US Army Corps
of Engineers®
Walla Walla District

FINAL

Lower Snake River Juvenile Salmon Migration Feasibility Report/ Environmental Impact Statement

APPENDIX C

Water Quality

February 2002

FEASIBILITY STUDY DOCUMENTATION

	Document Title
Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement	
Appendix A (bound with B)	Anadromous Fish Modeling
Appendix B (bound with A)	Resident Fish
Appendix C	Water Quality
Appendix D	Natural River Drawdown Engineering
Appendix E	Existing Systems and Major System Improvements Engineering
Appendix F (bound with G, H)	Hydrology/Hydraulics and Sedimentation
Appendix G (bound with F, H)	Hydroregulations
Appendix H (bound with F, G)	Fluvial Geomorphology
Appendix I	Economics
Appendix J	Plan Formulation
Appendix K	Real Estate
Appendix L (bound with M)	Lower Snake River Mitigation History and Status
Appendix M (bound with L)	Fish and Wildlife Coordination Act Report
Appendix N (bound with O, P)	Cultural Resources
Appendix O (bound with N, P)	Public Outreach Program
Appendix P (bound with N, O)	Air Quality
Appendix Q (bound with R, T)	Tribal Consultation and Coordination
Appendix R (bound with Q, T)	Historical Perspectives
Appendix S*	Snake River Maps
Appendix T (bound with R, Q)	Clean Water Act, Section 404(b)(1) Evaluation
Appendix U	Response to Public Comments

*Appendix S, Lower Snake River Maps, is bound separately (out of order) to accommodate a special 11 x 17 format.

The documents listed above, as well as supporting technical reports and other study information, are available on our website at http://www.nww.usace.army.mil/lsr. Copies of these documents are also available for public review at various city, county, and regional libraries.

STUDY OVERVIEW

Purpose and Need

Between 1991 and 1997, due to declines in abundance, the National Marine Fisheries Service (NMFS) made the following listings of Snake River salmon or steelhead under the Endangered Species Act (ESA) as amended:

- sockeye salmon (listed as endangered in 1991)
- spring/summer chinook salmon (listed as threatened in 1992)
- fall chinook salmon (listed as threatened in 1992)
- steelhead (listed as threatened in 1997).

In 1995, NMFS issued a Biological Opinion on operations of the Federal Columbia River Power System (FCRPS). Additional opinions were issued in 1998 and 2000. The Biological Opinions established measures to halt and reverse the declines of ESA-listed species. This created the need to evaluate the feasibility, design, and engineering work for these measures.

The Corps implemented a study (after NMFS' Biological Opinion in 1995) of alternatives associated with lower Snake River dams and reservoirs. This study was named the Lower Snake River Juvenile Salmon Migration Feasibility Study (Feasibility Study). The specific purpose and need of the Feasibility Study is to evaluate and screen structural alternatives that may increase survival of juvenile anadromous fish through the Lower Snake River Project (which includes the four lowermost dams operated by the Corps on the Snake River—Ice Harbor, Lower Monumental, Little Goose, and Lower Granite Dams) and assist in their recovery.

Development of Alternatives

The Corps' response to the 1995 Biological Opinion and, ultimately, this Feasibility Study, evolved from a System Configuration Study (SCS) initiated in 1991. The SCS was undertaken to evaluate the technical, environmental, and economic effects of potential modifications to the configuration of Federal dams and reservoirs on the Snake and Columbia Rivers to improve survival rates for anadromous salmonids.

The SCS was conducted in two phases. Phase I was completed in June 1995. This phase was a reconnaissance-level assessment of multiple concepts including drawdown, upstream collection, additional reservoir storage, migratory canal, and other alternatives for improving conditions for anadromous salmonid migration.

The Corps completed a Phase II interim report on the Feasibility Study in December 1996. The report evaluated the feasibility of drawdown to natural river levels, spillway crest, and other improvements to existing fish passage facilities.

Based in part on a screening of actions conducted for the Phase I report and the Phase II interim report, the study now focuses on four courses of action:

- Existing Conditions
- Maximum Transport of Juvenile Salmon

- Major System Improvements
- Dam Breaching.

The results of these evaluations are presented in the combined Feasibility Report (FR) and Environmental Impact Statement (EIS). The FR/EIS provides the support for recommendations that will be made regarding decisions on future actions on the Lower Snake River Project for passage of juvenile salmonids. This appendix is a part of the FR/EIS.

Geographic Scope

The geographic area covered by the FR/EIS generally encompasses the 140-mile long lower Snake River reach between Lewiston, Idaho and the Tri-Cities in Washington. The study area does slightly vary by resource area in the FR/EIS because the affected resources have widely varying spatial characteristics throughout the lower Snake River system. For example, socioeconomic effects of a permanent drawdown could be felt throughout the whole Columbia River Basin region with the most effects taking place in the counties of southwest Washington. In contrast, effects on vegetation along the reservoirs would be confined to much smaller areas.

Identification of Alternatives

Since 1995, numerous alternatives have been identified and evaluated. Over time, the alternatives have been assigned numbers and letters that serve as unique identifiers. However, different study groups have sometimes used slightly different numbering or lettering schemes and this has led to some confusion when viewing all the work products prepared during this long period. The primary alternatives that are carried forward in the FR/EIS currently involve the following four major courses of action:

Alternative Name	PATH[1/] Number	Corps Number	FR/EIS Number
Existing Conditions	A-1	A-1	1
Maximum Transport of Juvenile Salmon	A-2	A-2a	2
Major System Improvements	A-2’	A-2d	3
Dam Breaching	A-3	A-3a	4

[1/] Plan for Analyzing and Testing Hypotheses

Summary of Alternatives

The **Existing Conditions Alternative** consists of continuing the fish passage facilities and project operations that were in place or under development at the time this Feasibility Study was initiated. The existing programs and plans underway would continue unless modified through future actions. Project operations include fish hatcheries and Habitat Management Units (HMUs) under the Lower Snake River Fish and Wildlife Compensation Plan (Comp Plan), recreation facilities, power generation, navigation, and irrigation. Adult and juvenile fish passage facilities would continue to operate.

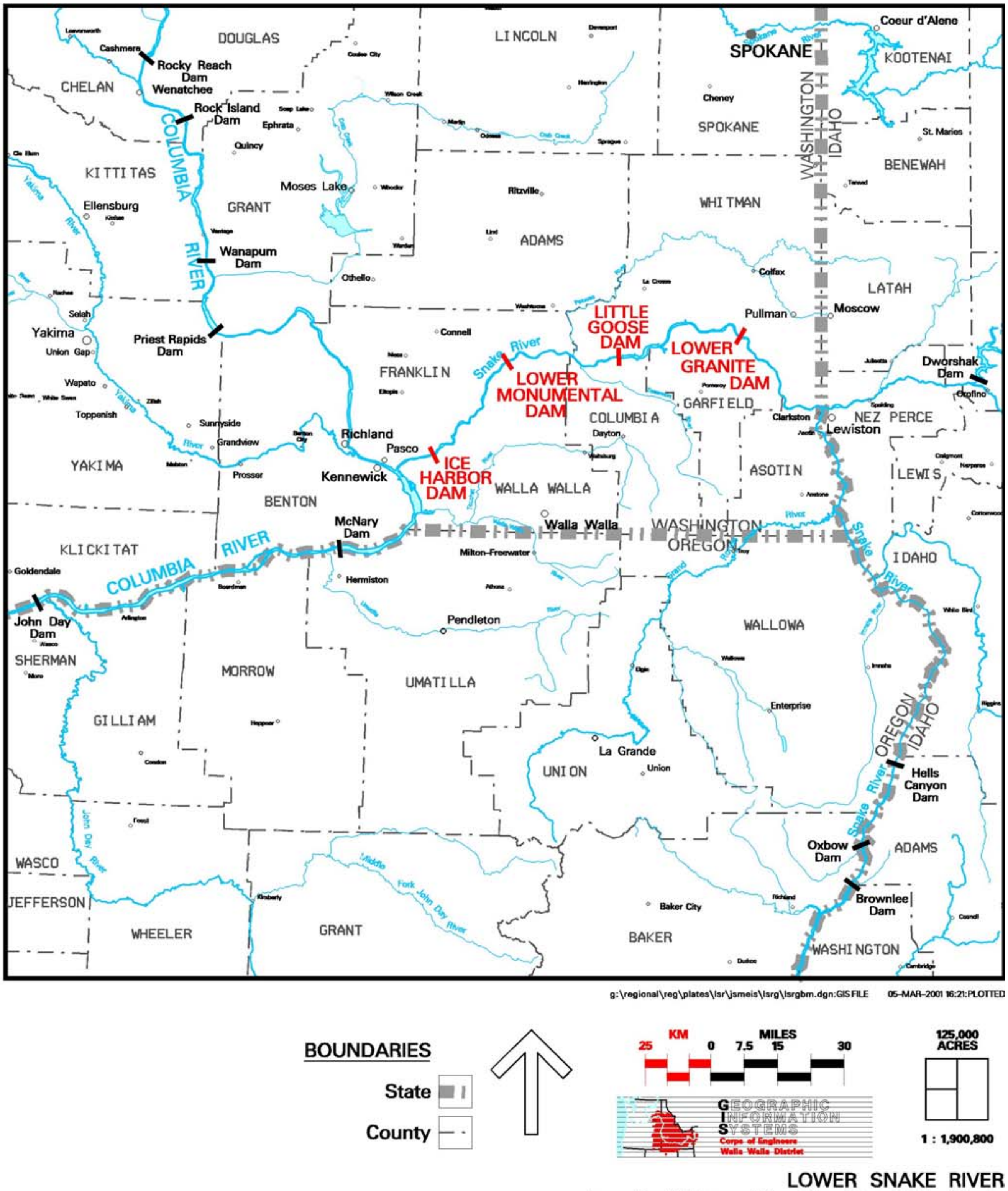
The **Maximum Transport of Juvenile Salmon Alternative** would include all of the existing or planned structural and operational configurations from the Existing Conditions Alternative. However, this alternative assumes that the juvenile fishway systems would be operated to maximize fish transport from Lower Granite, Little Goose, and Lower Monumental and that voluntary spill would not be used to bypass fish through the spillways (except at Ice Harbor). To accommodate this maximization of transport, some measures would be taken to upgrade and improve fish handling facilities.

The **Major System Improvements Alternative** would provide additional improvements to what is considered under the Existing Conditions Alternative. These improvements would be focused on using surface bypass facilities such as surface bypass collectors (SBCs) and removable spillway weirs (RSWs) in conjunction with extended submerged bar screens (ESBSs) and a behavioral guidance structure (BGS). The intent of these facilities would be to provide more effective diversion of juvenile fish away from the turbines. Under this alternative, an adaptive migration strategy would allow flexibility for either in-river migration or collection and transport of juvenile fish downstream in barges and trucks.

The **Dam Breaching Alternative** has been referred to as the "Drawdown Alternative" in many of the study groups since late 1996 and the resulting FR/EIS reports. These two terms essentially refer to the same set of actions. Because the term drawdown can refer to many types of drawdown, the term dam breaching was created to describe the action behind the alternative. The Dam Breaching Alternative would involve significant structural modifications at the four lower Snake River dams, allowing the reservoirs to be drained and resulting in a free-flowing yet controlled river. Dam breaching would involve removing the earthen embankment sections of the four dams and then developing a channel around the powerhouses, spillways, and navigation locks. With dam breaching, the navigation locks would no longer be operational and navigation for large commercial vessels would be eliminated. Some recreation facilities would close while others would be modified and new facilities could be built in the future. The operation and maintenance of fish hatcheries and HMUs would also change, although the extent of change would probably be small and is not known at this time.

Authority

The four Corps dams of the lower Snake River were constructed and are operated and maintained under laws that may be grouped into three categories: 1) laws initially authorizing construction of the project, 2) laws specific to the project passed subsequent to construction, and 3) laws that generally apply to all Corps reservoirs.



LOWER SNAKE RIVER
Juvenile Salmon Migration Feasibility Study

REGIONAL BASE MAP

US Army Corps of Engineers®
Walla Walla District

Final
Lower Snake River Juvenile Salmon Migration Feasibility Report/ Environmental Impact Statement

Appendix C
Water Quality

Produced by

Normandeau Associates

and

U.S. Army Corps of Engineers
Walla Walla District

Produced for

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February 2002

FOREWORD

Appendix C was prepared by Normandeau Associates, Inc. in conjunction with the U.S. Army Corps of Engineers' (Corps) study team. Foster Wheeler Environmental Corporation was a contributor to the technical portions of the appendix. This appendix is one part of the overall effort of the Corps to prepare the Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement (FR/EIS).

The Corps has reached out to regional stakeholders (Federal agencies, tribes, states, local governmental entities, organizations, and individuals) during the development of the FR/EIS and appendices. This effort resulted in many of these regional stakeholders providing input and comments, and even drafting work products or portions of these documents. This regional input provided the Corps with an insight and perspective not found in previous processes. A great deal of this information was subsequently included in the FR/EIS and appendices; therefore, not all of the opinions and/or findings herein may reflect the official policy or position of the Corps.

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ACRONYMS AND ABBREVIATIONS

ABA	attached benthic algae
AET	apparent no-effect
AFODW	ash-free, oven dry weights
AMPA	aminomethylphosphonic acid
B	barometric pressure
BGS	behavioral guidance structure
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
CCC	criteria continuous concentration
cfs	cubic feet per second
CLR	Columbia River
CLW	Clearwater River
CoC	chemicals of concern
Comp Plan	Lower Snake River Fish and Wildlife Compensation Plan
Corps	U.S. Army Corps of Engineers
CRFMP	Columbia River Fish Mitigation Program
CRiSP	Columbia River Salmon Passage
DGAS	Dissolved Gas Abatement Study
DGS	dissolved gas supersaturation
DO	dissolved oxygen
Ecology	Washington State Department of Ecology
EIS	Environmental Impact Statement
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
ESBS	extended submerged bar screen
FCRPS	Federal Columbia River Power System
Feasibility Study	Lower Snake River Juvenile Salmon Migration Feasibility Study
FGE	fish guidance efficiency
flip lips	spillway flow deflectors
FLUSH	Fish Leaving Under Several Hypotheses
FMS	fixed monitoring station
FPE	fish passage efficiency
FR/EIS	Lower Snake River Juvenile Salmon Migration Feasibility Report/ Environmental Impact Statement
FTU	Formazin turbidity units
GBD	gas bubble disease
GBT	gas bubble trauma
HMU	Habitat Management Unit
IHR	Ice Harbor
ISAB	Independent Scientific Advisory Board
kcfs	thousand cubic feet per second
LGO	Little Goose
LGR	Lower Granite

ACRONYMS AND ABBREVIATIONS

LM	Lower Monumental
LSR	lower Snake River
MASS1	Modular Aquatic Simulation System 1D
mg/kg	milligrams per kilograms
mg/L	milligrams per liter
MOP	minimum operating pool
MW	megawatt
NCFAP	National Center for Food and Agricultural Policy
NEPA	National Environmental Policy Act
NGVD	National Geodetic Vertical Datum
NMFS	National Marine Fisheries Service
NO_2	Nitrate
NO_3	Nitrite
NMFS	National Marine Fisheries Service
NPDES	National Pollutant Discharge Elimination System
NTU	Nephelometric Turbidity Unit
NYDEC	New York Department of Environmental Conservation
ODEQ	Oregon Department of Environmental Quality
PAH	polynuclear aromatic hydrocarbons
ppb	parts per billion
ppm	parts per million
ppt	parts per trillion
RBM10	River Base Model 10
RM	River Mile
RSW	removable spillway weir
SBC	surface bypass collector
SCS	System Configuration Study
SNR	Snake River
SOR	System Operation Review
TCDD	tetrachlorodibenzo-p-dioxin
TCDF	tetrachlorinated dibenzo furan
TDG	total dissolved gas
TDGMS	total dissolved gas monitoring station
Temp.	temperature
TEQ	toxicity equivalence quantity
TGP	total gas pressure
TKN	total kjeldahl nitrogen
TMDL	total maximum daily load
TMT	Technical Management Team
TN	total nitrogen
TP	total phosphorous
TPH	total petroleum hydrocarbons
TSS	total suspended solids

ACRONYMS AND ABBREVIATIONS

Turb.	turbidity
USGS	U.S. Geological Survey
WAC	Washington Administrative Code
WRC	Water Research Center
WQRRS	Water Quality River Reservoir Systems

ENGLISH TO METRIC CONVERSION FACTORS

To Convert From	To	Multiply By
LENGTH CONVERSIONS:		
Inches	Millimeters	25.4
Feet	Meters	0.3048
Miles	Kilometers	1.6093
AREA CONVERSIONS:		
Acres	Hectares	0.4047
Acres	Square meters	4047
Square Miles	Square kilometers	2.590
VOLUME CONVERSIONS:		
Gallons	Cubic meters	0.003785
Cubic yards	Cubic meters	0.7646
Acre-feet	Hectare-meters	0.1234
Acre-feet	Cubic meters	1234
OTHER CONVERSIONS:		
Feet/mile	Meters/kilometer	0.1894
Tons	Kilograms	907.2
Tons/square mile	Kilograms/square kilometer	350.2703
Cubic feet/second	Cubic meters/sec	0.02832
Degrees Fahrenheit	Degrees Celsius	(Deg F –32) x (5/9)

UNIT DEFINITIONS

parts per million (ppm) ≅ mg/L
parts per billion (ppb) ≅ µg/L
parts per trillion (ppt) ≅ ng/L

Executive Summary

This Water Quality Appendix was prepared as part of the Lower Snake River Juvenile Salmon Migration Feasibility Study (Feasibility Study). This document describes existing water quality conditions within the study reach (a 225-kilometer [140-mile] corridor through the four lower Snake River reservoirs to the mouth of the Snake River as it enters the Columbia River). Also included were areas above the study reach on the lower Snake River and Clearwater River to establish base conditions for the water entering the upper reservoir (Lower Granite) and areas below the lower Snake River in the McNary reservoir on the Columbia River.

This document describes four alternative action plans being evaluated to improve fish passage through the lower Snake River reservoir system and anticipated impacts on water quality associated with each alternative. Primary water quality concerns include temperature, total dissolved gas, dissolved oxygen (DO), sediment accumulation and chemical contamination, nutrient levels, and trophic ecology. This analysis included a review and comparison of available water quality data collected from various locations on the lower Snake River going back to 1960, detailed data collected from key locations between 1994 and 1997, and other data collected to assess the effects of dredging. The more recent data were used to model biological productivity in a near-natural lower Snake River, which could be compared with productivity in the reservoir system as it is being currently operated and to characterize chemical contamination in accumulated sediments.

Existing Conditions (Alternative 1)

For this alternative, the reservoir projects would continue to be regulated under current operating plans in place or under development at the time of this report.

General Water Quality Conditions:

- The quality of Snake River water flowing into the Lower Granite reservoir is described as excellent (Class A).
- The quality of the Clearwater River water flowing into the Lower Granite reservoir is considered exceptional and special resource water.
- The quality of Snake River water within the lower Snake River reservoir system is described as excellent (Class A) to good (Class B) depending on the primary use and specific location.

Summer Water Temperatures (June through September):

- Snake River water temperatures above the Lower Granite reservoir typically reach a maximum of 23 degrees Celsius (°C) (73 degrees Fahrenheit [°F]) each year and normally exceed a base temperature of 20°C (68°F) approximately 60 days each year.
- Clearwater River water temperatures above the Lower Granite pool typically reach a maximum of 21°C (70°F) and normally exceed a base temperature of 20°C (68°F) approximately 5 days each year.
- Snake River water temperatures below the lower Snake River reservoir system typically reach a maximum of 23°C (73°F) each year and exceed a base of 20°C (68°F) approximately 60 days each year.

- Typical day and night temperature fluctuations above the Lower Granite pool on the Snake River are approximately 0.5 to 1.5°C (0.9 to 2.7°F) and 1.5 to 3.0°C (2.7 to 5.4°F) on the Clearwater River.
- Water temperatures within the reservoir system have a 1 to 2°C (1.8 to 3.6°F) smaller day and night temperature fluctuation than upstream inflow to the Lower Granite reservoir.
- Typical day and night temperature fluctuations on the Snake River below the reservoir system are approximately 0.4 to 1.0°C (0.7 to 1.8°F).
- Temperatures at any point within the lower Snake River reservoir system are typically zero to 2°C (0 to 3.6°F) warmer or cooler than the Snake River water flowing into the reservoir system at the Lower Granite reservoir depending on the time of year, location, flow conditions, current flow augmentation and temperature control operations, and voluntary spill/power operations.
- Average water temperatures within the reservoir system warm slower by approximately 1 week and cool slower by approximately 2 weeks than the Snake River water flowing into the Lower Granite reservoir.
- Flow augmentation with cold water from the Dworshak reservoir on the North Fork Clearwater River is effective in reducing water temperatures in the Lower Granite reservoir and has benefited fall chinook salmon rearing in the Snake River.

Total Dissolved Gas:

- The TDG levels typically reach 115 percent to 130 percent each year during the annual spring period (May or June) when Snake River flows exceed power plant capacities (approximately 130,000 cubic feet per second [cfs]) at the dams and large amounts of water are released through the spillbays (involuntary spill).
- The TDG levels typically reach 115 percent to 120 percent each year and are operated to those levels as water is released through the spillbays to assist in fish passage (voluntary spill).

Dissolved Oxygen:

- The DO levels in the Snake River are too low in the late summer/fall low-flow period when biochemical oxygen demand depletes oxygen near the bottom in deep water.
- The DO levels in the period August through November are typically the lowest of the year and in the range of 6 to 10 mg/L. Low DO levels make the deep areas of the reservoirs uninhabitable by fish, which reduces access to cooler ground water upwellings. Without these cool water refugia, the ability of fish to tolerate high water temperature is affected.
- The DO levels in the reservoir system (December through July) typically improve because higher river flows are circulating the deep-water areas and water temperatures are low.

Sediment:

- The lower Snake River reservoir system accumulates an average of approximately 2.3 to 3.1 million cubic meters (3.0 to 4.0 million cubic yards) of sediment each year (primarily within the Lower Granite reservoir and the Palouse River and Tucannon River deltas). Since the construction of the lower Snake River reservoirs, approximately 76.5 to 115 million cubic meters (100 to 150 million cubic yards) of sediment have accumulated within the system.

- These sediments have accumulated chemical contaminants, which were analyzed by intensive sampling that was conducted as part of this study and in other studies. Contaminants of particular concern were identified as ammonia, nitrogen, manganese, heavy metals, pesticides, herbicides, and dioxin.
- Chemical contaminants are locked in the sediments as long as they remain undisturbed.

Productivity:

- Productivity within the reservoir system is based on phytoplankton and typically varies with the season. There is usually low productivity in the winter and higher productivity during the growing season from the end of April or May through September or October.

Maximum Transport of Juvenile Salmon (Alternative 2)

For this alternative, the existing condition configurations of Alternative 2 would be used except Lower Granite, Little Goose, and Lower Monumental projects would be operated to maximize fish transport, and voluntary spill would not be used at those projects.

General Water Quality Conditions:

- No change from the existing conditions.

Summer Water Temperatures (June through September):

- No change from the existing conditions.

Total Dissolved Gas:

- No change from the existing conditions for the high-flow, forced spill periods.
- Without voluntary spill, TDG levels would be reduced to or below 110 percent at Lower Granite, Little Goose, and Lower Monumental except for the forced spill periods.

Dissolved Oxygen:

- No change from the existing conditions.

Sediment:

- No change from the existing conditions.

Productivity:

- No change from the existing conditions.

Major System Improvements (Alternative 3)

For this alternative, the existing condition configurations of Alternative 2 would be used. In addition, at Lower Granite, major improvements such as SBC facilities, extended submerged bar screen (ESBS), and BGS system would be used to collect fish. Then, fish would be delivered to upgraded transportation facilities. Each project could get combinations of training walls and additional redesigned spillbays.

General Water Quality Conditions:

- No change from the existing conditions.

Summer Water Temperatures (June through September):

- No change from the existing conditions.

Total Dissolved Gas:

- Through combinations of training walls and additional redesigned spill bays at each project, TDG levels could be reduced to 110 percent or less during periods of no forced spill. During periods of forced spill, TDG levels could be reduced from 130 percent levels to approximately 115 percent.

Dissolved Oxygen:

- No change from the existing conditions.

Sediment Quality:

- No change from the existing conditions.

Productivity:

- No change from the existing conditions.

Dam Breaching (Alternative 4)

For this alternative, the earthen embankment sections of the four dams would be removed, allowing the reservoirs to be drained and resulting in a near-natural river that would remain unimpounded. This action would span a period of 9 years with the first 3 to 4 years focused on the engineering and design processes. The embankments of the four dams would be breached during two construction seasons in the process. Construction work dealing with mitigation and restoration of the various facilities adjacent to the former reservoirs would last for 3 to 4 years.

During the first 3 years before dam breaching, additional water quality sampling would be performed, and monitoring plans and requirements would be developed for the dam breaching period. During the dam breaching periods, water quality monitoring would be conducted throughout the lower Snake River reach and into the McNary reservoir. The monitoring effort is expected to cost a total of approximately $9.6 million for the periods before, during, and after dam breaching.

General Water Quality Conditions:

- The period before dam breaching would result in no change in the water quality classifications.

- The period during dam breaching and for a period of approximately 1 to 3 years after breaching, water quality classifications would change from excellent (Class A) and good (Class B) to poor as resuspended sediments and contaminants within the four lower Snake River reservoir area are transported downstream into the McNary reservoir.

- From 3 years after dam breaching and into the future, the water quality conditions would improve and be similar to those conditions currently found in the Snake River above the existing Lower Granite reservoir pool. This classification would be excellent (Class A). The Clearwater River would mix with the Snake River water and may improve the resultant water slightly.

Summer Water Temperatures (June through September):

- Analyses of the effects of dam breaching on water temperature are based on both empirical data and model simulations. The results of these two analysis methods vary slightly but are in agreement.

- The U.S. Environmental Protection Agency (EPA) provided its water temperature modeling expertise and resources to evaluate the effects of the reservoirs using its RBM-10 model to simulate 1980, 1984, 1988, 1994, 1995, and 1997 conditions with and without the reservoirs at Snake River RM 10 (Ice Harbor) and RM 107 (Lower Granite).
- Empirical data indicate that water temperatures within the study reach after dam breaching would be similar to those found on the Snake River above the existing Lower Granite pool. The maximum summer water temperature expected each year would typically reach 23°C (73°F) and would exceed a 20°C (68°F) benchmark temperature approximately 60 days (which are the approximate conditions found within the existing reservoirs dependent upon location and operations). Fluctuations between day and night water temperatures would typically be approximately 0.5 to 1.5°C (0.9 to 2.7°F) within the water column and 1 to 2°C (1.8 to 3.6°F) at the water surface. Spring water temperatures after breaching will warm faster (approximately 1 week) than the existing reservoir temperatures and will cool faster (approximately 2 weeks) in the late summer than the existing reservoir temperatures.
- RBM-10 simulations indicate approximately the same maximum summer water temperatures of approximately 22 to 23°C (72 to 73°F) with and without the dams. The number of days that a benchmark temperature of 20°C (68°F) would be exceeded at RM 107 in an average flow year would be 46 days for the reservoir condition and 44 days for the near-natural river condition. At RM 10 the computed number of days exceeding 20°C (68°F) was 57 days for the reservoir condition and 46 days for the near-natural river condition.
- RBM-10 simulations show greater differences in the 1994, 1995, and 1997 simulations when Dworshak Dam augmentation with cold water was used to compute temperature differences between the existing condition and the near-natural river condition. In an average flow year, the number of days the temperature exceeded 20°C (68°F) at RM 107 goes from 64 with the dams to 59 without the dams.
- According to RBM-10 simulations, the effect of the dams on average temperature during the hot period of the year (June through August) is minimal with temperature going from 18.9°C (66°F) with the reservoirs in place to 19.1°C (66.4°F) for a near-natural river condition.
- The RBM-10 simulations showed that the near-natural river would warm up approximately 1 week sooner than the reservoir condition, and it would cool down about 2 weeks sooner than the reservoir condition.
- The RBM-10 simulations showed that daily water temperatures would fluctuate 1 to 2°C (1.8 to 3.6°F) more with the near-natural river condition as compared to the reservoir temperatures.

Total Dissolved Gas:

- After dam breaching, the TDG levels found in the lower Snake River reach would be similar to those found above the existing Lower Granite reservoir pool and would be 110 percent or less.

Dissolved Oxygen:

- After dam breaching, the summer low DO conditions experienced with reservoirs in place would not exist.

Sediment:

- Sediment and contaminant water quality conditions would be poor during the dam breaching and for approximately 1 to 3 years after breaching. From after 3 years and into the future, the sediment and contaminant conditions would continue to improve as sediments move through the lower Snake River reach and into the McNary reservoir.
- Extremely high suspended sediment levels would be experienced in the lower Snake River for at least 3 years following initial dam breaching. Aquatic life in the river, including various life stages of ESA listed fish stocks, would be heavily impacted.
- Chemicals of concern are ammonia (NH_3), Total DDT, manganese, and dioxin TEQ. Ammonia is expected to be most critical but dependent upon water temperature and pH; thus, timing of activities is very important. Cool water temperatures reduce the problems.
- The sediment and contaminant conditions within the McNary reservoir would worsen after lower Snake River dam breaching. The Snake River sediments and contaminants would be resuspended and would move into the McNary reservoir over time. Any of the existing Lower Granite reservoir problems associated with sediments and contaminants would be transferred to McNary.
- More data would need to be collected (primarily resuspension concentrations for the contaminants) to adequately define the water quality conditions that would be present during and for the first 2 years after dam breaching.

Productivity:

- For at least 3 years after dam breaching, the productivity of the lower Snake River would be degraded by the sediments moving through the river into the McNary pool. When equilibrium is reached in the river, primary and secondary productivity would improve and the lower Snake River would produce more suitable food for rearing salmonids than when impounded.

1. Introduction

The primary objectives of this appendix are to summarize the existing limnological/water quality, sediment quality, and primary productivity conditions throughout the lower Snake River and to describe the potential changes that may result from the various alternatives being considered. For each alternative evaluated, including the existing conditions alternative, a description of both anticipated short-term effects (i.e., construction and/or transition period) and long-term effects is provided.

Section 2.0 provides background information on the study reach, available water quality data, and information on a sediment quality study performed to provide some of the information for this evaluation.

Section 3.0 of this document contains a detailed description of the existing water quality conditions and aquatic biology in the lower Snake River system based on data collected since the start of construction of the lower Snake River reservoir projects. The study reach of the lower Snake River now consists of a series of large impoundments, in sharp contrast to its former "free-flowing" riverine system. As a result, the physical, chemical, and biological nature of this river reach has changed. The reduced flow velocities within these impoundments not only extend time of travel through this reach, but also result in increased sediment accumulation in the Lower Granite reservoir, lower turbidity, increased nutrient availability, and greater light penetration. This provides excellent growing conditions for algae, especially planktonic algae (i.e., phytoplankton). The aquatic food web has consequently changed in favor of planktonic consumers and fish species typical of open-water reservoir systems.

Section 4.0 describes the various proposed alternatives and the potential effects of these alternatives on the lower Snake River water quality and aquatic biology. The proposed dam breaching alternative has the greatest potential to affect water quality. A major short-term effect is the re-suspension and downstream movement of sediment accumulated behind the four dams. During the initial transition period following dam breaching, heavy sediment loads can be expected to move downstream, affecting water quality and substrate conditions in the next downstream impoundment. The duration and magnitude of this impact on beneficial uses of the lower Snake and Columbia River waters are discussed. Over the long term, however, major changes in lower Snake River water quality are not expected, because most of the influences on water quality relate to the land-based activities that occur within the watershed and existing discharges located along the river. The aquatic biology and associated food web, however, would likely change to reflect aquatic communities more typical of riverine environments.

In 1994, a similar Water Quality Appendix was prepared as part of the Columbia River System Operation Review (SOR) EIS process. The objectives of that document were quite similar in providing a description of existing conditions and an assessment of future changes associated with various alternatives including reservoir drawdown (Bonneville Power Administration [BPA], 1995). However, this earlier study differs in that it focused on the entire Columbia River basin, a broader range of alternatives, and was a description of existing conditions based primarily on pre-1990 water quality data. These data were collected as part of several long-term monitoring programs conducted by various state and Federal agencies and universities. In this Appendix, the description of existing water quality conditions is based on both the earlier data and the more recent data collected since 1994 by research teams from the University of Idaho, Washington State University, and NMFS under contract with the Corps.

The most recent water quality data contain a broader list of parameters, including biological productivity data (Normandeau, 1999a). In addition, given the number of sampling stations and selected locations, these data provide a more synoptic view of existing conditions throughout the lower Snake River. This detail reveals the effects that the lower Snake River reservoir system may have on water quality, as well as those attributable to upstream and tributary contributions. The effects of the lower Snake River on the Columbia River system can also be more closely evaluated with this database because sampling was conducted above and below the Snake River confluence. Comparisons of those data to other historical, long-term data are made wherever possible to identify any relevant historical trends. However, comparisons to natural free-flowing riverine conditions are difficult given that there is very limited pre-impoundment data available for the lower Snake River, aside from limited water quality data collected prior to completion of Lower Granite Lock and Dam in 1975.

Predictions of future water quality conditions under the proposed alternatives were based on the results of previous and recent modeling efforts, including the previous SOR modeling of sediment movement and the more recent modeling of potential biological productivity and temperature changes under the proposed “near natural reach” drawdown scenario (Normandeau, 1999a). This more recent effort used the Water Quality River Reservoir Systems (WQRRS) model to simulate changes in the hydrodynamics, temperature, and biological productivity under the proposed drawdown conditions. Additional temperature modeling has been conducted by the EPA (Yearsley, 1999) and the Pacific Northwest Laboratory (Perkins and Richmond, 1999). The analysis of potential impacts in this appendix focused on those parameters that are most likely to have an effect on anadromous fish, as well as those that are most likely to be affected by dam breaching.

2. Background

2.1 Project Study Area

The Snake River originates in western Wyoming at Yellowstone National Park, and flows approximately 1,609 kilometers (1,000 miles) through the states of Idaho, Washington, and Oregon to its confluence with the Columbia River near Pasco and Burbank, Washington. It is the largest tributary to the Columbia River and drains an area of approximately 282,000 square kilometers (109,000 square miles), including most of Idaho and portions of Oregon, Washington, Wyoming, Nevada, and Utah. The topography within the basin ranges from steep mountainous areas, mainly in the upper headwater areas, to extensive volcanic plateaus and plains that have been deeply incised by the river over geologic time. The Snake River flows through several different physiographic provinces including the Columbia Plateau/Basalt Plain, which extends east from the foothills of the Cascade Range in Washington and Oregon to western Idaho; the Snake River Plain, which extends from southeastern Oregon, across southern Idaho and northern Nevada and Utah; the Blue Mountains province, which extends from southeastern Washington to central Oregon; and, the Northern Rocky Mountains province, which encompasses much of Idaho and Wyoming (BPA, 1995). Elevations range from approximately 152 meters (500 feet) along the gorges of the lower Snake River in the Columbia Plateau physiographic province to more than 3,048 meters (10,000 feet) in the mountains (BPA, 1995). The geology primarily consists of basaltic and granitic rocks and to a lesser extent consolidated sedimentary rocks and alluvium. Soils within the drainage area of the Snake River generally consist of young alluvial materials along the lower terraces of the river and a fine wind-deposited loess in large areas of the uplands in the Columbia Plateau. In addition, areas of glacial outwash and lake-bed silts caused by past glacial activity can be found in the Columbia Plateau. Soils within the Rocky Mountain province include a variety of parent materials, including metamorphic rock, as well as deposits of glacial drift, outwash, and alluvium (BPA, 1995).

The study area includes the lower Snake River drainage basin, associated tributaries, and a portion of the middle Columbia River (see Figure 2-1). The lower Snake River, the primary focus of this study, consists of a 225-kilometer (140-mile) reach extending from the point of confluence with the Columbia River, upstream to the Clearwater River near Lewiston, Idaho. Between 1961 and 1975, the Corps completed construction and began operating four run-of-river lock and dam projects on this reach of the Snake River. These project uses include navigation, power generation, recreation, irrigation, and fish and wildlife. The following listing summarizes some key elements of the lower Snake River reservoir system projects.

