream studies with Potlatch whole effluent.

- Analysis of effects of resin acids in Potlatch effluent

h. Analysis of effects of resin acids in Potlatch effluent

In its analysis in the BE, EPA states that the permit implements BMPs that are a key component to control black liquor leaks and spills from bleached paper grade kraft and paper grade sulfite pulp and paper mills (see 40 C.F.R. section 430.03) that focus on the necessary infrastructure and process operations to minimize the discharge of black liquor to wastewater treatment systems. The EPA believes that the implementation of BMPs and proper operation and maintenance of the

ASB system virtually eliminates resin acids in the discharge. The EPA conducted studies of resin acids in pulp and paper mill effluents when developing effluent guidelines (EPA 1982). The studies show that the biological treatment system effectively removes 85% resin acids. Therefore, they anticipate that exposure to salmonid species would be minimal, and conclude that resin acids 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..

No data are available on current resin acid levels in Potlatch effluent, but concentrations of resin acids were measured in Potlatch effluent from the NCASI stream studies conducted between 1980 and 1993. In the initial study conducted in 1980, the total resin acid concentration in 100% effluent was 1142 ug/L, and in subsequent pre-conversion studies, concentrations ranged from 105-204 ug/L. Conversion to chlorine substitution and oxygen delignification did not seem to affect resin acid levels to any great degree. In the first post-conversion study, with chlorine substitution only, the resin acid concentration was 61 ug/L, while in the study with chlorine substitution and oxygen delignification, it was 205 ug/L. If current practices as similar to those adopted in 1993, then resin acids have not been eliminated from the discharges. Furthermore, toxicity data for resin acids suggests that if resin acid concentrations in 100% effluent are still in the 100-200 ug/L range, they could be high enough to have toxic effects on salmon, and could even be lethal for sensitive life stages or extended periods of exposure.

Based on analyses of 100% effluent, in NCASI stream studies with1.5-5% Potlatch effluent, estimated resin acid levels would have ranged from 5-10 ug/L for 5% effluent and 1.5-3 ug/L for 1.5% effluent. The concentrations in the 5% effluent are equivalent to the lowest levels at which effects are reported in the literature, while concentrations in the 1.5% effluent are below reported effect levels. This suggests that benchmark concentration of 1.5% recommended by EPA for direct effects of whole effluent on juvenile and adult salmon (EPA 2003) would also be protective for effects of resin acids. In fact, a report on the experimental stream studies (NCASI 1989) confirms that in one of the studies in which resin acid concentrations in the treatment streams ranged from 1.9-4.1 ug/L, no effects were observed on any pathological endpoints measures, including hematocrit, leucocrit, and liver somatic index. However, exposure to less dilute effluents (>5-100%) could have negative impacts on salmon health. In the analysis for WET, EPA estimated that BKME concentrations would remain above the toxicity benchmark for fish for 15 miles beyond the discharge (EPA 2003). It follows that resin acid concentrations could also remain above the toxicity benchmark for a substantial distance beyond the discharge. This suggests that resin acid in effluent would be likely to adversely affect salmonids.

Moreover, resin acid concentrations have not been measured in either the Snake or the Clearwater Rivers in the Action Area, so the environmental baseline is unknown. Even if current discharges are low, resin acid could be present in sediments from historical discharges. Also, resin acids have been reported to undergo biological hydrogenation/dehydrogenation and/or decarboxylation to produce retene (Bright et al., 1999), which accumulates in sediments and is resistant to biodegradation than the resin acids themselves. Because of the lack of baseline data, it is difficult to know the extent to which current discharges resin acid from the mill could contribute to historical levels already in the environment.

2.3.3.8 Phytosterols

a. Introduction

Phytosterols (plant sterols) are triterpene compounds that are important structural components of plant membranes (Moreau et al. 2002). More than 200 different types of phytosterols have been reported in plant species (Moreau et al. 2002). They are frequently found in sediments that are enriched in plant-derived organic matter (Hassett and Lee 1977; Ghosh et al. 1990; Fattore et al. 1996; Mudge et al. 1999) and are present as wood-product derivatives in effluents from pulp mills using a wide variety of processes, including those that are elemental chlorine free (Cook et al. 1996).

The most common phytosterols in pulp mill effluents include beta-sitosterol, stigmasterol, stigmastanol, and campesterol, with beta-sitosterol usually being present at the highest concentrations. Concentrations can be relatively high. In one survey of 22 U.S. pulp and paper mills, total sterol concentrations ranged from 71 to 535 ug/L, with beta-sitosterol being the major plant sterol at all the sampling sites (Cook et al. 1996). Tremblay and Van Der Kraak (1999) report total sterol concentrations of 208 and 124 ug/L for two Canadian pulp mills, and beta-sitosterol concentrations of 86 and 79 ug/L. Data on phytosterol concentrations in sediments associated with pulp mills are limited; Leeming and Nichols (1998) report total sterol concentrations in the 30-110 ug/g dry wt range in sediments from the Upper Derwent Estuary in Tasmania that were closest to BKME discharges, and from 0.6 to 8.1 ug/g dry wt at sites further from mill sites. Reported phytosterol concentrations in sediments from other sites that are not specifically associated with pulp mills range from < 1 to ~30 ug/g dry wt (Hasset and Lee 1977; Mudge et al. 1999).

Phytosterols have received considerable attention from the medical community in recent years because of their cholesterol-lowering properties (Moreau et al. 2002). There is also some speculation that they may reduce the risk of certain cancers and have immunomodulatory properties (Atif et al. 2000; Bouic 2002), but this has not been fully confirmed.

In fish, they may have reproductive effects and could be responsible for some types of reproductive dysfunction observed in fish exposed to pulp mill effluents (Lander et al. 1994; Tana et al. 1994).

In addition to phytosterols, isoflavone plant compounds, such as genistein have recently been identified in pulp mill effluents at concentrations in the 10-25 ug/L range (Kiparissis et al. 2001). Laboratory studies with mammalian models have shown that genistein is an estrogen agonist (Birt et al. 2001), but its effects on teleost fish are not well-studied, and the extent to which it could contribute to estrogenic activity in pulp mill effluents in unclear.

b. Direct Lethal Effects

No information was available on lethal effects of phytosterols on adult or juvenile fish, but increased mortality of eggs larvae is reported at beta-sitosterol concentrations of 10 ug/L and above in brown trout (Lehtinen et al. 1999). At an exposure concentration of 20 ug/L, Lehtinen et al. (1999) observed a 28% mortality of unhatched trout roe.

c. MFO Activity

Phytosterols are generally not considered strong CYP1A inducers, although there are a few reports of increased MFO activity in fish exposed to physterols. Tremblay and Van Der Kraak (1999) report a statistically significant but small (less than 1-fold) increase in MFO activity in trout exposed to beta-sitosterol at concentrations of 75 ug/L and above, and Lehtinen et al. (1999) reports increased EROD activity in female lake trout exposed to beta-sitosterol. On the other hand, Lehtinen et al. (1992) found no significant EROD induction in rainbow trout after a 2-week in vivo exposure to beta-sitosterol in a mesocosm experiment, and MacLatchy and Van Der Kraak (1995) found no increase in EROD activity in goldfish injected with beta-sitosterol. In studies with whole effluents, no correlations have generally been found between beta-sitosterol concentrations in BKME and EROD activity (Hewitt et al. 1996).

d. Reproductive and Developmental Effects

Several studies indicate that exposure to phytosterols may disrupt reproductive processes in fish. Beta-sitosterol, the most widely studied of the phytosterols present in BKME, has been shown to affect the fish reproductive system through several modes of action.

There is some evidence that beta-sitosterol may have estrogenic activity. It in vitro studies, it was able to bind to the rainbow trout estrogen receptor and induced the production of vitellogenin (the estrogen-inducible egg yolk protein) in primary cultures of trout liver cells (Tremblay et al. 1995; Tremblay and Van Der Kraak 1998). Beta-sitosterol has also shown estrogenic activity in human breast cancer cell lines (Mellanen et al. 1996). In live juvenile rainbow trout, it induced expression of the vitellogenin gene in trout liver and induced plasma vitellogenin production (Mellanen et al. 1996; Tremblay and Van Der Kraak 1998). Tremblay and Van Der Kraak (1998, 1999) observed vitellogenin production in sexually immature rainbow trout exposed to beta-sitosterol for 21 days at concentrations of 25 ug/L and above. Nakari and Erkomaa (2003a,b) report vitellognin induction in zebrafish exposed to a phytosterol preparation containing mainly with beta-sitosterol at similar exposure concentrations.

Beta-sitosterol may also decrease plasma sex steroid levels and in vitro gonadal steroid production in fish. Tremblay and Van Der Kraak (1998, 1999) found that trout exposed to a 75 ug/L concentration of beta-sitosterol for three weeks had significantly reduced plasma levels of testosterone. Pregnenolone levels declines at levels as low as 25 ug/L and cholesterol levels at 10-25 ug/L. However, beta-sitosterol exposure had no effect on gonad size, liver size, or condition of exposed trout in these experiments. Beta-sitosterol has similar effects in male goldfish, reducing the plasma levels of reproductive sex steroids after intraperitoneal injection (MacLatchy and Van Der Kraak 1995) or waterborne exposures at concentrations as low as 75 ug/L (MacLatchy et al. 1997; the dosage range was 75-1200 ug/L for this study). More recently, Leusch and MacLatchy (2003) found that in adult male goldfish exposed for 24-31 days to beta-sitosterol (55% of a phytosterol mixture or 96% pure; 150 microg/g; Silastic implant), the size of the mitochondrial pool of reactive cholesterol was decreased in the testes, suggesting beta-sitosterol is impeding cholesterol transfer across the mitochondrial membrane. Because cholesterol is a precursor for sex steroids, this could account for the reduction in steroid hormone synthesis associated with beta-sitosterol exposure. However, in another study, Dube and MacLatchy (2001), using a toxicity identification evaluation approach, found that plant phytosterols were probably not responsible for depressed testosterone levels they observed in mummichog exposed to BKME.

Phytosterols may have developmental effects on eggs and larvae exposed to these compounds either directly or through maternal transfer. Lehtinen et al. 1999 exposed maturing brown trout (*Salmo trutta lacustris*) of both sexes to 10 and 20 ug/L doses of phytosterols (mainly beta-sitosterol) for 4.5 months before spawning. Eggs from preexposed females were artificially fertilized with milt from preexposed males in clean water, and then incubated in clean water until hatching. Yolk sac fry were followed until swim-up, and mortality as well as deformities was recorded. In the phytosterol-exposed fish, there was a increased dose-dependent egg mortality, smaller egg size, lower mean weight of the yolk sac stage larvae, and a higher prevalence of deformed larvae. These effects were first observed at 10 ug/L and were greater at 20 ug/L, with 28% mortality of unhatched roe at the 20 ug/L concentration. Effects on larvae and fry were similar whether or not males were exposed to beta-sitosterol, indicating the effect was linked to female exposure. A dose-dependent increase in phytosterols in the roe was also observed. Exposed female fish exhibited higher plasma estradiol, higher 7-EROD, and slower egg maturation, whereas exposed males showed accelerated sexual maturation.

Mattsson et al. 2001 exposed viviparous blenny, *Zoarces viviparous* L., to phytosterols from oogenesis to parturition at concentrations of 0, 10, 20, and 30 ug/L and from breeding to parturition at concentrations of 0, 10, and 20 ug/L. After parturition the offspring were reared either in clean or in water containing phytosterol at the same concentration to which the parents were exposed, and their reproductive function was tested after they matured. The results showed that phytosterol exposure to the parental generation affected embryological development of the larvae before hatching, and also altered levels of circulating hormones in the parent fish. The larvae contained higher levels of phytosterols than controls, indicating that there was maternal transfer of these compounds. Exposure during parturition appeared to have a greater effect on larvae than exposure after parturition. Effects observed included stimulation of larval growth at 10 ug/L during parturition, with other embryological effects at the higher concentrations.

Phytosterols may also cause changes in sexual behavior or masculinize female fish. In controlled exposures to beta-sitosterol, female mosquitofish were masculinized, showing changes in gonopodium and anal fin development (Howell and Denton 1989; Denton et al. 1985). Similarly, Krotzer 1990 found that female mosquitofish (*Gambusia affinis affinis*) exposed to degraded products of a 65% stigmastanol-30% B-sitosterol showed masculinized behavior, following and thrusting their gonopodia at non-treated females and larger masculinized females. Nakari and Erkomaa (2003a,b) report altered sex ratios in zebrafish exposed continuously across three generations to a phytosterol preparation with beta-sitosterol as the main component. In generation F1, the predominant sex was male, while in generation F2 it was female. These impacts were observed at an exposure concentration of 8.2 ug/L.