Project	RM	Date of Pool Raise	Hydraulic Height (feet)	Total Storage (acre-feet)	Usable Storage (acre-feet)	Normal Operating Range NGVD*
Ice Harbor	9.7	Nov. 27, 1961	105	406,900	24,900	437 to 440
Lower Monumental	41.6	Jan. 6, 1969	103	376,300	20,000	537 to 540
Little Goose	70.3	Jan. 25, 1970	101	565,200	48,900	633 to 638
Lower Granite	107.5	Feb. 14, 1975	105	483,800	43,600	733 to 738

* NGVD = National Geodetic Vertical Datum

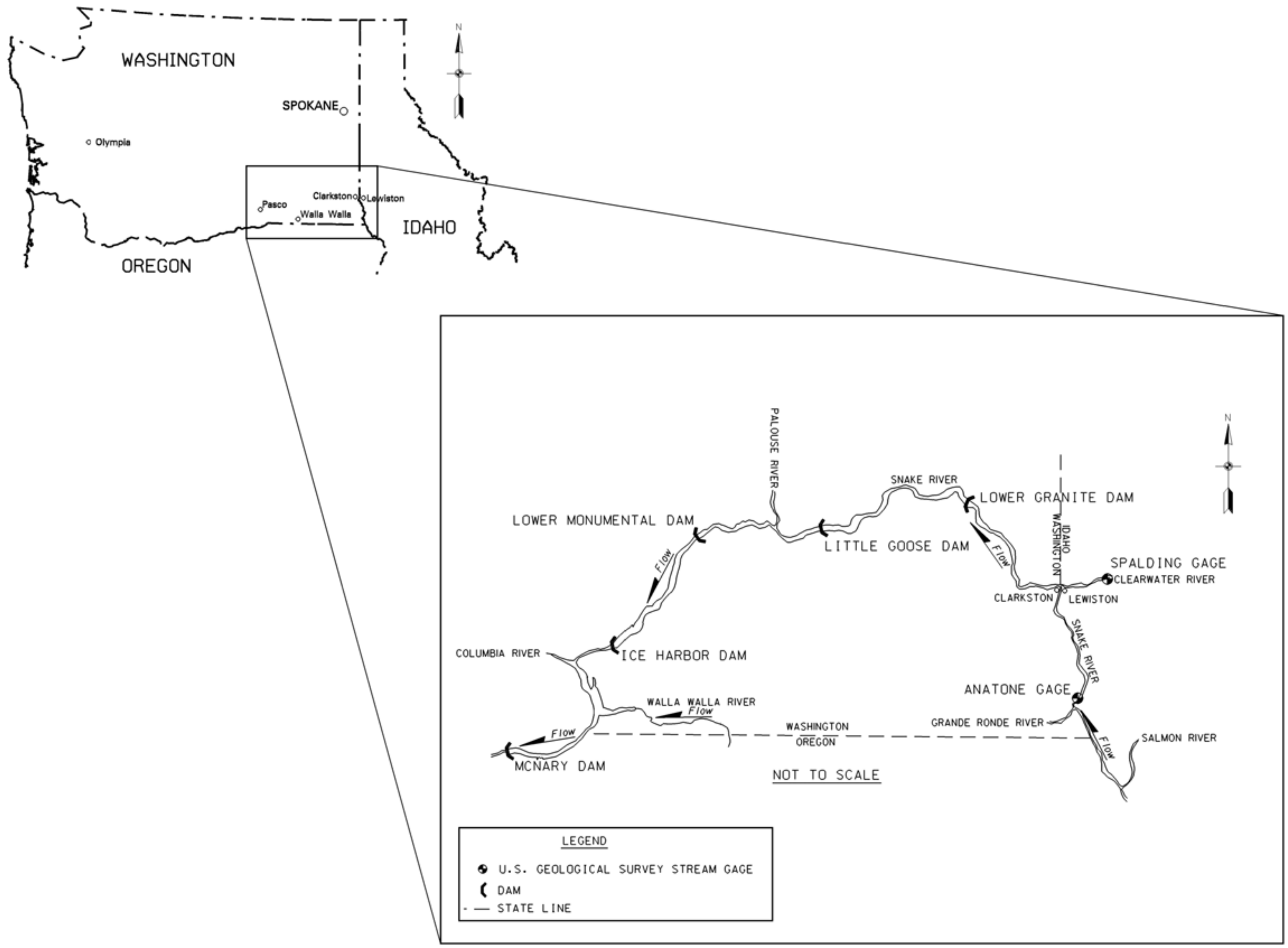


Figure 2-1. Lower Snake River Study Area

Source: Developed by Normandeau

The hydraulic height is considered the difference between the maximum forebay and minimum tailwater in the normal operating range. Total storage is computed at the top of the normal operating range at low flow. Usable storage is the storage in the normal operating range at low flow.

The reservoir pool behind each of these four dams is approximately 30 meters (100 feet) deep at each damsite and approximately 5 meters (16 feet) deep at the upper end of the pool. The length of each of these pools ranges from 45 to 65 kilometers (28 to 45 miles). Lower Granite pool extends 39.3 miles upstream on the Snake River and 4.6 miles on the Clearwater River.

McNary Lock and Dam is a run-of-river project on the Columbia River and was constructed by the Corps. It began operating in 1956 and is operated by the Corps for navigation, power generation, recreation, irrigation, and fish and wildlife. The dam is located on the Columbia River at river mile (RM) 292, which is approximately 51 kilometers (32 miles) below the mouth of the Snake River. The reservoir pool from the dam extends a length of approximately 98 kilometers (61 miles) and has a total reservoir storage of 1,600 million cubic meters (1.3 million acre-feet) as compared to 2,200 million cubic meters (1.8 million acre-feet) for all four Snake River projects. The pool is normally operated between elevations 102 and 104 meters (335 and 340 feet) at the dam and uses approximately 228 million cubic meters (185,000 acre-feet) of the total storage for that 1.5-meter (5-foot) range. The McNary Lock and Dam project is included in this appendix to identify potential water quality changes or sediment resuspension within the McNary pool that might take place as a result of some of the alternatives being considered on the Snake River reach.

2.2 Summary of the Available Long-term Water Quality Data

The previous Columbia River SOR, an earlier interagency study, provided an extensive review of the available long-term water quality data collected in the Columbia and Snake river basins prior to 1990 (BPA, 1995). There are eight long-term monitoring stations along the lower Snake River used by a number of state and Federal agencies going back as far as 1975 (Table 2-1). The sampling locations, frequency, and number of years sampled varies among the various agencies. Much of this monitoring focused on a few key parameters including temperature, pH, conductivity, turbidity, dissolved oxygen (DO), and TDG supersaturation. The Corps monitored these parameters, as well as Secchi transparency, within each of the reservoirs at a limited frequency of one to four times a year. Occasionally, other parameters such as hardness, total suspended solids (TSS), turbidity, and nutrient levels were measured. The EPA and the individual states conducted ambient water quality monitoring programs to primarily assess compliance status and trends. The Washington State Department of Ecology (Ecology) sampled intensively (i.e., up to 10 samples per year) in 1975 for these same parameters plus fecal coliform bacteria. The U.S. Geological Survey (USGS) samples about once a year at two long-term monitoring stations on the lower Snake River (Anatone [RM 167] and Burbank [RM 2.2], Washington) and one on the Clearwater River at Spalding, Idaho (RM 11), where similar parameters were tested. The University of Washington and the University of Idaho analyzed pre-impoundment water quality at the Lower Granite Lock and Dam area from 1970 to 1972 (Falter et al., 1973). Limited data were collected on various toxics including heavy metals, pesticides, and other organic compounds.

Table 2-1. Summary of Long-term Water Quality Monitoring Data for Various Sampling Locations throughout the Project Area (page 1 of 2)

River/Location	River Mile	Agency	Sampling Period	No. of Years	Sampling Frequency	Parameters[1/]
			Columbia River			
McNary Dam, Tailwater	291	Corps	1993-pres	7+	Apr-Sep (Cont)	B, TDG, Temp, DO
McNary Dam, Forebay	292	Corps	1984-pres	15+	Apr-Sep (Cont)	B, TDG, Temp, DO
McNary Pool	295/306 295	Corps EPA	1975-90 1974-76	9 2	2-3/yr 9-10/yr	Conventional parameters except TSS and TN Temp and DO only
Above Snake River Confluence	326	Corps	1997-98	9	2-3/yr	Conventional parameters except TSS and TN
Richland, WA	340	USGS EPA	1979-90 1975-92	12 18	1/yr 6-9/yr	Conventional parameters except nutrients Temp, DO, Cond, Turb, pH, TP, and OP
			Snake River			
Burbank, WA	2.2 8.7	USGS	1960-69, 1972-78; 1979-90	16	1/yr	Conventional parameters
Ice Harbor Dam, Tailwater	6.0	Corps	1991-pres	9+	Apr-Sep (Cont)	B, TDG, Temp, DO
Ice Harbor Dam, Scroll Case	9.7	Corps	1961-pres	39+	Continuous	Temp (100 foot depth)
Ice Harbor Dam, Forebay	9.7	Corps	1984-pres	15+	Apr-Sep (Cont)	B, TDG, Temp, DO
Ice Harbor Pool	18.0	Corps EPA Ecology	1975-90 1975 1975-90	9 1 15	3/yr 5/yr 6-10/yr	Conventional parameters except TSS and TN Conventional par except TSS, Turb, Hardness Conventional parameters except nutrients
Lower Monumental Dam, Tailwater	40.6	Corps	1991-pres	9+	Apr-Sep (Cont)	B, TDG, Temp, DO
Lower Monumental Dam, Scroll Case	40.6	Corps	1971-pres	29+	Continuous	Temp (100 foot depth)
Lower Monumental Dam, Forebay	41.6	Corps	1984-pres	15+	Apr-Sep (Cont)	B, TDG, Temp, DO
Lower Monumental Pool	44.0	Corps EPA	1975	1	5/yr	Temp, Cond, DO, pH, Turb
Little Goose Dam, Scroll Case	69.5	Corps	1972-pres	27+	Continuous	Temp (100 foot depth)

Table 2-1. Summary of Long-term Water Quality Monitoring Data for Various Sampling Locations throughout the Project Area (continued) (page 2 of 2)

River/Location	River Mile	Agency	Sampling Period	No. of Years	Sampling Frequency	Parameters[1/]
Snake River (continued)						
Little Goose Dam, Tailwater	69.5	Corps	1978-1992	9+	Apr-Sep (Cont)	B, TDG, Temp, DO
Little Goose Dam, Forebay	70.3	Corps	1984-pres	15+	Apr-Sep (Cont)	B, TDG, Temp, DO
Little Goose Pool	83.0	Corps EPA	1984-pres 1975	9 1	1/yr 5/yr	Temp, Cond, DO, pH, Turb
Lower Granite Dam, Tailwater	106.7	Corps	1991-pres	9+	Apr-Sep (Cont)	B, TDG, Temp, DO
Lower Granite Dam, Scroll Case	107.5	Corps	1975-pres	25+	Continuous	Temp (100 foot depth)
Lower Granite Dam, Forebay	107.5	Corps	1984-pres	15+	Apr-Sep (Cont)	B, TDG, Temp, DO
Lower Granite, Lower Pool	106.5	Corps USGS EPA	1978-89 1975-78 1975-77	9 4 4	1-2/yr 1/yr up to 25/yr	Conventional parameters Temp & Cond mostly Temp, DO, Cond, Turb, pH, TP, and OP
Lower Granite, Upper Pool	120.0	Corps USGS	1978-92 1974-77	9 3	1-2/yr 1/yr	Temp, Cond Conventional parameters except TSS, TP, and OP
Anatone, WA	167.0	USGS Corps	1974-pres 1999	20+ 1	1/yr Apr-Sep (Cont)	Temp and Cond; other par less frequently B, TDG, Temp, DO
Clearwater River						
North Fork	0.5	Corps	1993-pres	7+	Apr-Sep (Cont)	B, TDG, Temp, DO
Peck	4.2	Corps	1996-pres	3	Apr-Sep (Cont)	B, TDG, Temp, DO
Spalding, ID	11.0	USGS	1974-pres	20+	1/yr	Temp and Cond mainly; other parameters less frequently
Lewiston	37.4	Corps	1996-pres	3	Apr-Sep (Cont)	B, TDG, Temp, DO

1/ Conventional parameters consist of temperature (Temp.), conductivity (Cond.), dissolved oxygen (DO), pH, TSS, turbidity (Turb.), total nitrogen (TN), nitrate and nitrite (NO_2 and NO_3), total phosphorus (TP), and orthophosphate (OP). Other parameters include TDG measured continuously (TDG), and barometric pressure (B).
Source: Developed by Normandeau

2.3 Summary of Recent Water Quality Data Collected within the Lower Snake River

In 1994, the Corps initiated an extensive sampling program throughout the lower Snake River basin with the assistance of research teams from Washington State University, NMFS, and the University of Idaho. The primary goal of this sampling program was to provide a more complete synopsis of the existing limnological and biological productivity conditions above, below, and throughout the lower Snake River reach and to assess the effects, if any, that the various dams have on water quality. Sampling was conducted both in the impoundments and in the "free-flowing" reaches and major tributaries. Sampling was also conducted in the Columbia River above and below the Snake River confluence (Table 2-2). Initially, in 1994 and 1995, data were collected on a monthly or bi-weekly basis within the lower Snake River system. The sampling frequency was increased in 1997 to bi-weekly monitoring through the growing season in both the lower Snake River and portions of the Columbia River. An extensive suite of parameters was sampled during these investigations, including many of the same conventional parameters used in the long-term monitoring studies such as pH, alkalinity, conductivity, DO, nutrients, TSS, and turbidity. Various anions and cations were also monitored including chloride, silica, sulfate, calcium, magnesium, sodium, and potassium. In addition, biochemical oxygen demand was also measured at selected locations, as well as various biological parameters including chlorophyll *a*, phytoplankton, zooplankton, attached benthic algae (ABA), and other primary productivity indicators.

A range of hydrological conditions was encountered during the recent sampling program, including a relatively dry year in 1994 (ranging from 11 thousand cubic feet per second [kcfs] to 93 kcfs and ranked near the lowest 10 percent of annual flows); an average year in 1995 (ranging from approximately 15 kcfs to 149 kcfs); and a wet year in 1997 (ranging from approximately 15 kcfs to 225 kcfs), based on historical streamflow data. The comparison of water quality conditions collected during a range of hydrologic conditions will assist in estimating how future conditions might be different, if at all, under various hydrologic conditions.

Researchers at Washington State University recently compiled much of these recent and long-term water quality data for the Columbia and Snake river basins into a computerized database using the Microsoft Access (Version 2.0) Program. This database was added to the existing NMFS database and represents the principal resource used in preparing a description of existing water quality conditions for this Appendix.

2.4 Description of Relevant Water Quality Sampling Station Locations

Figure 2-2 illustrates the locations of various water quality sampling stations throughout the study area. As many as 13 sampling stations were established along the mainstem of the lower Snake River. Upstream and downstream stations bracketed each of the four dams, accounting for eight stations. Other key sampling stations include those representing the major tributary inputs to the lower Snake River, as well as two additional stations in the upper portions of the Lower Granite reservoir at RM 118 and 129. Stations SNR-140 and SNR-148 represented water quality conditions in the upper areas of the lower Snake River reach, upstream of the Clearwater River confluence and downstream of the middle Snake River reach. Station 148 is located in a "free-flowing" zone of the river that extends another 257 kilometers (160 miles) upstream. The water quality at this location is still somewhat influenced by the biological and limnological conditions of the upstream impoundments as part of that from the Hells Canyon Dam Complex, as well as inputs from the Grande Ronde River and other tributaries.

Table 2-2. Sampling Stations in the Clearwater River, Lower Snake River, Columbia River, Palouse River, and Tucannon River in 1997 Page 1 of 2

Station Name	River	RM	Reach	Reach Type		Purpose
CLW-11	***Clearwater***	~ 11	Spalding	Free-flowing	PP/Limno/ABA	Free-flowing Clearwater River, little controlled.
CLW-1		~ 1	Lewiston	Free-flowing	Limno	Free-flowing Clearwater River before it merges with the Snake River. Included in previous studies and complements the upstream primary productivity site. Also useful for eliciting any changes between stations.
SNR-148	***Snake***	~ 148	Asotin	Free-flowing	PP/Limno/ABA	Free-flowing Snake River, little controlled.
SNR-140		~ 140	Lewiston/ Clarkston	Free-flowing	Limno	Free-flowing Snake River used in previous studies. Analogous benefits as the Clearwater 1 station.
SNR-129		~ 129	Lower Granite reservoir	Transition zone	Limno	Visited in previous studies and represents the transition between riverine and lacustrine environments.
SR-118		~ 118	Lower Granite reservoir	Reservoir	PP/Limno/ABA	Represents the location in Lower Granite pool where complete mixing of the inflowing Snake and Clearwater Rivers has occurred. Previously visited and part of the primary productivity study.
SNR-108		~ 108	Above Lower Granite Dam	Reservoir	Limno	Site close to the forebay that was included in previous studies and located at deepest part of the reservoir.
SNR-106 SNR-105		~ 106/105	Below Lower Granite Dam	Free-flowing/ reservoir mix	PP/Limno/ABA	Hybrid of free-flowing/ reservoir, but more riverine.
SNR-83		~ 83/81	Little Goose reservoir	Reservoir	PP/Limno/ABA	Only station that has consistently been sampled in the Little Goose reservoir, and was included in the primary productivity study.
SNR-66		~ 68/67	Below Little Goose Dam	Free-flowing/ reservoir mix	PP/Limno/ABA	Hybrid of free-flowing/ reservoir, but more riverine.
SNR-50	***Snake***	~ 52/50	Lower Monumental reservoir	Reservoir	PP/Limno/ABA	Snake River impoundment.
SNR-40		~ 40/37	Below Lower Monumental Dam	Free-flowing/ reservoir mix	PP/Limno/ABA	Hybrid of free-flowing/ reservoir, but more riverine.

Table 2-2. Sampling Stations in the Clearwater River, Lower Snake River, Columbia River, Palouse River, and Tucannon River in 1997 (continued) Page 2 of 2

Station Name	River	RM	Reach	Reach Type		Purpose
SNR-18		~ 18	Ice Harbor reservoir	Reservoir	PP/Limno/ABA	The only site that has routinely been sampled in the Ice Harbor reservoir.
SNR-6		~ 6	Below Ice Harbor Dam	Free-flowing/ reservoir mix	PP/Limno/ABA	Hybrid of free-flowing/ reservoir, but more riverine.
CLR-397	***Columbia***	~ 410/397	Priest Rapids reservoir	Reservoir	PP/Limno/ABA	Impoundment and unconfounded by pollution inputs.
CLR-369		~ 369	Hanford reach	Free-flowing	PP/Limno/ABA	True free-flowing river with similar gradient to the lower 140 miles of the Snake River.
CLR-326		~ 326	McNary reservoir	Transition zone	Limno/ ABA	Similar to RM 129 on the Snake River in that it is at the upper end of the McNary reservoir and in the transition between riverine and lacustrine environments. Also upstream from the confluence of the Snake River.
CLR-306		~ 306	McNary reservoir	Reservoir	Limno	Impoundment receiving Snake River flows and sampled in the past by the Corps.
CLR-295		~ 295	McNary reservoir	Reservoir	PP/ Limno	The closest station to McNary Dam that has traditionally been sampled by the Corps.
PAL-6	***Tributaries***	~ 6	Palouse River	Free-flowing	Limno	Has relatively small flow volume compared to the lower Snake River, but can have extremely high concentrations of nitrate and suspended solids.
TUC-1		~ 1	Tucannon River	Free-flowing	Limno	Has less discharge than the Palouse River, but water quality unknown at beginning of study.

Notes: PP-Primary Productivity Sampling
Limno-Limnological Sampling
ABA-Attached Benthic Algae Sampling
Source: Developed by Normandeau

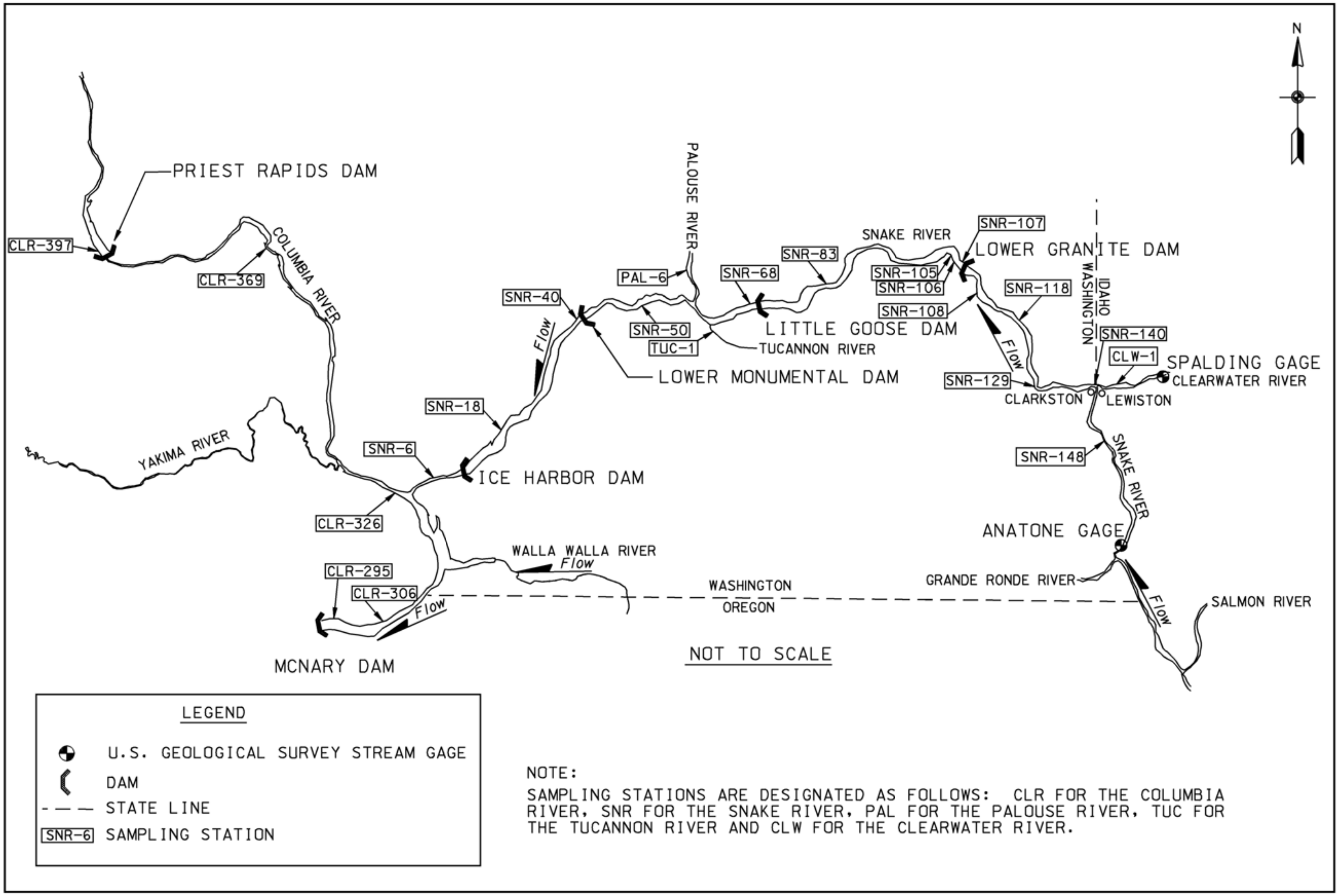


Figure 2-2. Limnology/Primary Productivity Sampling Stations

Source: Developed by Normandeau

Within the Columbia River, Stations CLR-295 and CLR-306 were located in the lower end of the McNary reservoir, while Station CLR-326 was located just above the Snake River confluence and Station CLR-397 was just above Priest Rapids Dam.

Several free-flowing stream sections were sampled to compare water quality conditions between impounded and nonimpounded reaches. In the Columbia River, a large free-flowing section near Hanford was sampled at Station CLR-369. Stations CLW-11, PAL-6, and TUC-1 were used to represent conditions in the Clearwater, Palouse, and Tucannon Rivers, respectively.

2.5 Sediment Quality Study

To assess the potential impacts from sediment transport associated with the drawdown alternative, a study of existing sediment conditions was initiated in 1997. The study was a two-phase effort and encompassed the collection of sediment samples from all four reservoirs. During the first phase (Phase 1), sediment samples were collected and analyzed to determine the grain size of the materials. During the second phase (Phase 2), additional sediment samples were collected from selected locations and submitted for chemical analyses. A summary of the methodology used to analyze existing sediment quality is presented below, and details are provided in Normandeau (1999b).

During Phase 1, sediment samples were collected along transects established across the reach of the lower Snake River and upstream of each dam. Three additional transects were sampled in the McNary reservoir for a total of 54 transects. Sampling during Phase 1 focused on identifying those locations within the study reaches where the river bed sediment consisted primarily of very fine sand 0.002 to 0.005 inch (0.062 to 0.125 millimeter) and silt/clay-size (less than 0.002 inch [less than 0.062 millimeter]) particles (CH2M HILL, 1998). These locations were to be revisited during Phase 2 for the collection of sediment samples for the analysis of inorganic and organic chemical constituents. Only those areas where fine-grained sediments are present were of interest because it is assumed that only the fine-grained sediments will be eroded and transported by the free-flowing water if the dam breaching alternative is implemented and because any organic or inorganic contaminants of concern would be most likely concentrated in the finer-grain-size fraction due to their physio-chemical properties.

Phase 2 of the study involved collection of sediment core samples from the areas identified in Phase 1 as having the highest percentage of fine particles. At each of the sediment sampling locations, river water samples were also collected. The river water samples were collected to perform elutriate tests and to determine existing water quality conditions.

The sediment samples were analyzed for a variety of parameters including metals, semivolatiles, herbicides, pesticides, organics, mercury, and nutrients. Elutriates were prepared for ambient pH. The results were used for the sediment evaluation to estimate impacts to water quality.

3. Affected Environment

3.1 Water Resources

The Snake River is the largest tributary to the Columbia River, composing about 42 percent of the overall drainage area and about 18 percent of the water flow in the Columbia River system. From its origin in Yellowstone National Park to its confluence with the Columbia River near Pasco, Washington, the river flows slightly more than 1,609 kilometers (1,000 miles). The Snake River watershed is about 282,000 square kilometers (109,000 square miles), comprising most of Idaho, the eastern part of Oregon, and lesser parts of Washington, Wyoming, Nevada, and Utah. The topography within the basin ranges from steep mountainous areas, mainly in the upper headwater areas, to extensive volcanic plateaus and plains that have been deeply incised by the river over geologic time. The geology primarily consists of basaltic and granitic rocks and to a lesser extent consolidated sedimentary rocks and alluvium. Annual precipitation ranges from 20 centimeters (8 inches) or less in much of the lower plains to more than 203 centimeters (80 inches) in the higher mountains (Laird, 1964).

Historically, the average annual flow for the lower Snake River segment is 49.8 kcfs. Peak monthly flows average around 115 kcfs in June compared to an average seasonal monthly low flow of around 20 kcfs in September. The highest historical flow was 409 kcfs recorded in 1894, and the lowest historical flow was 10 kcfs and 6.6 kcfs recorded in 1931 and 1958, respectively (the latter occurring during construction of Brownlee Dam).

The Clearwater River, the largest tributary to the lower Snake River segment, historically contributes about 39 percent of the combined flow in the lower Snake River reach (BPA, 1995). The USGS maintains a gaging station at Spalding, Idaho, 18 kilometers (11 miles) upstream from the Snake River. The peak monthly flow generally occurs in May and averages between 45 and 50 kcfs. The low-flow period typically occurs in September when the average monthly flows generally range between 3 and 5 kcfs. Flows from the mainstem Clearwater River, along with the recent additional releases from Dworshak Dam on the North Fork of the Clearwater River can compose close to 50 percent of the lower Snake River flows during periods of low flow.

The other principal tributaries to the lower Snake River include the Palouse River and the Tucannon River, which empty into the Snake River at RM 60 and RM 65, respectively. The Palouse River is the larger of the two tributaries, but total flow contributions from these two tributaries are relatively minor and generally make up less than 1.5 percent of the Snake River flow. The relative low-flow contributions from these tributaries do not appear to have a major effect on overall water quality in the lower Snake River, although they may be locally significant.

3.2 Water Quality

3.2.1 General Description

The water quality of the Snake River reach is described by others to be excellent (Class A) to good (Class B) depending on the primary use and specific location (BPA, 1995). The middle section of the Snake River, above the Hells Canyon reservoir complex, has been reported to show poorer water quality conditions than the lower sections below Brownlee Dam. The EPA has described the middle Snake River reach as having marginal water quality due primarily to nonpoint pollution sources such as irrigation return flow and runoff from grazing areas. The Idaho Department of Health and Welfare has also reported an increasing trend in bacteria, nutrients, and suspended sediment concentrations as the river flows from Marsing (RM 240) to Weiser (RM 351.3). Although both the bacteria and sediment levels were noted to decrease as the river flows through the Hells Canyon reservoir, elevated nutrient levels continued to be of concern downstream along with occasional low DO levels (BPA, 1995). Agriculture in the drainage basin has an impact on water quality both in terms of reduced flows due to irrigation withdrawals and increased nutrients, salts, sediments, and pesticides from the return flow. Agriculture represents the largest nonpoint source of pollution and uses the largest amount of surface water within the basin. In 1964, more than 1,133,160 hectares (2.8 million acres) were estimated to be under irrigation (Laird, 1964), and irrigated agriculture is by far the largest segment of economic activity in the basin.

As the river flows through the Lewiston-Clarkston area (RM 140), the river water quality is potentially affected by the discharge of urban runoff and secondary-treated wastewater effluent. The sources of these discharges are a pulp mill and municipal wastewater treatment plants at Lewiston, Idaho, and Clarkston, Washington.

The water quality of the Clearwater River is considered exceptional, better than the lower and middle portions of the Snake River. The Clearwater River, which contributes as much as 50 percent of the lower Snake River flow during low-flow periods, generally has a beneficial effect on the lower Snake River water quality. The USGS and recent Corps data indicate that the Clearwater River is quite low in dissolved solids, nutrients, and productivity and lacks any inorganic and organic contaminants (BPA, 1995). This is attributable to the largely granitic geology and minimal development or agriculture within its watershed. The Dworshak Dam, located on the North Fork of the Clearwater River, is a relatively large dam with a structural height of 219 meters (717 feet) and a reservoir with a storage capacity of approximately 3,700 million cubic meters (3 million acre-feet). The reservoir thermally stratifies every year with a summer thermocline at about 12 to 15 meters (40 to 50 feet) deep (BPA, 1995). Water temperatures below this depth remain constant throughout the year at about 4 to 5°C (39 to 41°F). The Corps has initiated controlled flow releases of this deep cooler water from about July through August to reduce water temperatures in the lower Snake River. This added flow from the Clearwater River has had some effect on the water quality and biological productivity in the lower Snake River (Normandeau, 1999a).

Within the lower Snake River, water temperatures and TDG generally represent the primary water quality concerns related to fisheries, particularly anadromous fish. Numerous studies and measures have been implemented over the years to alleviate elevated levels of both parameters. Spillway deflectors (flip-lips) have been installed at the face of spillways to reduce dissolved gas entrainment caused by the plunging effect into the stilling basins. As discussed earlier, cooler-temperature water from within the Dworshak reservoir has been released during the summer months for temperature control. These flow augmentations generally lowered temperatures 2 to 3°C (3.6 to 5.4°F) in the Clearwater River and possibly 1 to 2°C (1.8 to 3.6°F) in the Lower Granite reservoir, with diminishing effects downstream on the Snake River (Normandeau, 1999a; Karr et al., 1997). However, a relatively large release of water from the Dworshak reservoir in 1994 (a low-flow year) lowered temperatures in the Clearwater River by as much as 10°C (18°F) and lowered temperatures for some measured points within the Lower Granite reservoir by as much as 5°C (9°F) at 6 meters depth (Bennett et al., 1997).

Other water quality concerns include turbidity and nutrient levels resulting from irrigation return flows. Increased nutrient concentrations can lead to greater productivity in the impoundments, reduced water clarity, and lower dissolved oxygen levels in the bottom waters. Extensive algal blooms have been noted to periodically occur throughout the lower Snake River and up into the Clearwater River (Normandeau, 1999a).

3.2.2 Water Quality Standards

The states of Washington and Oregon have established surface water quality standards. Oregon has water quality standards in the Columbia River for that portion of McNary reservoir, which is within the State of Oregon. In addition, the State of Idaho also has water quality standards for the uppermost portion of the Lower Granite reservoir, above the Clearwater River confluence. Each of the state standards is typically based on, and sometimes more stringent than, the EPA water criteria that were developed for the protection of aquatic life and beneficial water uses.

According to the State of Washington's four-tiered water classification system, which ranges from Class AA (extraordinary) to Class C (fair), the lower Snake River is currently classified as Class A (excellent). Beneficial uses for Class A waters include water supply (domestic, industrial, agricultural); stock watering; fish and shellfish rearing, spawning, and harvesting; wildlife habitat; recreation (primary contact); and commerce and navigation. The lower Snake River is listed as "water quality limited" for total dissolved gas and water temperature under Section 303(d) of the Clean Water Act.

In protecting the various beneficial uses of surface waters, the states have imposed numerical standards for several key parameters including temperature, dissolved oxygen, TDG saturation, turbidity, pH, and fecal coliform bacteria. These are in addition to the Federal aquatic life criteria and primary and secondary drinking water criteria that are also referenced and incorporated into the State Water Quality Standards (Washington Administrative Code [WAC] 173-201A; Rev: November 18, 1997). The numerical standards for each key water quality parameter are depicted in Table 3-1 and discussed below. A detailed discussion of existing water quality in the lower Snake River within the context of the key water quality parameters begins in Section 3.2.3.

3.2.2.1 Temperature

Washington's water quality standards specify that water temperatures in the lower Snake River shall not exceed 20°C (68°F) as a result of human activity. In addition, temperature increases due to human activity in the lower Snake River (i.e., below the Clearwater River) shall not exceed t = 34/(T+9) °C where t = change in temperature and T = background temperature. For example, if the background temperature were 20°C (68°F), then the maximum allowable temperature increase due to human activity would be 1.17°C (2.1°F). Above the Clearwater River (RM 139.3), increases over 0.3°C (0.5°F) caused by human activity from a single source are not allowed, and increases over 1.1°C (2°F) from all activities are not allowed when the background stream temperature is over 20°C (68°F)

Oregon also allows no water temperature increases in the Columbia River, outside an assigned mixing zone, when the stream water temperature is at or above 20°C (68°F). When the river temperature is 19.7°C (67.5°F) or less, the Oregon standard dictates that no more than a 0.28°C (0.5°F) increase is allowed due to a single-source discharge. No more than a 1.1°C (2°F) increase is allowed by all sources when the stream temperature is 19°C (66°F) or less. Idaho's specific temperature criteria for salmonid spawning calls for a maximum water temperature set at 13°C (55°F) with daily averages no greater than 9°C (48.2°F) for the Clearwater River from its mouth to RM 15, which is designated as a "special resource water." The Idaho standard for temperature in the mainstem Snake River from the confluence of the Clearwater River (RM 139) to Asotin, Washington at RM 147 calls for a maximum water temperature of 22°C (72°F) with daily averages no greater than 19°C (66°F).

Table 3-1. Water Quality Standards in Oregon, Idaho, and Washington

Parameters	Oregon	Idaho	Washington
Temperature	≥ 20°C (68°F): No increase, single source ≤ 19.7°C (67°F): Increase < 0.28°C (.5°F), single source < 19°C (66°F): Increase < 1.1°C (2°F), all sources	Maximum instantaneous temp: 22°C (72°F)[1] Daily average: < 19°C (66°F) Maximum instantaneous temp.: 13°C (55°F)[2] Daily average: < 9°C (48°F)	Temp.: ≤ 20°C (68°F) Temp. < 34 (T[3] + 9) °C
Dissolved Oxygen	≥ 90 percent saturation	≥ 6.0 milligrams per liter, 30-day mean ≥ 4.7 milligrams per liter, 7-day mean ≥ 3.5 milligrams per liter L, minimum ≥ 6.0 milligrams per liter or 90 percent saturation (salmonid spawning)	≥ 8 milligrams per liter
Total Dissolved Gas	≤ 110 percent saturation[4,5] ≤ 105 percent saturation[6]	≤ 110 percent	< 110 percent[4] < 120 percent, during salmon migration[5]
Turbidity	≤ 10 percent increase	≤ 5 nephelometric turbidity unit (NTU) increase[7] < 10 NTU increase[8] < 50 NTU[9] < 10 NTU[10]	≤ 5 NTU increase[7] < 10 NTU increase[8]
pH	6.5 – 8.5	6.5 – 8.5	6.5 – 8.5
Fecal coliform	100 organisms/100 millileter[11]	100 organisms/100 milliliter[11]	100 organisms/100 milliliter[11]

1/ Standards for the Snake River; Clearwater River Confluence (RM 139) to Asotin, Washington (RM 147)
2/ Standard for the Clearwater River; mouth (RM 0) to Potlatch River Confluence (RM 15). This reach is designated as a “special resource water.”
3/ T = Background Temperature
4/ Except when stream flow exceeds 10-year, 7-day average flood frequency
5/ Waivers to 120 percent in tailrace and 115 percent in forebay of downstream dam, with 125 percent maximum for 1 to 2 hours during voluntary spills
6/ In hatchery-receiving waters and when depth < 2 feet.
7/ If background is ≤ 50 NTU.
8/ If background instantaneous measure is > 50 NTU.
9/ Instantaneous, outside mixing zone.
10/ 10 consecutive days
11/ Geometric Mean
Source: Developed by Normandeau

3.2.2.2 Dissolved Oxygen

In Washington, dissolved oxygen concentrations for Class A water must be equal to or greater than 8 milligrams per liter during all times of the year. Oregon specifies at least 90 percent saturation for its portions of the Columbia River. Idaho requires the following minimum limits: at least 6 milligrams per liter (30-day mean); 4.7 milligrams per liter (7-day mean); 3.5 milligrams per liter instantaneous minimum); and 6 milligrams per liter or 90 percent of saturation (whichever is greater) for salmonid spawning purposes.

3.2.2.3 Total Dissolved Gas Supersaturation

According to Oregon Administrative Rules, Chapter 340, relating to Water Quality Control, the standards for "the concentration of TDG relative to atmospheric pressure at the point of sample collection shall not exceed 110 percent of saturation, except when stream flow exceeds the 7-day, 10-year average flood. However, for hatchery-receiving waters and waters less than two feet in depth, the concentration of TDG relative to atmospheric pressure at the point of sample collection shall not exceed 105 percent of saturation." Also, an adjacent portion of the rules states that, "the liberation of dissolved gases, such as carbon dioxide, hydrogen sulfide, or other gases, in sufficient quantities to cause objectionable odors or to be deleterious to fish and other aquatic life, navigation, recreation, or other reasonable uses made of such water shall not be allowed."

According to Chapter 173-201 WAC, the State of Washington water quality standards for surface waters, the Columbia River, from the mouth to the Washington-Oregon border, "shall not exceed 110 percent of saturation at any point of sample collection." The water quality criterion for TDG does not apply when the stream flow exceeds the 7-day, 10-year average flood frequency. In addition, a special exemption for salmonid migration has been promulgated that allows 115 to 120 percent during specific periods.

In recent years, the states of Oregon and Washington have been granting, at NMFS's request, standards waivers to allow the voluntary spill for fish passage to occur. The 110 percent limit was adjusted to 120 percent in the tailrace and 115 percent in the forebay of the next dam downstream, with a maximum of 125 percent TDG 1 hour average. The Oregon and Washington waivers have applied to the March 23 to August 31 period.

3.2.2.4 Turbidity

Washington and Idaho specify that increases in turbidity levels shall not exceed 5 Nephelometric Turbidity Units (NTU) when the background level is 50 NTU or less, and no more than a 10 NTU increase is allowed when background is more than 50 NTUs. In addition, the State of Idaho allows for a mixing zone and requires that the instantaneous level below this mixing zone not exceed 50 NTUs instantaneous measurement, or 25 NTUs for 10 consecutive days. Oregon simply specifies that no more than a 10 percent increase over background is allowed.

3.2.2.5 pH

All three states require pH levels to be within 6.5 and 8.5 pH units.

3.2.2.6 Bacteria

Fecal coliform bacteria levels must be less than a geometric mean value of 100 organisms/100 milliliters in all three states.

3.2.3 Summary of Existing Water Quality/Limnology Conditions

The following sections provide a synopsis of the relevant physical, chemical, and biological parameters that can be used to characterize water quality/limnological conditions within the lower Snake River system. This synopsis is based on all available data with current-condition evaluations utilizing the recent sampling data collected by Washington State University and University of Idaho between 1994 and 1997.

3.2.4 Hydrology and Meteorological Conditions

Figures 3-1 and 3-2 present combined monthly mean discharges for the Snake River at Lower Granite Lock and Dam from 1975 through 1977 (period before flow augmentation and cold water releases), and from 1994 through 1997 (after special operations began), along with historical monthly averages from 1974 through 1999. In 1997, average monthly flows were considerably higher than the historical monthly averages, especially between May and July.

Monthly average flows ranged from a high of about 170 kcfs in May to a low of approximately 25 kcfs in November and December. Flows in August and September of 1997 were nearly twice as high as the historical average flows of 20 to 25 kcfs for these months. In 1995, the mean monthly flows were very close to the historical monthly averages for the first half of the year and reflect slightly wetter conditions during the summer and fall months. In 1994, average monthly flow levels were consistently below the historical averages with a high of about 75 kcfs during May and a low of around 10 kcfs for much of August and September. The August and September flow levels were nearly 50 percent lower than historical averages for these months. The average flow data for 1975 through 1977 contained 2 years that had above-average flows (1975 and 1976), and 1 year (1977), that was primarily below average. Although the flow rates at Ice Harbor Lock and Dam varied from Lower Granite Lock and Dam, the same seasonal flow pattern and annual variability can be seen in Figures 3-3 and 3-4. These figures depict the flow data at Ice Harbor Lock and Dam for the same time periods.

Figure 3-5 illustrates the monthly mean air temperatures recorded at the Nez Perce weather station in Lewiston, Idaho, during the years of 1994 through 1997 in comparison to historical monthly averages. Differences in mean monthly temperatures between sampling years are not as dramatic relative to the fluctuations in flow conditions. In 1994, the mean monthly air temperatures were fairly close to normal except during the critical months of July, August, and September, when slightly warmer temperatures were recorded. Both 1995 and 1997 had slightly below-normal mean monthly air temperatures in June and July. August was also cooler than normal in 1995, whereas in 1997 both August and September were slightly warmer than normal. Based on this comparison, the air temperatures in 1994 most likely had the greatest influence on peak water temperatures in the lower Snake River, followed by 1997, and then 1995.

3.2.4.1 Water Temperature

Introduction

Temperature represents one of the most important characteristics of river water. It affects other physical properties, such as dissolved oxygen, and also influences the chemical and biological reactions that take place in aquatic systems (Calow and Petts, 1992). Transfers of heat energy fundamentally determine the temperature of a river or reservoir. Energy inputs are short-wave solar radiation and long-wave atmospheric radiation, condensation and precipitation, conduction, and through advection of heat from groundwater, upstream, and tributary inflows. Heat energy is lost from the system through reflection of solar and atmospheric radiation, back radiation from the water surface, evaporation, and as the heat content of water leaving a reach. These processes, along with ranges of values for those that occur at the air-water interface, are illustrated in Figure 3-6.

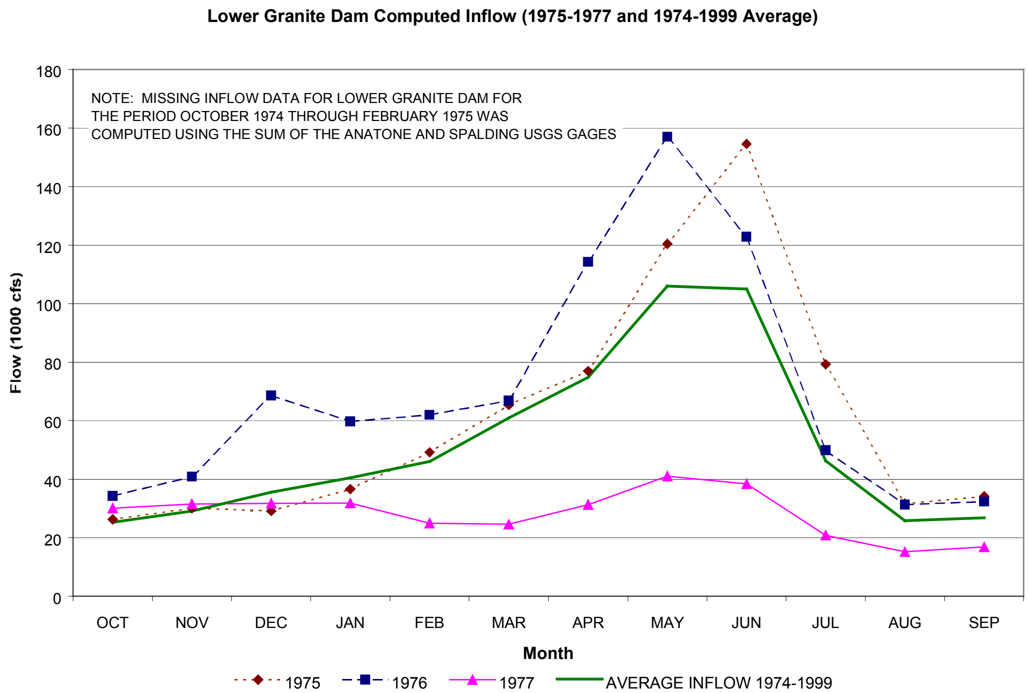


Figure 3-1. Monthly Mean Flow Data at Lower Granite Dam for the Years 1975 through 1977

Source: Developed by Normandeau

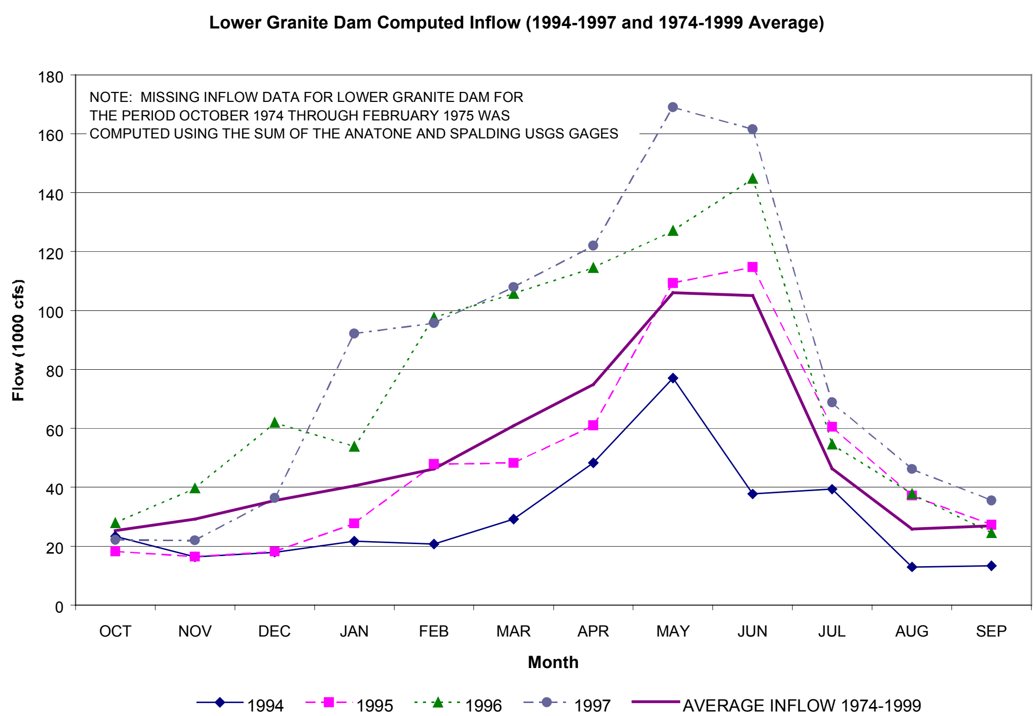


Figure 3-2. Monthly Mean Flow Data at Lower Granite Dam for the Years 1994 through 1997

Source: Developed by Normandeau

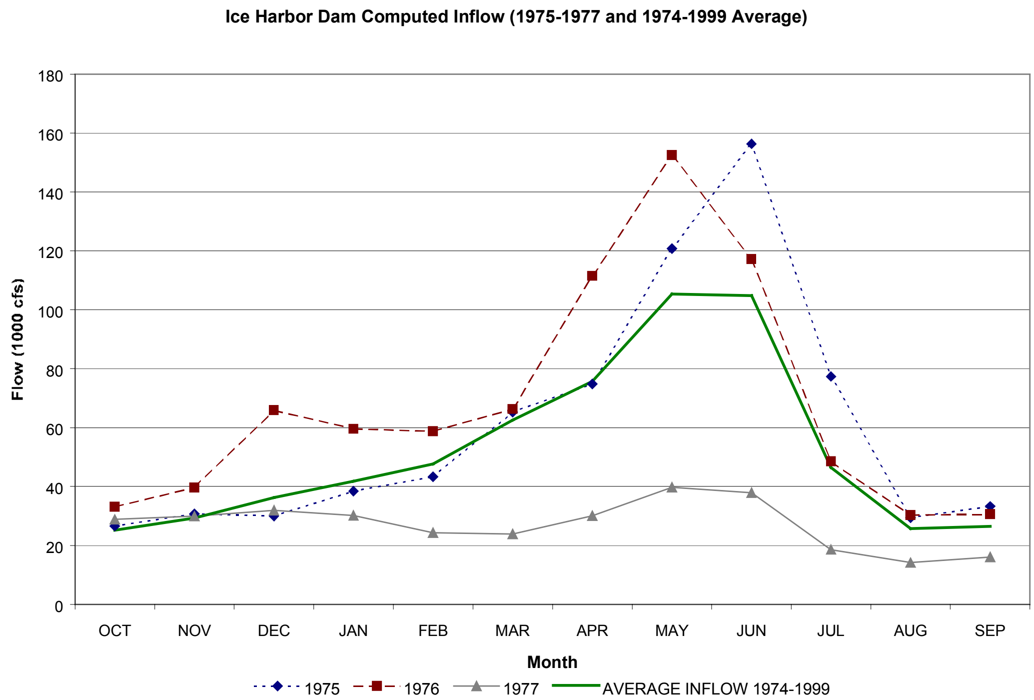


Figure 3-3. Monthly Mean Flow Data at Ice Harbor Dam for the Years 1975 through 1977

Source: Developed by Normandeau

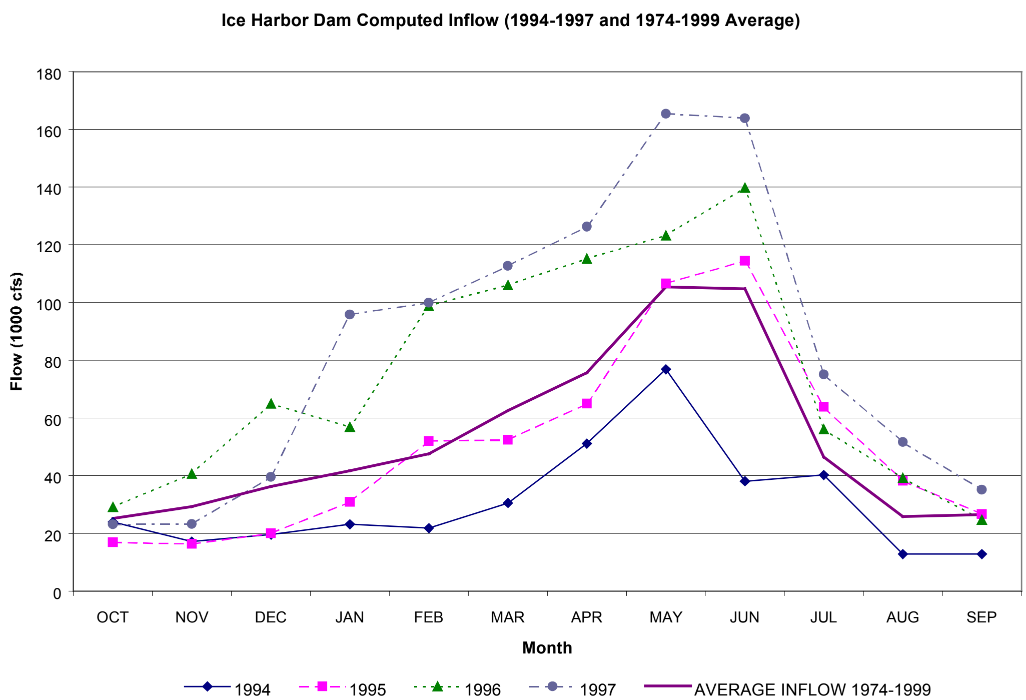


Figure 3-4. Monthly Mean Flow Data at Ice Harbor Dam for the Years 1994 through 1997

Source: Developed by Normandeau

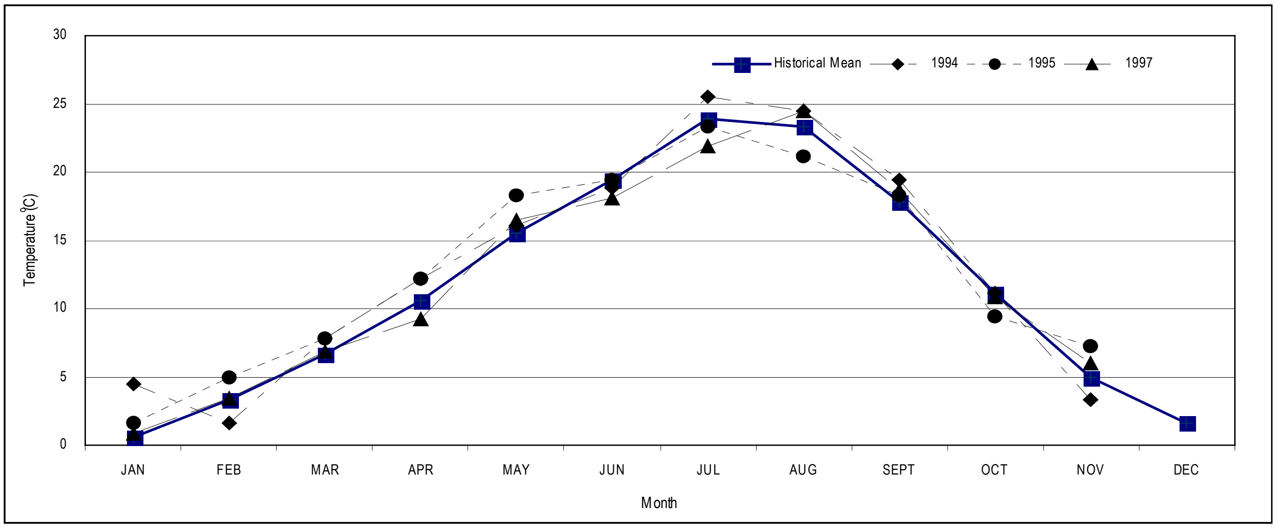


Figure 3-5. Comparison of Average Monthly Air Temperatures during 1994, 1995, and 1997 to the Historical Monthly Averages Recorded at the Nez Perce Weather Station in Lewiston, Idaho

Source: Developed by Normandeau

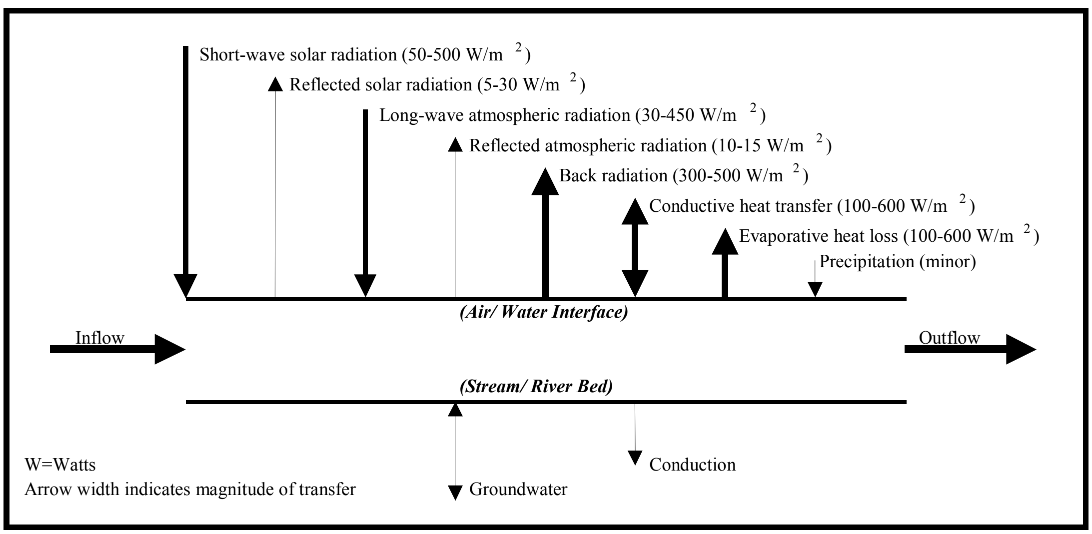


Figure 3-6. Major Sources of Heat Loss and Gain to a River System (adapted from Martin and McCutcheon, [1999])

Source: Developed by Normandeau

Short-wave radiation is that segment of the solar input that is not absorbed by the atmosphere (i.e., it is mainly visible light). The incident energy depends on the altitude of the sun, atmospheric conditions (e.g., cloudiness) and shading provided by the regional topography. A portion of this light, typically less than 10 percent, is reflected but varies with the solar angle (Rinaldi et al., 1979). The remainder is converted into heat within the river since it has the unique ability to penetrate below the surface. Short-wave radiation is, therefore, the major natural source of heat to most water bodies. The degree that it penetrates depends on a combination of color, concentrations of dissolved and inorganic suspended material, and biological components such as algae. Attenuation of the light in the water column follows an exponential decay so that most of the energy is absorbed within the first few meters, but short-wave radiation can penetrate to considerable depths if the water is clear (Wetzel, 1983).

A second source of heat gain and loss is long-wave radiation. The difference in solar radiation at the top of the atmosphere versus the solar radiation at the water surface, is the radiation absorbed by the clouds and atmosphere. This atmospheric heat is in turn reflected at longer wavelengths. Since the absorption coefficient of air for infrared radiation is low, its magnitude varies directly with temperature and atmospheric moisture content and to a lesser extent ozone and carbon dioxide (Martin and McCutcheon, 1999). Atmospheric long-wave radiation is often the greatest source of heat at the water surface on cloudy days. About 3 percent of the long-wave radiation is reflected, and the rest is available for heating. However, this input contributes only to surface warming, and the heat must be transferred into the main body of the river by wind-or water-induced turbulence. The water body also emits long-wave radiation, which represents a heat loss (Martin and McCutcheon, 1999). Since the absorption coefficient for this type of energy in water is very high, only a very thin surface layer emits back radiation (Rinaldi et al., 1979). As such, the surface water temperature is the variable that controls this process.

Energy can also be exchanged between the river and the atmosphere through heat conduction (exchange of sensible heat). Pure heat conduction (i.e., heat transport through molecular or atomic collisions) is, however, important only at the air-water interface. Within the two media, heat is transported mainly through convection, which occurs in the form of both large-scale movement (e.g., advection) and turbulent mixing (eddy diffusion). The convection of air is primarily forced by the wind. Therefore, the flux of sensible heat through the water surface depends not only on the air and water temperature, but also on wind velocity. Furthermore, wind-wave interactions dominate turbulent transport of heat away from the surface by changing the character of that interface. Waves increase the surface area available for exchange and thereby enhance the rate of transport. Even if no wind is blowing, turbulent mixing is still the main transport mechanism for sensible heat in air since the river flow generates air turbulence through frictional contact with the air.

Evaporation of water from the river also represents a transport of heat because the change in the state of aggregation requires energy (Martin and McCutcheon, 1999). The amount of evaporation is proportional to the water vapor pressure gradient between the water and the atmosphere (i.e., the gradient is between the saturated vapor pressure at the temperature of the water surface and the actual vapor pressure at the temperature of the air). The same processes that influence the transport of sensible heat govern the dispersion of the evaporated water. Under calm conditions, equilibrium between the vapor pressure of the water and that of the air immediately above the water surface is quickly attained. The rate of diffusion of vapor away from the surface layer then controls the evaporation rate. However, wind-induced turbulence in both the water and atmosphere enhances the transport. Therefore, the transfer rate is a function of the wind speed near the water surface.

The addition of heat from rain alone is usually negligible. Any contribution of heat to the river originates from the change in air temperature, wind, and disruption of the water surface that are associated with precipitation.

Heat exchange with the riverbed is another potential avenue that can influence the temperature of the water body (Martin and McCutcheon, 1999). The rate is determined by the heat conductivity of the soil. Since this rate is typically very small and the temperature gradients that occur are moderate, this mechanism of heat transfer is usually very minor. Furthermore, large rivers that involve significant volumes of water will not be as sensitive to the bottom flux as compared to shallow rivers or small streams.

The relative importance of each component of the thermal budget in a large river versus a small stream is, to a large extent, site specific. Examples of the particular differences between these systems, along with a few generalities, are provided below.

Small streams have a larger surface area for a given volume of water than a large river does, and this can have several important implications. First, everything else being equal, the processes that occur at the air/water interface will be highlighted. For example, direct solar radiation will tend to cause a greater temperature increase during the day. At night when the air temperature decreases, the loss of heat from the water surface will also be greater per unit volume of water and the amplitude of diel fluctuation increases. Second, small streams tend to be influenced to a greater extent by streambank vegetation. If the riparian zone has an overstory of trees, then they will generally provide more shading to small streams than they would to a large river simply due to the distance between banks. Trees significantly reduce the effects of direct solar radiation and help to cool the water, and that is one reason they are often planted as part of stream restoration projects. Shading also minimizes heat loss via evaporation; and since the temperature difference between the air and water is less, the conductive heat loss will also be reduced. It should also be mentioned that although riparian trees reduce evaporation, evapotranspiration is greater and can noticeably alter the daily flow regimen of a small stream. Third, the effect of groundwater additions or losses can be greater in a small stream. In fact, one of the reasons that there is any flow in many small streams during the summer is groundwater discharge. Groundwater tends to have a more uniform temperature throughout the year (i.e., cooler in the summer and warmer in the winter) than does surface water, and is one of the motivating factors why salmonids often place their redds in areas of discharge. Conversely, water from the stream can enter the hyporheic zone (the porous medium under and next to a stream that is saturated with water), and in some circumstances may "disappear" into the streambed during summer low-flow conditions simply because the substrate is very porous. This groundwater/surface water interaction can be rapid enough so that stream water may be completely exchanged with porewater within a reach of a few miles. Even in an intermediate-size river, such as the Spokane River, Washington, the hyporheic zone is extensive and a vital component of the local hydrology.

Several features set a large river apart from a small stream in terms of the influence of the various components of the heat budget. One of these is the larger mass of water. The exchange rates per unit surface area could be the same, but the larger volume of water in the river will moderate any changes. The lower Snake River is wider than upstream reaches, and this provides for a larger air/water interface and consequently heat exchange. Although the total amount of heat added via direct solar radiation or long-wave radiation is greater, the larger volume of mixed water that has a relatively large specific heat moderates local effects. Overall, however, the temperature of a river will tend to increase from the headwaters to the mouth during the summer. It should also be noted that cumulative effects usually increase the concentrations of soluble ions in a downstream progression as well. The thermal benefits of riparian vegetation as discussed above are also reduced in a large river. When present, the moderating effects are noticeable near shore and can add complexity, and therefore habitat alternatives, to the aquatic system. However, riparian vegetation is so minimal along the lower Snake River that any effects on temperature are insignificant. Orographic effects, on the other hand, can provide shading from direct solar radiation, but not long-wave radiation, in some reaches of the lower Snake and middle Snake River (e.g., Hells Canyon). Wind is another driving variable that is more important in a large river. This occurs not only because the trees along a small stream can act as a windbreak, but also because the fetch (length of water that the wind can travel across unimpeded) is greater for a large river. As such, the energy associated with the wind not only leads to mixing of the water, but also displaces the air/water interface farther away from its equilibrium point. Finally, interactions between groundwater and surface water do occur in most large rivers, but the percentage of the total stream volume involved is less than in a smaller stream, and this also minimizes the thermal impacts.

In summary, the most relevant quantities that determine the temperature of a river are the convection characteristics of the water body and the meteorological parameters that affect the energy transfer through the water surface. The importance of the meteorological parameters has led to the concept of the equilibrium temperature, which is defined as the temperature that a completely mixed water column has if the net heat flux through its surface is zero. The equilibrium temperature is determined uniquely by the meteorological conditions.

Water temperature is one of the more critical parameters affecting fish migration behavior during the April through September adult and juvenile salmonid migration periods. The optimal temperature range during the summer juvenile and adult migration period is generally recognized to be between 10 to 20°C (50 to 68°F) (BPA, 1995). Historic water summer temperatures in the Snake River basin typically exceed these optimal ranges.

Existing Water Temperature Conditions—Empirical Data Evaluation

Available empirical water temperature data were evaluated to determine or quantify water temperature conditions that anadromous fish encounter while passing through the lower Snake River reservoir system using the current operating conditions for the reservoirs. Those conditions include the current flow augmentation, water temperature control, and voluntary spill criteria for each of the reservoirs. The study reach analyzed is from the mouth of the Snake River and continues upstream to the Anatone gaging station above Lewiston on the Snake River and to the Spalding gaging station above Lewiston on the Clearwater River. Summer temperatures were of most concern because of the potential impact on fish. Where possible, data were summarized and displayed in two different ways to evaluate whether the summary methods would influence the conclusion results. The first method used was to average daily maximum water temperature data by days for the period of record and then graph the results. The second method used was to summarize the number days each year that the daily maximum water temperatures exceeded a base temperature of 20°C (68°F). A base temperature of 20°C (68°F) was used because that is the base used in the Washington State Water Quality Standards, as indicated in the following paragraph.

Washington's water quality standards specify that water temperatures in the lower Snake River shall not exceed 20°C (68°F) as a result of human activity. In addition, temperature increases due to human activity in the lower Snake River (i.e., below the Clearwater River) shall not exceed $t = 34/(T+9)$ °C where t = change in temperature and T = background temperature. For example, if the background temperature is 20°C (68°F) then the maximum allowable temperature increase due to human activity would be 1.2°C (2.2°F). Above the Clearwater River (RM 139.3), increases over 0.3°C (.5°F) caused by human activity from a single source are not allowed, and increases over 1.1°C (2°F) from all activities are not allowed when the background stream temperature is over 20°C (68°F).

The evaluation was divided into the following five parts and then a conclusion summary was prepared.

- Evaluate water temperatures above the lower Snake River reservoir system
- Evaluate water temperatures below the lower Snake River reservoir system
- Compare water temperatures above and below the reservoir system
- Compare water temperature station data above and below the reservoir system against station data within the reservoir system
- Evaluate temperature effects of the Dworshak reservoir releases.

Data used for this evaluation came from several sources. Figure 3-7 displays the water temperature data sites and period of records available just above, below, and within this study reach.

Data collected by the USGS are from its annual "Water Resources Data" publications for Washington and Idaho. The EPA has consolidated much of the historic data prior to 1995 and posted it on its Web site: (http://www.epa.gov/r10earth/offices/oea/temperature/Tdata.html).

The EPA information included data on the Snake River from Hells Canyon to the Burbank site near the Columbia-Snake River confluence, Clearwater River, and local tributaries.

Data from the Corps total dissolved gas monitoring stations (TDGMS) can be found at the following Corps Web site: (http://www.nwd-wc.usace.army.mil/TMT/tdg_data/months.html).

Data have also been collected by universities and other agencies and can be found in reports that they have produced.

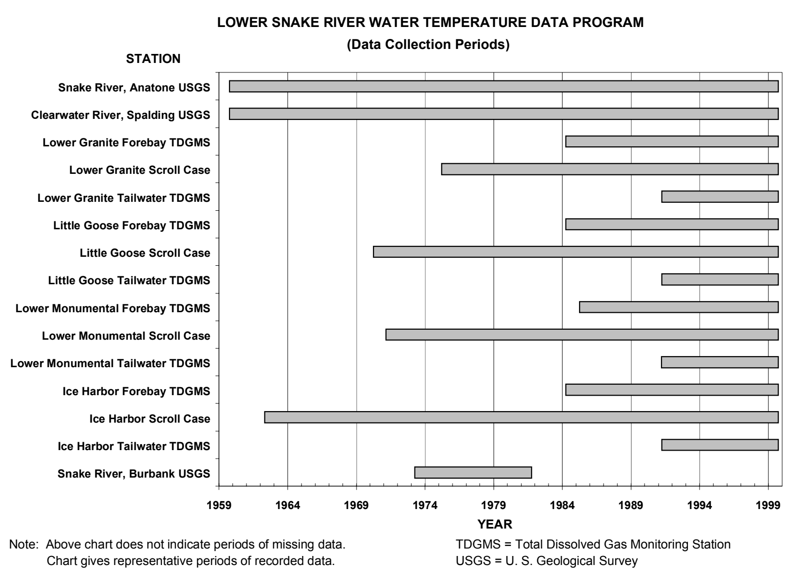


Figure 3-7. Lower Snake River Water Temperature Data Collection Periods
Source: Developed by the Corps

Water Temperatures Above the Lower Snake Reservoir System

Water temperature data in the lower Snake River upstream of the study reach have been collected through routine monitoring programs since 1959. Construction of the lower Snake River reservoir projects began in 1956 at the Ice Harbor Dam site. To support construction and future project data requirements, the USGS began collecting water temperature data at the following gaging stations:

- The river gaging station near Anatone, Washington (Snake River RM 167.2, approximately 13 kilometers [8 miles] east of Anatone)
- The river gaging station near Clarkston, Washington (Snake River RM 134)
- The river gaging station near Spalding, Idaho (Clearwater River RM 11.6, approximately 18 kilometers [11 miles] above Lewiston, Idaho).

The flow data, water temperature data, and sediment and other data collected at these stations would be used to represent input parameters to the lower Snake River reservoir system as it would be developed and operated. The data collected by the USGS at these three stations are published in its annual Water Resources Data Report for Washington and Idaho. The quality of these data is controlled by the USGS using its standards, and the record is considered to be very good and representative of the river at these locations. The Snake River station near Clarkston was discontinued in 1964; therefore, only the Anatone and Spalding station data were used in this evaluation. The annual USGS publications list maximum and minimum temperatures for each day of the period. During this period of record, there are a number of changing reservoir conditions upstream from both the Anatone and Spalding gages. The Idaho Power Company Hells Canyon reservoir complex (Brownlee, Oxbow, and Hells Canyon) construction was completed on the Snake River in 1967. Dworshak Dam construction was completed in 1973 on the North Fork of the Clearwater River (approximately 56 kilometers [35 miles] above the Spalding station). By 1974, upstream reservoir development above Lewiston, Idaho, was completed on both the Snake and Clearwater Rivers and the upstream reservoirs were being operated under their normal operating criteria (as defined before special reservoir operations began for threatened and endangered fish species).

Where possible for this evaluation, data graphs were plotted beginning in 1974 to best represent the current level of upstream reservoir development. Listings of data by year included data before 1974 if they were available for a more complete record listing. Data were analyzed by comparing each year in the record and also by preparing period averages for each record. Figure 3-8 displays an average of the daily maximum water temperature data values collected at the Anatone and Spalding stations for the period 1974 through 1999.

As shown by the graph, the Anatone station summer water temperatures are typically higher than the Spalding station temperatures by approximately 2 to 5°C (3.6 to 9°F). An average of the maximum daily water temperatures at the Spalding station do not normally exceed 20°C (68°F) while an average of the maximum daily temperatures at Anatone normally reaches 23°C (73°F) each year and exceeds 20°C (68°F) for a period of approximately 60 days each summer.

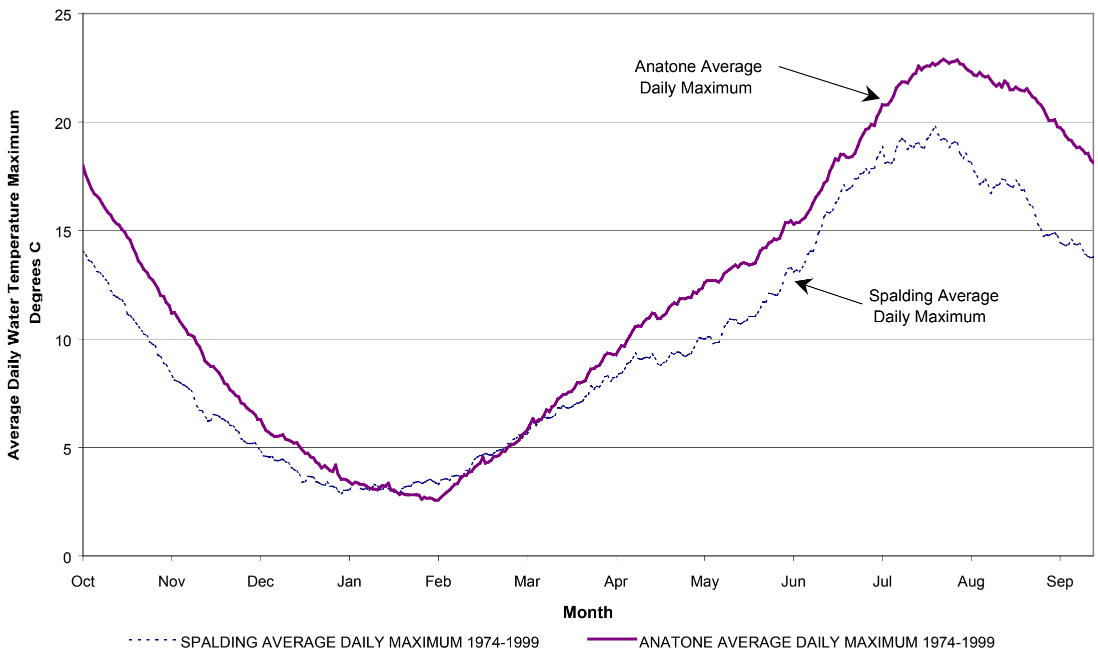


Figure 3-8. Average Daily Maximum Water Temperature for Spalding and Anatone, 1974 through 1999

Source: Developed by the Corps

Table 3-2 compares the number of days each year (1972 through 1999) that daily maximum water temperatures at the Anatone and Spalding stations have exceeded 20°C.

From the table, Anatone water temperatures exceeded 20°C (68°F) during the summer each year the data were available for the 1972 through 1999 period, and the exceedance each year averaged approximately 60 days, which agrees with the graph summary of Figure 3-8. The Spalding water temperatures exceeded 20°C (68°F) most of the summers by an average of only a 15-day duration. Looking at Figure 3-8 and Table 3-2 yields a little different conclusion of what is expected at Spalding based on the procedure used for summarizing a data record. The graph indicates that water temperatures above 20°C (68°F) are not normal, while the table indicates that there are normally days each year above 20°C (68°F). This comparison demonstrates that the table summary is very sensitive to the base temperature being used. The Spalding record summary (1974 through 1999) includes the Dworshak temperature control operation (1995 through 1999) and is lowering the daily averages of the record. This demonstrates that conclusions can readily be different based on methods and period of records being used to evaluate data.

Figure 3-9 displays an average difference in the daily maximum and daily minimum water temperatures at the Anatone and Spalding stations for the 1995 through 1999 period to reflect relative differences between night and day temperature fluctuations.

Table 3-2. Number of Days for Anatone and Spalding in which Daily Maximum Water Temperatures Exceeded 20°C, 1972 through 1999

Year	Snake River near Anatone, WA		Clearwater River at Spalding, ID	
	Days over 20 °C	Days Missing	Days over 20 °C	Days Missing
1972	49	16	6	0
1973	57	0	25	0
1974	34	0	0	0
1975	+	97	10	27
1976	43	0	8	0
1977	60	0	16	0
1978	35	0	22	0
1979	79	0	1	0
1980	51	0	9	0
1981	72	0	6	0
1982	47	0	2	0
1983	47	0	0	16
1984	---	137	+	44
1985	---	137	13	0
1986	80	19	17	0
1987	83	0	39	0
1988	76	0	26	0
1989	71	7	16	0
1990	88	0	52	0
1991	68	0	36	0
1992	64	2	41	3
1993	39	0	+	109
1994	88	0	22	0
1995	66	0	5	0
1996	60	0	0	0
1997	63	0	13	0
1998	82	0	8	0
1999	61	0	0	0

Notes:
1. Days Missing based on the period of 1 June to 15 October (137 total days)
2. Missing days may skew the maximum temperature and number of days over 20°C
3. --- = Data missing for indicated periods
4. + = Insufficient Record

Source: Developed by the Corps

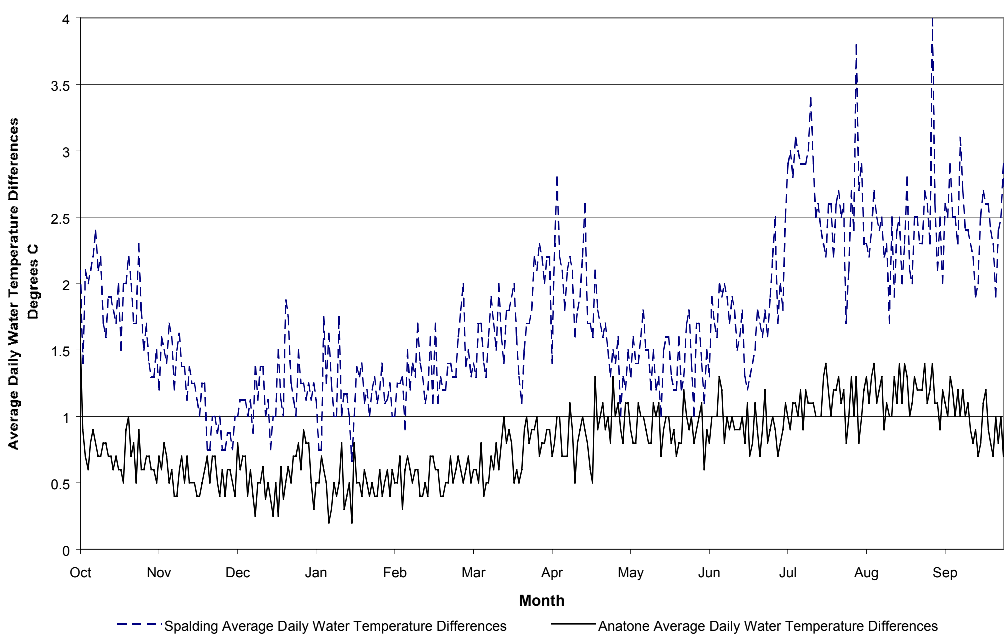


Figure 3-9. Average Daily Water Temperature Difference Values for Spalding and Anatone Sites, 1995 through 1999

Source: Developed by the Corps

The difference in the average maximum and minimum daily fluctuations appears to be higher for Spalding (2 to 3.5°C [3.6 to 6.3°F] in the summer) as compared to for the Anatone station (0.5 to 1.5°C [.9 to 2.7°F]).