It is possible that genistein, an isoflavone plant derivative that has been reported in pulp mill effluents, could contribute to reproductive and developmental problems in fish exposed to BKME. Genistein has been shown to have estrogenic activity in fish or cause other reproductive problems. Intraperitoneal injections of genistein and to yearling Siberian sturgeon (*Acipenser baeri*) induced vitellogenesis (Pelissero et al. 1991). In Japanese medaka (*Oryzias latipes*) exposed intraperitoneally to genistein, vitellogenin was not induced in males and females, but plasma 17 beta-estradiol was increased in exposed females and plasma testosterone levels were reduced in exposed males (Zhang et al. 2002). When genistein-enriched diets were fed to rainbow trout (*Oncorhynchus mykiss*) for one year, gametogenesis and reproductive efficiency were reduced (Bennetau-Pelissero et al. 2001).

In a study by Kiparissis et al. 2003, waterborne exposures of Japanese medaka (*Oryzias latipes*) to genistein, 1,000 µg/L, caused a low incidence (i.e., 12%) of gonadal intersex in male medaka. The ovaries of female medaka showed delayed oocyte maturation, atretic oocytes, an enlarged ovarian lumen, proliferation of somatic stromal tissue, and primordial germ cells; responses were concentration dependent. In treatments with 1,000 ug/L genistein, 72% of male medaka showed feminized secondary sex characteristics. However, these concentrations are much higher than those measured in pulp mill effluents (10-15 ug/L; Kiparissis et al. 2001) so it is not clear that such effects would occur in wild fish near pulp mills.

e. Metabolic Effects

A number of studies have shown that plant sterols can lower plasma cholesterol levels, and these substances have even been used to manage cholesterol levels in people at risk for heart disease (Katan et al. 2003). Studies have shown that these substances can also affect plasma cholesterol and lipid levels and metabolism in fish and wildlife. MacLatchty et al. (1997) and Tremblay and Van Der Kraak (1998, 1999) have both shown reductions in serum cholesterol levels in rainbow trout and goldfish exposed to beta-sitosterol at doses in the 10-75 ug/L range. Gilman et al. (2003) exposed male brook trout (*Salvelinus fontinalis*) and goldfish (*Carassius auratus*) to a phytosterol mixture (72% beta -sitosterol) with an intraperitoneal implants, and found that low-density lipoprotein cholesterol and triglyceride levels decreased significantly, 43 and 50%, respectively, in phytosterol-treated fish. Similarly, Nieminen et al. 2002 found that phytosterols affected plasma lipid levels and metabolism in the European polecat. In polecats exposed to phytosterol, both liver glycogen content and kidney glucose-6-phosphatase activity increased, but liver lipase esterase activity decreased. Although total serum cholesterol did not change, low-density lipoprotein levels increased and the high-density lipoprotein-cholesterol ratio decreased. Whether such changes would have adverse effects on fish or wildlife health is not clear.

f. Mutagenic and Genotoxic Effects

There is no evidence that phytosterols have genotoxic or mutagenic effects. Wolfreys and Hepburn 2002 assessed the mutagenic potential of phytosterols and phytosterol esters in a bacterial mutation assay and an in vitro chromosome aberration assay. In addition, an in vitro mammalian cell gene mutation assay and two in vivo mutagenicity studies (i.e., rat bone marrow micronucleus and liver unscheduled DNA synthesis (UDS) assays) were conducted on phytosterol esters only. Phytosterols and phytosterol esters did not show any evidence of mutagenic activity in any of these assays.

g. Effects on Immune Function

Some studies suggest that phytosterols may have anti-inflammatory and immunoregulatory properties, and they have been proposed as treatments for immune system abnormalities (Bouic 2002). However, no data are available on their immunomodulatory effects on fish.

h. Indirect Effects

Invertebrate prey organisms that feed on organic matter can take up phytosterols, and concentration have been measured in muscle and hepatopancreas of several benthic species (Shetty and Sherma 1990; Napilitano et al. 1993; Avery et al. 1998). Consequently, there is the potential for dietary uptake of these compounds by salmonids.

However, little information is available on the health effects of phytosterols on invertebrates. Some studies have shown that phytosterols are not acutely toxic to *Daphnia magna* at concentrations up to their solubility maximum (ISO 6341:1996). Czech et al. 2001 studied the effect of beta-sitosterol on reproduction of the gastropod snail, *Lymnaea stagnalis*. Recently matured adult snails were exposed to beta-sitosterol at 1, 10 and 100 ng/L for 7-12 weeks. Exposed snails showed atrophy of the albumen gland, but there was no effect on shell height and weight or mortality of adult snails, or fecundity or fertility of adult snails or their offspring.

i. Toxicity Benchmarks

Based on the studies reviewed above, the LOEC for phytosterols is 10 ug/L; exposure at this level is associated with egg mortality and developmental abnormalities in early life stages, and metabolic alterations such as changes in cholesterol levels in juvenile and adult fish. There are not sufficient data to identify toxicity benchmarks for effects on salmon prey.

j. Analysis of Effects of Phytosterols in Potlatch Effluent

To our knowledge, no data are available on phytosterol concentrations in Potlatch effluent. Phytosterols have not been tested for in either the Snake or the Clearwater Rivers in the Action Area, so the environmental baseline is unknown.

The current Idaho and Washington water quality standards do not specifically address phytosterols, and proposed final permit does not limit phytosterols because according to EPA (EPA 2003), their levels should be effectively controlled through the effluent limitation for BOD. In their analysis in the BE, EPA states that through BMPs to control leaks and spill of black liquor that contains phytosterols (see 40 C.F.R. section 430.03), and appropriate reductions of BOD in the wastewater through biological treatment (e.g., activated sludge treatment (AST) or an ASB system, compounds such as phytoserols, terpenes, resin acids, fatty acids, phenols, formic acid, acetic acid, sacharinic acids and other small organic acids can be virtually eliminated from kraft mill wastewaters (TAPPI 1975), substantially lowering the toxicity of the effluent.

However, to our knowledge, no supporting data on the levels of these compounds in effluent are available to confirm that this is true. Moreover, in their analysis of WET, EPA determined that 100% Potlatch effluent would be likely to have adverse effects on listed salmonids and their prey, and would require dilution to a concentration of approximately 1.5-3% to reach a non-toxic level (EPA 2003). This indicates that current practices to reduce BOD are not completely effective in eliminating effluent toxicity, although their effectiveness in eliminating phytosterols is unknown. In the BE (EPA 2003), EPA states that in its opinion, the implementation of BMPs and proper operation and maintenance of the ASB system virtually eliminates phytosterols in the discharge,

so exposure to salmonid species would be minimal, and they would not be adversely affected by these compounds. However, in the absence of any supporting data on concentrations of phytosterols in effluent, this cannot be confirmed.

2.3.4 Organic Pollutants: Summary of effects

2.3.4.1 Effects on habitats from waterborne organic contaminants

a. Adverse Effects of Effluents on Habitats: Hydrodynamic modeling of the downstream extent of water benchmarks exceedences

The purpose of the modeling exercises was to estimate the extent of Snake River habitat adversely affected by waterborne contaminants contained in the Potlatch pulp mill effluent discharged from Outfall 001 through a diffuser into the Snake River. The intent was to predict realistic scenarios that, based on recent history, can reasonably be expected to occur in the project area. The evaluations are not intended to represent worst-ever cases, but to capture conditions that occur with a predictable frequency over a 10-year time period. A 10-year time period encompasses about 2 –3 generations of salmonids that use the habitats, and two NPDES 5-year permit cycles. In the past, NPDES permits have often been administratively extended so that the statutory "5-year" permits may effectively control discharges for 10 or more years. For example, as of February 2002, the Potlatch Lewiston Mill is authorized to discharge under a 1992 NPDES permit.

An important limitation of this modeling effort is that it is limited to predicting waterborne effects only. That is, it predicts the distance and time required for waterborne effluents to be diluted by the ambient river flows to meet benchmarks to protect listed salmonids. It does NOT predict the extent of potential sediment deposition or contamination or possible food web based effects to aquatic life resulting from possible sediment contaminations.

The CORMIX version 4.2GT was used for modeling (Internet http://www.cormix.info/). The CORMIX consists of a series of software subsystems for the analysis, prediction, and design of aqueous discharges into watercourses, with emphasis on the geometry and dilution characteristics of the initial mixing zone, including the evaluation of regulatory requirements (Jirka et al. 1996). The CORMIX is probably the most widely used of several dilution models for effluent dispersion that EPA has supported.

Jirka et al. (1996) remind their readers that hydrodynamic modeling by any known technique is not an exact science. However, extensive comparison with field and laboratory data has shown that the CORMIX predictions on dilutions and concentrations (with associated plume geometries) are reliable for the majority of cases and are accurate to within about ±50% the standard deviation of model predictions. In addition, CORMIX displays a high degree of flexibility in predicting a wide variety of flow possibilities, including various flow patterns and boundary interactions. As a further safeguard, CORMIX will not give predictions whenever it judges the design configuration as highly complex and uncertain for prediction.

b. Selection of modeling scenarios

The selection of modeling scenarios was designed to run realistic chronic exposures to ESA listed salmonids with the vicinity of the diffuser in the Snake River. The scenarios considered included the lowest monthly-average flow that occurred in the last 10-years during times when salmon smolts would present (May-July), the lowest monthly-average flow that occurred in the last 10-years at any time of the year, and the lowest 7-day average flow occurring at any time of the year during the last 10 years (7Q10). The lowest flows-at-any-time of the year scenarios were included because some sub-yearling chinook are assumed to rear year-round in the reservoir. Also, because after near-field turbulent mixing ends, predictions of dilution are based on passive diffusion and buoyancy become sensitive to small differences in model inputs, another purpose was to compare various concentrations (i.e. multiples of benchmarks) to estimate the spatial extent of adverse habitat modification and the time that drifting organisms would be in plume concentrations.

For the modeling to be comparable to an extensive set of modeling conducted by Kristine Koch, EPA (EPA 2003), except as noted, the same ambient, effluent, and discharge configurations were used. Various baseline concentration assumptions, flows and benchmarks were used.

Options for flows to conduct additional modeling with that were considered were:

Lowest anytime monthly average: 9765 cfs which occurred in July 1992. This is the minimum long-term (30-day average) flow occurring in the last 10-years. This could be appropriate for analyzing effects that fraction of sub-yearling chinook that might remain in LGR for some extended period. 10-years was selected because it is assumed to relate to current climate and dam operation practices. This was nearly identical to a scenario of 9693 cfs that EPA (2003; K. Koch, personal communication) modeled using horizontally stratified Snake River and 1Q10 flows by month (Table 14). Therefore there was no need to make redundant model runs just for that purpose.

Table 14. Flows calculated for EPA in 2003 BE:

	July	August	September	October	December
1Q10	11,116	9970	9693	11,822	12,660
7Q10	12,461	10,577	11,028	12,861	13,252

7Q10. The 7Q10 is the flow statistic traditionally used by EPA along with chronic aquatic life criteria for calculating effluent limits under critical conditions. This flow statistic is assumed to correspond reasonably well with the frequency and averaging periods for chronic criteria (4-day average concentrations). The 7Q10, the lowest consecutive seven-day average flows in a 10-year recurrence interval, are assumed to correspond with the duration of exposure that could begin causing long-term sublethal type effects, and is at an infrequent enough return frequency to be uncommon, assuming that aquatic populations can tolerate periodic exceedences of criteria without harm (EPA 1997). The lowest 7Q10 calculated by EPA (2003) was 10,577 cfs (Table 14).

Lowest monthly average occurring during May, June, or July smolt outmigrant period in the last 10-years ("Smolt 30Q10"): July 2001, 15010 cfs, which is very similar to the June-December 2001 (7-month average) flows of 15178. This is to account for a somewhat conservative but realistic conditions that chinook smolts would encounter during their May-July outmigration in a dry year. This scenario of sustained low-flows occurring during the May-July time frame during a dry year is not a worst case but commonly occurring situation. However, this scenario was rejected for basing estimates of effects because while most juvenile salmon might leave Lower Granite after this time, we considered it possible that at least some sub-yearling chinook could be in the reservoir year-round. Likewise, migrating adult salmon could be in Lower Granite at other times of the year. Therefore, we selected the lowest monthly average flows that occurred at any time during the last 10-years as the critical, chronic condition to use to model the extent of effects of constituents of Potlatch's effluents.

Lowest monthly average occurring anytime during the last 10-years ("30Q10": We assumed that this condition would sufficiently represent the reasonable, worst case for estimating chronic effects of Potlatch effluents on the aquatic environment of the horizontally stratified LGR. The lowest monthly-average Snake River flow occurring in the last 10-years was 10,810 cfs which occurred during August 1994. Curiously, this value is nearly identical to the 7Q10 value of 10,577 cfs. These values are so similar that modeling each separately would provide little information. Instead, the 7Q10 values are relevant for estimating minimum monthly average flows as well. We refer to this flow as a "30Q10."

An important note is that under each of these scenarios, the Snake and Clearwater Rivers were assumed to be horizontally stratified, and the effluent plume would be retained and diluted in the Snake River flows only. This scenario is illustrated in Figure 4. In other words, for modeling purposes the Clearwater River flows are completely ignored and the Snake River current is only 335 m wide from its west bank to its approximate boundary with the Clearwater River flows.