In 1992, flow augmentation regulation from the Dworshak reservoir was implemented in an attempt to improve lower Snake River conditions for fish. Each year thereafter, the Dworshak augmentation flow procedure was adjusted and was then standardized in 1995 for downstream flow and temperature control. Since 1995, Dworshak normally releases 1,500 million cubic meters (1.2 million acre-feet) of cold water from July through the end of August for downstream flow augmentation and temperature control. The impact of this regulation change is displayed in Figure 3-10 for the Spalding station. A comprehensive discussion of the effects of Dworshak is included in the following section.

From the graph, it appears that the July through August Clearwater River temperatures at Spalding are normally reduced by approximately 2 to 5°C (3.6 to 9°F) because of this operation. From the data shown in Table 3-2, the temperature exceedances would appear to be reduced from a 15-day average duration (1974 through 1999) to approximately a 5-day duration (1995 through 1999) because of the temperature control operation.

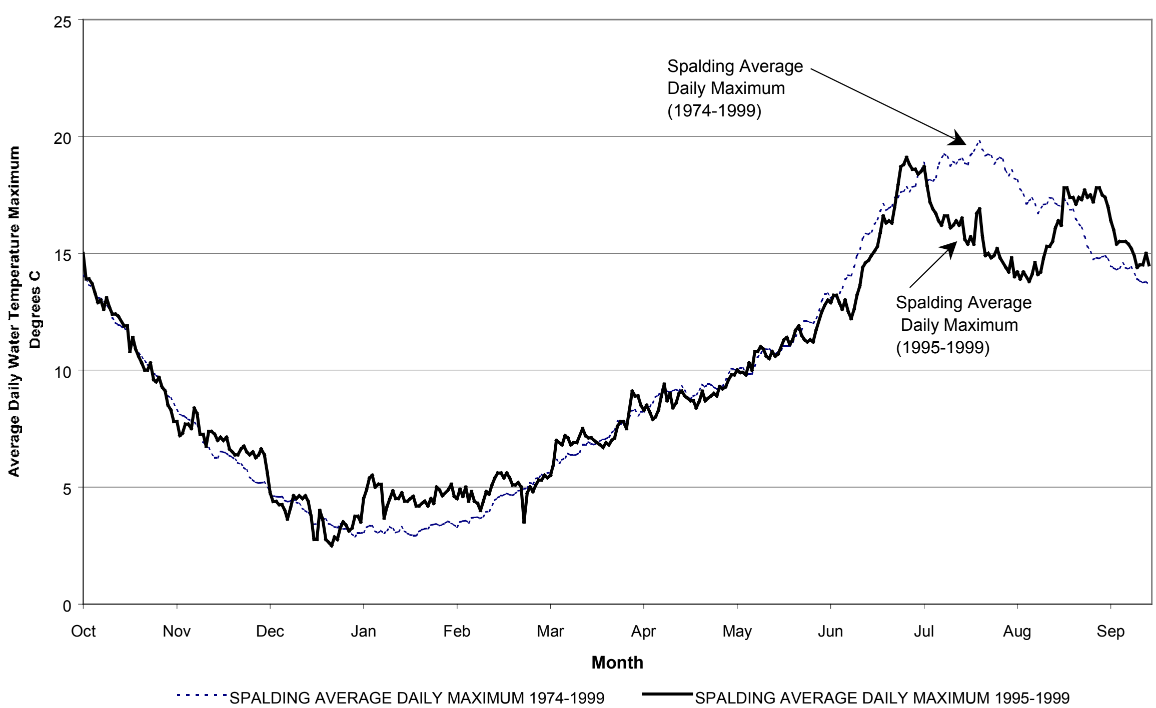


Figure 3-10. Average Daily Maximum Water Temperature (°C)
Source: Developed by the Corps

When evaluating the empirical temperature from Anatone and Spalding, there are some general conclusions that might be made to characterize temperature conditions above the lower Snake River reservoir system:

- Summer (July through September) maximum daily temperatures of the Snake River at the Anatone site are generally greater than those of the Clearwater River at Spalding site by approximately 1 to 4°C (1.8 to 7.2°F). The normal summer daily temperature variation in maximum and minimum daily values is approximately 0.5 to 1.5°C (0.9 to 2.7°F) at Anatone and 2 to 3.5°C (3.6 to 6.3°F) at Spalding.
- The Anatone maximum daily summer water temperatures normally exceed 20°C (68°F) by approximately 60 days per year and are expected to reach approximately 23°C (73°F) each year on the average.
- Maximum daily water temperatures on the Clearwater River at Spalding normally exceed 20°C (68°F) by approximately 15 days per year without the current Dworshak augmentation and temperature release regulation. Since implementation of the Dworshak summer flow augmentation and temperature release regulation in 1995, temperatures exceeding 20°C (68°F) at the Spalding site have averaged approximately 5-days duration per year. Further, the overall mid-July through September water temperatures appear to be reduced by approximately 2 to 5°C (3.6 to 9°F).

Water Temperatures Below the Lower Snake Reservoir System

Available water temperature station data that might somewhat represent temperature conditions below Ice Harbor Dam are:

- Snake River at Burbank (1973 through 1981). This station was located on the Snake River at RM 2.2 near Burbank, Washington, and approximately 4 kilometers (7 miles) below Ice Harbor Dam. This station depth is about 5 meters (15 feet). Data were collected by the USGS.
- The TDGMS station (1991 to present). This station is located on the north bank of the Snake River approximately 4.7 kilometers (2.9 miles) downstream from Ice Harbor Dam. The depth of the probe is approximately 5 meters (15 feet). Data were collected by the Corps.
- Ice Harbor Dam scroll case temperature gage (1961 to present). This gage is located on the Unit No. 1 cooling system water supply pipe, located at the south end of the dam. The inlet of the pipe is located at a depth of approximately 30 meters (100 feet). Data were collected by the Corps.

After reviewing the data and the record lengths for these stations, the Burbank site data and Ice Harbor Lock and Dam scroll case data from 1973 through 1981 (1974 through 1981 for the graphs) were used to evaluate water temperature conditions below the lower Snake River reservoir system before the use of the current flow augmentation and temperature control regulation at Dworshak. Available Ice Harbor scroll case data and Ice Harbor tailwater TDGMS data (1995 through 1999) were used to evaluate conditions after the Dworshak regulation for augmentation and temperature control. Figure 3-11 displays an average of the maximum daily values for the Burbank station and the average daily values for the Ice Harbor scroll case gage. Figure 3-12 displays an average of the maximum daily values for the Ice Harbor TDGMS tailwater station and the scroll case gage.

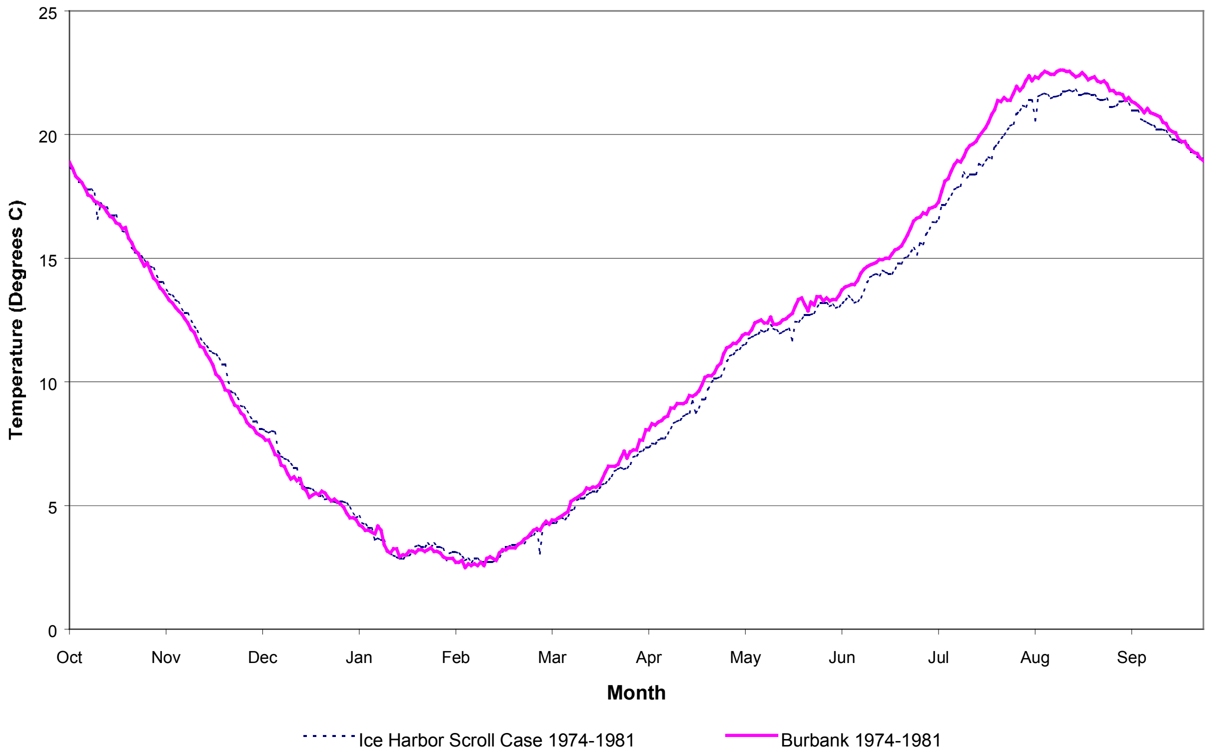


Figure 3-11. Average Maximum Temperatures in Degrees Celsius, 1974 through 1981
Source: Developed by the Corps

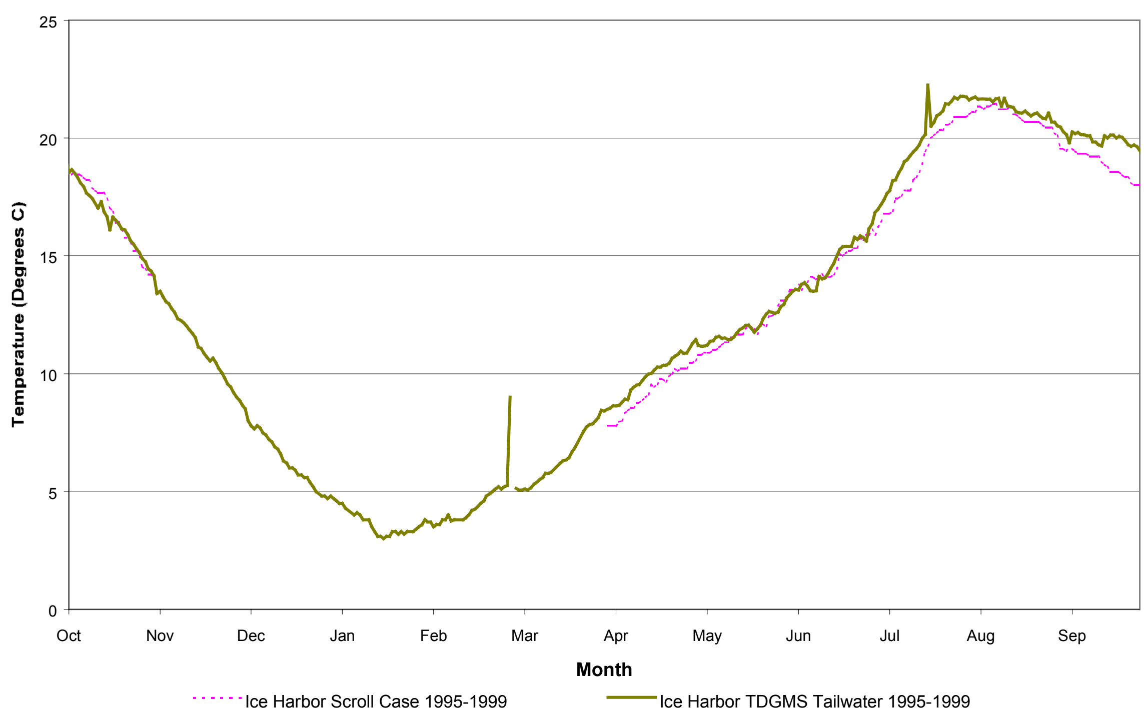


Figure 3-12. Average Maximum Temperatures in Degrees Celsius, 1995 through 1999
Source: Developed by the Corps

Both Figures 3-11 and 3-12 show that the Ice Harbor scroll case water temperatures are lower than either the Burbank station data (by 1 to 3°C [1.8 to 5.4°F]) or the Ice Harbor TDGMS tailwater data (0 to 2°C [0 to 3.6°F]) during the summer period. When the Ice Harbor TDGMS site current data were compared with the discontinued Burbank station (two different periods of record), it appeared that the Burbank water temperatures might have been slightly warmer.

Figure 3-13 shows average daily differences between the maximum and minimum values for the Ice Harbor TDGMS tailwater station and the Burbank station.

Table 3-3 lists the number of days when water temperatures exceeded 20°C (68°F) for these stations and periods.

Table 3-3 shows (as did Figures 3-11 and 3-12) that the Ice Harbor scroll case water temperatures are lower than either the Burbank station or the Ice Harbor TDGMS tailwater data during the summer period. The average number of days each year that the scroll case temperatures exceeded 20°C (68°F) is 45 versus 60 for the Burbank station and 35 versus 56 for the Ice Harbor TDGMS tailwater site. The difference in the scroll case average yearly summer exceedances for the two periods summarized was a 10-day decrease (from 45 days for the 1973 through 1981 period down to 35 days for the 1995 through 1999 period). Perhaps, it is just due to a different period of record with different air temperatures and other conditions or, perhaps, it is a changed density gradient due to the Dworshak augmentation and temperature control operation and/or the voluntary spill operations at the projects to pass fish.

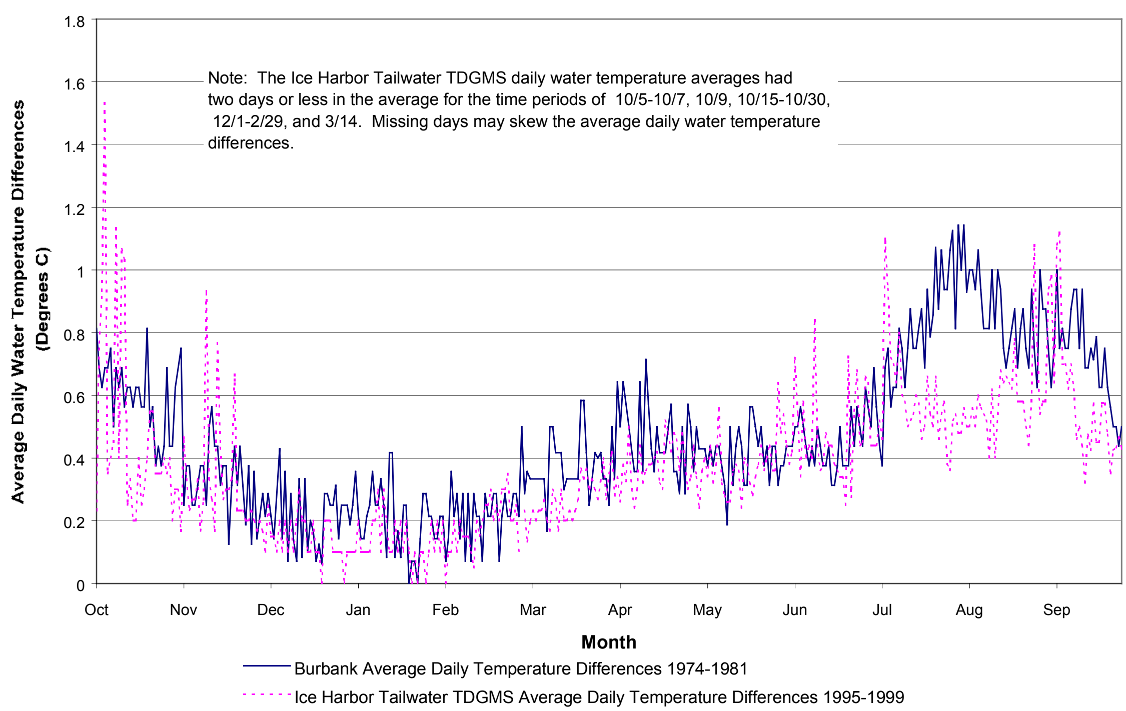


Figure 3-13. Average Daily Water Temperature Differences at Burbank and Ice Harbor Dam
Source: Developed by the Corps

Table 3-3. Days in which Water Temperatures Exceeded 20°C (68°F) at Burbank and Ice Harbor Dam, 1973 through 1981, 1995 through 1999

Year	Ice Harbor Scroll Case		Burbank USGS	
	Days over 20°C	Days Missing	Days over 20°C	Days Missing
1973	42	0	69	0
1974	46	0	59	5
1975	29	0	54	0
1976	44	0	51	0
1977	43	0	64	4
1978	28	0	46	0
1979	74	0	79	0
1980	48	0	53	0
1981	55	0	65	0

Notes: Burbank - USGS Gaging Station 13353200 Snake River at Burbank, WA.
Days Missing - Days missing that could be in excess of 20 Degrees C for the record ending September 30

Year	Ice Harbor			
	Days over 20°C Scroll Case	Days Missing	Days over 20°C TDGMS Tailwater	Days Missing
1995	18	0	48	0
1996	41	0	48	0
1997	44	0	57	0
1998	52	0	84	0
1999	22	0	42	0

Notes: TDGMS - Total Dissolved Gas Monitoring Station
Days Missing - Days missing that could be in excess of 20 Degrees C for the record ending September 30

Source: Developed by the Corps

When evaluating the Burbank, Ice Harbor TDGMS, and scroll case data, there are several conclusions that might be made:

- The Ice Harbor scroll case data appear to be colder than either the Burbank data or the Ice Harbor TDGMS data as indicated by summarizing the data through an average graphical plot and a temperature exceedance summary. This leads to questions of how well the data may represent well-mixed river temperature conditions. Perhaps, it does not represent an average river temperature, but rather represents only one temperature point location at a depth of approximately 100 feet within the Ice Harbor reservoir pool.

- An average of the daily maximum summer temperatures of the scroll case gage appears to be 1 to 3°C (1.8 to 5.4°F) cooler than the Burbank station and 0 to 2°C (0 to 3.6°F) cooler than the TDGMS station. The TDGMS station depth is about 5 meters (15 feet), and the scroll case pipe intake is at a 30-meter (100-foot) depth.

- The average number of days each year that the scroll case temperatures exceeded 20°C (68°F) is 45 versus 60 for the Burbank station and 35 versus 56 for the Ice Harbor TDGMS tailwater site. It does not appear that there is a big difference in the exceedance days between the Burbank station and the

TDGMS tailwater station even though they are different periods of record. It is unclear why there is a 10-day decrease in the scroll case days of exceedance; 45 days for the 1973 through 1981 period down to 35 days for the 1995 through 1999 period.

- A comparison of the average daily fluctuation between maximum and minimum water temperatures shows that there is an approximate variation in the daily maximum and minimum water temperatures at the Burbank station of 0.4 to 1.2°C (0.7 to 2.2°F) each day during the summer and approximately a 0.3 to 0.8°C (0.5 to 1.4°F) variation in the Ice Harbor tailwater TDGMS station.

Water Temperature Comparison Above and Below the Lower Snake River Reservoir System

Water temperature conditions above the lower Snake River reservoir system were compared to conditions below the reservoir system using available data stations and summaries already discussed. Figure 3-14 displays an upstream versus downstream temperature summary comparison for 1974 through 1981, and Figure 3-15 displays a 1995 through 1999 comparison. Table 3-4 displays the number of days each year that exceeds 20°C (68°F) at each site.

Data in Figures 3-14 and 3-15 and Table 3-4 do not indicate that summer temperatures (peak and duration) below the reservoir system are substantially different from the summer temperatures (peak and duration) of the Snake River above the reservoir system at the Anatone data site. The Anatone temperatures (1974 through 1981) may be slightly cooler than Burbank temperatures, and the Anatone temperatures (1995 through 1999) may be slightly warmer than the Ice Harbor TDGMS tailwater data. Ice Harbor scroll case temperatures were consistently cooler than any of the other data. This is not unexpected since the scroll case intake pipe is at a depth of 100 feet, and the other gages are at depths of approximately 5 meters (15 feet).

Figures 3-14 and 3-15 do appear to show that the timing of the warming and cooling between the upper reach and lower reach may be different. It appears that the upper reach (Anatone station) temperatures may warm faster in the spring and summer and cool faster in the fall and winter than the lower reach (Burbank and Ice Harbor TDGMS tailwater) temperatures.

Table 3-4. Days in which Water Temperatures Exceeded 20°C (68°F) at Burbank, Ice Harbor Dam, and Anatone, 1973 through 1981 and 1995 through 1999

	Ice Harbor				Burbank		Anatone	
	Scroll Case		TDGMS Tailwater		USGS			
Year	Days Over 20 °C	Days Missing	Days Over 20 °C	Days Missing	Days Over 20 °C	Days Missing	Days Over 20 °C	Days Missing
1973	42	0	No Data	No Data	69	0	57	0
1974	46	0	No Data	No Data	59	5	34	0
1975	29	0	No Data	No Data	54	0	+	97
1976	44	0	No Data	No Data	51	0	43	0
1977	43	0	No Data	No Data	64	4	60	0
1978	28	0	No Data	No Data	46	0	35	0
1979	74	0	No Data	No Data	79	0	79	0
1980	48	0	No Data	No Data	53	0	51	0
1981	55	0	No Data	No Data	65	0	72	0
1995	18	0	48	0	No Data	No Data	66	0
1996	41	0	48	0	No Data	No Data	60	0
1997	44	0	57	0	No Data	No Data	63	0
1998	52	0	84	0	No Data	No Data	82	0
1999	22	0	42	0	No Data	No Data	61	0

Notes: TDGMS - Total Dissolved Monitoring Station
Days Missing - Days missing that could be in excess of 20°C for the records ending September 30
Burbank - USGS Gaging Station 13353200 Snake River at Burbank, WA.
+ Insufficient Record
Source: Developed by the Corps

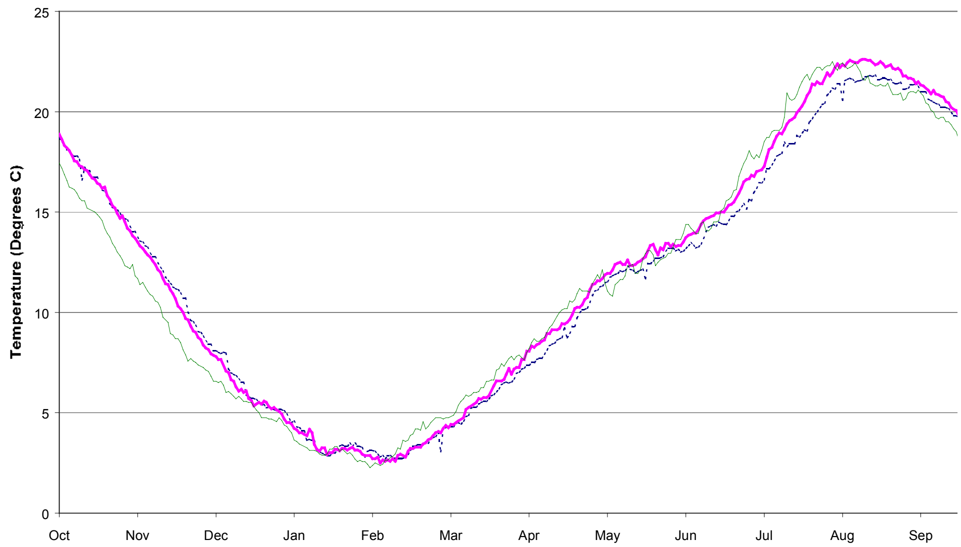


Figure 3-14. Average Maximum Temperatures in Degrees Celsius, 1974 through 1981
Source: Developed by the Corps

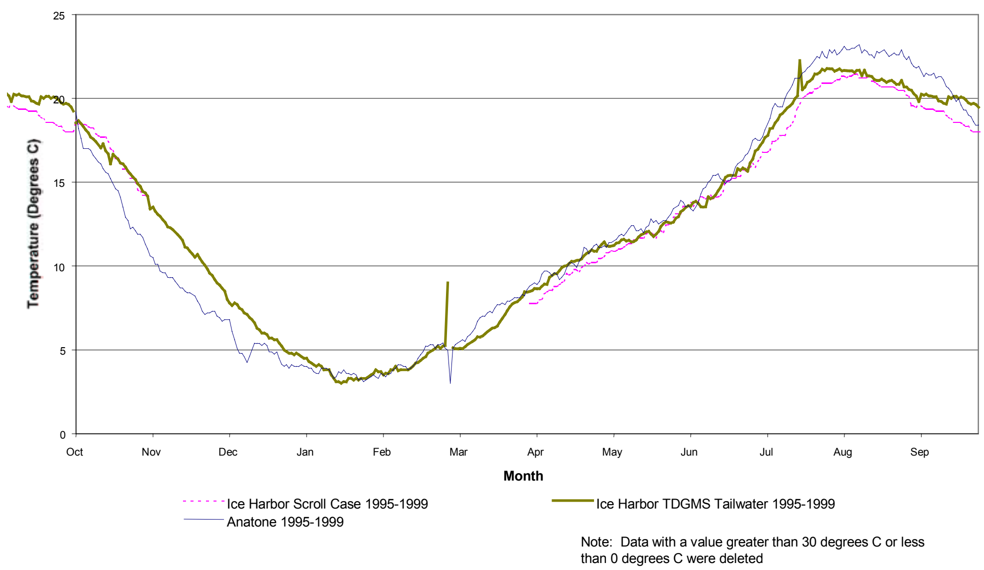


Figure 3-15. Average Maximum Temperatures in Degrees Celsius, 1995 through 1999
Source: Developed by the Corps

Figure 3-16 displays an average fluctuation difference between daily maximum and minimum water temperature data. This graph also shows little difference in the upstream and downstream reaches.

Data collected at these sites are for those particular points within the river and may or may not accurately represent a well-mixed water temperature and average cross-section condition for these two reaches of the Snake River. If enough quality-controlled data could be collected over time throughout the reaches, it may then be possible to determine how representative and accurate the existing data are.

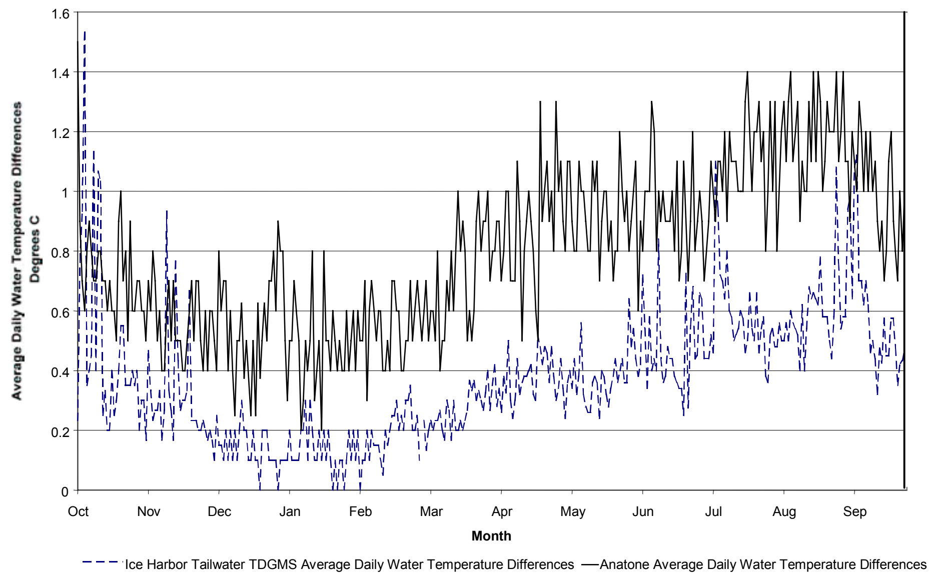


Figure 3-16. Average Daily Water Temperature Difference Values for Ice Harbor Tailwater TDGMS and Anatone Sites, 1995 through 1999

Source: Developed by the Corps

When comparing the reach above the lower Snake River reservoir system with the reach below the reservoir system, there are several conclusions that might be made:

- The Anatone, Burbank, and Ice Harbor TDGMS data do not indicate that summer temperatures (peak and duration) below the reservoir system are substantially different than the summer temperatures (peak and duration) of the Snake River above the reservoir system at the Anatone station.
- The Anatone, Burbank, and Ice Harbor TDGMS data do appear to show that the timing of the warming and cooling between the upper reach and lower reach may be different. It appears that the upper reach (Anatone station) temperatures may warm earlier in the spring and summer and cool earlier in the fall and winter than the lower reach (Burbank and Ice Harbor TDGMS tailwater) temperatures.
- The data show an average summer temperature fluctuation difference between an average of the daily maximum and minimum water temperature data of 0.8 to 1.4°C (1.4 to 2.5°F) for the upstream and 0.4 to 1.0 °C (0.7 to 1.8°F) for downstream reaches.

Water Temperatures Within the Lower Snake Reservoir System

Water temperature data collected within the lower Snake River reservoir system primarily consists of TDGMS temperature data for each of the four reservoir projects at both a forebay and tailwater site and at a temperature gage on the scroll case cooling water pipe for each project. The TDGMS tailwater sites for each project are typically 1.1 to 5 kilometers (0.7 to 3 miles) downstream from the dam and the probe is at approximately a 5-meter (15-foot) depth. The TDGMS forebay sites are typically on the upstream face of the dam, and the probe is at approximately a 5-meter (15-foot) depth. The scroll case gages at each dam are typically on the cooling water intake pipe for unit 1 with the inlet at a depth of approximately 30 meters (100 feet). These data have been collected by the Corps. The scroll case data have not used any specific quality-control structure as compared to the TDGMS data. Scroll case data that have been collected are published in the *Corps Annual Fish Passage Report* for that particular year. The Corps and others have collected temperature profile data for several locations within the reservoir system. Table 3-5 summarizes, for 1995 through 1999, the number of days that summer temperatures exceed 20°C (68°F) for the TDGMS forebay and tailwater stations and scroll case gages at each project. The data from this period should best represent the current operational conditions being used for the reservoir regulation (this would include upstream flow augmentation, upstream temperature control, and voluntary spill at the projects being used for fish passage). This period is also when the TDGMS data had the highest degree of quality control and consistency of station operation.

Table 3-5. Summer Days in which Water Temperatures at Corps Dams Exceeded 20°C (68°F), 1995 through 1999

Year	Ice Harbor Days over 20 °C					Lower Monumental Days over 20 °C					Little Goose Days over 20 °C					Lower Granite Days over 20 °C				
		TDGMS					TDGMS					TDGMS					TDGMS			
	Scroll Case	Tailwater	Days Missing	Forebay	Days Missing	Scroll Case	Tailwater	Days Missing	Forebay	Days Missing	Scroll Case	Tailwater	Days Missing	Forebay	Days Missing	Scroll Case	Tailwater	Days Missing	Forebay	Days Missing
1995	18	48	0	82	4>	23	53	1	77	4	26	54	0	49	27	0	38	3	61	0
1996	41	48	0	33	6	41	38	0	49	1	53	47	13>	54	0	23	35	0	39	0
1997	44	57	0	67	0	28	51	0	66	0	57	21	20>	49	17>	26	11	24	56	0
1998	52	84	0	85	0	75	84	0	88	0	82	65	13>	86	2	36	64	3	89	0
1999	22	42	0	47	0	29	29	0	45	0	45	13	0	42	0	0	2	0	55	0

Notes: TDGMS - Total Dissolved Gas Monitoring Station
Days Missing - Days missing that could be in excess of 68 °F (20 °C) for the record ending September 30
\> = Indicates days missing that could exceed 68 °F (20 °C) end of data to September 30

Source: Developed by the Corps

Based on the data in the table, it would appear that the summer forebay temperatures are almost consistently warmer than either the tailwater or scroll case data. The summer tailwater temperatures are almost consistently warmer than the scroll case temperature data. Perhaps these different locations may represent an existing density gradient near the damsite:

- Deep-level temperatures (scroll case sites at an approximate 30-meter [100-foot] depth)
- Near-surface temperatures (forebay sites at a depth of approximately 5 meters [15 feet])
- Mid-range temperatures (tailwater sites at a 5-meter [15-foot] depth with water coming from the power units and over the spillbays as part of the voluntary spill program).

It appears that the Lower Monumental reservoir temperatures may be lower than the Little Goose reservoir temperatures in the summer. The Palouse and Tucannon Rivers flow into the Lower Monumental reservoir pool and may be affecting temperatures within the reservoir.

Figures 3-17 and 3-18 show water temperature profiles that were collected in July 1997 and August 1994 at various sites within the lower Snake River reservoir system. These data demonstrate that the range in water temperatures can vary substantially within the reservoir pools depending on location, time of year, flow levels, and particular reservoir operations.

The longest water temperature record available at the dams is for the scroll case data. Those data sets have been summarized and are displayed in Table 3-6 and Figures 3-19 through 3-22.