There was no obvious relationship between river flow and pool elevation, i.e. diffuser depth (Figure 3) so depth was assumed constant at different flows. Depths and diffuser configuration were estimated from the mid-range of "as-built" engineering drawings provided to NOAA Fisheries by Potlatch and pool elevations at July 2001 which corresponded with the "smolt 30Q10" flows. The "as builts" showed the diffuser lay between elevations 694 an 702 feet which gives mid-range depths of about 36 feet when the pool elevation was about 734 feet during September 1994.

c. Whole effluent toxicity (WET) modeling

Whole-effluent toxicity was modeled using various TU multiples to estimate minimal, moderate, and severe effects of effluent on aquatic ecosystems. Effluent percent dilutions had to calculated as "toxic units" because the CORMIX model assumes a high - low concentration of pollutants from effluent to ambient waters. The WET modeling is used to estimate AOX effects extent as well, since AOX is an complex mixture best represented by whole-effluent toxicity. Also Noton 1991 reported that AOX was a convenient tracer of BKME in rivers. Upstream was assumed to have zero background toxicity.

d. Chlorinated Phenolic Compounds (CPCs) modeling

The 12 CPCs regulated in the NPDES permit were modeled using a sum TU approach. This is used because they have similar modes of toxicity, and if they were analyzed individually, may underestimate risk to aquatic ecosystems. For an individual compound, its toxic unit is its estimated allowable effluent concentration divided by its benchmark. The sum TU are simply the sum of the 12 unitless fractions. Effluent concentration were estimated by EPA (2003) based on limitations in the plant's fiberlines, assuming the fiberlines are the sole possible source of these CPCs. Following that assumption that these CPCs are peculiar to the pulp mill effluent, CPCs concentrations in the upstream Snake River are assumed to be zero.

e. Dioxin and Furan Modeling

Dioxin was modeled using different benchmarks and assumed background concentrations as a way to bound high and low estimates. The EPA (2003) and NOAA Fisheries (this document) independently derived different waterborne benchmarks using different approaches. The EPA (2003) made extensive calculations based on applying available site information to established developed food chain models using a BSAF approach, where NOAA Fisheries used a simpler approach based on toxicity testing. Both approaches have strengths and limitations, and so for this exercise, both benchmarks were modeled to provide a range of potential dispersion scenarios. The values independently estimated by EPA and NOAA Fisheries are relatively similar, ~ a factor of 3X, which are quite close considering the uncertainties in the data needed to derive them. Because the values are reasonably supportive of each other, NOAA Fisheries concurs with the use of EPA's benchmark of 0.063 pg/L. Both benchmarks are used in the CORMIX simulations in order to consider the sensitivity of the model predictions to small changes in input parameters, as with the use of 7Q10 and "smolt 30Q10" flow values.

Necessary assumptions of upstream concentrations are a major source of uncertainty in estimating plume dispersion to meet endpoints, since no data on ambient dioxin or furan water concentrations in the Snake River were located. This is important, because as upstream concentrations approach or exceed benchmarks, then there is no "assimilative" capacity of the water body to dilute additional loading.

Because the benchmark for furans (2,3,7,8 TCDF) that NOAA Fisheries estimated would be higher than the permitted effluent concentrations, the benchmark would be met at the diffuser and no modeling of distance downstream to meet benchmarks are needed (29.8 pg/L benchmark vs. 11.6 pg/L effluent limit). Consultants to Potlatch calculated dioxin and furan benchmarks using the same BSAF approach as used by EPA but making different environmental assumptions and estimated TCDF values that were intermediate to those estimated by EPA and NOAA Fisheries (Potlatch's values were 0.32 pg/L 2,3,7,8 TCDD and 1 pg/L 2,3,7,8 TCDF, Ginsburg 2004).

While no study area or regional data were located for 2,3,7,8 TCDD or 2,3,7,8 TCDF background levels in water. However, dioxins/furans are widespread in the environment and background concentrations cannot be assumed to be zero. Dioxin and furan data at environmentally relevant concentrations are limited because current analytical detection limits for dioxin congeners for water samples collected using conventional sampling techniques are about two orders of magnitude higher than EPA's water quality standard for 2,3,7,8 TCDD.

Three different background assumptions were used to cover a range of possibilities: (1) an arbitrary assignment of a 2/3 fraction of the calculated allowable dioxin load for the Snake River (0.004 pg/L, EPA 2003), (2) average North American background levels from surface waters that are used as drinking water sources (0.00056 pg/L as TEQs, EPA 2000), and (3) background levels from the Ohio River measured using advanced high volume, low detection level methods (0.033 pg/L). The latter data were used in the modeling because they were the only report of low-level monitoring of ambient concentrations of 2,3,7,8 TCDD in a large river that was located (Dinkins and Heath 1998). Since no regional or study area data have been located, ambient data from Ohio River sites that were not directly influenced by local dioxin sources were used as background estimates for plume modeling. This is probably a conservative estimate of dioxin concentrations in the Snake River since the Ohio River valley is assumed to be more urban, industrialized, and populous than the Snake River basin upstream of Potlatch's discharges. There are no pulp and paper mills located in the Snake River basin upstream of Potlatch. Land use in the Snake River basin above Potlatch is primarily agricultural, range, and mountains and forest.

For the Cormix modeling, the North American background values from waters used as drinking water sources are assumed to represent the lower range of possible ambient values, and the Ohio River background values represent the upper range.

f. Results

The following modeling scenarios used 62 CFS of effluent into 10,577 cfs of the horizontally stratified Snake River. In this scenario the Snake and Clearwater are treated as running side-by-side, without mixing for a considerable distance downstream. The Snake/Clearwater discontinuity is treated as the “right bank” of the river for modeling purposes, and the percentage of the channel occupied by the plume at different concentrations is just for the ribbon of flow from the upper Snake River, not the full Snake River from true bank to bank.

Table 15. Predicted extent of whole-effluent toxicity (WET) of pulp mill effluents under the 7Q10 scenario

Condition	Distance downstream (m)	Plume width when benchmark is reached	Percent of estimated Snake River channel	Downstream drift time (minutes)
Distance and time a drifting organism in the Snake River would spend above severe effects (10% effluent or 6.7 toxic units (TUs))	2.2	120	35%	<1
Distance and time a drifting organism in the Snake River would spend above moderate effects benchmark (5% effluent or 3.3 TUs)	10.5	100	30%	1
Distance and time a drifting organism in the Snake River would spend above minimal effects benchmark (1.5% effluent or 1 TU)	9530	335	100%	27 hours

Near-field region results	
Concentration at the edge of the zone of initial dilution (ZID)	1.6
Distance downstream to edge of ZID (m)	45
Depth of plume beneath the surface at edge of ZID (m)	4.5
Time for a passively drifting organism to pass through zone of initial dilution (minutes)	4

The near-field region is of interest because it is a zone of strong turbulent mixing caused by the initial velocities and configuration of the effluent. This is the area that more or less corresponds to EPA practices and guidance allowing for a small "toxic dilution zone" aka "zone of initial dilution" where the CMC (acute criteria) may be exceeded without resulting in lethality to organisms passing through the effluent plume. In the near-field region, mixing and dilution is very rapid due to turbulence caused by the high velocity outfalls shooting into the river water. Beyond this zone, further mixing and dilution of the effluent plume may be very slow and under calm conditions the plume may persist as a narrow ribbon of flow as final mixing due to buoyant spreading, passive dilution, and wind-driven circulation. Figure 11 illustrates the patterns of initial rapid mixing and slow final mixing.

In this scenario, full vertical mixing occurs within 5X the river depth of 9.14 m (~45 m), then it re-stratifies due to buoyancy. Complete lateral mixing of the plume is predicted to occur at 1215m downstream (i.e. the plume would be spread fully across the horizontally stratified Snake River side of LGR).

Table 16. Predicted extent of chlorinated phenols contained in pulp mill effluents using a sum toxic units approach under a scenario of typical low-flows occurring during a dry year. Effluent concentration for direct effects is 14 TU.

Condition	Distance downstream (m)	Plume width when benchmark is reached	Percent of estimated Snake River channel	Downstream drift time (minutes)
Distance and time a drifting organism in the Snake River would spend above a NOEC (1 TU) benchmark	3.5	115	34%	<1

Near-field region results	
Concentration at the edge of the zone of initial dilution (ZID)	0.3
Distance downstream to edge of ZID (m)	45
Depth of plume beneath the surface at edge of ZID (m)	4.5
Time for a passively drifting organism to pass through zone of initial dilution (minutes)	4

Benchmark for direct effects is reached before leaving the near-field region of mixing.

Table 17. Predicted extent of allowable 2,3,7,8 TCDD concentrations in the pulp mill effluent under scenarios of a 7Q10 flow and modeled with different benchmarks and flows.

	Benchmark	Assumed background concentration scenarios	Effective benchmark (Benchmark minus background)	Distance downstream (m)	Plume width when benchmark is reached	Percent of stratified Snake River "channel"	Downstream drift time (hours)
Scenario: Snake River horizontally stratified with Clearwater River during August 7Q10 low-flows (10,577 cfs)							
Distance and time a drifting organism in the Snake River would spend above a NOEC benchmark)	0.063 pg/L (EPA 2003)	0.004 pg/L (EPA 2003, 2/3 of loading allowed in TMDL)	0.059 pg/L	477	309	92 %	1.6
Distance and time a drifting organism in the Snake River would spend above a NOEC benchmark	0.2 pg/L (this report)	0.033 pg/L (Ohio R background)	0.167 pg/L	10	100	30 %	0

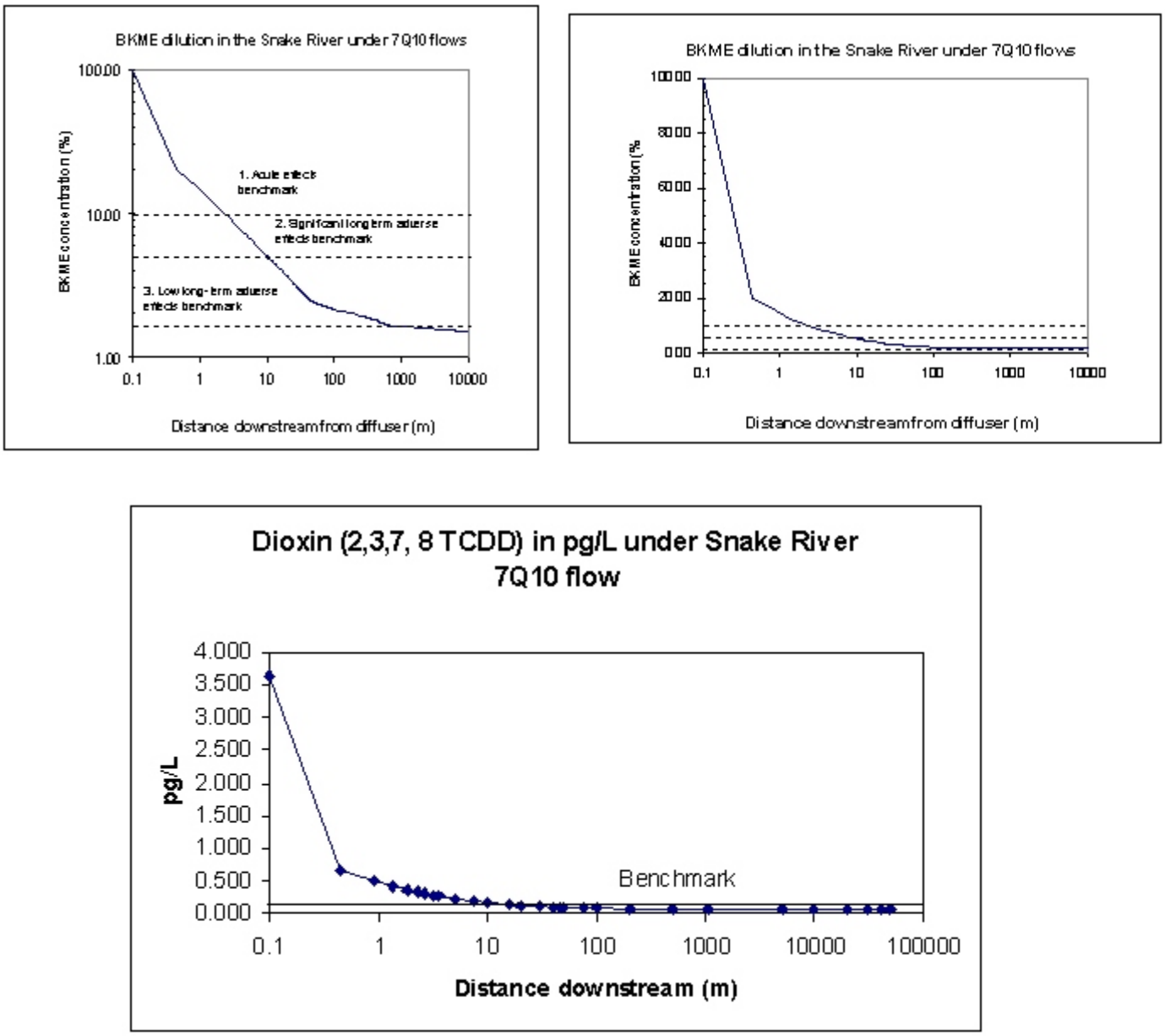


Figure 11. Dilution of the effluent plume and its concomitant reduction in chemical concentrations is initially very rapid close to the diffuser. Rapid mixing and dilution in this "near-field region" is caused by turbulence of the high velocity jets shooting into the river. After this zone of initial dilution, much slower further mixing results from passive diffusion and buoyant spreading of the plume across the river channel.

g. Conclusions

These waterborne adverse modifications to habitat do not address potential bioaccumulative effects or potential sediment contamination.

If benchmark exceedences of factors greater than about 2X or 3X were considered to result in considerable adverse effects to those habitats due to waterborne contaminants in the effluent plume, then the affected area are estimated to be limited to fairly small portions of the LGR. These areas are less than 10 m downstream by about 120 m across the active Snake River current, which is about 35% of the width of the stratified flows.

The potential for more subtle waterborne effects associated with BKME, analyzed as whole effluetn, extends further downstream about 9.5 km and encompasses the full width (>50%) of the Snake River current portion of the channel. These subtle effects include possible benthic community alterations, and increased juvenile growth rates associated with biostimulatory effects of dilute pulp mill effluent (Table 7).

2.3.4.2 Summary of fate and effects of organic contaminants

This section summarizes and integrates information presented in earlier sections on individual constituents. It does not recite information listed earlier.

The fate and effects of organic contaminants and related TSS has not been characterized sufficiently to define the extent and severity of potentially contaminated sediments or biota. However, the available information supports some inferences about the potential effects and expected extent of contamination from the pulp mill effluents.

Total Suspended Solids (TSS)

So long as the suspended solids remain suspended, concentrations from Potlatch's effluents are quickly diluted to levels where significant adverse direct effects to salmonids are probably minimal beyond the immediate vicinity of the diffuser. However, when the suspended particles reach quiescent areas of the reservoir, they will eventually settle out of the water column and accumulate on the bottom. In areas with significant accumulation, the diversity of the benthic community may be reduced. Thus reducing available prey to juvenile salmonids. The LGR has been reported to have higher amounts of fine grained, highly organic sediments than reservoirs further downstream; lower benthic invertebrate diversities; and poorer juvenile salmon feeding success. Potlatch's TSS discharges are likely to contribute to those conditions.

The extent of effects to habitats was estimated using the following reasoning and comparisons.

- The thalweg (center of the mainchannel) have a filament of higher velocities than occur at areas near the banks, and the flows in thalweg have greater competence to carry suspended solids than lower velocity areas near the banks (Cunningham 1993, PNNL 2002). TSS was assumed to settle out in quiescent areas of the reservoir that are close to the main channel.

- At median and lower flows (<=50th percentile flows), flows along the bank areas outside the thalweg are not expected to mobilize fine grained sediments. At very high flows

(10th percentile flows) velocities in LGR are high enough to move suspended solids a great distance. However, in the protected waters occurring in off-channel, shallow areas velocities are too low to mobilize fines (PNNL 2002).

- Nearshore, low velocity, moderately sloping shallow areas are the most preferable reservoir habitat type for juvenile chinook salmon and are the least preferable habitat type for predatory fish such as smallmouth bass, walleye, and pikeminnows (Key et al. 1996). These littoral areas correspond with likely depositional areas for Potlatch's TSS.

- Most of the littoral habitat in LGR is upstream of river mile RM 124.

Based on the above findings, most of the likely effects of Potlatch's TSS on juvenile salmon habitat in LGR occur between the outfall at RM 139 and about RM 124 (Figures 12 and 13). However, significant accumulations of sediment have been located further downstream near RM 117 (PNNL 2002). So it is possible that TSS is carried further downstream than just the habitats shown in Figure 13.

Dioxins and Furans

Dioxins and furans are hydrophobic and are thought to be sorbed onto TSS. Therefore, factors that affect TSS transport and deposition should be directly related to dioxin and furan distributions as well. Most data reviewed for dioxins and furans in sediment were well below the no-effect benchmarks. However, the sediment data are fairly sparse and it is difficult to make determination of how the release of Potlatch Mill effluent may affect listed salmonids via this route of exposure. Therefore, additional environmental monitoring of sediment will be essential for such an evaluation.

Similar to the case with CPCs, dioxin and furan levels in white sturgeon and largescale suckers collected in LGR by EPA (2002) were all below fish tissue benchmarks. Again, the monitoring was not specifically designed to detect effects from a discrete source, so the possibility of sediment contamination and uptake into the aquatic food web cannot be discounted.

Figure 14 illustrates the means by which dissolved and particulate-sorbed phases of bioaccumulative compounds in the effluents (e.g. dioxins, furans, and CPCs) may become bioavailable in the aquatic environment.

Chlorinated Phenols (CPCs)

CPCs may accumulate and persist in the sediment and the food web. CPCs may be dissolved into solution from the effluents or be sorbed to particulate organic carbon. If LGR habitats were contaminated by CPCs, the large littoral habitat area located on the south bank of the Snake River between RM 139 and 137 would be the most likely area. No data on CPCs in sediment were located, and apparently none exist. No CPCs were detected in white sturgeon and largescale suckers collected in LGR by EPA (2002). Those data were collected as part of a Columbia River basin study and was not intended to detect effects from discrete point sources such as Potlatch. Without negative results of monitoring that was specifically designed to detect effects of Potlatch's effluents, it is presummend that bioaccumulation of CPCs into the food web is possible and could have a meaningful effect on salmonid health.

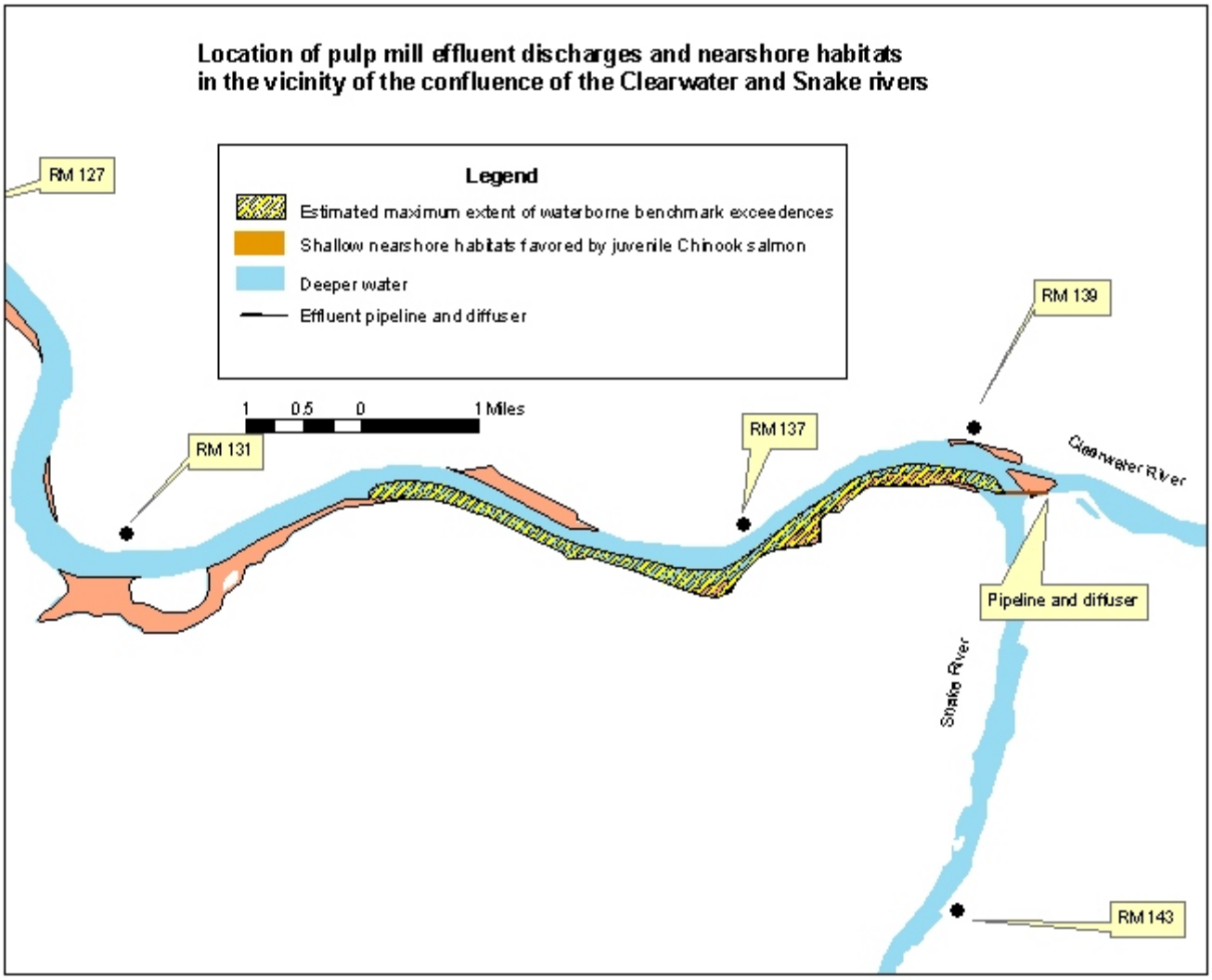


Figure 12. Under low flow conditions (7Q10 and 30Q10), the maximum extent of waterborne benchmark exceedences are predicted to be up to about 9.5 km (6 miles) downstream. Most chronic effects to salmonids are not predicted to extend more than 10m beyond the diffuser (not visible at this scale). Drawing is based on modeling results presented is this report and information presented in EPA (2003) and Cook et al (2003).

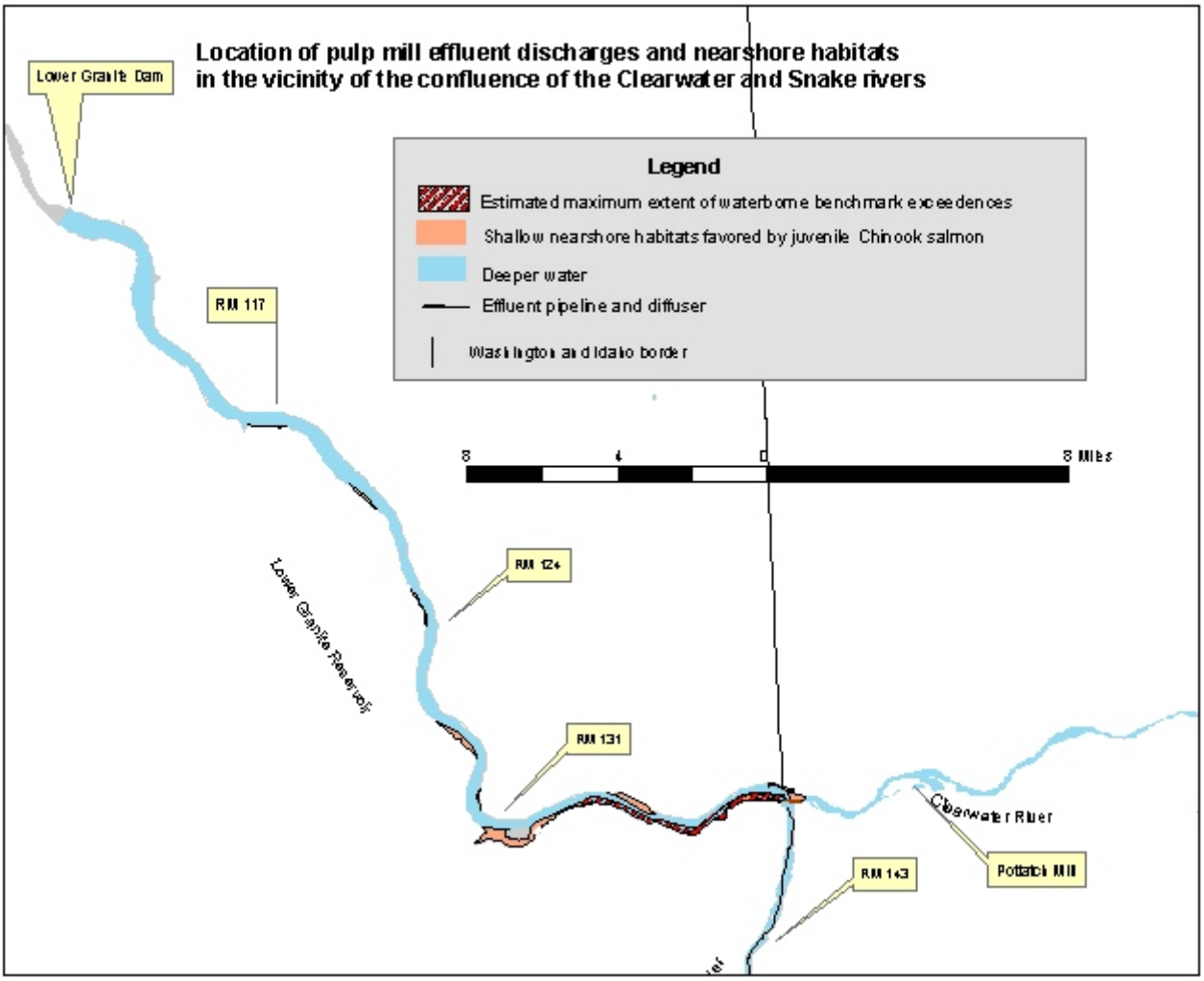


Figure 13. Overview of pulp mill effluent plume in relation to the nearshore habitats in LGR that are critical for juvenile Chinook salmon. Drawing is based on modeling results from Section V of the BO and information presented in and Cook et al (2003).