Based on the scroll case data over the period of record, the special flow augmentation and temperature control operations at Dworshak may be lowering summer maximum season temperatures and the number of days each summer that water temperatures exceed 20°C (68°F) at the scroll case gages. This seems to

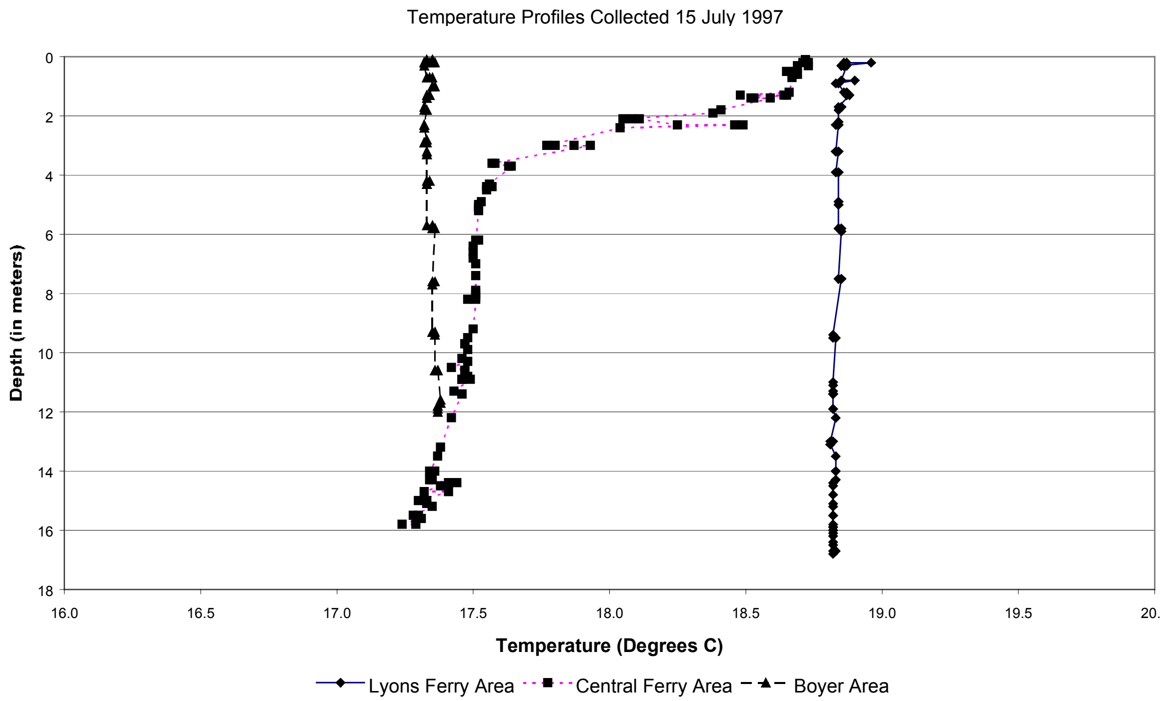


Figure 3-17. Temperature Profiles, Sites in the Lower Monumental and Little Goose Reservoirs
Source: Developed by the Corps

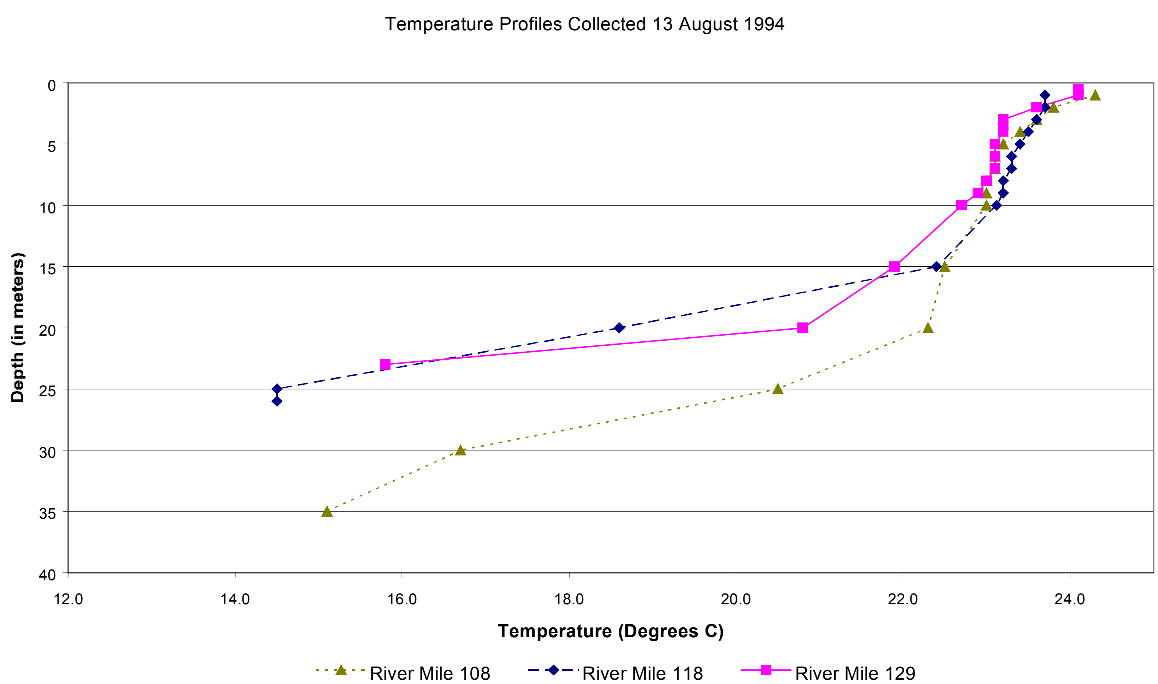


Figure 3-18. Temperature Profiles, Three Sites in the Lower Granite Reservoir
Source: Developed by the Corps

Table 3-6. Maximum Water Temperatures at Corps Dams (Data Measured at Scroll Case)

Year	Ice Harbor					Lower Monumental					Little Goose					Lower Granite				
	Degrees F	Degrees C	Days over 68	First Day	Last Day	Degrees F	Degrees C	Days over 68	First Day	Last Day	Degrees F	Degrees C	Days over 68	First Day	Last Day	Degrees F	Degrees C	Days over 68	First Day	Last Day
62	76	24.44	60	16 July	13 Sept.															
63	76	24.44	71	13 July	21 Sept.															
64	72	22.22	47	15 July	30 Aug.															
65	75	23.89	42	21 July	31 Aug.															
66	75	23.89	60	14 July	11 Sept.															
67	76	24.44	75	12 July	30 Sept.															
68	75	23.89	54	9 July	9 Sept.															
69	73	22.78	57	19 July	13 Sept.															
70	73	22.78	61	13 July	11 Sept.	74	23.33	53	10 July	3 Sept.										
71	74	23.33	54	25 July	16 Sept.	75	23.89	54	22 July	13 Sept.	76	24.44	54	18 July	9 Sept.					
72	73	22.78	36	9 Aug.	13 Sept.	73	22.78	39	5 Aug.	13 Sept.	73	22.78	42	1 Aug.	12 Sept.					
73	72	22.22	42	22 July	7 Sept.	72	22.22	43	25 July	5 Sept.	74	23.33	46	13 July	2 Sept.					
74	72	22.22	46	30 July	13 Sept.	71	21.67	48	27 July	12 Sept.	74	23.33	51	23 July	14 Sept.					
75	71	21.67	29	28 July	31 Aug.	70	21.11	33	31 July	1 Sept.	70	21.11	37	25 July	30 Aug.	76	24.44	35	21 July	25 Aug.
76	71	21.67	44	30 July	16 Sept.	70	21.11	41	7 Aug.	7' Sept.	71	21.67	38	28 July	13 Sept.	72	22.22	51	18 July	10 Sept.
77	73	22.78	43	27 July	7 Sept.	71	21.67	35	27 July	11 Sept.	72	22.22	26	10 Aug.	4 Sept.	76	24.44	49	28 June	5 Sept.
78	72	22.22	28	3 Aug.	8 Sept.	72	22.22	38	30 July	5 Sept.	72	22.22	29	30 July	27 Aug.	75	23.89	35	20 July	10 Sept.
79	73	22.78	74	19 July	30 Sept.	73	22.78	67	24 July	28 Sept.	74	23.33	64	22 July	24 Sept.	74	23.33	59	17 July	17 Sept.
80	72	22.22	48	31 July	16 Sept.	71	21.67	40	24 July	2 Sept.	73	22.78	43	22 July	3 Sept.	74	23.33	39	21 July	28 Aug.
81	73	22.78	55	29 July	30 Sept.	74	23.33	55	1 Aug.	24 Sept.	73	22.78	61	23 July	21 Sept.	78	25.56	64	17 July	18 Sept.
82	72	22.22	35	14 Aug.	17 Sept.	72	22.22	52	26 July	15 Sept.	73	22.78	49	29 July	15 Sept.	74	23.33	46	26 July	12 Sept.
83	73	22.78	40	8 Aug.	16 Sept.	74	23.33	42	5 Aug.	17 Sept.		No Data		---	---	74	23.33	41	30 July	9 Sept.
84	73	22.78	60	20 July	17 Sept.	73	22.78	49	26 July	12 Sept.		No Data		---	---	74	23.33	46	23-Jul	6 Sept.
85	75	23.89	51	17 July	5 Sept.	73	22.78	54	10 July	1 Sept.		No Data		---	---	74	23.33	49	7 July	28 Aug.
86	75	23.89	73	9 July	19 Sept.	74	23.33	52	9 July	20 Sept.		No Data		---	---	74	23.33	62	30 June	12 Sept.
87	72	22.22	81	4 July	22 Sept.	71	21.67	71	12 July	20 Sept.		No Data		---	---	73	22.78	74	26 June	15 Sept.
88	72	22.22	53	27 July	17 Sept.	72	22.22	50	25 July	12 Sept.		No Data		---	---	73	22.78	85	25 June	20 Sept.
89	71	21.67	50	25 July	12 Sept.	71	21.67	49	25 July	11 Sept.		No Data		---	---	74	23.33	47	13 July	28 Aug.
90	73	22.78	70	24 July	1 Oct.	73	22.78	59	30 July	26 Sept.		No Data		---	---	77	25.00	77	3 July	18 Sept.
91	74	23.33	49	1 Aug	18 Sept.	74	23.33	44	5 Aug.	17 Sept.	76	24.44	55	23 July	16 Sept.	76	24.44	55	12 July	12 Sept.
92	71	21.67	43	16 July	10 Sept.	71	21.67	50	10 July	13 Sept.	72	22.22	49	4 July	10 Sept.	72	22.22	25	* 1 July	* 28 Aug.
93	68	20.00	0	---	---	68	20.00	0	---	---	72	22.22	40	8 Aug.	29 Sept.	69	20.56	8	18 Aug.	5 Sept.
94	70	21.11	18	16 July	5 Aug.	71	21.67	30	* 13 July	* 20 Sept.	72	22.22	28	* 8 July	* 2 Oct.	73	22.78	32	* 17 July	* 11 Sept.
95	70	21.11	18	25 July	11 Aug.	70	21.11	23	19 July	10 Aug.	72	22.22	26	16 July	9 Aug.	68	20.00	0	---	---
96	70	21.11	41	23 July	1 Sept.	70	21.11	41	20 July	29 Aug.	71	21.67	53	12 July	2 Sept.	70	21.11	23	22 July	16 Aug.
97	71	21.67	44	21 July	5 Sept.	71	21.67	28	3 Aug.	8 Sept.	71	21.67	57	1 Sept.	26 Sept.	71	21.67	26	21 Aug.	17 Sept.
98	73	22.78	52	* 19 July	* 8 Oct.	73	22.78	75	17 July	30 Sept.	72	22.22	82	12 July	1 Oct.	70	21.11	36	* 10 July	* 25 Sept.
99	70	21.11	22	5 Aug	26 Aug	73	22.78	29	17 Jul	15 Aug	74	23.33	45	* 18 Jul	* 3 Sept	68	20.00	0	---	---

Notes: Highest temperatures usually occur in August at all dams, but with unseasonably warm weather, may occur in late July or with prolonged hot weather, in September.

Blanks for Little Goose (1983-90) are for years when data was not reported.

* Temperatures over 68 °F occurred between 2 periods.

Ice Harbor Lock and Dam
1998 19 July-5 Sept., 46 days over 68 °F
3 Oct.-8 Oct., 6 days over 68 °F

Lower Monumental Lock and Dam
1994 13 July-21 July, 9 days over 68 °F
31 Aug.-20 Sept., 21 days over 68 °F

Little Goose Lock and Dam
1994 8 July-4 Aug., 28 days over 68 °°F
24 Aug.-2 Oct., 40 days over 68 °F
1999 18 July-18 Aug., 32 days over 68 °F
21 Aug.-3 Sept., 13 days over 68 °F

Lower Granite Lock and Dam
1992 1 July, 1 day over 68 °F
5 Aug.-28 Aug., 25 days over 68 °F
13 Aug.-11 Sept., 29 days over °68 F
1998 10 July-July 22, 5 days over 68 °F
7 Aug.-25 Aug., 7 days over 68 °F
2 Sept.-25 Sept., 24 days over 68 °F

Source: Developed by the Corps

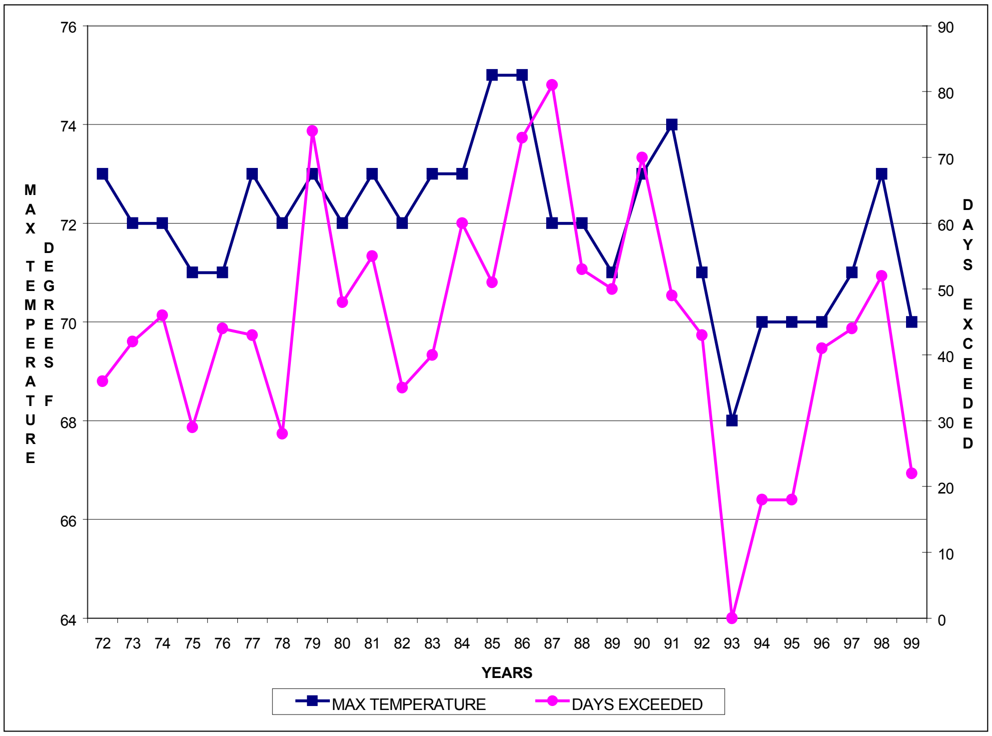


Figure 3-19. Days Exceeding 68°F (20°C)–Ice Harbor Dam Scroll Case Temperature
Source: Developed by the Corps

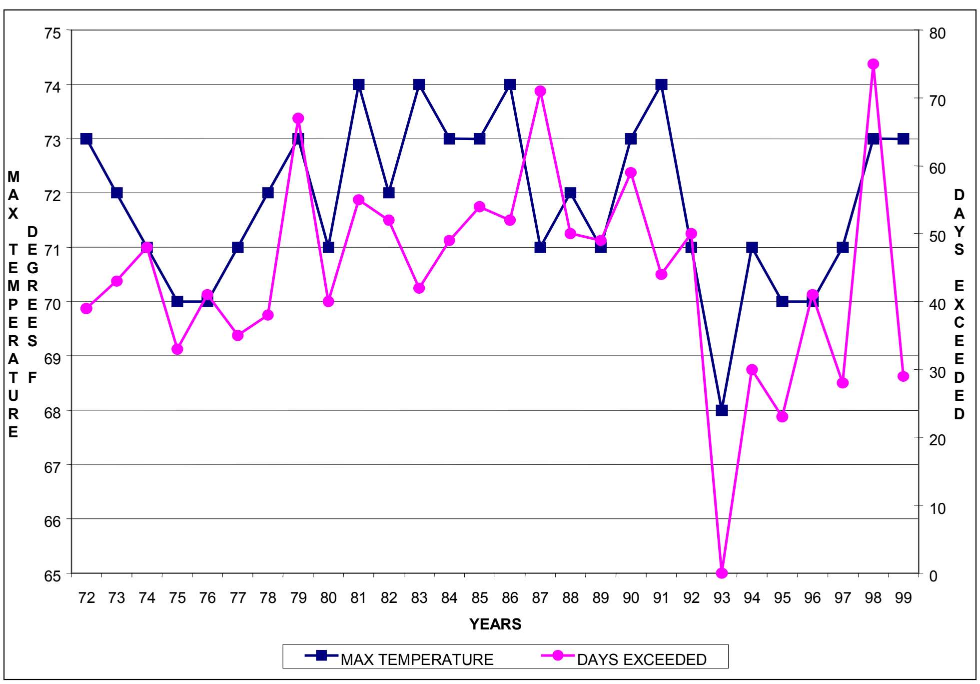


Figure 3-20. Days Exceeding 68°F (20°C)–Lower Monumental Dam Scroll Case Temperature
Source: Developed by the Corps

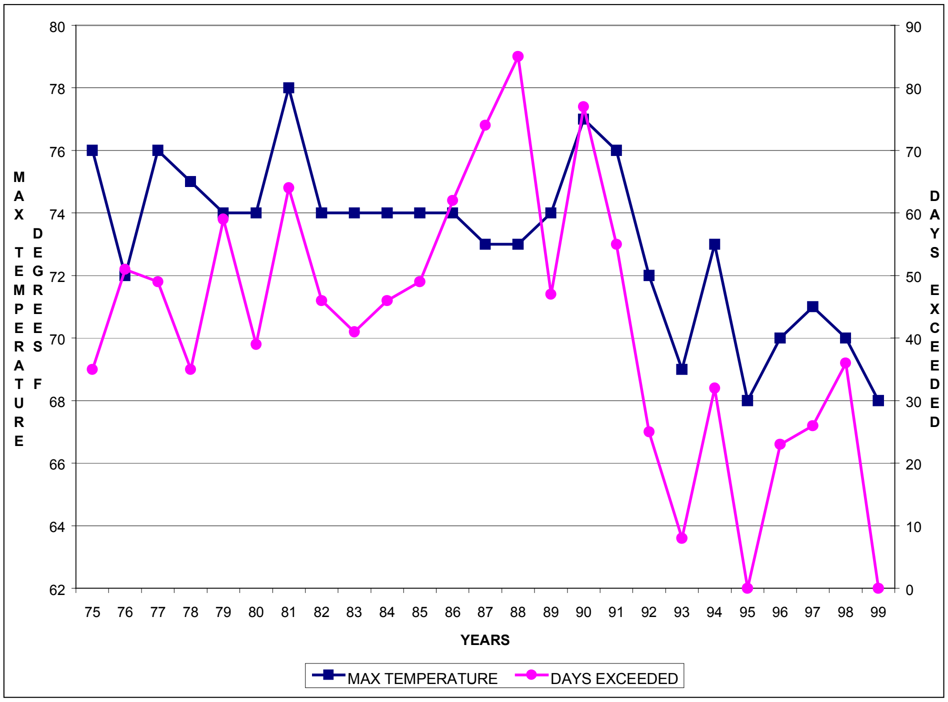


Figure 3-21. Days Exceeding 20°C (68°F)–Little Goose Dam Scroll Case Temperature
Source: Developed by the Corps

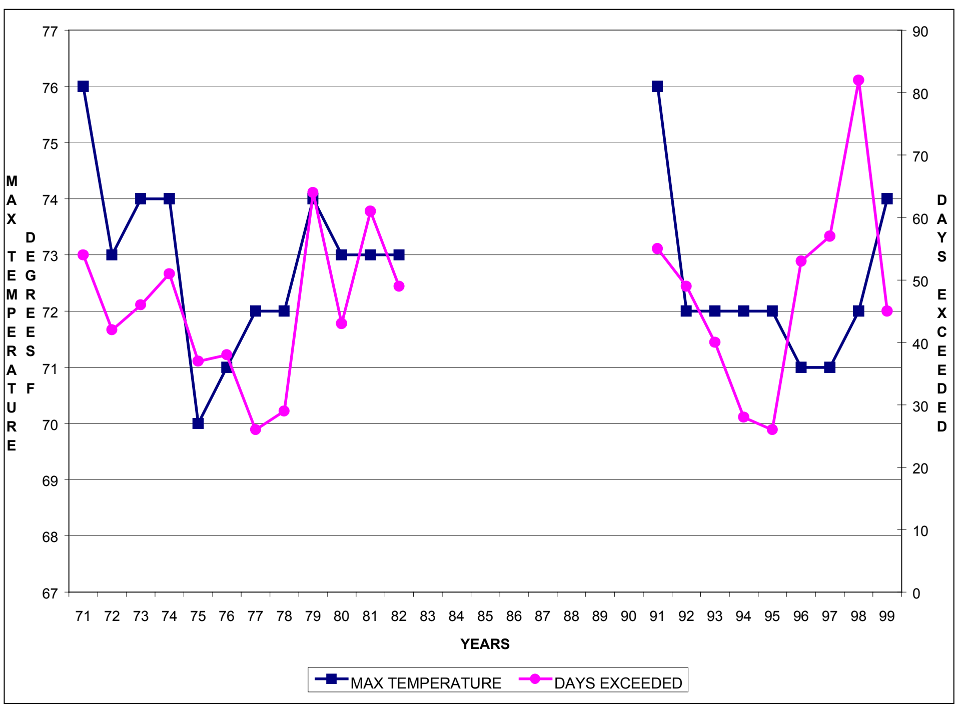


Figure 3-22. Days Exceeding 20°C (68°F)–Lower Granite Dam Scroll Case Temperature
Source: Developed by the Corps

be most noticeable at Lower Granite Lock and Dam as would be expected since Lower Granite is the most upstream reservoir.

When comparing data stations and gages within the lower Snake River reservoir system, several trends were found:

- Summer forebay temperatures at the dams were almost consistently warmer than either the tailwater or scroll case data. The summer tailwater temperatures at the dams were almost consistently warmer than the scroll case temperature data.
- The Lower Monumental reservoir temperatures appear to be lower than the Little Goose reservoir temperatures in the summer. The Palouse and Tucannon Rivers flow into the Lower Monumental reservoir pool and may be affecting temperatures within the reservoir.
- Water temperature profile data for various sites within the reservoir system demonstrate that the water temperatures can certainly vary substantially or almost not at all within the reservoir pools depending on location, time of year, flow levels, and particular reservoir operations.
- Based on the scroll case data, it would appear that the special flow augmentation and temperature control operations at Dworshak may be changing the density gradient at some locations within the reservoirs and may be affecting the number of days each summer that water temperatures exceed 20°C (68°F) at the project scroll case gages. This is most noticeable at Lower Granite Lock and Dam.
- Within the lower Snake River reservoir system, temperatures both warmer and cooler than those above or below the reservoirs can be found depending on the specific location, time of year, depth, flows, reservoir operations, and other factors.

Temperature Effects of Dworshak Reservoir Releases

The Dworshak reservoir is a 3 million-acre-foot storage project located on the North Fork of the Clearwater River approximately 56 kilometers (35 miles) above the Spalding gage. The dam has a multi-level outlet structure so water can be withdrawn at selected depths. When the reservoir is thermally stratified, the dam operators maintain the temperature of the outflowing water at about 10 to 12°C (50 to 54°F). This is necessary since a Federally operated fish hatchery downstream utilizes water from the outlet.

Figure 3-23 shows the average temperature of the water discharged from the reservoir between 1977 through 1980 and 1993 through 1998. The 1977 through 1980 data were obtained from a temperature sensor located on the downstream side of the dam on the pier nose of powerhouse bay number two. Readings were usually taken once per day between Monday and Friday. The 1993 through 1998 graph is based on hourly data collected downstream near the fish hatchery. The illustration shows that during both time intervals the mean temperature of the outlet water during early May was from 4 to 5°C (39 to 41°F). Average temperatures gradually increased until the latter part of June and then fluctuated within the target range through September. Overall, significant differences are not apparent between the two time periods. However, interannual variability does exist. Data from 1994 show that outlet temperatures reached the target range by June 1, decreased to 6°C during the July drawdown period, and subsequently returned to the 10 to 12°C (50 to 54°F) range (Figure 3-24). As a comparison, outlet water temperatures during the 1997 summer drawdown were within the target range, but increased to almost 16°C (61°F) in early September.

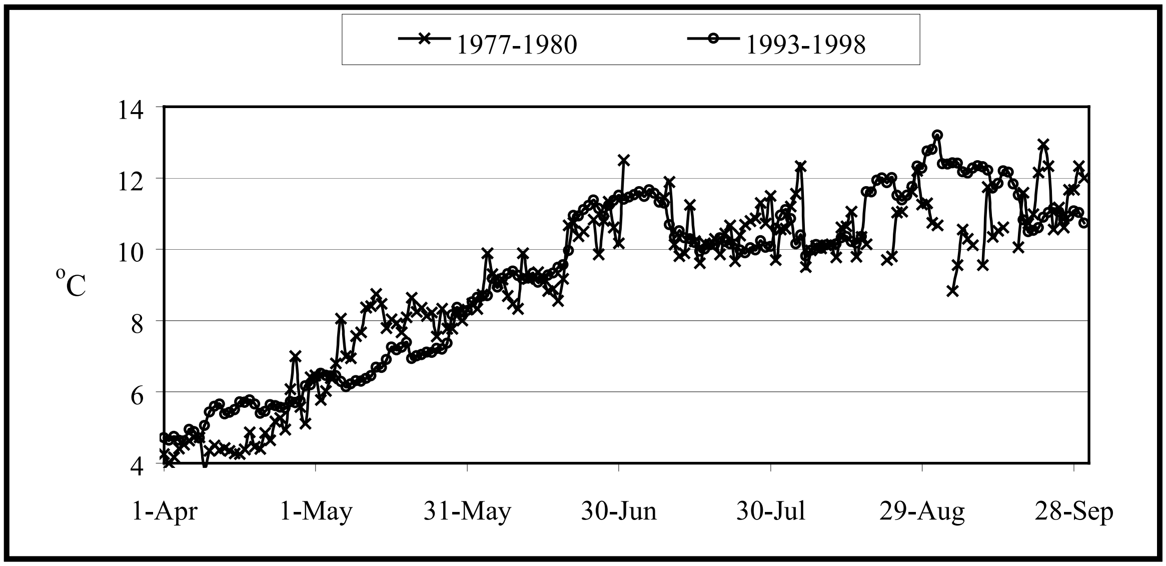


Figure 3-23. Average Temperatures in the Dworshak Reservoir Outlet, 1977 through 1980 and 1993 through 1998
Source: Developed by Normandeau

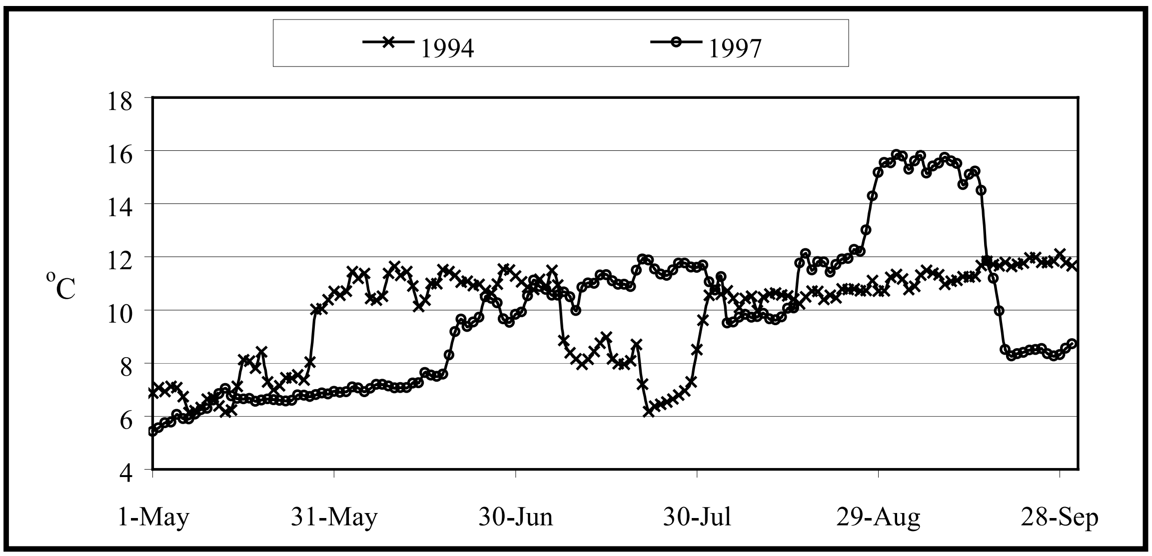


Figure 3-24. Daily Temperatures in the Dworshak Reservoir Outlet, 1994 and 1997
Source: Developed by Normandeau

The timing and quantity of discharges from Dworshak Dam began to change in the early 1990s. The Dworshak flood control operation has remained constant but flow targets for streamflow augmentation were established for the Lower Granite reservoir downstream to facilitate anadromous fish migration in the lower Snake River. Some augmentation water is usually released during April, May, and June. More importantly from a temperature standpoint, however, is the increase that occurred during July and August of recent years when up to 75 percent of the discharge in the Clearwater River originated from the Dworshak reservoir. The proportion was as high as 90 percent during 1994, which was a low-flow year for the entire drainage basin. Finally, the percentage of the flow in the Clearwater during September that originated from Dworshak dropped dramatically during the latest 6-year interval. Historically, almost 80 percent of the flow in the Clearwater River during that month originated from the Dworshak reservoir.

These September releases were necessary to generate power and to lower the pool elevation for the required winter flood control operation.

The changes in system operation at the Dworshak reservoir to summer releases (Figure 3-25) have also changed the thermal regimen of the downstream Clearwater River. Figure 3-26 shows mean temperatures for the Spalding gaging station for the 1974 through 1990 (the selector gates went into operation in late 1973) and 1993 through 1998 periods. The two time intervals are almost indistinguishable from the beginning of the year through late June and from early October to the end of the year. The primary differences occur between early July and late September. Mean temperatures peaked close to 19°C (66°F) during the last week of July and first week of August in the historical period before declining towards the winter lows. During the 1993 through 1998 period, three maximum and two minimum points were reached. The three peak averages of about 17°C (63°F) occurred during the first weeks of July, August, and September. The two troughs averaged 14°C (57°F) and were recorded during the latter parts of July and August. Figure 3-26 also shows that the water temperatures during September 1993 through 1998 have been 1 to 3°C (1.8 to 5.4°F) higher than during the historical period as a result of the 30 percent reduction in reservoir discharges.

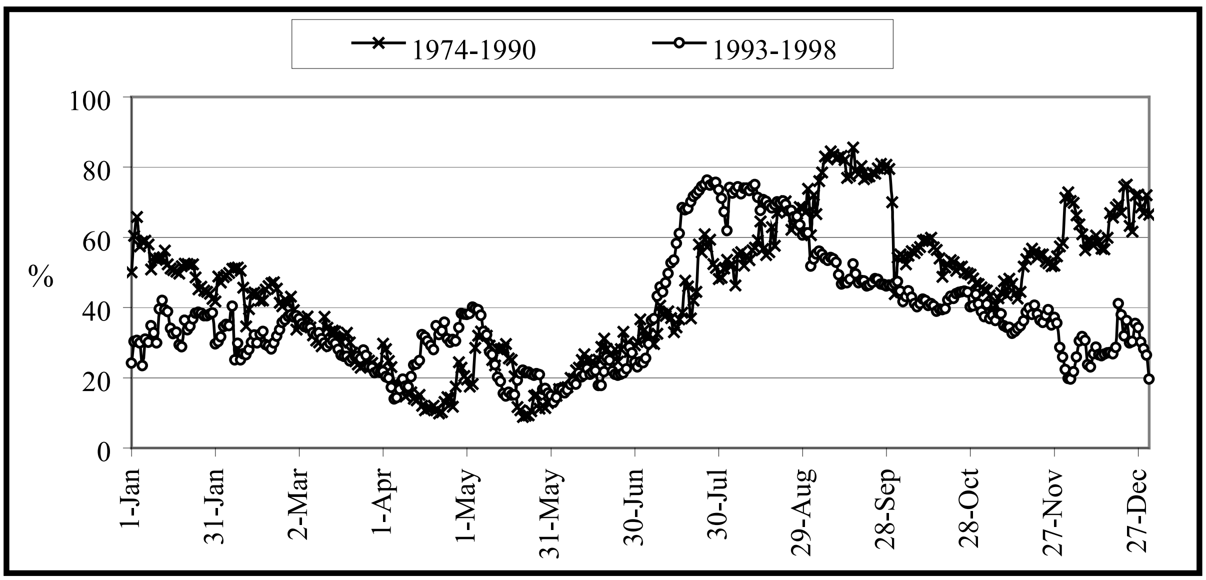


Figure 3-25. Flow in the Clearwater River at Spalding, Idaho–Percentage Due to Discharge from the Dworshak Reservoir, Idaho
Source: Developed by Normandeau

The changing temperature regimen in the Clearwater River has also influenced observed temperatures in the lower Snake River. The period of record for temperature information in the Lower Granite reservoir at one location is not as extensive as the one at Spalding, Idaho. However, data collected at a depth of 4.6 meters (15 feet) in the forebay of Lower Granite Lock and Dam from 1988 through 1998 are available. Additionally, Dr. David Bennett at University of Idaho provided thermistor recordings collected from a depth of approximately 5 meters (16 feet) at RM 110.5. A comparison of the forebay data with that collected 5 kilometers (3 miles) upstream shows that both sets follow the same seasonal trends, but differ slightly with respect to magnitude (Figure 3-27). From April 1 to May 1 the readings are equal, between May 1 and July 1 the forebay temperatures are 0.5 to 1.0°C (0.9 to 1.8°F) warmer, and remain 1 to 2°C (1.8 to 3.6°F) higher from July 1 through September. However, the data also indicate that the 1993 through 1998 mean surface temperatures during July and August were up to 3°C (5.4°F) cooler near the dam than they had been between 1988 through 1990. Additionally, average surface temperatures were up to 2°C (3.6°F) warmer during September of the most recent period. These shifts are believed to be primarily due to the cooler water originating from the Dworshak reservoir during the recent interval, and the relative volume of water that the Clearwater River contributed to the lower Snake River.

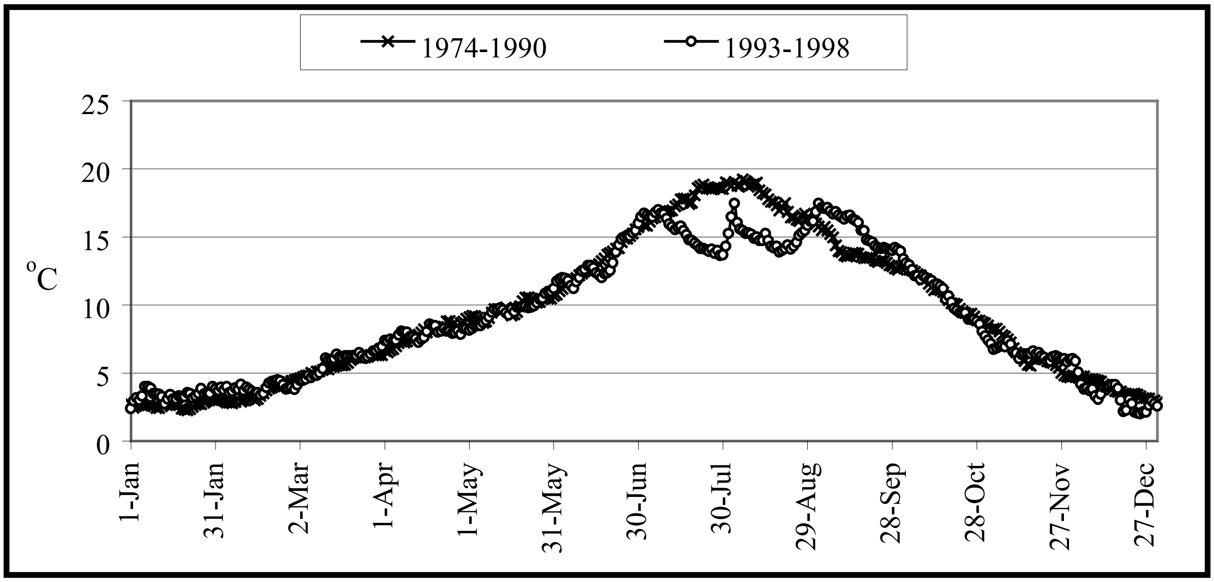


Figure 3-26. Average Temperatures in the Clearwater River at Spalding, Idaho 1974 through 1990 and 1993 through 1998

Source: Developed by Normandeau

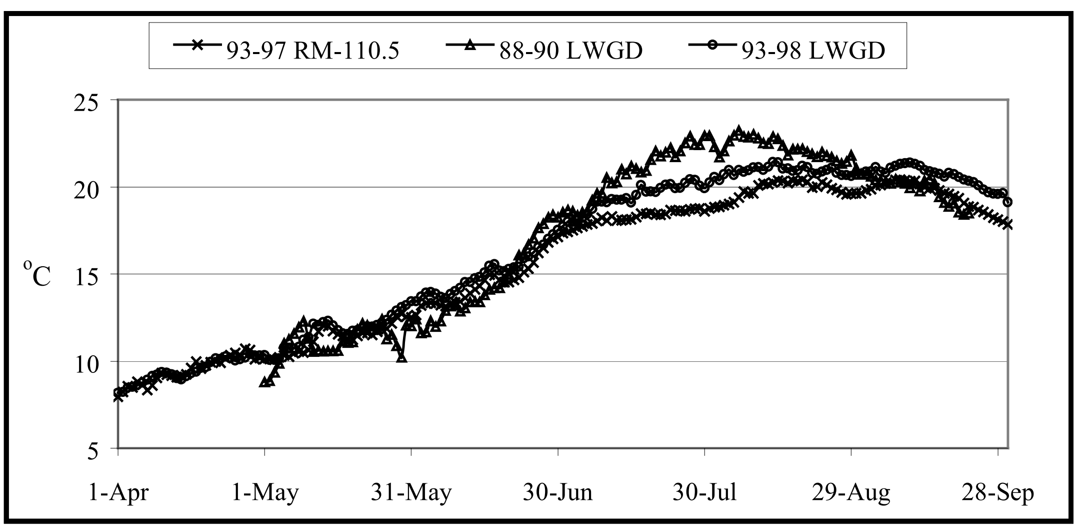


Figure 3-27. Average Near-surface Temperatures from the Lower Snake River at RM 110.5 during 1993 through 1997 and in the Forebay of Lower Granite Lock and Dam during 1988 through 1990 and 1993 through 1998

Source: Developed by Normandeau

Figure 3-28 illustrates the changes that have occurred with respect to the relative importance of the Clearwater River to discharge, and consequently temperature, in the lower Snake River. The Clearwater River accounted for 3 to 7 percent more flow during late April and early May of the recent interval and up to 9 percent less from the latter part of May and through June. These changes, however, did not have a significant impact on temperatures in the lower Snake River (Figure 3-27) since the free-flowing Snake River at the USGS gaging station at Anatone, Washington, has historically only been 2 to 3°C (3.6 to 5.4°F) warmer during those months. With the implementation of summer flow augmentation, the average contribution of the Clearwater River has risen to as high as 45 percent (Figure 3-27), or an increase of up to 19 percent. At the same time, the mean temperature in the Clearwater River decreased so that it was running 6 to 8°C (10.8 to 14.4°F) cooler than the Snake River instead of the 2 to 3°C (3.6 to 5.4°F) difference that normally occurred. An analysis of temperature data collected from the free-flowing Snake River at Anatone also revealed that the mean daily values for the 1974 through 1990 interval were essentially equivalent to those calculated for the 1993 through 1998 period.

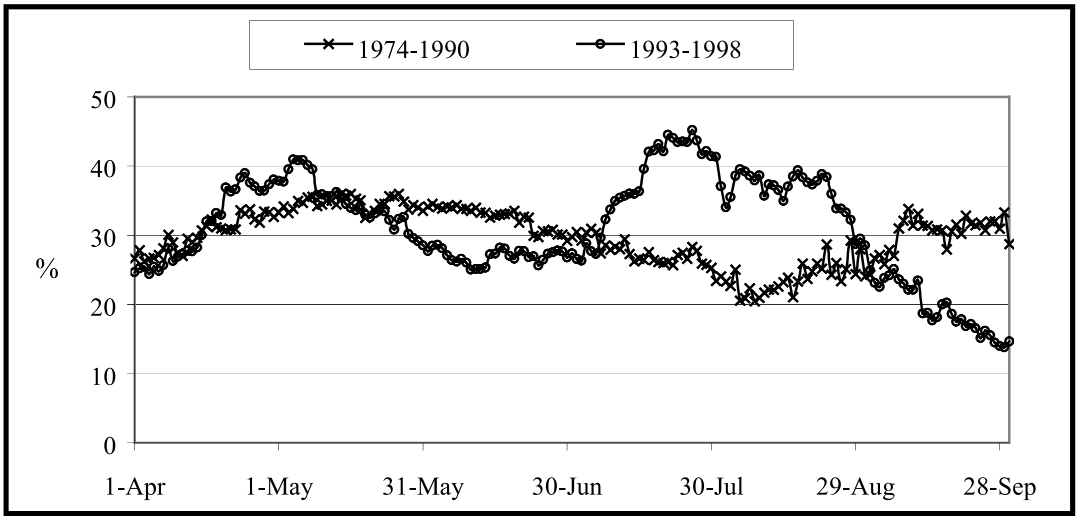


Figure 3-28. Discharge in the Lower Snake River at Lower Granite Lock and Dam—Average Percentage that Originated from the Clearwater River during 1974 through 1990 and 1993 through 1998

Source: Developed by Normandeau

The lower Snake River reservoirs do not stratify thermally to the extent that the Dworshak reservoir or as many lakes do. Significant temperature differences between the surface and bottom waters are generally rare in running waters. A frequently used rule-of-thumb is that a water body has to have a mean depth greater than 10 meters (30 feet) and a mean annual hydrologic residence time in excess of 20 days before strong thermal stratification develops. The mean depths of the lower Snake River reservoirs are greater than 10 meters (30 feet), but the average annual residence time for each pool is about 5 days. The calculated retention time can approach 20 days during the summer and fall of low-flow years, such as 1994, and vertical temperature differences do develop. However, wind- and flow-induced turbulent diffusion, along with convective mixing, prevent that from happening most of the time

Four sets of temperature profiles are shown in Figure 3-29 for RM 108 using data from 1975, 1977, 1994, and 1997 and may illustrate some of the potential thermal influence factors that the summer

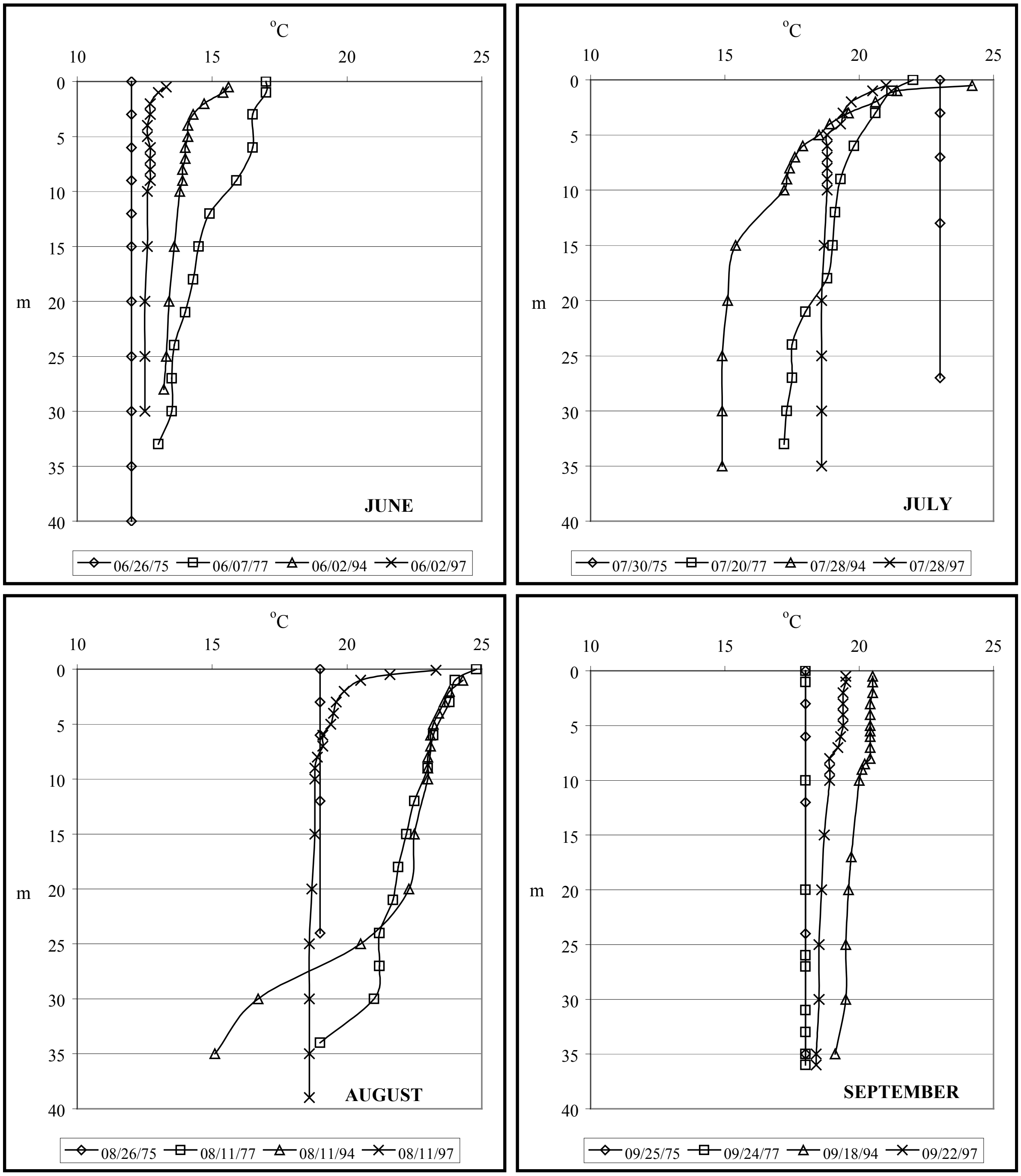


Figure 3-29. Temperature Profiles at Snake River RM 108 during June, July, August, and September of 1975, 1977, 1994, and 1997

Source: Developed by Normandeau

releases from the Dworshak reservoir may have on the lower Snake River. Additionally, these four years represent a broad spectrum of runoff conditions. The 1977 water year had a very low mean runoff of 18.9 kcfs at the Anatone gaging station-the lowest for the period of record from 1958 through 1991. The 1994 water year was also comparatively low, with a mean runoff of 21.1 kcfs in the same free-flowing reach of the Snake River. The 1997 water year was the highest for the period of record (1958 through 1998) with a mean discharge of 59 kcfs. The 1975 water year was intermediate at 45.4 kcfs, but still higher than the 1958 through 1998 mean of 35.9 kcfs. Temperature differences between the surface and bottom of the profiles were minor during June of each year. The 1975 profile was isothermal at 12°C (54°F) while the 1977 water temperatures were the warmest of those illustrated and showed a 4°C (7.2°F) difference between the surface and 33 meters (108 feet). The 1994 and 1997 temperatures were intermediate and the differences between the top and bottom of the profiles were 2 and 1°C (3.6 and 1.8°F), respectively. During late July, the influences of surface warming by short-wave radiation and cold water from Dworshak are evident. The 1975 profile was still reported at isothermal at 23°C (73°F). A distinct thermocline did not develop in 1977, but there was a 4°C (7.2°F) difference between water at 33 meters (108 feet) and 1 meter (3 feet) while the surface water reached 22°C (72°F). The low inflow, and, therefore, longer residence time probably enhanced this thermal difference. The greatest vertical difference shown was in 1994, and this was also the year when Dworshak releases composed up to 90 percent of the discharge in the Clearwater River and runoff in the entire basin was at a near-record low. While the water at 0.5 meter (1.6 feet) reached 24°C (75°F), it was only 15°C (59°F) below a depth of 15 meters (49 feet). The 1997 profile was almost isothermal; 19°C (66°F) between 35 meters (115 feet) and 4 meters (13 feet) and warming to 21°C (70°F) near the surface. The August temperature profiles for 1975 and 1997 were very similar. Both were near 19°C (66°F) throughout the water column, except for a 4°C (7.2°F) increase in the upper 5-meter (16-foot) layer during 1997. The 1977 and 1994 profiles were also very similar during the August sampling dates with temperatures of approximately 24°C (75°F) near the surface and decreasing to 19 and 15°C (66 and 59°F) at the base of the profile in 1977 and 1994, respectively. September profiles present almost complete mix with 18°C (64°F) throughout in 1975 and 1977 and approximately 20 and 19°C (68 and 66°F) in 1994 and 1997, respectively. Superimposed on the annual temperature regime of rivers are variations that take place from day to day and within a 24-hour period. These diel fluctuations are typically greater for a small river than for a large river or reservoir. This is simply because the smaller water body has a relatively larger surface area and streambed contact, along with less mass, than does a larger system. Additionally, water has a very high specific heat-it requires more heat energy to raise the temperature of a unit mass of water by 1°C (1.8°F) than most other substances. This means that a larger body of water has a greater buffer capacity to temperature changes. A comparison of diel temperatures in the Clearwater River and lower Snake River illustrates this effect. Two time periods (early June and August 1997) and two locations (Snake River RM 110.5 and Clearwater River RM 4) are used as examples in Figure 3-30. During both time intervals, the Clearwater River showed diel variability of approximately 2°C (3.6°F) whereas the daily fluctuations in the reservoir were less than 1°C (1.8°F).

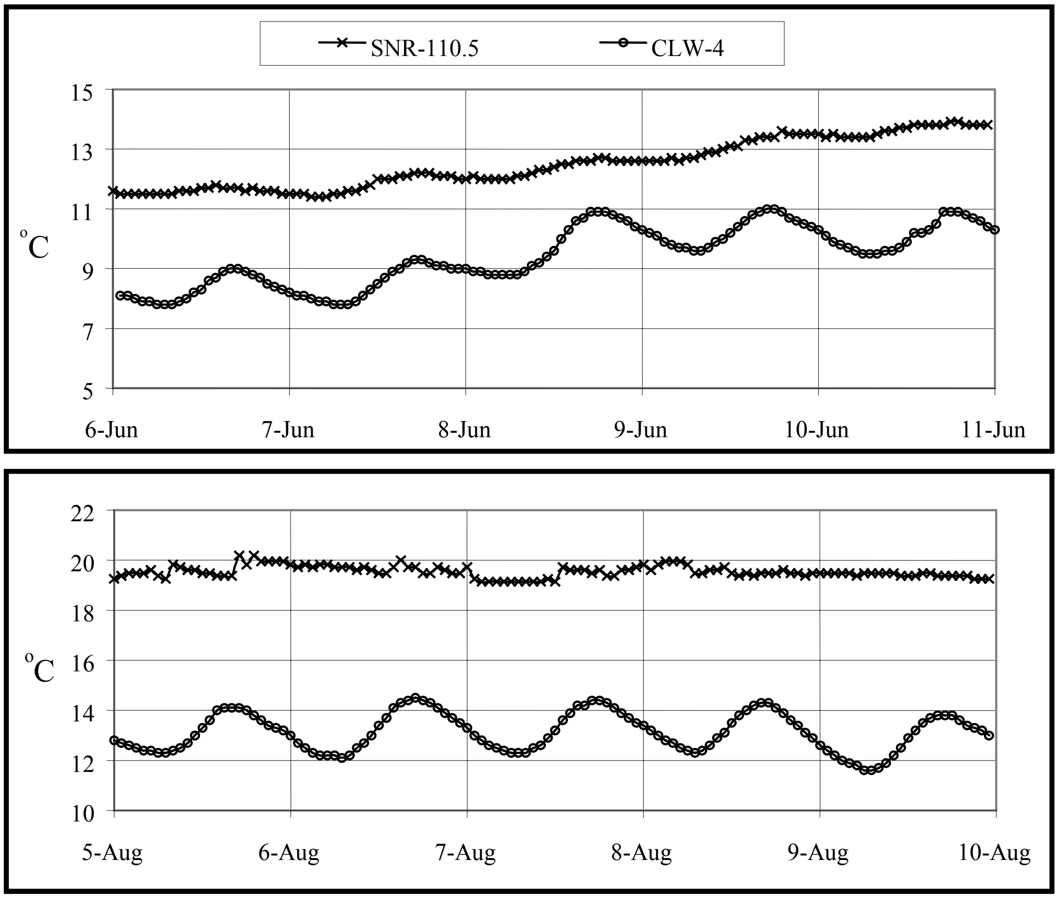


Figure 3-30. Hourly Diel Temperatures Collected Near the Surface in the Lower Snake River at RM 110.5 and in the Clearwater River at RM 4 during June and August 1997
Source: Developed by Normandeau

Water Temperature Evaluation Conclusion Summary

The following bulleted items highlight the findings and conclusions of this water temperature evaluation that was based on existing empirical data (both the data displayed and additional data available but not shown in this report) and current reservoir regulation practices for flow augmentation, temperature control, and voluntary spill for fish:

- Anadromous fish traveling through the lower Snake River reach above the Lower Granite reservoir should encounter water temperatures similar to those found at the Anatone gaging station. Water temperatures in April, May, and June during the beginning of the fish movement season should not be warmed to levels that are of concern. On the average, summer temperatures (June through September) should be expected to exceed 20°C (68°F) for a period of approximately 60 days each year. The maximum daily seasonal temperature expected each year during the exceedance period should be approximately 23°C (73°F). During the June through September period each year, typical fluctuations each day between night and day (daily maximum and daily minimum) water temperatures should be approximately 0.5 to 1.5°C (0.9 to 2.7°F). Temperatures within the river cross-section should be quite consistent and reflect a well-mixed river for uniform temperature conditions (based on the July 1999 study done by Idaho Department of Environmental Quality) (Essig, 1998).

- Anadromous fish traveling through the Clearwater River reach above the Lower Granite reservoir should encounter water temperatures similar to those formerly found at the Spalding gaging station. Water temperatures in April, May, and June during the beginning of the fish movement season should not be warmed to levels that are of concern. On the average, summer temperatures (June through September) should be expected to exceed 20°C (68°F) for a period of approximately 5 days each year. The rest of the summer season water temperatures should average much cooler (2 to 5°C [3.6 to 9°F]) than the normal before the Dworshak summer augmentation and temperature control operation was started. The maximum daily seasonal temperature expected each year during the exceedance period should be approximately 21°C (70°) or higher. During the June through September period each year, typical fluctuations each day between night and day (daily maximum and daily minimum) water temperatures should be approximately 1.5 to 3.0°C (2.7 to 5.4°F). Temperatures within the river cross-section should vary within the cross-section and not necessarily be well mixed until the flow has moved into the downstream portions of the Lower Granite reservoir (based on the July 1999 study done by Idaho Department of Environmental Quality) (Essig, 1998). This is discussed further in the next section.

- Anadromous fish traveling through the Snake River reach below the Ice Harbor reservoir should encounter water temperatures similar to those formerly found at the discontinued Burbank station and at the current Ice Harbor TDGMS tailwater station. Water temperatures in April, May, and June during the beginning of the fish movement season should not be warmed to levels that are of concern. On the average, summer temperatures (June through September) should be expected to exceed 20°C (68°F) for a period of approximately 60 days each year. The maximum daily seasonal temperature expected each year during the exceedance period should be approximately 23°C (73°F). During the June through September period each year, typical fluctuations each day between night and day (daily maximum and daily minimum) water temperatures should be approximately 0.4 to 1.0°C (0.7 to 1.8°F). Temperatures within the river cross-section should be quite consistent and reflect a well-mixed river for uniform temperature conditions.

- Comparing the reach above the reservoir system with the reach below the reservoir system:

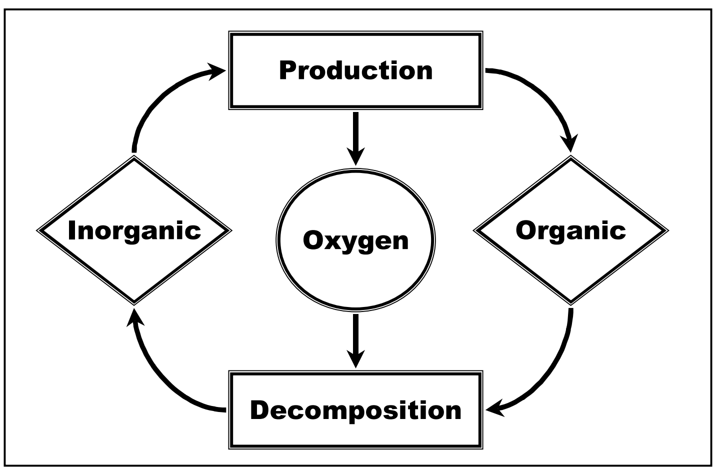
  1. Summer daily maximum water temperatures expected for average current reservoir operation. (Above the four LSR reservoirs = 23°C [73°F].) (Below the four LSR reservoirs = 23°C [73°F].)
  2. Average number of days each year above 20°C for average current reservoir operation. (Above the four LSR reservoirs = 60 days) (Below the four LSR reservoirs = 60 days)
  3. Average fluctuation between the summer daily maximum and daily minimum water temperatures for average current reservoir operation. (Above the four LSR reservoirs = 0.5 to 1.5°C [0.9 to 2.7°F].) (Below the four LSR reservoirs = 0.4 to 1.0°C [0.7 to 1.8°F].)
  4. The reach above the reservoir system warms earlier and cools earlier (1 to 3 weeks) than the reach below the reservoir system.

- Anadromous fish traveling through the lower Snake River reservoir system should encounter more variations in water temperatures than found in either the reach above or below the reservoir system. Daily temperatures in the summer at any location within the reservoirs could easily be 0 to 2°C (0 to 3.6°F) warmer or cooler than temperatures above or below the reservoir system based on observed temperature profile data. Actual temperatures encountered at any location should be dependent on that particular location, the time of year, climatic conditions, flow conditions, current augmentation and temperature control operations, and voluntary spill/power operations. Surface water temperatures within the reservoir pools should have less diel temperature fluctuation than found in either the Clearwater River or the Snake River above Lewiston.

- It appears that cold water releases from the Dworshak reservoir in July and August may be changing some of the temperature profiles found within the Lower Granite reservoir such that parts of the reservoir pool may be or resemble stratified sections at some locations.

3.2.4.2 Dissolved Oxygen

Dissolved oxygen is critical to the ecology of both riverine and reservoir systems. Foremost among the functions that it performs is the sustenance of most biological life. Nearly as important, oxygen is the key element in many chemical processes in water. Through oxidation and reduction reactions, the concentration of oxygen has the ability to influence the concentration of many dissolved substances in water. These chemical processes include the decomposition of organic matter, the cycling of nutrients and the transformation and transport of toxic substances within the water column and between the sediments and the water column, among others. A simplified diagram of the role of dissolved oxygen in aquatic systems is presented in Figure 3-31. The biochemical processes of photosynthesis and respiration by living organisms provide a means by which the aquatic community can regulate the amount of oxygen in the aquatic environment, within limits. Most organisms cannot survive with too little oxygen while the solubility of oxygen generally limits the maximum amount that can be dissolved in water under most conditions. Supersaturation of water with oxygen does occur during periods of intense photosynthetic activity and as a result of dissolution of oxygen under high hydrostatic pressure in the plunge pools of high head dams (Bowie et al., 1985). Both of these special situations occur, at times, in the lower Snake River.



$$106CO_2 + 16NO_3^- + HPO_4^{2-} + 122H_2O + 18H^+ \text{ (+ trace elements; energy)}$$

Photosynthesis $\downarrow\uparrow$ ***Respiration***

$$(C_{106}H_{263}O_{110}N_{16}P_1) + 138O_2$$

algal protoplasm

Figure 3-31. A Simplified Relationship of Oxygen to the Production/Decomposition Cycle in Natural Waters. As Indicated by the Equation, Photosynthesis or Production Creates Oxygen, and Respiration or Decomposition Depletes Oxygen from a Lake (adapted from Reckhow and Chapra [1983])

Source: Developed by Normandeau

The primary sources of dissolved oxygen to river and reservoir systems include the dissolution of oxygen from the atmosphere at the air water interface (reaeration) and production of oxygen as a byproduct of photosynthesis by attached and free-floating aquatic plants (Bowie et al., 1985). The solubility of oxygen in water is inversely proportional to temperature such that cold water can hold more oxygen per unit volume than warm water. Reaeration rates are generally determined by turbulence at the air water interface. This turbulence can occur as a result of the physical characteristics of the channel such as slope and roughness producing turbulent fast water, falls, and rapids. This reaeration process generally dominates in riverine reaches of rivers. For water quality modeling purposes, reaeration in rivers is often defined empirically as a function of water depth and velocity (Bowie et al., 1985). Reservoir reaches typically have a larger surface area than riverine systems. Reaeration in these reaches is typically a function of wind over the water surface (Reckhow and Chapra, 1983). Wind causes waves and turbulence at the air-water interface, further promoting reaeration at the surface.

Dissolved oxygen is removed from riverine and reservoir systems through a variety of processes. Foremost among those processes are the processes of biological respiration and decomposition of organic matter. Aerobic respiration (respiration in the presence of oxygen) is the process by which living organisms gain energy by breaking down organic molecules in the presence of oxygen and producing carbon dioxide as a byproduct. Decomposition of nonliving organic matter (both dead organisms and waste products from live organisms) by bacteria in the presence of oxygen follows a similar biochemical reaction. Decomposition that occurs in the water column is often referred to as biochemical oxygen demand while decomposition in the sediments is referred to as sediment oxygen demand. Other potential losses of oxygen from aquatic systems are related to the discharge of oxygen demanding substances and degassing at the air-water interface (generally occurs when water is supersaturated with oxygen).

Spatial variability in DO includes longitudinal, vertical, and temporal components. Longitudinal variability of DO in river and reservoir systems can occur in response to a variety of conditions in a river or reservoir system. Typical depletion of dissolved oxygen 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 dissolved oxygen concentration, and the reaeration potential of the reach downstream of the source (Bowie et al., 1985). Longitudinal variability in dissolved oxygen concentrations can also be related to the locations of areas of sediment oxygen demand, stands of ABA or 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. Dams have the ability to change the lateral dissolved oxygen distribution because they have the ability to dramatically alter flow patterns. Spilling water over the top of a dam has the potential to significantly raise the dissolved oxygen concentration downstream while passing water through turbines may have little influence on the dissolved oxygen concentration downstream.

DO does not typically vary vertically in riverine environments because they are typically well mixed. However, in reservoirs the potential exists for dissolved oxygen depletion at depth. This 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 dissolved oxygen depletion through the settling into and decomposition of organic matter in the deep layers. Sediment oxygen demand is also exerted on these deep layers of the water column. The slow movement of water in deep reaches of a reservoir relative to shallow sections allows fine particles of silt and organic matter to settle out. The potential for oxygen depletion at depth is higher in slow, deep, biologically productive reservoirs. Dissolved oxygen concentrations can also be elevated near the surface of reservoirs in the presence of intense algal blooms. Blooms have the potential to supersaturate the water column with oxygen to concentrations well above saturation (Wetzel, 1983).

Dissolved oxygen dynamics vary over temporal time scales ranging from seasonal to hourly. Seasonal variability in dissolved oxygen concentrations in a system is typically related to water temperature in northern temperate climates. Dissolved oxygen solubility increases as water temperatures drop (Wetzel, 1983). During the winter, reduced water temperatures and subsequent increased solubility of oxygen result in higher dissolved oxygen concentrations. In the summer, increased water temperatures have the opposite effect, resulting in lower dissolved oxygen concentrations. Increased water temperatures accelerate the decomposition of organic substances, resulting in higher oxygen consumption during periods when water is warmer. Changes in river flow also have the potential to result in changes in dissolved oxygen concentrations, particularly in river systems with fixed continuous discharges of oxygen-demanding substances. Periods of high river flow offer more dilution and thus reduce the magnitude of the dissolved oxygen sags experienced downstream of discharges. Periods of low flow can increase the magnitude of sags downstream.

The growing season has different dissolved oxygen dynamics than the nongrowing season. During the late fall, winter, and spring, respiration and decomposition oxygen demands are balanced primarily by reaeration. During the summer and early fall (the growing season), oxygen is added to the system by reaeration as well as photosynthesis by phytoplankton, ABA, and macrophytes (Wetzel, 1983). Additional oxygen losses present during the growing season include respiration from these same aquatic plants. The addition of photosynthesis and respiration from aquatic plants during the growing season results in variability in dissolved oxygen concentrations related to a diurnal cycle of daylight and night. There is generally a net production of oxygen in surface waters (in the lighted or photic zone) during daylight hours with peak oxygen concentrations occurring in mid-to-late afternoon while there is a net loss of oxygen at night with minimum surface concentrations occurring immediately before dawn. The magnitude of this diurnal fluctuation can be as much as 2 to 4 milligrams per liter of oxygen or higher. The diurnal influence of aquatic plants on the dissolved oxygen regime typically peaks in August although algal blooms can strongly influence dissolved oxygen concentrations at other times during the growing season. Decomposition of aquatic plants and algae at the end of the growing season exerts an additional demand on the oxygen resources of river and reservoir systems.

The lower Snake River is classified as Class A water according to WAC-173-201A. These standards stipulate that the dissolved oxygen content of freshwater shall exceed 8 milligrams per liter. The sources of information used to evaluate the degree that this standard is met in the lower Snake River system were:

- 1969 through 1971 and 1975 through 1977 data collected by researchers at Washington State University and University of Idaho under contract with the Corps. Sampling stations ranged from RM 6 below Ice Harbor Lock and Dam to RM 154 on the Snake River and up to RM 9 on the Clearwater River. Most of the data were collected during the summer and fall although monitoring did continue through the 1976/1977 winter. The azide modification to the Winkler Method was utilized during the first study period. An electrometric probe was used during the second interval and calibrated using air standardization and the Winkler Method.

- The next major set of reservoir dissolved oxygen data was collected from 1994 through 1998. The 1994 through 1996 data were collected by the State of Washington Water Research Center (WRC), while the 1997/1998 study was a collaborative effort between the WRC and the Corps. The monitoring stations visited varied slightly from year to year, but encompassed the ones established in previous investigations to maintain consistency. Most of the measurements were taken between May and October on a biweekly or monthly schedule although monthly sampling did continue through the winter of 1994/1995. Hydrolab Surveyor® 3 or 4 and YSI 6,000 instruments were pre- and post-calibrated each field day using the saturated air method along with Winkler titrations. Any questionable field results were also verified with Winkler titration kits before leaving the site.

- The NMFS collected near-shore oxygen data as part of a 1994/1995 study in the Lower Granite reservoir. Three locations were monitored on a biweekly to monthly schedule. The NMFS also used a Hydrolab Surveyor® 3 that was calibrated during each field event according to the manufacturer's recommendations.

- The Corps also collects dissolved oxygen data as part of its overall water quality-monitoring program. Its primary focus has been on fixed stations above and below the four lower Snake River dams and in the Clearwater River up to the Dworshak reservoir. Oxygen data, presented as millimeters mercury, are available from early 1995 to the present at many of the stations. Data are not available from all stations year-round, with the primary data gaps occurring during the winter months. However, these stations are the only ones within the system that provide hourly information. The majority of the measurements have been taken with Hydrolab® instrumentation and regularly maintained by Corps personnel. The Corps has also periodically collected profile data within the reservoirs, but the frequency has varied throughout the years.

- The USGS has collected oxygen data at its gaging stations, although the intensity and frequency has varied. Composite sampling was completed at RM 2.2 from 1977 to 1990 on a monthly to quarterly basis. Monthly monitoring was also completed at RM 9.7 from 1967 to 1972 (with the exception of the 1970 and 1971 water years); at RM 106.5 from July 1975 to July 1977; at RM 132.9 between the latter part of 1971 and mid-1972; and finally at the Anatone, Washington, gaging station during the 1976 through 1980 water years. Its methods included both an electrometric probe and the Winkler titrations.

- The Ecology has several long-term monitoring stations throughout the state, and one of them is located at RM 139.2. Monthly data are available from December 1990 to the present. Prior to that, monthly to quarterly data were collected from late 1963 to late 1969 with the exception of 1966 and 1967. Its methods follow those used by the USGS.

The concentrations of dissolved oxygen in the reservoirs change from one year to the next, as well as seasonally. Figure 3-32 illustrates water column averages for June through September data at three reservoir stations (RM 18, RM 83, and RM 108) during 1975, 1977, 1994, and 1997. The graph shows that the average dissolved oxygen concentration at RM 18 remained near 8 milligrams per liter between 1975 and 1977, yet decreased at RM 83 and RM 108. The decline from 8.9 milligrams per liter to 7.3 milligrams per liter at RM 108 was the only change that was statistically significant at the 95 percent confidence interval, based on the available data. Perhaps more interesting are the comparisons between the data collected during the 1970s versus that from the 1990s. Both 1977 and 1994 were low-flow years. The increases from 7.5 milligrams per liter to 9.6 milligrams per liter at RM 83 and from 7.3 milligrams per liter to 9 milligrams per liter at RM 108 were significant. The shift from 8.1 milligrams per liter to 8.8 milligrams per liter at RM 18 during the same 17-year interval, however, is not statistically significant. A comparison of the 1975 and 1997 data failed to show any significant differences in average concentrations. Additionally, the only major shift between 1994 and 1997 occurred at RM 83 when the mean decreased from 9.6 milligrams per liter to 8.6 milligrams per liter.

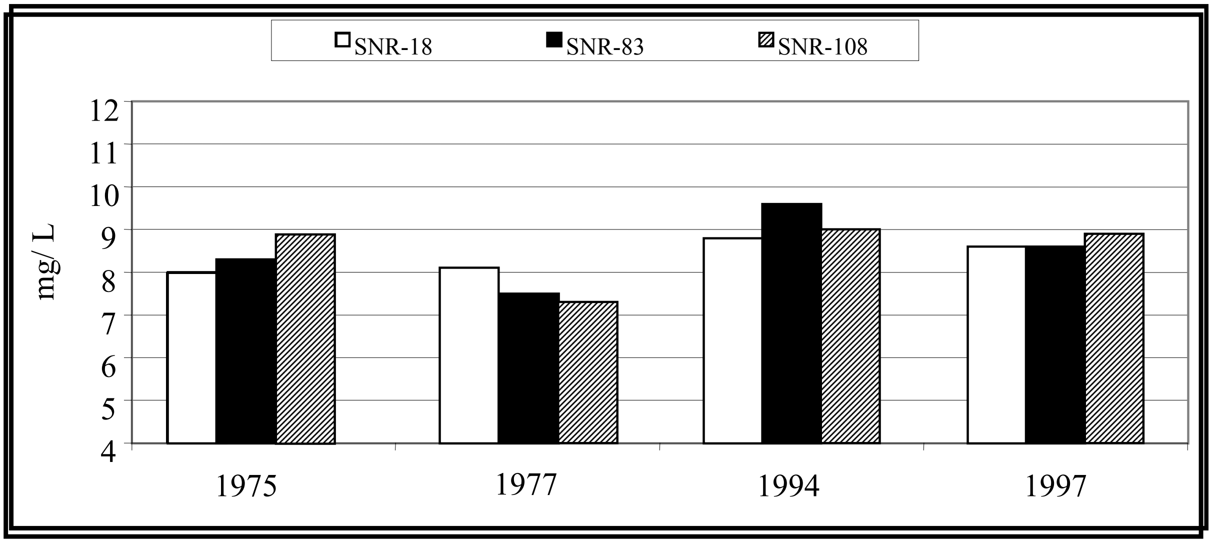


Figure 3-32. Average Water Column Dissolved Oxygen Concentrations at SNR-18, SNR-83, and SNR-108 during June through September of 1975, 1977, 1994, and 1997
Source: Developed by Normandeau

The reasons that the average seasonal values display interannual fluctuation are apparent when some of the vertical profiles from each site are examined. Figures 3-33 through 3-35 show that the dissolved oxygen concentrations throughout the water columns at RM 108, RM 83, and RM 18 were uniform and greater than 8 milligrams per liter at the beginning of June during all 4 years. The July data at RM 18 still displayed constant oxygen distributions that were at, or greater than, the standard (Figure 3-33). Farther upstream at RM 83, the July profiles begin to change. The 1994 and 1997 profiles were still relatively uniform with mean concentrations of 9.9 and 8.4 milligrams per liter, respectively. The 1977 profile still had an average concentration of 8.7 milligrams per liter, but exhibited vertical differences with 9.9 milligrams per liter at the surface and 7.8 milligrams per liter at 28 meters (92 feet). In the next upstream reservoir, these up-and-down differences were more apparent. The July 1977 data showed supersaturated conditions near the surface with a concentration of 11.6 milligrams per liter at 3 meters (10 feet) that was probably due to a high rate of algal photosynthesis. Concentrations, however, progressively declined to 6.1 milligrams per liter at 33 meters. The 1994 data at RM 108 also showed a similar profile. The late-July 1975 and 1997 profiles in the Lower Granite reservoir were very similar, reasonably uniform, and had means of 8.4 and 8.5 milligrams per liter, respectively. August is typically the month when oxygen concentrations have shown increased variability, and deficits, throughout the water columns. A high degree of vertical fluctuation was again shown in 1977 at all three sites during that month. Concentrations at the bottom of the water column at RM 108 were as low as 2.4 milligrams per liter, but rose to 8.3 milligrams per liter near the surface to produce a calculated mean of 6.6 milligrams per liter. Positive heterogrades (oxygen increases below the water surface) were again detected at RM 18 during August 1977 and at RM 108 during 1975 and 1997. The August 1994 data from RM 108 also show the presence of a negative heterograde (a decrease in oxygen concentration at depth). Oxygen concentrations were slightly greater than 10 milligrams per liter at the surface, decreased to 8.2 milligrams per liter in the 6- to 10-meter (20- to 33-foot) interval, rose to 9 milligrams per liter between 15 to 20 meters (49 to 66 feet), and then decreased to 6.8 milligrams per liter at 35 meters (115 feet). September is generally the month when the oxygen profiles resumed a more uniform profile if deviations occurred during the summer. Concentrations can still decrease near the sediment-water interface as they did in 1994 at RM 108 when concentrations approached 2 milligrams per liter near the sediments.