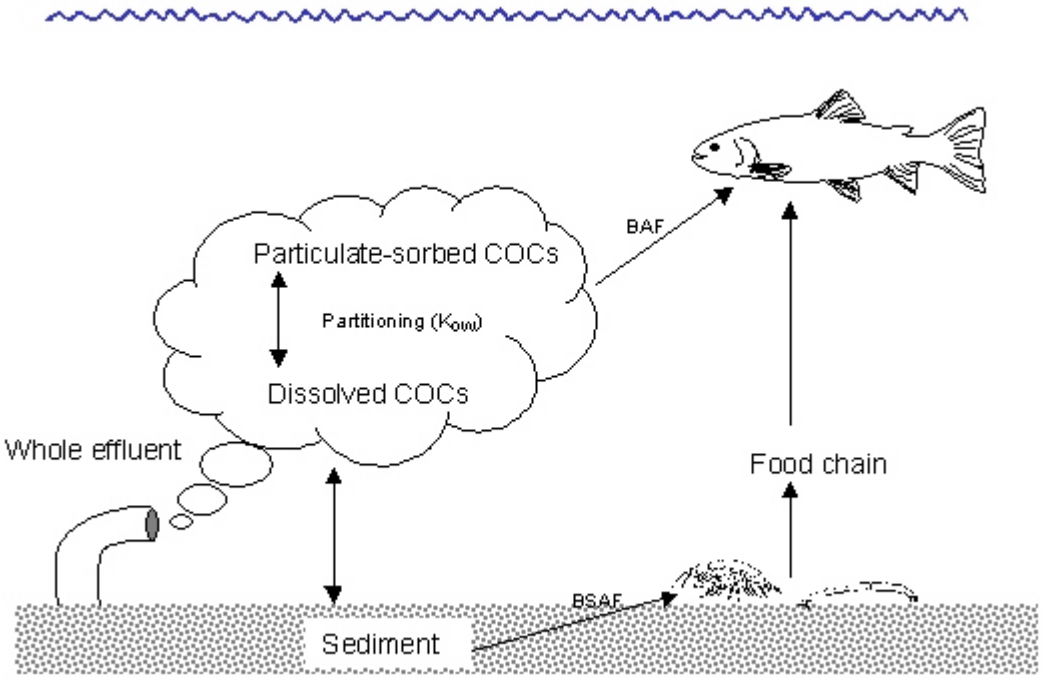


Figure 14. Possible routes of exposure and effects of organic contaminants contained in pulp mill effluents in LGR.

Resin acids, phytosterols, and retene

Few data are available on concentrations of several non-regulated organic contaminants that can be harmful to aquatic life in trace amounts including retene, resin acids, and phytosterols. Because few data were available, concentrations from literature about other rivers that receive pulp mill effluents were used as a first estimate of whether benchmarks are likely to be exceeded. Concentrations of retene that were measured in effluents from mills that use processes that would produce similar or higher concentrations than Potlatch. Assuming they were similar, then effects on salmonids from waterborne retene would be low beyond the near-field region (<50m) of the mixing zone. However, retene could still accumulate and persist in sediments and enter the food chain. Based on similar inferences, the extent of effects from resin acids is assumed to roughly correspond with that of BKME. Resin acids in sediments may be transformed into retene which

may be persistent in sediments. No data on phytosterol concentrations in effluent, receiving water, or sediment were available, and data from other similar pulp mill operations were insufficient to make any comparative inferences.

Bioavailability of organic contaminants of concern

The pathways and mechanisms by which CPCs, dioxins, and furans enter the Lower Granite habitats and could result in exposure and effects to anadromous salmonids include:

- Organic chemical contaminants are released from the plant effluent through its outfall and as pond seepage,
- Some fraction of the contaminants are sorbed to suspended particulates in the effluent, and the remainder is dissolved in water column,
- The contaminants may enter the aquatic food chain by exposure across the gill to dissolved fractions, through direct ingestion of particles, or the particles settle onto the substrate where they may be bioavailable to benthic organisms,
- Contaminated benthic organisms are eaten by forage fish or juvenile salmonids,
- Exposure of salmonids to contaminants results in elevated tissue residues or biomarkers of exposure (e.g. liver enzymes), and
- Exposure to contaminants may result in adverse fish health measures such as increased probability of mortality, reduced growth or impaired reproductive success (e.g. measured condition factors or hormone levels).

These pathways are illustrated in Figure 14.

Summary of likely effects of TSS and organic chemical contaminants

A summary of concerns regarding Potlatch's discharges into the LGR follow in Table 18 (the summary is highly abbreviated, refer to the evaluations of individual constituents for more details).

Table 18. Summary of concerns regarding TSS and organic contaminants.

Constituent	Types of Effects that are Likely	Estimated spatial extent of effects
TSS	Deposition in quiescent off-channel shallow habitats that are also the preferred habitat of juvenile Chinook in the reservoir. Substrate alteration may be depressing salmonid prey availability.	Indefinite. However, if assumptions about sediment transport are correct, much settling would occur in shallow habitats between the outfall and RM 127.
BKME (WET) toxicity	Subtle changes in fish biochemistry or condition factors due to organic enrichment of the river by dilute (1 - 2%) pulp mill effluent	<9.5 km downstream
	Numerous chronic effects to salmonids that are clearly adverse are likely at higher BKME (>5.0%) concentrations	<11 m
	Acute exposures may start killing salmonid fry at >10% BKME	<3 m
CPCs	Numerous chronic effects to salmonids	< 5m
	Sediment-sorbed CPCs may settle out and result in bioaccumulation and sublethal effects	Indefinite. However, if assumptions about sediment transport are correct, much settling would occur in shallow habitats between the outfall and RM 127.
Dioxins and furans (TCDD/TCDF)	Waterborne TCDD could result in bioaccumulation, food chain transfer, and reduced reproductive potential in salmonids	10 - 500 m
	TSS-sorbed TCDD/TCDF could result in bioaccumulation, food chain transfer, and reduced reproductive potential in salmonids	Indefinite. However, if assumptions about sediment transport are correct, much settling would occur in shallow habitats between the outfall and RM 127.
Retene	Numerous chronic effects to salmonids	No data, but assuming that Potlatch's effluent concentrations are no higher than those from other pulp mills, waterborne effects are likely low beyond near-field region of the mixing zone. If these assumptions are accurate, <50m
	Sediment-sorbed compounds may settle out and result in bioaccumulation and sublethal effects	No data, perhaps similar to TSS and CPCs.
Resin acids	Waterborne effects are assumed to be accounted for by BKME analysis	Not estimated
	Sediment-sorbed resin acids may transform to retene and persist in sediments	No data, perhaps similar to TSS and CPCs.
Phytosterols	Waterborne effects are assumed to be accounted for by BKME analysis.	Not estimated

A reasonable scenario exists for exposure and direct or indirect effects to salmonids from the effluents. It is likely that if sufficient data were collected, some of these assumptions would be rejected. A robust monitoring strategy is needed that would elucidate these unknowns in addition to meeting other ESA needs, such as tracking take of listed fish.

A monitoring and assessment plan has been designed with four elements to achieve the goal of estimating the likelihood of harmful effects. These are:

1. Characterize the chemical and physical components of the action area: Potlatch effluent from discharge point to LGD,

2. Determine if contaminants from the Potlatch effluent are bioaccumulated by endangered fish species,

3. Characterize the potential effects to biota in the action area, and if desired,

4. Develop or modify contaminant benchmarks for the protection of salmonids, their prey, and habitat.

Each of these four elements is related to characterizing the action area, reducing uncertainties in the BA and Opinion, and determining through *in situ* studies if species are or will be harmed.

The monitoring and assessment planns is appended in Appendices 5 and 6.

2.3.5 Interdependent and Interrelated Activities

NOAA Fisheries has identified one interrelated or interdependent activity that is essentially required by the NPDES renewal and depends on the renewal for its justification. This activity is the withdrawal of about 70 cfs of Clearwater River flow for purposes of mill operation, processing, treatment, and discharge of mill waste. This rate of withdrawal represents about 5% to 13%of the Clearwater River at lowest flows (550 cfs to 1,500 cfs) and is diverted only for a short distance (2 to 3 miles) before being returned to the Snake River at the mill's outfall.

Because water quality of the Clearwater River is typically colder and more oxygenated than that of the Snake River, withdrawing Clearwater flow, reduces its cooling increment on the Snake River. During most seasons and years the affect of this withdrawal is difficult to measure; however, during seasons when Snake River temperatures exceed those suitable for cold water salmonid life stages, the net effect of removing 70 cfs of 14ºC water and returning it to the river at 33ºC is a localized reduction in water quality for less than 1% of the Snake River.

NOAA Fisheries is not aware of other interrelated and interdependent activities that could significantly affect ESA-listed salmonids in the action area.

2.3.6 Effects on Critical Habitat

NOAA Fisheries designates critical habitat based on physical and biological features that are essential to the listed species. Essential features of designated critical habitat include substrate,

water quality, water quantity, water temperature, food, riparian vegetation, access, water velocity, space and safe passage. Effects to critical habitat from these categories would be similar to the effects described above in section 2.3.

2.3.7 Cumulative Effects

Cumulative effects are defined in 50 C.F.R. 402.02 as "those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation." Other activities within the watershed have the potential to impact fish and habitat within the action area. Future Federal actions, including the ongoing operation of hydropower systems, hatcheries, fisheries, land management activities, and issuance of incidental take authorization under section 10(a)(1)(B), are being reviewed through separate section 7 consultation processes. Past Federal actions have already been added to the environmental baseline in the action area.

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 are mostly associated with urban activities in the Lewiston-Clarkston area and state-level permitting of private activities. The most primary of these include: air deposition of pollutants from the Potlatch Mill stacks and from other sources in the Lewiston-Clarkston area; the existence and authorization of other mixing zone plumes in the Lewiston-Clarkston area; urban stormwater, agricultural runoff, and groundwater contamination of pesticides, herbicides, hydrocarbons, metals, temperature, organics, and nutrients; urban development, timber harvesting, and water withdrawals.

These activities are expected to continue in the future and are likely to increase as natural resources decline and human populations increase. The primary cumulative effects are from the multiple additive sources and inputs of chemical and organic compounds to receiving waters in the action area. Many biocides (pesticides) include similar chemical compounds, work in similar modes, and have similarly amplified adverse effects on listed-salmonids when multiple forms are loaded together in receiving waters. In the Snake River, water and sediment loads of these pollutants are not quantified nor is use or fate closely regulated or monitored. For example, pesticide use for private agricultural activities includes CWA exemptions without Federal regulation. There remains a high risk of these pollutants being present in significant background levels in the action area.

NOAA Fisheries is unaware of any other reasonably foreseeable effects of future state, tribal, local or private actions that need to be considered in this analysis.

2.3.8 Integration and Synthesis of Effects

The severity of potential effects to anadromous salmonids in the action area due to pollutants that are authorized to be released under this proposed action is a function of both the magnitude and duration of exposures to the pollutants of concern. Section 2.1.3 (Status of Listed Species and their use of the Action Area) describes the timing and duration of use of habitats that could be influenced by the effluents. Section 2.3 (Analysis of Effects) describes the derivation of fisheries benchmarks for listed species and predicted the distances downstream that the benchmarks might be exceeded. The analysis of effects section focused more on the potential magnitude of exposures than on the expected durations of exposures. In the following discussion, we attempt to integrate the previous analyses of durations of habitat usage by anadromous fish species with the predictions of the potential downstream extent of benchmark exposures.

The discussion begins with some important limitations, considerations, and uncertainties and concludes with species-by-species descriptions.