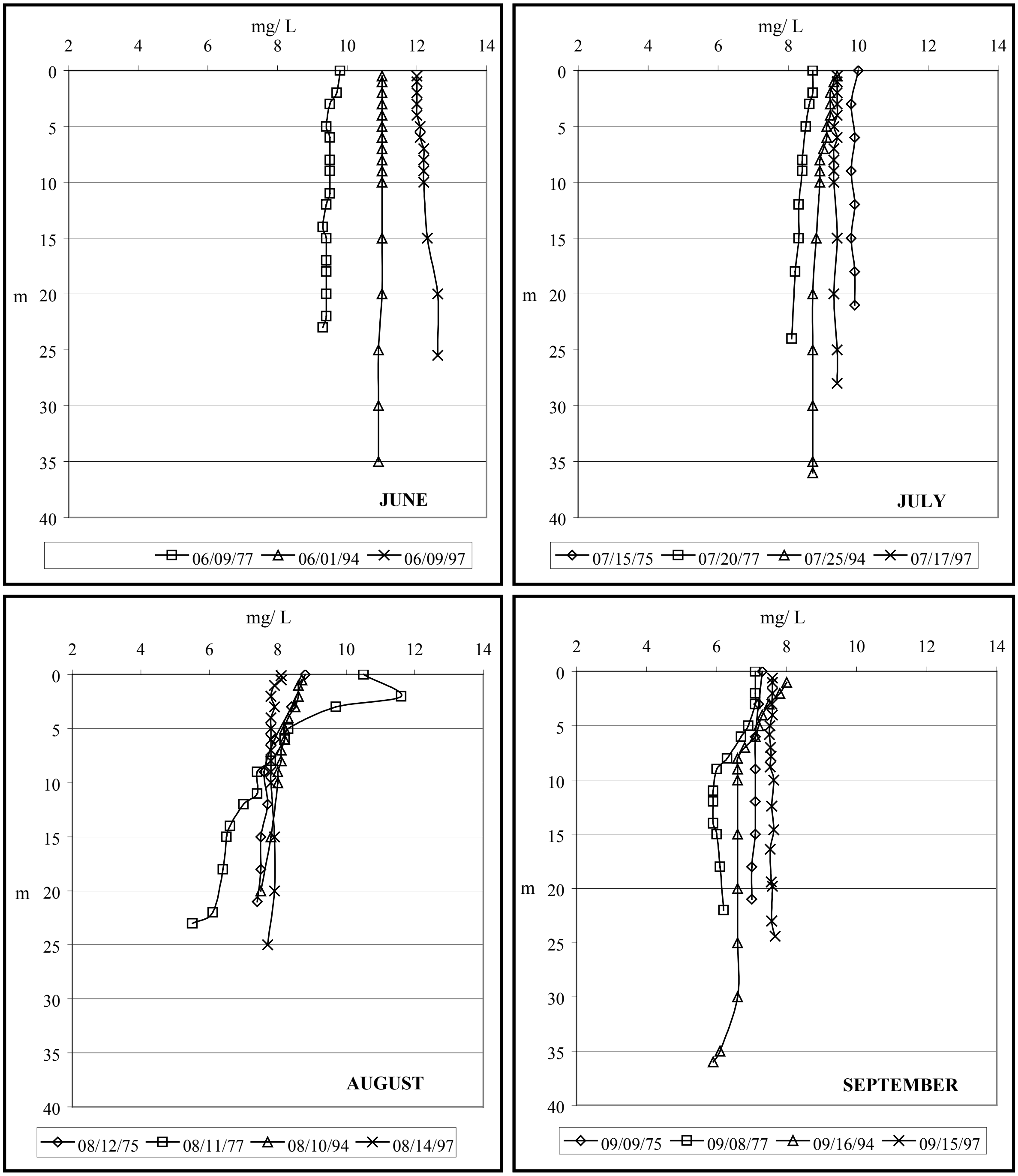


Figure 3-33. Dissolved Oxygen Profiles for the Sampling Station at SNR-18
Source: Developed by Normandeau

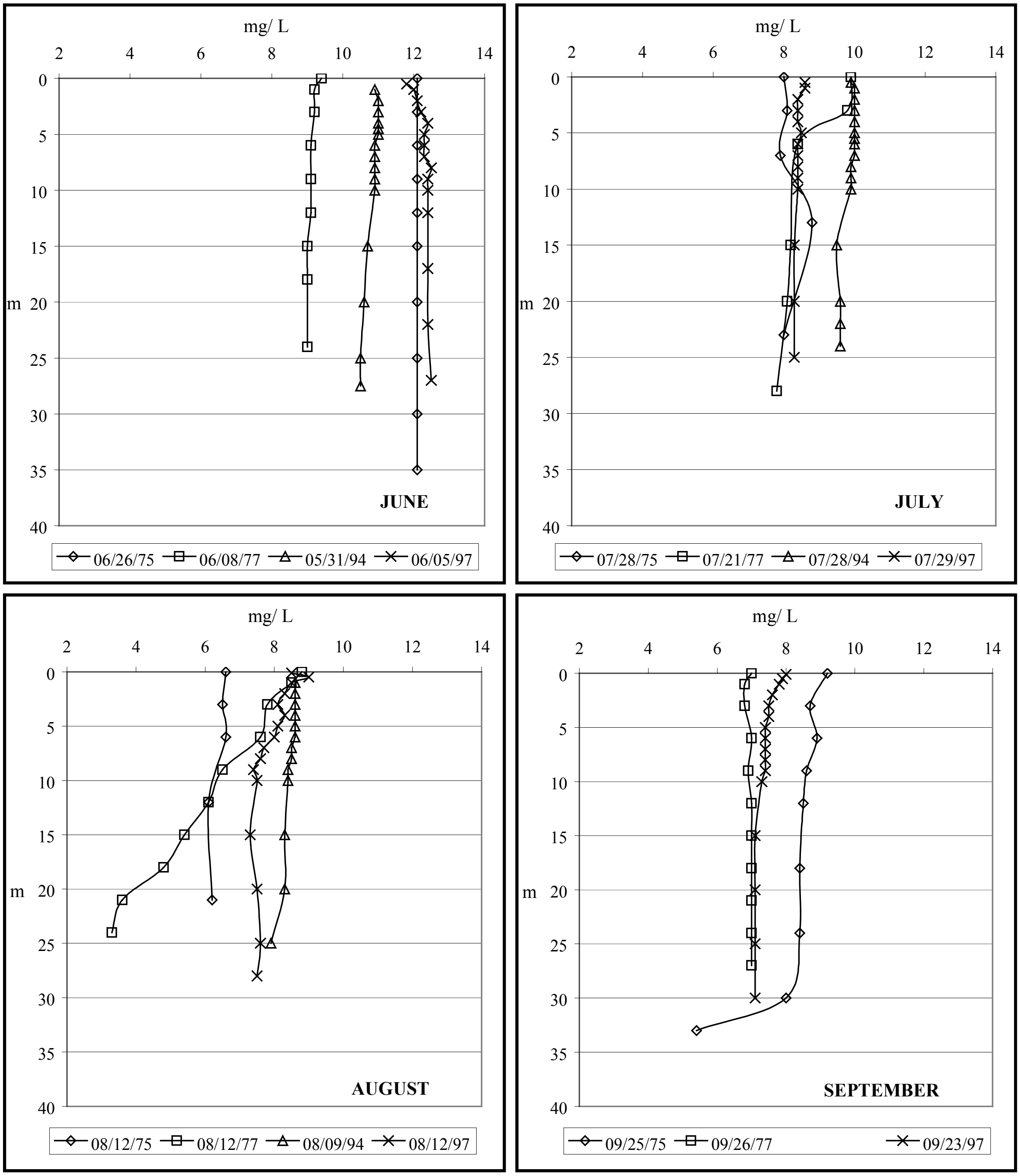


Figure 3-34. Dissolved Oxygen Profiles for the Sampling Station at SNR-83
Source: Developed by Normandeau

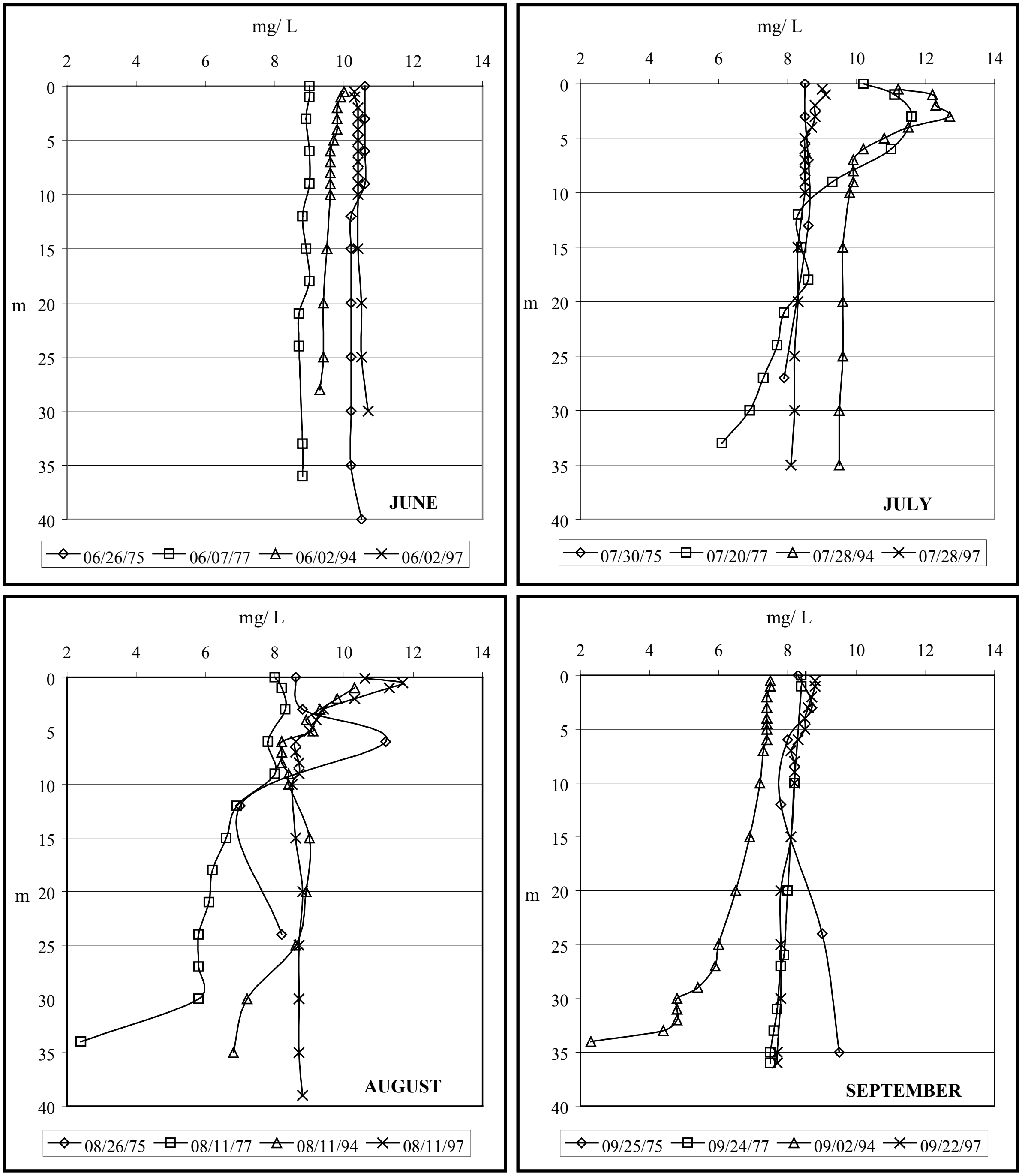


Figure 3-35. Dissolved Oxygen Profiles for the Sampling Station at SNR-108
Source: Developed by Normandeau

Conversely, there have been times when they increased with depth as shown by the 1975 data. Decreasing water temperatures during October, along with greater mixing, increases oxygen concentrations to the levels reached during the spring.

The concentrations of dissolved oxygen in the Snake River above Clarkston, Washington, are also of interest since this constitutes the primary inflow to the reservoirs. The 1976 through 1979 data shown in Figure 3-36 were collected at the USGS gaging station at RM 167.2. Corresponding data for recent years were not available, so the 1994 through 1997 information was gathered from the Ecology station at RM 132.9. Both graphs show the seasonal cycle that is typical in the river; higher concentrations in the winter, decreasing to August lows, and then increasing again as the water temperatures cool. The 1976 through 1979 data set shows more variability than the one for the later period, but this is probably a function of location. The USGS station is in a free-flowing reach while the Ecology site is in the transition zone between the lotic system and the reservoir. It should be noted that the dissolved oxygen concentrations did not go below the 8 milligrams per liter state standard in the 8 years shown, with the exception of April 20, 1978, when a value of 6.2 milligrams per liter was reported.

In contrast to water temperatures, the highest DO concentrations are typically observed during spring runoff and tend to decline with increasing temperature. The USGS data going back to 1975 indicate that low minimum DO levels of 2.3, 4.8, and 5.8 milligrams per liter have been recorded below Lower Granite Lock and Dam (RM 106.5), in the Clearwater River (RM 11.6), and upstream at Weiser, Idaho, respectively.

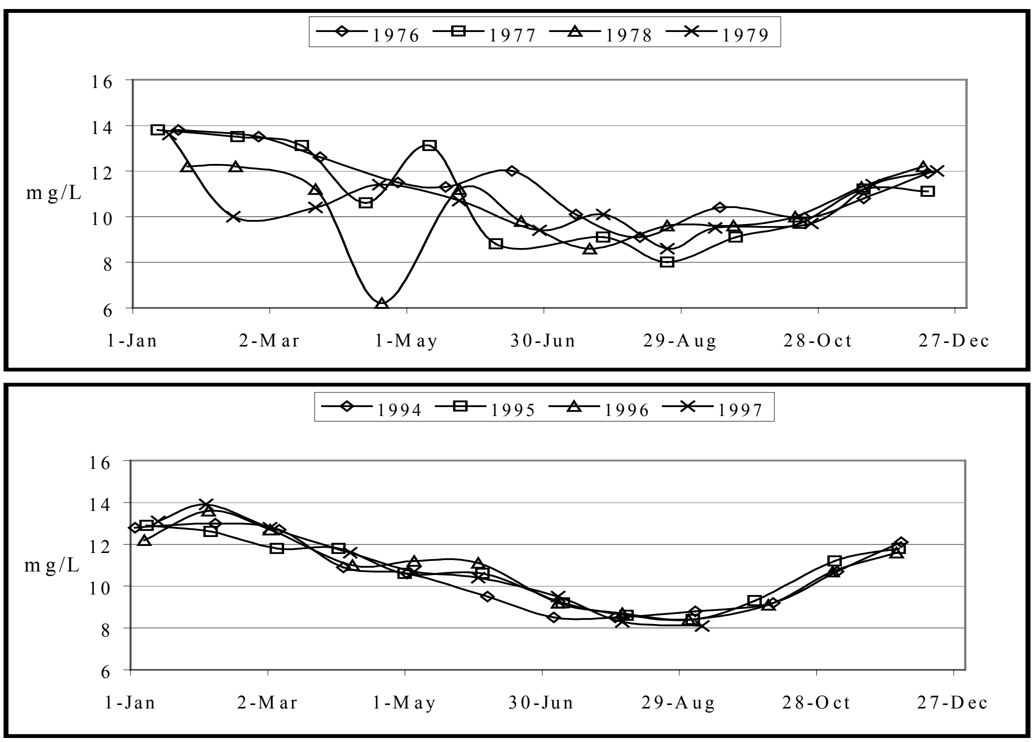


Figure 3-36. Dissolved Oxygen Data Collected at the USGS Gaging Station at SNR-167.2 during 1976-1979 at the Anatone Gaging Station and at the Ecology Monitoring Site at SNR-132.9 during 1994-1997

Source: Developed by Normandeau

Figure 3-37 presents a comparison of DO data throughout the project area for selected stations sampled in 1997. The values represent DO concentrations averaged over the entire water column. For the two Columbia River stations (CLR-326 and CLR-295), DO concentrations were consistently above 8 milligrams per liter over the entire sampling period. The lowest reading was 8.1 milligrams per liter in the bottom waters of the McNary reservoir during the mid-September sampling event, which equals the 90 percent saturation level specified by the Oregon water quality standards. The DO concentrations at all the stations were generally at their lowest levels during mid-September. During the early spill season, April through mid-July, DO levels are maintained by entrainment of air over spillways. As TDG supersaturation occurs, DO concentrations are supplemented.

Figure 3-38 depicts seasonal DO concentrations of surface waters (1-meter [3-foot] depth) collected before construction of Lower Granite Lock and Dam (1970 and 1971) from SNR-107 and post-construction (1995, a normal flow year) from SNR-108. These data indicate that during a free-flowing state prior to construction of the dam, DO concentrations followed a similar seasonal pattern with higher concentrations during the spring and declining concentrations throughout the summer into the fall.

The DO concentrations were slightly higher during the spring in 1971 than in 1995, but reached lower concentrations during late-July and late-August when compared to 1995. Peak water temperatures were measured on July 28, August 23, and September 9, 1971, and these sampling occasions were also three of the four lowest DO readings obtained that year.

At the lower Snake River stations, DO concentrations were for the most part above 8 milligrams per liter during 1997 except during one late-summer event at each station. The timing of the seasonal low level seemed to occur first upstream (SNR-140) and then progressively moved downstream. The average low concentration for these three Snake River stations during this one sampling event was about 7 milligrams per liter. A review of data collected in other years, particularly during the historically low-flow conditions in 1994, reveal only minor differences. During an early September sampling event, at Station SNR-108, the average DO concentration dipped to near 6.8 milligrams per liter but remained above 8 milligrams per liter for all other sampling events.

The lowest oxygen concentrations recorded during 1997 typically occurred during September. At station SNR-40 (not shown), which is in the tailwater section below Lower Monumental Lock and Dam, the lowest water column concentrations ranged from 6.7 to 6.8 milligrams per liter, or less than 75 percent saturation from the surface to the bottom with an overall depth of 8 meters (26 feet). The next downstream station, SNR-18, which is in the Ice Harbor reservoir, had DO concentrations ranging from 6.9 to 7.1 milligrams per liter, or roughly 76 to 79 percent saturation at depths above 20 meters (66 feet) during the same time interval. Readings at the Clearwater River stations (CLW-1 and CLW-11, not shown) were typically well above 8 milligrams per liter and 100-percent saturation, except during mid-September when surface concentrations dropped to 7.5 milligrams per liter or roughly 85 percent saturation at the surface. At the 1.0-meter (3.3-foot) depth, the DO concentration at the same station decreased to 6.3 milligrams per liter, or 71 percent saturation. The DO levels at both Clearwater River stations rebounded to 10 milligrams per liter or more by late September. Despite these relatively low DO values, it is still important to note that the median percent saturation at both Clearwater River sampling sites was over 100 percent during the 1997 growing season, and the analogous values at SNR-18 and SNR-40 were close to 90 percent.

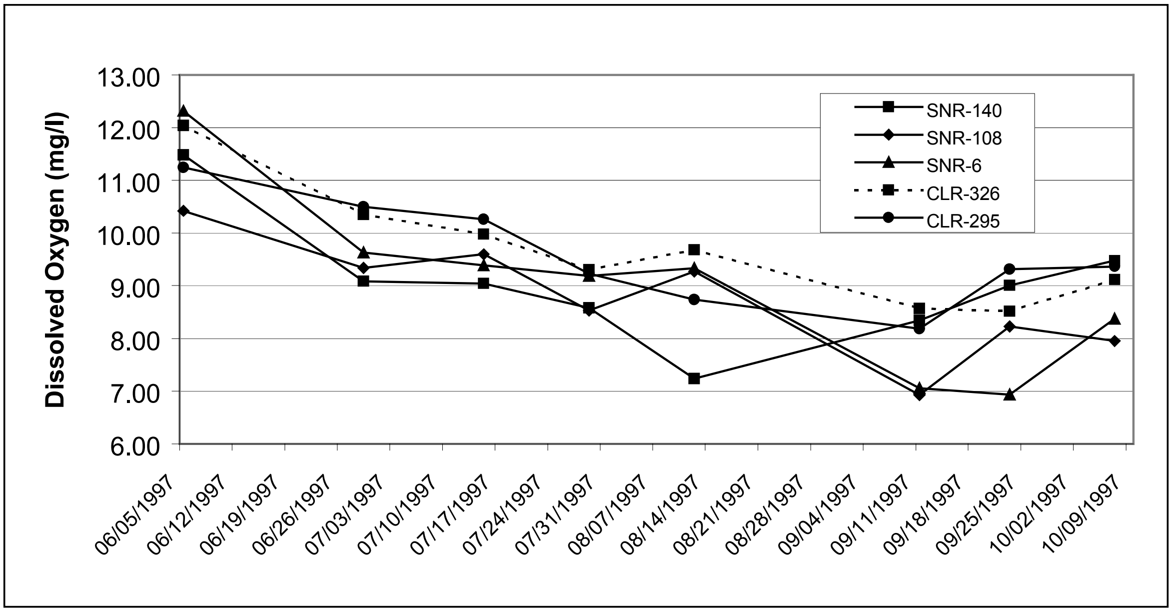


Figure 3-37. Dissolved Oxygen for Select Stations in 1997

Source: Developed by Normandeau

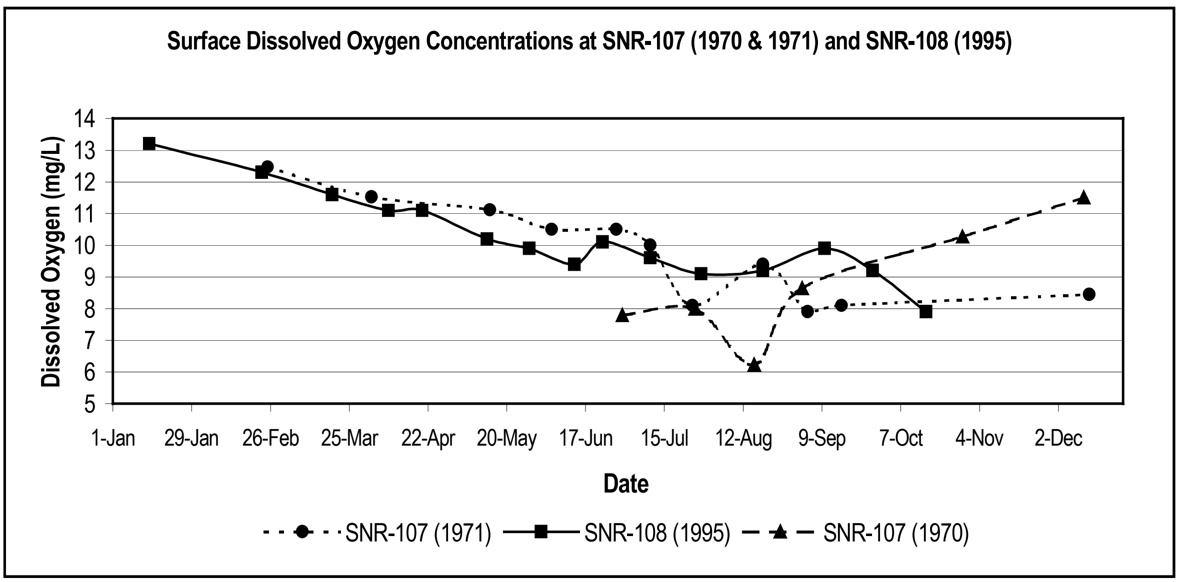


Figure 3-38. Surface Dissolved Oxygen Concentrations at SNR-107 (1970 and 1971) and SNR-108 (1995)

Source: Developed by Normandeau

3.2.4.3 Total Dissolved Gas Supersaturation

Nitrogen, oxygen, and argon compose about 78 percent, 21 percent, and 1 percent, respectively, of the elemental gases in dry air. When the pressure of every gas in the atmosphere reaches equilibrium with its dissolved form in water, the water is saturated. The pressures of gases in the air make up atmospheric pressure, and its counterpart in water is the TDG pressure. If the TDG pressure is greater than atmospheric pressure, the water is supersaturated.

3.2.4.4 Biological Effects of Total Gas Pressure on Fish and Aquatic Biota

Dissolved gas supersaturation (DGS) is an important issue that has received considerable attention in Canada and the United States. The DGS can lead to a physiological condition known as gas bubble trauma (GBT) in aquatic biota. GBT can be harmful or even fatal to aquatic organisms, as demonstrated by a number of significant fish kills in the United States portion of the Columbia and Snake Rivers (Weitkamp and Katz, 1980; Scholtz et al., 1998).

The DGS can produce a variety of physiological signs that are harmful or fatal to fish and other aquatic organisms (Renfro, 1963; Stroud et. al, 1976; Nebeker et al., 1976a, b, and c; Weitkamp and Katz, 1980; Cornacchia and Colt, 1984; Johnson and Katavic, 1984; Gray et al., 1985; Fidler, 1988; White et al., 1991; Fidler and Miller, 1997). As a class, these signs are referred to as GBT or gas bubble disease (GBD).

In general, the major signs of GBT that can cause death or high levels of physiological stress in fish include:

- Bubble formation in the cardiovascular system, causing blockage of blood flow, respiratory gas exchange, and death (Stroud et al., 1976; Weitkamp and Katz, 1980; Fidler, 1988 and 1998a).
- Overinflation and possible rupture of the swim bladder in some species of juvenile (or small) fish, leading to death or problems of overbuoyancy (Shirahata, 1966; Jensen, 1980; Fidler, 1988; Shrimpton et al., 1990a and b).
- Extracorporeal bubble formation in gill lamella of large fish or in the buccal cavity of small fish, leading to blockage of respiratory water flow and death by asphyxiation (Fidler, 1988; Jensen, 1988).
- Sub-dermal emphysema on body surfaces, including the lining of the mouth. Emphysema of the epithelial tissue of the mouth may also contribute to the blockage of respiratory water flow and death by asphyxiation (Fidler, 1988 and White et al., 1991).

Other signs of GBT in fish include exopthalmia and ocular lesions (Blahm et al., 1975; Bouck, 1980; Speare, 1990); bubbles in the intestinal tract (Cornacchia and Colt, 1984); loss of swimming ability (Schiewe, 1974); altered blood chemistry (Newcomb, 1976); and reduced growth (Jensen, 1988; Krise et al., 1990), all of which may compromise the survival of fish exposed to DGS over extended periods.

Each sign of GBT involves the growth of gas bubbles internal and/or external to the animal. For each sign of GBT, there is a threshold that must be exceeded before bubble formation or swim bladder overinflation will begin (Nebeker et al., 1976c; Fidler, 1988; Shrimpton et al., 1990a). Still, the activation of GBT signs is not an easily demonstrated cause-and-effect relationship. This is because bubbles that develop internal to the animal may form in several body compartments simultaneously, disrupting neurological, cardiovascular, respiratory, osmoregulatory, and other physiological functions (Stroud et al., 1976; Weitkamp and Katz, 1980; Fidler, 1988; Shrimpton et al., 1990a and b; Fidler, 1998a). Thus, depending on the total gas pressure (TGP), the depth of fish in the water column, and the duration of exposure, there may be multiple signs present in affected animals.

The GBT may also increase the susceptibility of aquatic organisms to other stressful factors such as bacterial, viral, and fungal infections (Meekin and Turner, 1974; Nebeker et al., 1976b; Weitkamp and Katz, 1980; White et al., 1991). All signs of GBT weaken fish, especially juvenile life stages, thereby increasing their susceptibility to predation (White et al., 1991). Consequently, GBT mortality can result from a variety of both direct and indirect effects caused by DGS.

The DGS can affect all aquatic organisms, including fish, invertebrates, and plants. This may lead to alterations in the food chain structure of an aquatic ecosystem. For example, GBT may increase or decrease the availability of a food source for a particular species (White et al., 1991). This may result in the redistributed populations of species either increasing in abundance through colonization or becoming locally extinct (Brammer, 1991). Changes such as this may affect the whole aquatic ecosystem structure.

When describing and analyzing the effects of DGS on fish, it is important to recognize that the signs and consequences of GBT are determined by the exposure history of the fish coupled to the physiological and environmental response of the animal. That is, the effects are expressed in terms of a classic dose-response relationship. In the case of GBT, the dose must include not only the TGP and temperature, but also the depth at which the exposure takes place (Fidler and Miller, 1997; Fidler, 1998a and b).

Physiological Response and Biological Consequences

Biological monitoring programs have failed to demonstrate a correlation between the external and internal signs of GBT observed in feral fish populations and corresponding levels of mortality. This results in uncertainties as to the effects of DGS on fisheries resources. Specifically, river physical and biological monitoring, including passive integrated transponder tag analyses, along with related research programs have been unable to quantify the acute, chronic, direct, and indirect effects of DGS on feral fish populations. Also, many aspects of DGS have not been adequately studied, such as its chronic effects on fish, and specifically the effects on invertebrates. Nor have any attempts been made to examine the effects of DGS on the overall ecology of aquatic communities.

An important distinction to be made in evaluation of TGP trauma effects is associated with the acute versus chronic effects of GBT. The acute effects of GBT usually occur at a change in pressure ΔP or $\Delta P'$ of 76 millimeters (3 inches) of mercury and greater, while chronic effects occur at a ΔP or $\Delta P'$ of less than 76 millimeters (3 inches) of mercury (Fidler and Miller, 1997). However, it is important to note that the distinction between acute and chronic effects cannot necessarily be associated with a particular sign of GBT or TGP.

The dynamics of the exposure environment, coupled with the behavior and physiology of the aquatic organism, will determine the biological impacts and not the ΔP levels alone. For example, in some situations, intermittent exposures to high ΔP may be relatively unimportant, compared to long-term exposures to lower levels of ΔP. It is necessary to acquire a detailed understanding of the cumulative biological effects of DGS on fish and aquatic biota under both steady-state and dynamic-exposure conditions (Krise, 1993).

Because of the dynamic nature of the exposure conditions, the physiological responses are also highly dynamic. For example, populations of fish that make frequent changes in depth over the exposure period may require much longer times to develop cardiovascular bubbles and mortality in the population than would be indicated from steady-state laboratory bioassay data. Similarly, for fish remaining at constant depth, changes in TGP over the exposure period may result in much longer times to any given level of mortality than would be indicated by steady-state laboratory bioassay data (Fidler, 1998).

Only a few studies have examined the acute effects of GBT in resident feral fish populations of rivers (Hildebrand, 1991; White et al., 1991; Lutz, 1995; Scholtz et al., 1998). These have been limited mostly to the observation of external signs of GBT and the incidence of mortality in resident fish captured in shallow water environments. The GBT signs and mortality have also been observed in resident fish held near the surface in net pens and monitored when TGP concentrations were above 115 to 120 percent (Ebel et al., 1975; Toner and Dawley, 1995; Schrank et al., 1997). As the Independent Scientific Advisory Board (ISAB) (1998) points out:

> *"Natural behavior of resident fish in the wild, such as occupancy of deeper layers, could mitigate the negative effects of supersaturated waters. However many resident fish are obliged to use shallow waters to carry out their life cycles. Early life stages of resident species have been shown to be especially susceptible to GBT above about 110 to 115 percent.* ***.*** *Food organisms such as cladocerans have also shown to develop bubbles in supersaturated water and lose the ability to swim normally. Few invertebrates in the food chain have been monitored adequately, although levels of gas saturation acceptable to fish have been assumed adequate for invertebrates. The monitoring of ecosystem components conducted so far has been inadequate to confidently relate DGS levels believed safe for selected species and ages of migrating salmonids to safety for the mainstem ecosystem as a whole."*

The TDG supersaturation has become a major regional concern due to the rapid decline in salmon runs and their listing as an endangered species. High levels of TDG cause mortality in juvenile and adult migratory fish, resident fish, and other organisms. Literature pertaining to the TDG supersaturation problem dates back to the beginning of the century. Dissolved gas control was a major issue in the Pacific Northwest during the late 1970s, and several of the eight Corps dams were retrofitted with spillway deflectors to reduce the level of high TDG supersaturation. These spillway deflectors were designed for involuntary spillway releases that occur when river discharges exceed powerhouse hydraulic capacities. Recently, operation of the Federal projects has changed with the requirement for voluntary spill to assist fish passage. The TDG supersaturation has resurfaced as a major regional concern due to increased voluntary spill to improve juvenile fish passage.

Several factors affect the fish susceptibility to GBT. Incidence of GBT in fish depends on concentration of supersaturation, duration of exposure to the fish, water temperature, general physical condition of the fish, and swimming depth maintained by the fish (Ebel and Raymond, 1976). Weitkamp and Katz (1980) also suggested that fish tolerance to supersaturation depends on their life history stage and follows two general trends. First, tolerance decreases from very high in the egg stage to very low in older-aged juveniles. Second, tolerance increases following the juvenile life stage, with adults having the most tolerant, free-swimming life stage (assuming avoidance behavior). They also reported that juvenile fish subjected to supersaturation may recover when returned to normal saturation concentrations, but adults do not recover.

The analysis of risk of physiological injury resulting from exposure to TDG supersaturation considers the exposure history of migrants during the whole of their passage through the Federal hydropower system. The reason is that, unlike processes responsible for physical injury that are specific to a particular location, TDG supersaturation, while generated at dams, is propagated throughout the system and constitutes a threat to fish health considerably beyond the point of generation. In addition, downstream transport and mixing processes for TDG supersaturation vary with each project. These variations translate into differences in the downstream environment from project to project. The convolution of the environmental consequences of these physical processes with project-to-project variation in fish passage behavior results in differences in fish health risk between projects with equivalent reductions in TDG supersaturation production. Therefore, the benefit to migrant health of Dissolved Gas Abatement Study (DGAS) alternatives cannot be assessed without considering the whole of the processes constituting the threat to fish health. In the case of TDG supersaturation, this includes the consequences of the transport and dispersion of dissolved gas throughout the hydropower system.

Bubble growth in the vascular system was often cited as a cause of mortality in the Columbia River studies (Weitkamp and Katz, 1980), but other symptoms (extra-corporeal bubbles in the gills and sub-dermal bubbles on the skin and mouth) were often present at the same time (Stroud et al., 1976, Stroud et al., 1975; and Meekin and Turner, 1974). It remains unclear whether the various symptoms act in concert or appear at different TDG concentrations. Fidler and Miller (1994) and White et al. (1991) suggest that physiological and behavioral responses occur concurrent with achieving TDG concentration thresholds at two tiers according to magnitude and duration of exposure. White et al. (1991) tested the results of a literature review analysis of over 1,000 records that suggested a lower mortality threshold occurs at a TDG supersaturation of 110 percent (1.1 atmospheres), while a higher mortality threshold occurs at 115 to 119 percent. Results with adult rainbow trout indicated that the lower threshold (110 percent) is associated with a combination of sub-dermal bubble growth in the mouth and extra-corporeal bubble growth between the gill lamella. The second phase of experiments included intravascular microscopic studies and confirmed the source of mortality for the lower threshold at a TDG supersaturation of 110 to 112 percent. The transition from a lower TDG threshold to an upper threshold involved a shift in the bubble-mechanisms that lead to death. At the lower threshold, sub-dermal bubbles in the mouth lining and extra-corporeal bubbles in the gill lamella act to block the exchange of respiratory gases. At a TDG of 115 percent, the extra-corporeal bubbles were fewer and could even disappear, thus increasing the time to mortality. This is due, possibly, to larger bubbles being dislodged by respiratory water flowing through the gills. At a TDG between 115 and 119 percent, intravascular bubble formation began and time to mortality decreased with increasing TDG concentrations. At these TDG concentrations, subdermal bubbles in the mouth lining were either small or absent. This observation indicates that the examination of external symptoms for GBT in sample fish is not representative of the impact on the population. Apparently, the rapid death caused by intravascular bubbles at these higher TDG concentrations does not allow time for sub-dermal bubbles to develop. This sequence to mortality correlated with data from the literature suggesting that during this transition from the lower to the higher threshold, a range of responses falsely appears to offer relief to the fish if only the external symptoms are evaluated.

Adult salmonids are physiologically more susceptible to the effects of TDG supersaturation than juveniles due to their more developed organs (White et el., 1991). This information is inconsistent with Weitkamp and Katz (1980), who reported that adult salmonids were more tolerant due to their free-swimming life stage. Although not reported, they may have assumed that adults avoid elevated TDG concentrations by sounding. One of the earlier reports on GBT was made by Westgard (1964), who reported that a 119 percent saturation produced blindness in 34 percent of the adult spring chinook salmon in the shallow McNary spawning channel in 1962. Blinded adults had difficulty spawning and had a pre-spawning mortality rate that was 82 percent higher than fish that were not blinded. Blindness and/or cranial blistering (referred to as "head burns" or "scalping") on adult salmonids occurred at Snake River dams immediately following the peak duration of forced spill during the high spring flows of 1993. Bjorn et al. (1994) outfitted and released 1,181 adult spring chinook salmon with radio transmitters at the John Day Lock and Dam and recaptured 255 of these at Lower Granite Lock and Dam. None of the spring chinook had external signs of GBT or "head burns" when captured, tagged, and released from John Day Dam, but when recaptured at Lower Granite Lock and Dam, 24 percent (62 of 255) had some degree of cranial burn after less than 15 days of exposure to supersaturated water. Ultimately, fish vitality of that segment of the passage population seemed compromised, and some of these adult fish may fail to spawn successfully when they reach their sub-basin spawning habitat.

Even when immediate mortality does not occur as a consequence of intermittent or low-concentration supersaturation, the physical condition of fish can become compromised. Accumulated internal damage would logically affect population vitality. No research has conclusively established whether fish actively or instinctively seek deeper water if it is available to avoid higher concentrations of DGS, or if fish that prefer deeper water habitat survive better than fish that inhabit shallow water. Fidler (1985) found that fish physiologically begin to lose control of the regulation of their swim bladder through the pneumatic duct at gas concentrations near 111 percent. Alderdice and Jensen (1985) recommended that, in order to prevent the acute mortality of juvenile salmonids and adult sockeye in the Nechako River, TGPs should be managed below 110 percent. This recommendation was based on their observations, wherein a significant amount of salmonids previously exposed to gas concentrations of 110 to 112 percent of barometric pressure swam within the shallow water depths of high supersaturation when deeper water was provided and directly available (supersaturation at or below 110 percent). The results of Alderdice and Jensen (1985) tend to support the physiological limitation presented by Fidler (1985) and fit within the boundary region estimates between chronic and acute GBT risk (108 to 116 percent TDG) modeled by Jensen et al. (1986).

Finally, Brammer (1991) found that the sensitivity of aquatic invertebrates to high gas saturation concentrations was expressed through increased buoyancy. The buoyancy caused involuntary drift, making these potential food items unavailable to salmonids during rearing or staging. He also observed that when gas supersaturation caused downstream displacement of invertebrates, the upstream migration, before egg laying, appeared to be compensatory. The current lack of invertebrate diversity below the Snake River dams (Bennett et a1., 1994; and Dorband, 1980) may be one ecological impact of hydrosystem operation during early 1979, when high quantities of flow were spilled. Such a base-level impact would have had great compensatory impacts to higher dependent trophic levels, including spawning and rearing salmonids. In the absence of more definitive research, the EPA's water quality criterion of 110 percent TDG supersaturation should serve as a limit for the conservative inriver management of GBT for listed salmonid stocks (Fidler and Miller, 1994). In 1994 and 1995, the states granted interim variances from the 110 percent TDG criteria during the juvenile outmigration. The reason for the variances was to allow far higher spill levels and potentially higher fish passage efficiency levels. The results of this action are not fully developed, but indications of gas bubbles were evident in juveniles during the spill actions.

The health risk of TDG exposure drops considerably as the water quality standard is approached. This results from the combined effect of two factors. The first is reduced risk of GBT upon exposure to TDG supersaturation conditions, and the second is concurrent reduction in the portion of the fish’s habitat at “effective” TDG levels above 110 percent. For example at 120 percent TDG, only fish located in the upper meter, or less, of the water column for extended periods are at any risk from GBT. In addition, the likelihood of developing GBT at TDG levels less than 120 percent appears to be low. The combination of these factors results in the risk to migrating juvenile salmonids of GBT from exposure to TDG supersaturation conditions to be likely very small at TDG supersaturation levels within 10 percent and perhaps as much as 15 to 20 percent of water quality standards. Because of direct linkage between risk and benefit, the low risk of GBT at these levels would also mean that the incremental benefits of gas abatement alternatives are also likely to be small. Initial engineering analyses have indicated that once flow deflectors are installed at Federal projects, the costs of incremental gains in gas reduction are likely to be high. The biological consequences of implementation include, but are not limited to, the potential for physical injury during passage through a gas abatement structure. It is not obvious at this time that the tradeoff between incremental reductions in TDG beyond those achievable with flow deflectors, and the benefits to fish health provided, would offset impacts to fish health during passage through gas abatement structures for some gas abatement alternatives.

Historical and recent research, in addition to observations of the condition of downstream migrants during periods when inriver TDG conditions exceed water quality standards, indicate that the onset, progression, and/or remission of GBD or GBT is a function of the physical dynamics of in-vivo gas bubbles. The data from these studies established that in juvenile chinook salmon and rainbow trout exposed to DGS, there is a long period during which nucleation site growth dominates the time to initiation of mortality in populations of these fish. This time can account for up to 80 percent of the total time to mortality at a TGP level of 140 percent and a water temperature of 15°C (59°F). The actual growth of tubular bubbles in gill filamental arteries accounts for only a small portion of the total exposure time to mortality.

The experiments also established that at a TGP level of 140 percent and a water temperature of 15°C (59°F) bubbles begin to develop in the hearts of juvenile chinook salmon and adult rainbow trout at about the same time they appear in the gill filamental arteries. The heart bubbles appear to develop in the sinus venosus and atrium and are ejected into the bulbous arteriosus where they can be seen visually or when the bulbous arteriosus is punctured. However, most of the gill filamental arteries of these fish fill rapidly with tubular bubbles just before death, giving a clear indication of their primary involvement in the mortality. The heart bubbles may contribute by way of reduced system pressures in the gill filamental arteries. This would be caused by the blockage of gill branchial arteries by bubbles from the heart. Also, at death, there are small bubbles in the posterior cardinal veins of some fish. However, these bubbles appear to develop late in the exposure period and are not likely to affect bubble growth processes in the heart of gill filamental arteries.