2.3.8.1 Synthesis considerations, limitations and uncertainties

Significance of no-effect or low-effect benchmark exceedences - One key consideration in synthesizing effects is the significance of exceeding a no-effect benchmark. Because threatened and endangered species are vulnerable to extinction risk, benchmarks for effects analysis were chosen conservatively to represent no-effect concentrations. This is in contrast to general water quality standards that allow for some adverse effects to the most sensitive genera so long as no valued species are lost and overall aquatic communities are protected (section 2.3.1.8, *Data Analysis for Threatened and Endangered Species*). Because safety factors were included in the derivation of the no- or minimal effects benchmarks , exceeding a no- or a nearly no-effect concentration does not automatically equate to resulting in an unsafe or observable adverse effects. Rather, in areas where the "safe" benchmarks are not exceeded, we are reasonably confident that adverse effects from the contaminant of concern are unlikely to occur. Where the "safe" benchmarks are exceeded, the safety factors are eroded, uncertainty is higher, and the possibility for crossing an indefinite threshold into conditions where the risk of adverse effects occur is greater. Using an analogy of highway safety, actions such as exceeding posted speed limits or passing on the shoulder may not always cause accidents, but may subjectively be judged to increase risks. Likewise with these effects analyses, less exposure is better, even if the increased risks of effects to anadromous salmonids at higher exposures cannot be directly quantified.

In the case of BKME, expected effects of BKME at the minimal effects benchmark include induction of biomarkers of exposure such as the reproductive hormonal changes, MFO liver enzyme induction, benthic community changes, and increased growth due to food stimulation from organic enrichment (section 2.3.3.2). These types of effects are not considered beneficial, but neither are they obviously predictive of more "important" adverse effects such as compromised disease resistance, or reduced reproductive success. Biochemical biomarkers are widely used in environmental monitoring and assessment as as early-warning indicators of contaminant exposure (Galloway et al. 2004; Pelley 2004; Van der Oost 2003 are a few recent examples). However, we are not aware of information linking biomarkers such as MFO induction to clearly detrimental effects in such as death, reduced reproductive success, or migratory disruption. This lack of information may simply be a data gap in our knowledge, rather the absence of true effects. For example, recent studies have shown that immune

suppression or increased susceptibility to disease are important effects that may occur after even brief exposure to contaminated sediments in urban estuaries (Arkoosh et al. 1998; 1998a). However, no comparable studies of immunotoxicity of pulp mill effluents with salmonids were located.

There have been several long-term tests of the growth, survival, and health of rainbow trout in natural-like experimental stream studies (ESS) using Potlatch effluents (section 2.3.3.2.g). In some studies, decreased survival of the trout was observed at the 1.5% effluent low-effects benchmark (Figure 8). Although lower survival occurred with 1.5% effluents in two 9-month duration tests, we assumed that the "final" ESS (using effluents that were more similar to present effluents than older tests) should be given greater weight than older tests. This study showed lower mortalities than earlier studies (Figure 8a, 8b). However, the differences between repeated studies of 1.2% effluent in Figure 8a were as large as the differences between the "final" ESS in which trout survival in 1.5% effluent was not lower than control survival and earlier tests in which trout survival was lower at 1.5% effluent. Thus while we believe that a 1.5% effluent concentration benchmark is supported by the strength of the evidence considered, we acknowledge the uncertainties in its selection. At the next higher effluent concentration that was frequently studied in the ESS program, 5% effluent, evidence that long-term exposures would likely be harmful to salmonids was less ambiguous.

Uncertainties relating to possible sediment or food chain pathways of contaminants - This section's integration of analyses of life history types and life stages of anadromous salmonids, spatial extent of potential waterborne benchmark exceedences, and the estimated extent of time the different life stages/history types focuses on the waterborne benchmark exceedence zones. The quantitative analyses focus on the waterborne benchmark exceedences because the dispersion of effluent-containing plumes of water into river water is readily modeled. Less amenable to quantitative estimates are other possible pathways for contaminants in effluents to enter the aquatic environment such as deposition of contaminants sorbed to suspended sediments, food web transfer, or seepage from ASB. There is a lack of accurate information on the actual concentrations of organic contaminants in the effluent and the dispersal of these contaminants (many are sorbed onto organic fines) over time and space within the LGR and therefore, the risk that some of these organic contaminants could accumulate in sediment or biota and pose a chronic health risk to ESA salmonids is not well understood. Present sediment data that Potlatch has collected and tissue data that EPA has collected do not indicate that NOAA's NOEC benchmarks of concern are reached in either sediment or tissue. However, these sediment data were somewhat limited in geographic scope and the fish data were not designed to evaluate effects from specific sources. Thus the currently available data are not sufficient to conclude the absence of effects of chemical contaminants of concern. These uncertainties are expected to be addressed through the monitoring and assessment plans (Appendices 5 and 6).

Uncertainties relating to habitat usage in LGR - To estimate durations of exposure to areas where benchmarks could be exceeded, assumptions were made of a roughly uniform rate of travel by fish between the fish traps at the head of the reservoir and when they pass LGD. This will probably be roughly accurate for some life histories and not accurate for others. Subyearling fall chinook and spring/summer chinook salmon are thought to preferentially use moderately sloped, nearshore areas of LGR, and nearshore areas are not randomly distributed throughout LGR. Rather, about 75% of the preferred nearshore habitat is located in the upper half of the reservoir. No simple method was apparent to us that could be used to reasonably account for the likely bias in usage toward the upper half of the reservoir by subyearling salmon and none was used. This probably resulted in underestimates of the durations of subyearling occurrence in

areas that may exceed benchmarks. These estimates of durations of use and further basis for the assumptions used are given in the following species-by-species discussion of the durations and possible consequences of benchmark exceedences.

2.3.8.2 Synthesis of effects by species

Snake River sockeye salmon. Sockeye salmon smolts and adults migrate through the action area during spring and summer months. Smolts primarily pass through the effluent mixing zone during May and early June when flows and DO levels are high and temperatures are low. Effluent dilution would be higher and contaminant concentrations would be lower in May and June than were the low-flow conditions that were analyzed in detail in section 2.3.4. The great majority (90%) of sockeye smolts pass through LGR in 15 days or less (Table 4). Assuming a roughly uniform rate of movement through the reservoir, sockeye smolts would take about three hours to pass through the <500m long area where the NOEC dioxin waterborne benchmarks may be exceeded.

Sockeye salmon adults return during summer when LGR temperatures are high and DO levels low. These summer conditions are similar to the low-flow scenarios that were used for modeling the extent of waterborne contaminants in section 2.3.4. Most adult sockeye salmon pass through LGR in seven days or less (Table 4). Assuming a roughly uniform rate of movement through the reservoir, sockeye adults could be exposed to the <500m long area where a dioxin waterborne benchmarks may be exceeded for up to about 1 hours. Similarly, sockeye adults could be exposed to BKME concentrations in excess of the lowest effect benchmark (1.5% effluent) for up to about one day. Continuing with the assumption of uniform movement rates, sockeye juveniles and adults would only spend about 1 minute in the portion of the near-field zone of initial effluent dilution (ZID)located <5m downstream of the diffuser in which BKME concentrations could be high enough that acute exposures on the order of >24 hours could result in lethality.

Similarly, sockeye smolts could be exposed to BKME concentrations in excess of the low effects benchmark (1.5% effluent) for up to about 2 days. This benchmark (also described as 1 toxic unit) was predicted to be slightly exceeded (i.e. by a factor of 1 - 1.6X) for up to 9.5 km downstream during low-flow conditions (Table 15; Figures 10 and 11). Potential low-level BKME induced effects that are possible in these concentration ranges would probably take about 3-weeks or more to start developing.

For sockeye salmon, we believe the assumption of a roughly uniform rate of travel through the reservoir without any periods of residency for rearing, resting or staging in the reservoir is reasonable. The LGR transit times for sockeye are among the shortest of the anadromous salmonids (Table 4), and they are believed to preferentially travel in the mid-channel areas with higher velocities (section 2.1.3.1).

NOAA fisheries benchmarks were developed assuming long-term (chronic) exposures. The durations of the chronic studies that were used to develop them varied, but were usually several weeks or more. In some cases the exposures were much longer, but in no case were short term (acute) studies of 4-days or less used to develop the fisheries benchmarks without adding safety factors to "adjust" the acute results a chronic no observed effect concentration. The estimated times when sockeye would be in areas where benchmarks are exceeded (<2 days) is considerably less than the length of time needed for chronic effects to manifest themselves. Adverse effects

that may develop after acute exposures of 24-96 hours are possible at concentrations expected within part of the ZID area extending about 5m or less downstream of the diffuser, if the fish stayed in the plume that long. Since the ZID occurs in an open-water section without structures or other features that might be attractive to fish, there are no apparent reasons why sockeye would preferentially stay in the effluent plume in the ZID. Even at concentrations that could result in lethality from contaminants of concern after 24-96 hour acute exposures, no effects are plausible following an exposures on the order of 2 minutes within that plume. More details on these waterborne exposure estimates are given in Table 19.

The calculations in Table 19 focused on chemical pollutants rather than stressors such as elevated temperature or reduced DO that could potentially result under the proposed permit limits. The area where temperatures are elevated enough (>1°C) by the effluent that potential avoidance might occur during warm conditions likely is limited to a plume that extends about 3m from the diffuser at river depths of around 6-8 m below the surface. For the potential effect of adult sockeye salmon avoidance of the plume, no estimates of duration within the plume need to be made (section 2.3.2.1). During the cooler times of the year that sockeye smolts could encounter the heated effluents, this area is only about 1 meter from the diffuser and is not considered a significant problem[2]. Within about 2 meters from the diffuser ports, temperatures of 5˚C or more above ambient could occur. If fish swam directly into heated water within two meters of the diffusers, the result would be a short-term, shock-like stress reaction.

Under baseline conditions, DO levels in LGR are sometimes unsuitable for migrating sockeye salmon. On August 6, 1997, from river miles 139 to 129, DO concentrations from near the surface to 6m of LGR ranged from 4.7 to 5.7 mg/L with corresponding temperatures ranging from 19 - 23°C. DO concentrations measured at the monitoring station near the head of the reservoir at river mile 144 on that day were more favorable, ranging from 6.6 to 7.4 mg/L at 22°C (Chapman and Hillman 2001). These conditions occurred during the upriver migration times for adult sockeye through LGR. In 1997, adult sockeye passed LGD from June 25 to August 26 with the peak at July 30 (EPA 2003 Table V-5). These DO and temperature data were collected weekly, so we consider it likely that these unsuitable conditions persisted for a few days around August 6, 1997. In the 42 sampling events conducted by Potlatch during the summers of 1997-1999, no other occurrences of similar conditions were observed. However, we assume similar unsuitable episodes could occur periodically during the summer, especially during periods of hot, calm weather. The conditions observed during the August 6, 1997 episode of low DO and high temperatures approached those combined DO and temperature conditions that have been reported to interrupt adult salmon migration. Materna (2001) describes events in which average daily minimum DO values in the 3.3 - 4.2 mg/L with temperatures of 19 - 22°C prevented migration of chinook salmon.

NOAA Fisheries made simple estimates of DO deficits resulting from Potlatch's actual BOD discharges. Based upon measured BOD concentrations in Potlatch's effluents and DO modeling reported in EPA (2003), NOAA Fisheries estimated that about 0.2 to 0.6 mg/L of these baseline conditions could be attributed to Potlatch's effluents (section 2.3.2.3). Following implementation of planned conservation measures to reduce TSS (Appendix 3), and assuming that the TSS reductions would reduce BOD proportionately, the DO reductions due to Potlatch's BOD discharges was estimated to be reduced to about 0.1 to 0.4 mg/L. Once Potlatch attains the

[2] In the earlier analysis of effects of temperature (section 2.3.2.1), faster fish travel rates were assumed since that analysis was focused on the possibility of thermal shock in the immediate vicinity of the diffuser jets in open water and fish were assumed to be moving at at least the speed of the water. In contrast, the calculations in this section may include periods of residency while rearing or holding occurs in addition to active migration times.

proposed BOD effluent limits, DO reductions in LGR due to their BOD discharges are predicted to be insignificant (<0.2 mg/L).

Compared to BOD, COD is much quicker to act on ambient DO and would be more likely to contribute to any depressions in DO within the first few miles downstream of outfall 001 than BOD (EPA 2003).

Table 19. Approximate durations that different life stages of anadromous salmonids would likely be exposed to waterborne contaminant concentrations in excess of "safe" benchmarks

Species	Life stage and life history type	Estimated time for 90% to pass through LGR (days)	Overall travel rate through LGR (km/day)	Minutes in ZID where acute effects could occur (<0.005 km downstream)	Hours in areas of potential dioxin exceedences (<0.5 km downstream)	Days in zone of potential BKME exceedences (<9.5 km downstream)
Sockeye	Smolt migrant	15	4.2	2	3	2
Sockeye	Adult migrant	7	9.0	1	1	1
Fall chinook	Rearing & smolts	60	1.1	7	11	9
Fall chinook	Juvenile overwintering	270	0.2	31	52	41
Fall chinook	Adult migrant	7	9.0	1	1	1
SS chinook	Subyearling	60	1.1	7	11	9
SS chinook	Smolt migrant	15	4.2	2	3	2
SS chinook	Adult migrant	7	9.0	8	1	1
Steelhead	Rearing & residualized	200	0.31	23	38	30
Steelhead	Smolt migrant	6	10.5	1	1	1
Steelhead	Adult overwintering	180	0.35	21	34	27
Steelhead	Adult migrant	7	9.0	1	1	1

Table notes: ZID – zone of initial dilution; SS chinook – spring/summer chinook salmon

Snake River fall chinook salmon. Snake River fall chinook salmon migrate into the action area as juveniles during spring, summer, and fall seasons. Adult fall chinook most commonly migrate through the action area in September through November (Section 2.1.3; EPA 2003, section V). In contrast to the sockeye, most juvenile fall chinook would likely encounter low-flow conditions

with higher temperatures and low DO, and most adults will experience cooler temperatures and higher DO levels than the juveniles.

The assumption for these calculated estimates of roughly uniform rates of movement through the reservoir without periods of extended residency for rearing, resting or staging in specific parts of the reservoir does not hold for fall chinook salmon as it did for sockeye salmon. Subyearling chinook salmon prefer nearshore habitats with relatively shallow gradients (Key et al. 1996, Curet 1993). These habitats do not occur uniformly throughout LGR, rather, most (~77%) occur in the upper half of LGR (Figure 2). Thus instead of a uniform travel rate, subyearling fall chinook probably spend the majority of their LGR time in the upper reservoir.

Most juvenile fall chinook rear for a period in LGR and then migrate seaward. For this life history type, a duration estimate in LGR of 60 days was used. These fish were estimated to pass the different zones of possible benchmark exceedence in about 1 hour to 9 days (Table 19). A small proportion (~9% of the fish sampled) of juvenile fall chinook salmon overwinter in Lower Granite Reservoir (Connor et al. in review). The time period that the overwintering juvenile fall chinook would be in LGR is uncertain, but for this exercise is assumed to be from about July to April, around 270 days. Using that duration with the simplifying assumption of an overall, roughly uniform seaward movement results in estimates of about 30-minutes in the ZID, 4-days in the dioxin potential-exceedence zone, and 41 days in the BKME potential-exceedence zone. These exposure durations for juvenile fall chinook salmon are long enough that they may be at increased risk of adverse chronic effects, especially for those fish that overwinter in LGR.

Adult fall chinook would pass through the areas where chemical benchmarks may be exceeded in less time than that expected to result in adverse effects. These times range from about 8 minutes to 1 day in the ZID and BKME exceedence zones respectively. The exposure durations for adult fall chinook salmon are too short to cause much risk of adverse effects, especially for those fish that overwinter in LGR.

Similarly to the case with sockeye salmon, adverse effects are considered unlikely when temperatures are elevated <1°C. These conditions are not expected to extend more than 6 - 8 m from the diffuser.

Most adult fall chinook salmon are probably not adversely affected by DO reductions from Potlatch's effluents since they would not be present in LGR in mid summer when DO levels are most likely to approach low values that kill or injure fish. Juvenile fall chinook that rear and migrate through LGR in mid-summer will encounter stressful periods of low DO combined with high temperatures. The likely response of juvenile fall chinook during such conditions is reduced activity and reduced growth (Materna 2001). The consequence of these responses may be migratory delays and reduced chances of survival.

Spring/summer chinook salmon. Most Snake River spring/summer chinook salmon migrate downriver into the action area as yearlings during spring and early summer. The great majority of wild yearling spring/summer chinook passage through LGR occurs during late March to early June, although the last migrating smolts pass through well into the fall on most years. Adult spring/summer chinook most commonly migrate upriver through the action area in April through July (Section 2.1.3; EPA 2003, section V). Thus, most juvenile spring/summer chinook salmon would likely encounter relatively high-flow conditions with cooler temperatures and high DO, and some late-returning adults spring/summer chinook may experience warm temperatures and lower DO levels than the juveniles.

Juvenile Snake River Spring/summer chinook salmon are commonly thought to have a "stream-type" life history where they overwinter near their natal streams and then migrate seaward as yearling smolts the following year. This has typically been considered to be one of the distinguishing characteristics between the fall and spring/summer chinook salmon (Healey 1991). However in the Snake River basin, some juvenile spring/summer chinook salmon have an "ocean-type" life history where the fry migrate seaward as subyearlings in their first year of life without overwintering in freshwater. In a 3-year study by Connor et al. (2001), of the total subyearling chinook salmon that were captured in the Snake River, tagged and recaptured at Lower Granite and Ice Harbor dams, 41% were of spring/summer chinook salmon, where the remainder (59%) would have been fall chinook salmon. The estimates of exposure durations above benchmarks used a 60-day LGR residency time for subyearling spring/summer chinook salmon and a 15-day LGR residency time for yearlings used (Table 19).

Adult spring chinook would pass through the areas where chemical benchmarks may be exceeded in less time than that expected to result in adverse effects with exposure durations. Subyearling and yearling spring/summer chinook exposures range from about 1 hour to 9 days (Table 19). For spring/summer chinook salmon, adverse effects are considered unlikely when temperatures are elevated <1°C. For late returning fish, these conditions are not expected to extend more than 6 - 8 m from the diffuser. For fish returning earlier in the year when it is cooler times, the extent would be less. Late returning adults may encounter similar summertime DO conditions as described for adult sockeye salmon.

Snake River steelhead. Most juvenile Snake River steelhead trout migrate seaward into the action area as juveniles during April through early June. Adult steelhead most commonly migrate upriver through the action area from July into November (Section 2.1.3; EPA 2003, section V). The last migrating juveniles leave LGR through October on some years. Thus, most juvenile steelhead would likely encounter relatively high-flow conditions with cooler temperatures and high DO, and early-returning portion of the adult steelhead run would experience warm temperatures and lower DO levels than most juveniles.

Most migrating juvenile and adult steelhead are estimated to pass the different zones of possible exceedences quickly, within a few minutes to 1 day (Table 19). Migrating adult or juvenile steelhead would pass through the areas where chemical benchmarks may be exceeded in less time than that expected to result in adverse effects.

In contrast to actively migrating steelhead, a significant portion of the adult steelhead migrate from the ocean to LGR, and then remain in LGR overwinter before moving upstream to spawn. In telemetry studies conducted from 1991-1995, 34% of the entire Clearwater River steelhead runs overwintered in the vicinity of the confluence of the Snake River with the Clearwater River (Bjornn et al. 2003). In this case the assumption of a roughly, uniform rate of movement through the reservoir clearly would be inappropriate, since the fish are disproportionately located near the discharge points in the confluence vicinity. Instead, estimating this overwintering time period as about 2 - 5 months (60 - 150 days), and assuming these fish spend their time evenly in the Snake and Clearwater influenced zones of the reservoir, these fish could be in the dioxin or BKME exceedence zones for times on the order of 30 - 75 days. No way to quantitatively estimate time in the ZID was obvious; instead we assumed that these overwintering fish would be frequently exposed for short time periods. It is conceivable that during the winter, the slightly warmer temperatures in the ZID could be attractive to adult steelhead.

While the effects of these likely exposures to overwintering adult steelhead are not easily

quanitifiable in terms such as death rates or reproductive success, these exposures are considered to add some risk of poorer health for these fish.

As was the case with the other species, adverse effects associated with temperature are considered unlikely when temperatures are elevated <1°C. These conditions are not expected to extend more than 6 - 8 m from the diffuser. Early-returning adults may encounter low summertime DO as described in the sockeye salmon discussion.

2.4 Conclusions

After reviewing the best available scientific and commercial information regarding the current status of the four listed ESU's discussed in this Opinion, the baseline for the action area, the effects of the proposed action, and cumulative effects, NOAA Fisheries concludes that EPA's proposed issuance of NPDES permit ID-000116-3 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.

2.5 Conservation Recommendations

Conservation recommendations are defined as "discretionary measures to minimize or avoid adverse effects of a proposed action on listed species or critical habitat or regarding the development of information" (50 C.F.R. 402.02). Section 7 (a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. NOAA Fisheries has worked with EPA and Potlatch to incorporate a number of measures to avoid or minimize adverse effects of the proposed permit activities, and does not identify additional, recommended conservation measures.

2.6 Incidental Take Statement

The ESA at section 9 [16 USC 1538] prohibits take of endangered species. The prohibition of take is extended to threatened anadromous salmonids by section 4(d) rule [50 C.F.R. 223.203]. Take is defined by the statute as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct." [16 USC 1532(19)] Harm is defined by regulation as "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation which actually kills or injures fish or wildlife by significantly impairing essential behavior patterns, including, breeding, spawning, rearing, migrating, feeding or sheltering." [50 C.F.R. 222.102] Harass is defined as "an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering." [50 C.F.R. 17.3] Incidental take is defined as "takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant." [50 C.F.R. 402.02] The ESA at section 7(o)(2) removes the prohibition from any incidental taking that is in compliance with the terms and conditions specified in a section 7(b)(4) incidental take statement [16 USC 1536].

Incidental take is defined as "any taking otherwise prohibited, if such taking is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity" [50 C.F.R. 17.3]. The ESA at section 7(o)(2) removes the prohibition from incidental taking that is in compliance with the terms and conditions specified in a section 7(b)(4) incidental take statement.

An incidental take statement specifies the impact of any incidental taking of endangered or threatened species. It also provides reasonable and prudent measures that are necessary to minimize impacts and sets forth terms and conditions with which the action agency must comply in order to implement the reasonable and prudent measures.

2.6.1 Amount or Extent of Take

The proposed action is reasonably certain to result in take of listed species because they are found in the area affected by the effluent during the time effluent is discharged in significant amounts. Despite the use of the best scientific and commercial data available, however, NOAA Fisheries cannot quantify a specific amount of incidental take of individual fish likely to result from this action. It is difficult to estimate amounts of lethal or sublethal take of individual fish from habitat-altering actions under the best of circumstances. In this instance there are additional uncertainties about the complex flow regime and contaminant concentrations and their effects on listed species in the action area which compound the difficulties of quantitative predictions. While NOAA Fisheries cannot quantify an amount of take, the information below describes the extent of the areas in which take is likely to occur.

Because of the uncertainties noted above, NOAA Fisheries developed a conservative approach to evaluate the potential for concentrations of contaminants to harm listed salmonids. As discussed in the effects section of this Opinion, NOAA Fisheries developed contaminant concentration benchmarks that would assist in determining the likelihood of harm to listed fish. These benchmarks are concentration levels at which there is no observable effect ("NOECs") on salmonids from long-term exposures. If measured levels of contaminants are below benchmarks, then it is assumed that take will not occur. NOAA Fisheries found that most life stages of listed salmonids are likely to be in portions of the action area where contaminant levels are above NOEC benchmarks for only short periods of time and that such short-term exposures would not harm listed salmonids. The take that will occur involves instances of long-term exposure of salmonids to chronic levels of contaminants within small portions of the lower Snake River.

The extent to which take will occur is described by the size of the affected area in combination with the presence of listed salmonids. The best available information indicates that take will occur as a result of chronic exposure to the following degraded water quality conditions:

1. *Temperature and DO*

Permit effluent limits allow water column concentrations that exceed NOAA Fisheries benchmarks for temperature and DO below each of 72 diffusers. A small zone of concentrated effects, or Zone of Initial Dilution (ZID), forms below each diffuser. The ZID that affects fish in short term exposures is very small and dependent on ambient river temperatures.

Temperature increases caused by release of Potlatch effluent are modeled to be 5˚C or less at a distance of two meters from the diffusers, 1˚C or less at three meters from the diffusers, and

0.3˚C or less at 45 meters from the diffusers during summer low flow periods. A few smolts will be harmed by effluent greater than or equal to 5˚C above ambient; therefore, the extent of take from the direct effects of the heated water is considered to be the area within two meters of the diffusers. For the few smolts encountering heated water within two meters of the diffusers, the result would be a short-term, shock-like stress reaction. A two-meter ZID comprises approximately 2-3% of the Snake River channel. Because smolts prefer to migrate over two meters off the bottom in large, deep rivers, it is expected that only a few smolts would be affected. Migrating adult and juvenile salmonids will tend not to occur in the 2-meter ZID, because of habitat preferences or seasonal timing.

Potlatch's effluents are believed to be significant contributors of biological and chemical oxygen demand (BOD and COD, respectively) and the frequently occurring DO sags in the lower Snake River and LGR. After Potlatch attains the final BOD effluent limits described in the BE, and has fully implemented the mitigation measures described in Appendices 3 and 4 of this Opinion, dissolved oxygen sags that could be attributed to BOD of the effluent will likely be indistinguishable from ambient. In the interim, take is expected from BOD-related DO increases modeled to occur within the lower Snake River. There is a general lack of adequate information on COD to make predictions on DO concentrations. The effect of COD on take is expected to be better defined through the monitoring required by the Terms and Conditions of this Opinion.

2. *Water-borne contaminants*

a. Dioxin - Predictions based on modeling describing water column concentrations of contaminants in the Snake River during low flow conditions indicate that take will likely result from juvenile salmonid exposures to chronic levels that exceed NOAA Fisheries NOEC benchmarks. This modeling indicated that juvenile salmonids residing within 500 meters downstream of the effluent plume could be exposed to dioxins above benchmark concentrations. Salmonids that are chronically exposed for several weeks or months to water-borne concentrations above benchmarks could accumulate enough dioxin to experience adverse effects. Therefore, the area within 500 meters downstream of the effluent source is considered the extent in which take is expected to occur.

Some early-arriving adult steelhead could spend enough time in the 500 meter zone downstream from the effluent such that accumulation of dioxin could cause adverse effects. Exposure by adults would not be expected until after LGR water temperatures cool in the fall or early winter. Water column models predict dioxins are diluted rapidly. Considering the residence time needed to bioaccumulate these contaminates in conjunction with low-flow river conditions, it is probable that few steelhead will experience chronic exposure. However, our knowledge of holding times of adult steelhead in the vicinity of the effluent diffusers is very limited. There is a risk that steelhead adults could bioaccumulate contaminates faster than predicted, or receive greater exposure than expected. These occurrences would result in more take of adult steelhead than predicted in this Opinion.

b. BKME - Concentrations of BKME of about 5% (3.3 toxic units) or greater are capable of causing chronic toxicity to salmonids. Under low flow conditions, the downstream extent of BKME concentrations greater than 5% would be about 11 meters. Therefore, take of juvenile and adult salmonids from BKME is expected to occur within this 11-meter zone below the diffusers. Due to the small size of this area, limited duration of the low flow conditions that create these concentrations, and because the vast majority of the listed salmonids are migrating

through rather than holding or rearing for extended periods of time, very few adult and juvenile salmon and steelhead are expected to occupy the 11-meter zone and be exposed to the high concentrations of BKME for sufficient time to be adversely affected.

3. *Monitoring*

Monitoring is a required element in this Incidental Take Statement and it must be performed to provide the information necessary to accurately predict the effects of effluent on listed salmonids in LGR. The required monitoring allows for the intentional capturing of some hatchery-reared juvenile salmon and steelhead and resident LGR fish. During capturing attempts, some listed fish could also be captured and subsequently die due to handling stress. Amounts of take from handling stress can therefore be estimated. It is expected that up to three wild juvenile steelhead and up to 15 wild juvenile spring/summer or fall chinook salmon would be killed in any one year due to this monitoring program. No sockeye salmon are expected to be taken. NOAA Fisheries authorizes the taking of three wild juvenile steelhead and 15 wild juvenile spring/summer or fall chinook salmon per year for purposes of complying with the required monitoring plan.

Elsewhere in this Opinion NOAA Fisheries has determined that based on the best information currently available, the amount of take resulting from discharges consistent with the proposed action is not likely to jeopardize the continued existence of listed salmonids or adversely modify designated critical habitat. Consequently, NOAA Fisheries authorizes incidental take resulting from Potlatch discharges as long as Potlatch is in compliance with all terms of the revised permit (including the compliance schedules for BOD and seasonal temperature) and the Conservation Measures specified in EPA's March 2, 2004, letter, and EPA and Potlatch are in compliance with the terms and conditions of this incidental take statement. NOAA Fisheries will review results of the monitoring annually to ensure that predicted levels of take are not exceeded. This incidental take statement may be revised if monitoring indicates higher levels of take than anticipated.

2.6.2 Reasonable and Prudent Measures

Reasonable and Prudent Measures (RPMs) are non-discretionary measures to minimize take that may or may not be part of the description of the proposed action. They must be implemented as binding conditions for the exemption in section 7(o)(2) to apply. The EPA has the continuing duty to regulate the activities covered in this incidental take statement. If the EPA fails to require the applicants to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, or fails to retain the oversight to ensure compliance with these terms and conditions, or if Potlatch fails to comply with this incidental take statement, the protective coverage of section 7(o)(2) may lapse.

The NOAA Fisheries believes that the following reasonable and prudent measures are necessary and appropriate to minimize take of listed fish resulting from implementation of the action. These reasonable and prudent measures would also minimize adverse effects on designated critical habitat.

The EPA and Potlatch shall:

1. Minimize the likelihood of take resulting from loading of all forms of oxygen demanding substances and nutrients into the LGR and the lower Snake River.

2. Minimize the impact of take resulting from outfall 001 suspended solids and associated effluent contaminants.

3. Better define the potential for take resulting from Potlatch's effluent below outfall 001, including from ASB seepage to the Clearwater River and LGR.

2.6.3 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the EPA and Potlatch must comply with the following terms and conditions, which implement the reasonable and prudent measures described above. These terms and conditions are non-discretionary.

1. To implement Reasonable and Prudent Measure 1:

   The EPA and Potlatch shall implement Conservation Measure 9 of Section VII.H of the BE including the amendments included in EPA's March 2, 2004 letter to NOAA Fisheries ("Potlatch Temperature Reduction Measures" and "Potlatch TSS Reduction Measures") as described in Appendices 3 and 4 of this Opinion.

   Because low DO levels could be the result of a combination of factors, including but not limited to Potlatch's effluent, in order to address DO deficits comprehensively and to allocate reductions fairly among sources, the lower Snake River should have a TMDL for DO. Before the next NPDES permit cycle for this facility, the EPA shall use all appropriate authorities to cause a TMDL(s) to be prepared for DO in the lower Snake River up to and including LGR. This TMDL(s) shall address all sources from both Idaho and Washington. EPA shall annually provide NOAA Fisheries with a status report on TMDL development and development completion schedules.

2. To implement Reasonable and Prudent Measures 2, EPA and Potlatch shall implement Conservation Measure 10 of Section VII.H in EPA's BE as amended by their March 2, 2004, letter to NOAA Fisheries ("Reduce TSS by 25%"), as described in Appendices 3 and 4 of this Opinion.

3. To implement Reasonable and Prudent Measure 3:

   a. The EPA and Potlatch shall implement Conservation Measure 12 of Section VII.H of the BE as amended by of EPA's March 2, 2004, letter to NOAA Fisheries (Monitoring and Assessment Plan) as described in Appendix 5 of this Opinion. Results of all monitoring and modeling for this Reasonable and Prudent Measures 3a. must be reported to NOAA Fisheries by January 15 of each year. The following additional elements shall be incorporated into the monitoring plan and annual reports:

      (1) Implement Opinion Appendix 6, addendum to the monitoring plan for monitoring the extent of take of ESA listed salmonids.

      (2) Measure the amount and concentration of contaminants that reach the groundwater from the ASB and eventually flow into either the Clearwater or Snake Rivers. Evaluate bioaccumulation risks to listed salmonids through sediment or food chain influences wherever ASB flows reach these rivers. EPA

shall submit a plan for these studies to NOAA Fisheries within 90 days of transmission of this Opinion and shall begin implementing the monitoring upon approval of NOAA Fisheries.

(3) Monitor COD and BOD to determine Potlatch-caused effects to the seasonally low DO levels in LGR and the lower Snake River. This information will better quantify take of salmonids in the Snake River below Outfall 001. EPA shall use best available models to predict effects of BOD and COD on listed salmonids. EPA shall submit a plan for these studies to NOAA Fisheries within 90 days of transmission of this Opinion and shall begin implementing the monitoring upon approval of NOAA Fisheries.

b. The required monitoring involves aquatic sampling, including the capture and collection of resident fish and hatchery produced salmon and steelhead. The EPA and Potlatch shall minimize take of listed salmonids during monitoring by requiring that all collection of fish specimens is done according to NOAA Fisheries approved fish rescue, salvage, and relocation methods described in Appendix 7. Any listed salmonids killed during the monitoring process must be adequately preserved and reported to NOAA Fisheries via fax or phone call within one day after a fish is killed. An annual report of take from monitoring must be provided to NOAA Fisheries by January 15 each year.

2.7 Reinitiation of Consultation

As provided in 50 C.F.R. 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) The amount or extent of taking specified in the Incidental Take Statement is exceeded, or is expected to be exceeded; (2) new information reveals effects of the action may affect listed species in a manner or to an extent not previously considered; (3) the action is modified in a way that causes an effect on listed species that was not previously considered; or (4) a new species is listed or critical habitat is designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, any operations causing such take must cease pending conclusion of the reinitiated consultation.

NOAA Fisheries will evaluate monitoring results annually to determine if new information indicates that the discharge may be affecting listed salmonids to an extent that was not anticipated in this Opinion. If effects not anticipated in the Opinion are revealed and linked to Potlatch effluent, then EPA must reinitiate consultation. NOAA Fisheries is particularly interested in the results of monitoring described in Appendix 6 because of the limited information currently available on these subjects and the potential to assess amount/extent of take with greater accuracy.

Failure to provide timely monitoring results to NOAA Fisheries may cause the Incidental Take Statement to expire. If EPA fails to provide specified monitoring information by January 15 each year, NOAA Fisheries may consider this inaction or delay to be a modification that causes an effect on ESA listed species or their habitats that was not previously considered and may cause the Incidental Take Statement to expire.

3. MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT

3.1 Statutory Requirements

The MSA, as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), established procedures designed to identify, conserve, and enhance EFH for those species regulated under a Federal fisheries management plan.

Pursuant to the MSA:

- NOAA Fisheries must provide conservation recommendations for any Federal or state action that would adversely affect EFH (§305(b)(4)(A));

- Federal agencies must provide a detailed response in writing to NOAA Fisheries within 30 days after receiving EFH conservation recommendations. The response must include a description of measures proposed by the agency for avoiding, mitigating, or offsetting the impact of the activity on EFH. In the case of a response that is inconsistent with NOAA Fisheries EFH conservation recommendations, the Federal agency must explain its reasons for not following the recommendations (§305(b)(4)(B)).

The EFH means those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity (MSA §3). 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 and may include aquatic areas historically used by fish where appropriate; 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, or growth to maturity" covers a species' full life cycle (50 C.F.R. 600.10). Adverse effect means any impact which reduces quality and/or quantity of EFH, and may include direct (e.g., contamination or physical disruption), indirect (e.g., loss of prey or reduction in species fecundity), site-specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 C.F.R. 600.810).

The EFH consultation with NOAA Fisheries is required regarding any Federal agency action that may adversely affect EFH, including actions that occur outside EFH, such as certain upstream and upslope activities.

The objectives of this EFH consultation are to determine whether the proposed action would adversely affect designated EFH and to recommend conservation measures to avoid, minimize, or otherwise offset potential adverse effects on EFH.

3.2 Identification of EFH

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 the PFMC 1999), and longstanding, naturally-impassable barriers (i.e., natural waterfalls in existence for several hundred years). Detailed descriptions and identifications of EFH for salmon are found in Appendix A to Amendment 14 to the Pacific Coast Salmon Plan (PFMC 1999). Assessment of potential adverse effects to these species' EFH from the proposed action is based, in part, on this information.

3.3 Proposed Actions

The proposed action and action area are detailed above in Sections 1.2 and 1.3 of this document. The action area includes habitats that have been designated as EFH for various life-history stages of chinook salmon.

3.4 Effects of Proposed Action on EFH

The effects on chinook salmon EFH are the same as those for ESA listed species and are described in detail in Section 2.2.1 of this document. The proposed action may result in adverse effects on a variety of habitat parameters. The primary habitat effects are: (1) near-field (mixing zone) water column increases in organic chlorines, nutrients, temperature, suspended solids and reductions in DO; (2) localized increases of AOX, organic chlorines, nutrients, and oxygen demanding substances in the water column and sediments of LGR; and (3) far-field increases of oxygen demanding substances and organic chlorines in the water column and sediments of the lower Snake River.

3.5 Conclusion

NOAA Fisheries concludes that the proposed action is likely to lead to baseline improvements in water quality for DO, nutrients, and temperature (localized mixing zone) and improvements in water and sediment quality for DO, nutrients, and organic contaminants, but will continue to adversely affect designated EFH for chinook salmon.

3.6 EFH Conservation Recommendations

Pursuant to section 305(b)(4)(A) of the MSA, NOAA Fisheries is required to provide EFH conservation recommendations for any Federal or state agency action that would adversely affect EFH. In addition to permit limitations and conservation measures proposed for the project (Introduction, section 1.2.4), and all of the reasonable and prudent measures and the terms and conditions contained in section 2.4.2, of the ESA portion of this Opinion are applicable to salmon EFH. Therefore, NOAA Fisheries incorporates each of those measures here as EFH conservation recommendations.

3.7 Statutory Response Requirement

Pursuant to the MSA (§305(b)(4)(B)) and 50 C.F.R. 600.920(j), Federal agencies are required to provide a detailed written response to NOAA Fisheries' EFH conservation recommendations

within 30 days of receipt of these recommendations. The response must include a description of measures proposed to avoid, mitigate, or offset the adverse impacts of the activity on EFH. In the case of a response that is inconsistent with the EFH conservation recommendations, the response must explain the reasons for not following the recommendations, including the scientific justification for any disagreements over the anticipated effects of the proposed action and the measures needed to avoid, minimize, mitigate, or offset such effects.

3.8 Supplemental Consultation

The EPA must reinitiate EFH consultation with NOAA Fisheries if the proposed action is substantially revised in a manner that may adversely affect EFH, or if new information becomes available that affects the basis for NOAA Fisheries' EFH conservation recommendations (50 C.F.R. 600.920(k)).

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