These studies also examined bubble growth in the heart and gill filamental arteries at other dissolved gas concentrations (percent TDG, percent TGP) and water temperatures as well as the response of vascular system bubbles to increasing hydrostatic pressures. The initial physiological response of fish to exposure to uncompensated TDG supersaturation conditions is the formation of bubbles in the fish’s vascular system followed by a period of growth which, depending on several factors, can eventually lead to development of GBT symptoms and death. Following the onset of development of GBT symptoms, the evolution of GBT is almost totally dependent on the depth of the fish relative to the level of TDG supersaturation. A small change in depth could mean the difference between the continued development of GBT symptoms or reduction in symptoms. The interplay between existing TDG supersaturation conditions, development of GBT symptoms, and variation in hydrostatic pressure resulting from changes in fish depth during normal migratory behavior affects fish mortality under migratory behavior conditions.

Uncompensated TDG supersaturation is that portion of TGP in excess of atmospheric level that is not compensated by hydrostatic pressure at the depth of the fish. Approximately 10 percent of TGP above 100 percent (atmospheric level) is compensated by each meter of depth. For example, a fish acclimated to a TDG pressure of 120 percent will not experience a gradient across its membranes and other tissues at depths of 2 meters (6 feet) or greater. Therefore, the physical conditions will not be present that would permit dissolved gas to come out of solution and begin the formation of bubbles.

Recent studies completed at Battelle’s Pacific Northwest National Laboratory studied juvenile salmonids and bluegills acclimated to TDG from 120 percent to 135 percent at a pressure of 15 pounds per square inch gage (equivalent to a depth of approximately 10 meters [33 feet]) and then exposed them to a pressure cycle typical of that juvenile fish experience when passed through hydroturbines in the Federal hydropower system (personal communication, Scott Abernethy, Pacific Northwest National Laboratory, July 2000). Necropsies performed on fish that died from exposure by this scenario showed ruptured swim bladders and bubbles in the heart, gills, and other internal organs. Injuries observed were typical for fish exposed to uncompensated levels of TDG supersaturation. This finding, if substantiated, could have implications for management of TDG for all hydroelectric projects in the Pacific Northwest.

Spill at the run-of-river dams on the lower Snake and Columbia rivers causes TDG supersaturation that frequently exceeds 110 percent of barometric pressure. Water passing through the spillways of the dams entrains air bubbles as it passes under the gates, flows over the spillway, and plunges into the stilling basin. Hydrostatic pressure forces the air bubbles into solution, thus raising TDG concentration in the water. As a convenience, dissolved gas pressures may be expressed as a percentage of barometric pressure (percentage of saturation). The TDG supersaturation is often mislabeled as "nitrogen supersaturation" because air is composed mostly of nitrogen, and nitrogen was believed to be the only gas that caused problems. While nitrogen does speed the problems of GBT, all of the dissolved gases in air participate in the process.

Variables that may determine dissolved gas concentrations on run-of-river dams include: 1) the total amount of spill; 2) the amount of spill per spillway bay; 3) the presence and effectiveness of spillway deflectors; 4) dissolved gas concentrations in the forebay; 5) water temperature; and; 6) the depth of the stilling basin relative to the tailwater elevation (i.e., the depth of spill plunge). While relationships among all of these variables have been hypothesized (Roesner and Norton, 1971), the significance of several variables is unknown at this point. However, it is known that spill volume and tailwater elevation are very significant factors and, therefore, are important in determining operational strategies.

Mathematical relationships, including simple linear regressions and those used in mechanistic computer models (e.g., GASSPILL), use spill discharge to explain most of the variation in TDG concentrations in a specific tailrace. Also important to note is that TDG in the reservoirs can be reduced by wind/wave action, and by shallow tailrace conditions. Other factors determining TDG supersaturation include total river discharge and downstream mixing of powerhouse and spillway discharges. In addition, water temperature and pressure are both important factors in determining gas solubility. For example, increasing the temperature of water decreases the volume of gas it will hold at equilibrium. Therefore, an increase in water temperature alone will produce supersaturation in water that is initially saturated (Colt, 1984). Similarly, pressure increases rapidly with depth as a result of greater hydrostatic pressures. This increase in hydrostatic pressure greatly enhances the capacity of deeper water to dissolve and retain gases in solution.

Dams slow and quiet the water, preventing the dissolved gases from equilibrating with the atmospheric air between dams. Consequently, supersaturated conditions can persist and accumulate over extended distances (Corps, 1992). Ebel (1969) reported that no equilibrium occurred between the four lower Columbia River System Dams. The problem dwells within the physical properties of various gases and water. At the project spillways, the depth of the plunge pool forces the entrained air into solution due to the increased surface area and hydrostatic pressure. With a low diffusion pressure and a low surface-to-volume ratio, the dissolved gas is slow to equilibrate with the atmosphere.

The operation of a powerhouse allows reductions in the amount of spill and may reduce TDG concentrations by diluting the higher dissolved gases created by spillway operations. However, as spill volumes increase, the dissolved gas levels downstream consistently increase. As the river flow passes each of the lower Snake and Columbia river dams, sequential spill will cause the level of dissolved gas in the river to be incrementally and cumulatively increased. The relationships among tailwater TDG levels, forebay TDG levels, spill discharge, and forebay TDG levels are known to be transmitted through the powerhouse. Even if forebay TDG is not significantly increased over the spillway, the reduced dilution provided by high-TDG powerhouse flows would result in accumulated TDG levels as the water flows down the system.

3.2.4.5 Existing Measures for Improved Salmonid Survivability

Several measures have been implemented within the project area to improve the downstream migration and survivability of juvenile salmonids. Among these measures are voluntary spillway releases, installation of flow deflectors, other spillway modifications, and transportation system improvements. These measures are discussed below.

Two major areas concerning fish survival relative to spillway passage need to be considered when evaluating the DGAS alternatives. The goals and objectives of the DGAS program are directed at reducing TDG concentrations caused by spilling at the lower Snake and Columbia river dams. By reducing concentrations of TDG, juvenile and adult salmonid survival in the system may increase. Potential alternatives were selected primarily based on their ability to reduce gas from existing conditions. Effects of spillway modifications include immediate losses specifically associated with passage through the spillway and stilling basin, whereas indirect mortality relates to factors causing death over a longer period of time. Bell and DeLacy (1972) and Ruggles and Murray (1983) comprehensively reviewed the relevant information to describe the factors affecting fish survival related to passage through spillways. They described seven ways in which fish may be injured at the spillways:

- rapid pressure change
- rapid deceleration
- shearing effects
- turbulence
- shearing force of fish on the water surface
- scraping and abrasion
- length of time juveniles spend in highly turbulent water.

Little information is available that describes the relative importance of each of these factors on fish survival. Bell and DeLacy (1972) and Ruggles and Murray (1983) reviewed the factors on fish response to fish impacts on structures, abrasion on rough concrete faces, various forces associated with deceleration, and the types of injuries fish sustained. Both reports described different spillway and stilling basin types and suggested important criteria that should be evaluated to minimize juvenile fish mortality.

Spillway Release

The Corps has been releasing water from the Columbia and lower Snake River projects as requested by NMFS Biological Opinion. These special spillway releases have been ongoing since 1994 and typically occur during the migration season from March through August. The volume of released water consists of up to 100 percent of the total river discharge. The specific requirements for the water releases for fish passage are spelled out in the NMFS Biological Opinion (1995 Biological Opinion and 1998 Biological Opinion). The start and end dates of this voluntary spill were determined by the Technical Management Team (TMT) based on seasonal monitoring information. Planning dates for the spring spill are April 3 to June 20 in the lower Snake River. Within the facility area, a planned summer spill between June 21 and

August 31 is required only at Ice Harbor Lock and Dam. Spill periods are for 24 hours a day at Ice Harbor Lock and Dam, and from 1800 to 0600 at Lower Monumental, Little Goose, and Lower Granite Locks and Dams.

The 1998 Biological Opinion also requires spring spill at all three Snake River collector facilities (Lower Monumental, Little Goose, and Lower Granite Locks and Dams) outside of the time windows mentioned above, "when seasonal average flows are projected to meet or exceed 85 kcfs." In addition, the 2000 Biological Opinion requires spilling directly up to spill discharge caps that correspond to the 120 percent TDG level below the spilling facilities. The spill discharge caps are set to not exceed 120 percent TDG.

Spillway Flow Deflectors

The spillway flow deflectors on the lower Snake River facilities are submerged flip-lips jutting out from the spillway faces, which force spilled water to skim over the surface of the tailwaters instead of plunging deep into the stilling basin. By minimizing the plunging of water into the stilling basin, the generation of TDG supersaturation is also minimized. Originally, spillway deflectors were designed to reduce TDG levels under forced spill conditions when tailwater elevations are high, as in the case of involuntary spills (e.g., when the river flow exceeds the capacity of the powerhouse). The fish passage spill program of the 1990s operates under varying tailwater elevations, including periods when the tailwaters are too low for the spillway deflectors to be effective. Consequently, the spillway discharge overrides the deflectors and plunges deep into the stilling basins, causing TDG to increase.

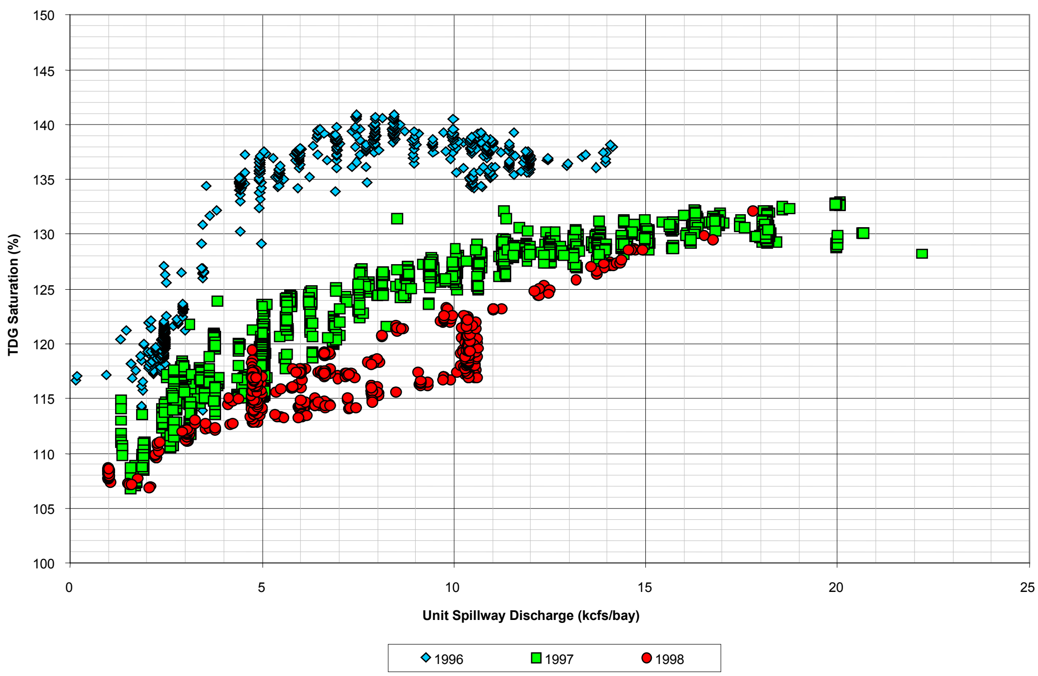
Spillway deflectors were installed in the 1970s at five of the facilities with varying degrees of success in TDG reduction. They were designed for optimal performance under conditions of large involuntary releases associated with high-flow events in the spring. The elevations of the deflectors were determined based on the tailwater elevations associated with those flood events, and, therefore, their performance under other tailwater conditions is not nearly as effective. Another parameter involved with the design of the deflectors is the amount of flow over the deflector. Generally, the deflectors perform adequately up to a point where the flow is able to override the deflector and establish a plunging flow in the stilling basin. At the time they were designed and installed, the spillway deflectors were viewed as an interim measure that would improve the TDG performance of the spillways until storage capacity and powerhouse capacity could be increased to greatly reduce the need to utilize spillways to pass spring flows. Beginning in the late 1980s, voluntary spillway releases were made for the improvement of juvenile fish passage. As a result, TDG levels increased again because the facility operations are not within the design range of the spillway deflectors.

The effectiveness of a flow deflector will improve if it can be designed to perform over a wider range of spill discharge and tailwater fluctuations. The ideal deflector generates a smooth, stable skimming flow across the water surface of the stilling basin. However, the hydraulic performance of existing deflectors is limited to a narrow range of tailwater elevations and unit spill discharges. The deflectors recently constructed at Ice Harbor Lock and Dam appear to perform better than other facilities in terms of gas production versus spill discharge. The new deflectors are 3.8 meters (12.5 feet) long with a 4.6-meter (15-foot) radius transition (curved surface from the spillway face to the horizontal surface of the deflector) and are set at an elevation that provides optimal performance during the more typical facility operations under the current voluntary spill program. Lower Granite and one deflector out of six at Lower Monumental were also constructed with a 4.6-meter (15-foot) radius. In contrast, the deflectors at Little Goose and McNary Locks and Dams do not have a radius fillet. Model studies and prototype evaluations indicate deflectors with a radius transition generate a smoother, more stable surface jet. The pier walls between spillbays at Ice Harbor Lock and Dam were also extended to the end of the deflectors.

These modifications are relatively low-cost and could provide some benefits by reducing TDG production.

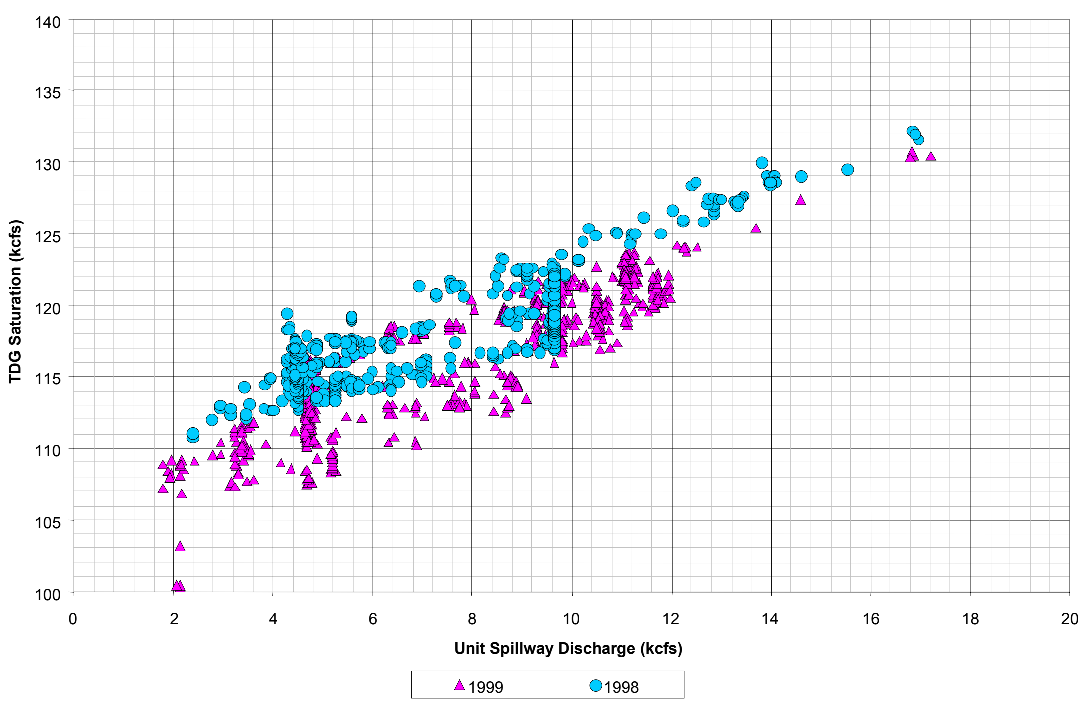
Figure 3-39 shows the percent decrease in TDG saturation per unit spillway discharge (kcfs/bay) with the addition of flow deflectors at Ice Harbor Lock and Dam for 1996 through 1998. Prior to 1996, Ice Harbor Lock and Dam was factored out as the most spill-limited dam on the lower Snake River and the dam with the highest production of percent TDG for either voluntary spill-for-fish and involuntary spill. These limits were partially attributable to the lack of structures in the 10 spillways at Ice Harbor to reduce TDG production, such as flow deflectors. The most expedient method to increase the spill volume at Ice Harbor to meet the NMFS request of 120 percent TDG was to design and construct flow deflectors in as many spillbays as deemed necessary without compromising good tailwater hydraulic conditions for adult salmon entrance into the ladders. In the winter of 1996/1997, flow deflectors were scheduled to be installed in 8 of the 10 spillbays, but high flows in January terminated construction, with deflectors installed in only 4 spillways, and leaving 2 additional spillbays inoperable. The configuration of four spillbays with deflectors, two spillbays inoperable, and four spillbays without deflectors in 1997 increased the spill cap for 120 percent TDG to around 45 kcfs from around 25 kcfs without any spillbays with deflectors prior to 1996. The spill cap for the Federal and state water quality standard of 110 percent changed to around 11 kcfs in 1997 from around 5 kcfs prior to 1996 with very little change in spill pattern for adult attraction to the ladders. In 1998, the original construction for deflectors in 8 of the 10 spillbays was completed and resulted in increasing the spill cap for 120 percent TDG to around 75 kcfs by flattening out the TDG production curve. The spill cap for the Federal and state water quality standard of 110 percent changed to around 19 kcfs in 1998 from around 11 kcfs in 1997 with very little change in spill pattern for adult attraction to the ladders. In 1999, construction to complete deflectors in all 10 spillbays resulted in increasing the spill cap for 120-percent TDG to around 90 to 110 kcfs by flattening out the TDG production curve even further (Figure 3-40). The spill cap for the Federal and state water quality standard of 110 percent changed to around 25 to 30 kcfs in 1999 from around 19 kcfs in 1998 with very little change in spill pattern for adult attraction to the ladders. Spill up to 100 percent of the flow when flow was below 110 kcfs during the night was tested beginning in 1999 for percent TDG and tailwater egress of smolts. The Ice Harbor Lock and Dam stilling basin is the only spillway structure on the lower Snake River that has baffle blocks for hydraulic energy dissipation, and, hence, increased potential for incidence of salmonid smolt physical injury and mortality beyond the 2 percent generally assumed for other spillways on the lower Snake River. Spillway passage survival and/or adult passage delay under high spills exceeding 100 kcfs or 100 percent of river flow has not been adequately monitored or tested to date in order to assess the actual acute or cumulative risk to salmonid passage survival.

Spillway elevations are set to provide ideal skimming flow conditions for target spill discharges and associated tailwater elevations. Since the construction of flow deflectors on Snake and Columbia River facilities in the 1970s, system-wide operations of Corps dams have changed. The flow deflectors were designed to reduce peak TDG concentrations generated by high involuntary spill releases when total river flows exceed powerhouse capacity. Current operations of the Corps facilities require a portion of the total river flow to be spilled for juvenile fish passage. The ratio of spill to powerhouse flows is in the biological opinion. The biological opinion also states that Corps reservoirs are to be operated at minimum operating pool (MOP). The MOP, combined with the requirement of spill flows at much lower total river flows, reduces the submergence and compromises the hydraulic performance of the deflector.



Note: No deflectors in 1996, 4 of 10 installed in 1997, 8 of 10 installed in 1998.

Figure 3-39. Total Dissolved Gas Measured Below Ice Harbor Dam, 1996 to 1998
Source: Developed by the Corps



Note: No deflectors in 1996, 4 of 10 installed in 1997, 8 of 10

Figure 3-40. Total Dissolved Gas Measured Below Ice Harbor Dam, 1998 to 1999
Source: Developed by the Corps

Facility-specific operations, for a design range of total river flows, must be established to optimize the deflector elevation. Given the percent-spill requirement and design range of total river flow, the tailwater elevations and unit spill discharges are easily ideal submergence and deflector elevation can then be determined from physical spillway model studies and prototype evaluations.

The Little Goose and Lower Monumental spillways have deflectors on six of the eight spillbays, with the potential for the future addition of deflectors on the outer spillbays (spillbays 1 and 8). Though these end-bay deflectors were not installed due to adult fish passage concerns, it is now believed that adding the end-bay deflectors may further reduce TDG levels without adverse impacts on adult fish passage.

Spillway Deflector Modifications

Modifications to spillway deflectors have been implemented as a means of further reducing TDG concentrations in the lower Snake River. Among the modifications used are pier extensions, which were added to the flow deflectors at Ice Harbor Lock and Dam. These pier extensions extend the downstream face of the existing piers flush to the downstream edge of the flow deflector and prevent the sidewall flow from directly impacting the flow deflector and plunging into the basin. The sidewall flow rises from the corners of the spillway gates and rides 1.8 to 2.4 meters (6 to 8 feet) above the surface of the spillway discharge jet. As the sidewall flow reaches the end of the pier walls, it expands abruptly. The two jets, one from each side of the wall, converge. The lower portion of the combined jet impacts the exposed section of the deflector immediately below the pier. The upper portion reaches beyond the deflector and plunges into the stilling basin. The extension forces the expansion of sidewall flow to occur further out away from the deflector, where the flow becomes intercepted by the much more dominant deflected surface flow, preventing it from plunging into the basin.

3.2.4.6 Dissolved Gas Abatement Study

The DGAS is a part of the Columbia River Fish Mitigation Program. The DGAS is in response to the 1995 NMFS Biological Opinion on Operation of the Federal Columbia River Power System. The goal of the DGAS is to identify means to reduce TDG at the eight Corps facilities on the Lower Snake and Columbia Rivers to the extent economically, technically, and biologically feasible. To date, gas abatement alternatives have been identified and evaluated for potential application at the eight study facilities. Additionally, numerical modeling tools have been developed to help evaluate the complex issues related to gas abatement through more than 300 miles of river. The next step for DGAS is to evaluate the alternatives and potential implementation scenarios using the numerical modeling tools.

The DGAS includes two parts: a Phase I reconnaissance level report and a Phase II feasibility level report. The Phase I reconnaissance level report was completed in April 1996 (Corps, 1996). The Phase II report will present and document the results of the DGAS Phase II feasibility-level investigations. The 60 percent draft report presents the status of major components of the DGAS as of December 1998. Most of the background information surrounding dissolved gas issues is discussed in the April 1996 DGAS Phase I Technical Report. This document continues where the Phase I Report ended and will be finalized when the study is complete in the year 2001.

The Phase II gas abatement effort includes a complex system-wide evaluation of alternatives and can be broken into five main tasks: alternative investigations, prototype construction, numerical modeling, biological research, and water quality research. Alternative investigations include the design and localized evaluation of alternatives. Prototype construction includes the construction and testing of an alternative to validate assumptions and estimates developed in the alternative investigations. Numerical modeling includes the development and use of numerical modeling tools to evaluate biological and water quality benefits of gas abatement alternatives throughout the river system. Biological research includes field and laboratory research required to validate assumptions made in the design of alternatives and to calibrate and validate the numerical modeling tools. Water quality research includes field research required to investigate gas production of the existing structures and the alternatives and to investigate transport and mixing characteristics of the river system as needed to develop the numerical modeling tools. Additionally, the program's review by the ISAB is also summarized, as this review played a significant role in changing the scope of this study.

3.2.4.7 Total Suspended Solids

Table 3-7 describes the mean TSS concentrations (milligrams per liter) and 95-percent confidence intervals for a select number of sample sites within the project area, covering a period of up to 24 years. These data indicate that in most cases the average concentrations for each site either increased slightly or were relatively similar. However, it is noteworthy that the 18 milligrams per liter mean determined for SNR-148 in 1997 was due, in large part, to the 65 milligrams per liter value that was observed during early June when runoff was close to maximum; otherwise, concentrations were less than 10 milligrams per liter most of the time. In the free-flowing reach of the Clearwater River (CLW-1), average concentrations have decreased from 15 milligrams per liter to 4 milligrams per liter between 1976 and 1997. Information for the years between 1976 and 1997 is not present in the current database, but the trend was similar to the one observed in the free-flowing Snake River. Similarly, the 1997 mean value of 20 milligrams per liter at SNR-83 was elevated as a result of the 70 milligrams per liter concentration determined at 1 meter (3 feet) on June 29 and due to a near-surface film; the concentration at 12.5 meters (41.0 feet) was only 13 milligrams per liter.

It is generally thought that larger particles transported by the rivers settle out in the transition zone in the vicinity of Lewiston, Idaho, and downstream into the Lower Granite reservoir. Finer material that passes the Lower Granite reservoir remains suspended. As such, the data suggest that there may have been a decrease in the larger fraction of the suspended solids transported by the in-flowing rivers, yet the amount of fines that travel down through the series of dams has remained about the same. Occasionally, elevated concentrations near the surface occur in the reservoirs as a result of localized algal blooms, port operations, and tributaries.

Typically, TSS concentrations are highest during the spring freshet and then decline as flows diminish through late summer and into the fall. Figure 3-41 presents the 1997 observed TSS levels averaged over depth for selected stations within the project area. The highest TSS levels were observed during the early June sampling event and then dramatically decreased with most stations having less than 10 milligrams per liter for the remainder of the season. Based on the measured data, the upstream Columbia River stations (CLR-369 and CLR-397) and the Clearwater River stations (CLW-1 and CLW-11) had the lowest levels in the study area, with maximum levels only reaching 164 milligrams per liter and seasonal medians of 2 milligrams per liter. Below the Snake River confluence, in the McNary reservoir, peak levels generally ranged from 21 to 32 milligrams per liter at Station CLR-295. At Station CLR-306 (not shown) one particularly high readings of 40 milligrams per liter was measured at a depth of 20 meters in mid-July. During this same event, the corresponding turbidity level was 6 NTUs and the TSS levels at the 1- and 10-meter (3.3- and 33-foot) depths were both 6 milligrams per liter. The 1997 data showed that the highest TSS levels were frequently found at the 20-meter (66-foot) depth or greater and these concentrations were often as much 10 milligrams per liter or more than those measured at the 1-meter (3.3-foot) depth.

Table 3-7. Average and 95 Percent Confidence Intervals for Growing Season TSS Concentrations (milligrams per liter) at 1 Meter for Selected Sampling Sites and Years

	1975		1976		1977		1994		1995		1996		1997	
Site	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI
SNR-18										3		3		4
	ND	ND	ND	ND	ND	ND	ND	ND	10		11		9	
										17		20		14
SNR-83				4		8				0		4		(2)
	ND	ND	8		14		ND	ND	6		10		20	
				12		21				12		16		41
SNR-108				2		7				3		2		3
	ND	ND	6		11		ND	ND	4		8		8	
				10		15				6		13		14
SNR-118										2				< 1
	ND	ND	ND	ND	ND	ND	ND	ND	6		ND	ND	10	
										9				20
SNR-129				3		9				3		2		1
	ND	ND	24		14		ND	ND	7		15		9	
				46		19				11		29		16
SNR-148				9		20								(2)
	ND	ND	19		27		ND	ND	ND	ND	ND	ND	18	
				30		34								37
CLW-1				4		3								0
	ND	ND	15		8		ND	ND	ND	ND	ND	ND	4	
				27		13								8

Note: ND=not detected
Source: Developed by Normandeau

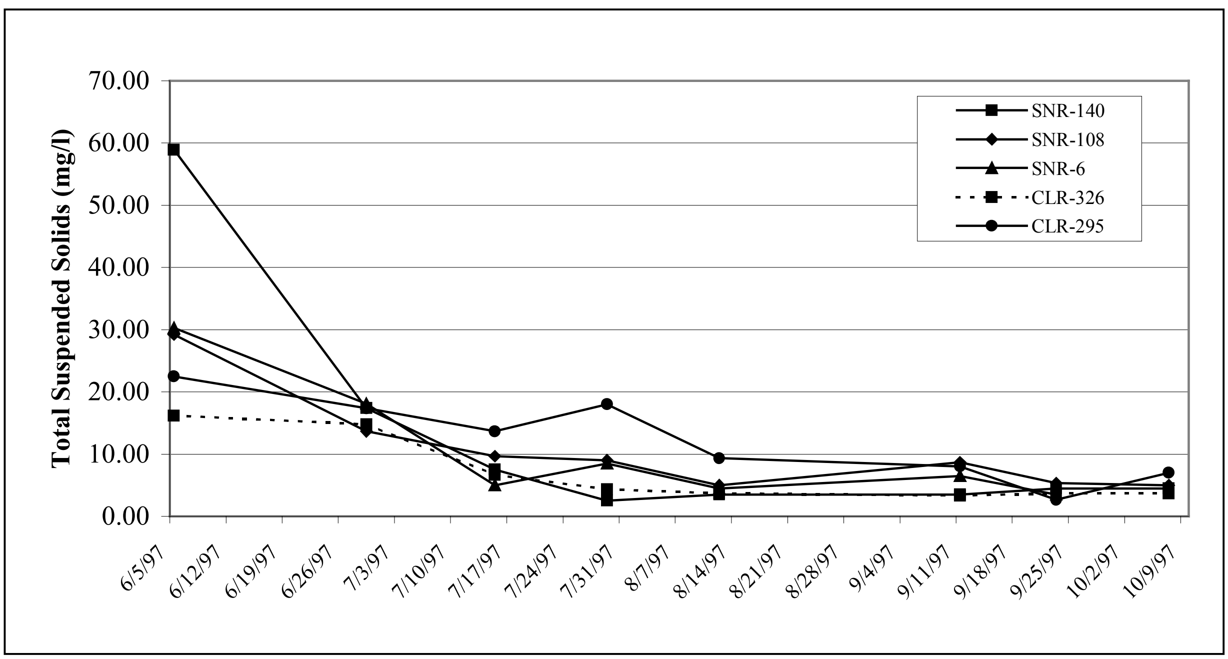


Figure 3-41. The TSS for Selected Sampling Stations in 1997
Source: Developed by Normandeau

Within the lower Snake River, the upstream station (Station SNR-140) had peak TSS levels of 60 and 65 milligrams per liter at the surface and bottom depths, respectively. Station SNR-129, at the uppermost portion of the Lower Granite reservoir, appeared to have the highest peak level of 72 milligrams per liter at a depth of 20 meters (66 feet) and an average concentration of nearly 50 milligrams per liter throughout the water column. Discharge in the lower Snake River was around 175 kcfs at the time, and the average TSS level throughout the remaining portions of the system was about 30 milligrams per liter with no distinct differences between the impounded and nonimpounded reaches. Again, the highest levels were generally observed at the greater sampling depths. By the June 29 sampling date, the average TSS level declined to just below 20 milligrams per liter, except at Station SNR-83 (not shown), which had an unusually high level of 70 milligrams per liter at the surface and much lower levels below. This high TSS level was likely the result of patchy conditions that often occur on the reservoirs. For the remainder of the sampling season, TSS levels were consistently below 20 milligrams per liter and most often below 10 milligrams per liter.

Peak TSS levels in the Palouse and Tucannon Rivers were generally much higher during 1997 than in the mainstem. The Palouse River had a particularly high concentration of 1,035 milligrams per liter in early June as compared to a seasonal low level of 13 milligrams per liter. The Tucannon River recorded the second-highest level of 130 milligrams per liter during the spring runoff period (early to mid-June) but levels remained below 10 milligrams per liter after mid-August. The high TSS levels in these two tributaries appeared to have little effect on the observed levels in the lower Snake River. The difference in TSS levels observed at stations SNR-40 and SNR-50, which are located downstream of these tributary inputs, were typically less than 5 milligrams per liter and the peak downstream level was no more than 26 milligrams per liter.

There are no state water quality standards for TSS. However, turbidity standards in Idaho and Washington limit increases to 5 NTUs when the background is less than 50 NTUs except when the flood exceeds the 7-day, 10-year flood frequency (Table 3-1). Turbidity levels of the Snake River exceeded state water quality standards in June 1997 at most stations (Table 3-8).

None of the TSS concentrations observed in 1997 would have lethal effects on adult or juvenile salmon (Newcombe and Jensen, 1996). Concentrations of 25 milligrams per liter for 4 hours have been shown to reduce feeding rate; higher concentrations up to 1,000 milligrams per liter have shown no deleterious effects on adult salmon other than coughing and apparent stress. One study showed 50 percent mortality of juvenile coho salmon at 509 milligrams per liter TSS (Newcombe and Jensen, 1996).

3.2.4.8 Nutrients In Water

Inorganic Nitrogen

Of the various soluble inorganic forms of nitrogen, nitrate plus nitrite ($NO_3 + NO_2$) was the principal component, often comprising more than 90 percent of the soluble fraction. Nitrate nitrogen concentrations exhibited inter-annual variations at several of the sites, but long-term trends were not apparent. However, two important issues were identified regarding the inorganic nitrogen species. First, nitrate concentrations were consistently greater than ammonia values at almost all stations. Second, the lowest nitrate concentrations were consistently identified in Clearwater River samples, and higher values were usually determined for the free-flowing Snake River.

Table 3-8. 1997 Turbidity Measurements (NTU[1/]) in Surface Waters at Selected Snake River Stations

Date	SNR-18	SNR-83	SNR-108	SNR-118	SNR-129	SNR-140
June 2 to June 9, 1997	16	17	18	17	17	20
June 28 to July 1, 1997	5	9	3	5	5	8
July 3, 1997	7	--	--	--	--	--
July 14 to July 19, 1997	4	3	3	2	2	3
July 28 to July 31, 1997	4	2	2	3	3	2
August 11 to August 14, 1997	5	3	2	2	2	2
September 8 to September 11, 1997	3	2	1	2	2	2
September 15, 1997	3	--	--	--	--	--
September 22 to September 25, 1997	3	2	2	2	2	2
October 6 to October 9, 1997	2	3	3	2	2	2

1/ Formazin Turbidity Units (FTUs) are equivalent to NTUs
Source: Developed by Normandeau

In 1997, the Clearwater River stations had a median NO_3 concentration of 0.03 milligrams per liter. In comparison, the two upstream lower Snake River stations, SNR-140 and SNR-148, had median NO_3 levels that were much higher, ranging between 0.33 and 0.35 milligrams per liter, while the median NO_3 levels throughout the lower Snake River reach ranged from 0.13 to 0.19 milligram per liter. These data suggest that the high levels contributed from the middle Snake River reach are slightly diluted by the low levels in the Clearwater River, resulting in moderately high NO_3 levels in the lower Snake River. The two principal tributaries in the lower Snake River reach, the Palouse and Tucannon Rivers, had relatively high median levels of 1.38 and 0.21 milligrams per liter, respectively. The Columbia River stations generally had lower NO_3 levels than those observed in the Snake River, with median levels ranging between 0.07 and 0.13 milligram per liter.

Total nitrogen (total-N) levels, which include both the inorganic and organic components, were relatively high in the Snake River stations. Figure 3-42 illustrates the temporal and spatial variability in total-N levels throughout the project area during the 1997 season. Total-N levels at the upstream Snake River station (SNR-140) were generally higher than those observed at the other sampling in total-N levels throughout the project area during the 1997 season. Total-N levels at the upstream Snake River station (SNR-140) were generally higher than those observed at the other sampling stations. In general, concentrations decreased throughout the lower Snake River, but were still higher than those observed in the Columbia River. In the spring and summer, the total-N levels increased from about 0.30 to 0.60 milligrams per liter at the lower Snake River stations compared to concentrations ranging from 0.20 to 0.40 milligram per liter at the Columbia River stations. The total-N levels increased considerably in the fall, with peak levels at the lower Snake River stations reaching 0.8 to 1.1 milligrams per liter 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 as well as agricultural harvesting in the watershed. Early fall rains after prolonged dry periods also increase nutrient concentrations. A corresponding increase in TSS levels was not detected. The seasonal pattern of nitrogen concentrations is also apparent in nitrate data collected in 1971 at SNR-107, prior to construction of Lower Granite Lock and Dam and SNR-108 in 1995 (Figure 3-43). Nitrate levels were generally highest in spring and fall, likely due to the lower biological uptake during the nongrowing season. Concentrations of nitrate were generally similar during the growing season for the 1971 and 1995 data.

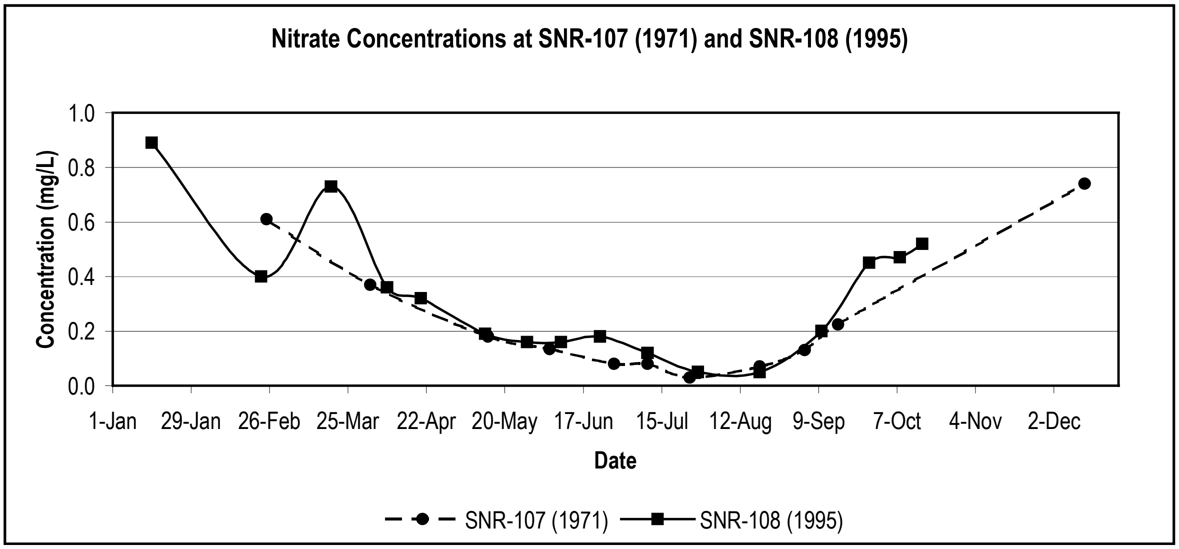


Figure 3-42. Total Nitrogen for Selected Sampling Stations in 1997
Source: Developed by Normandeau

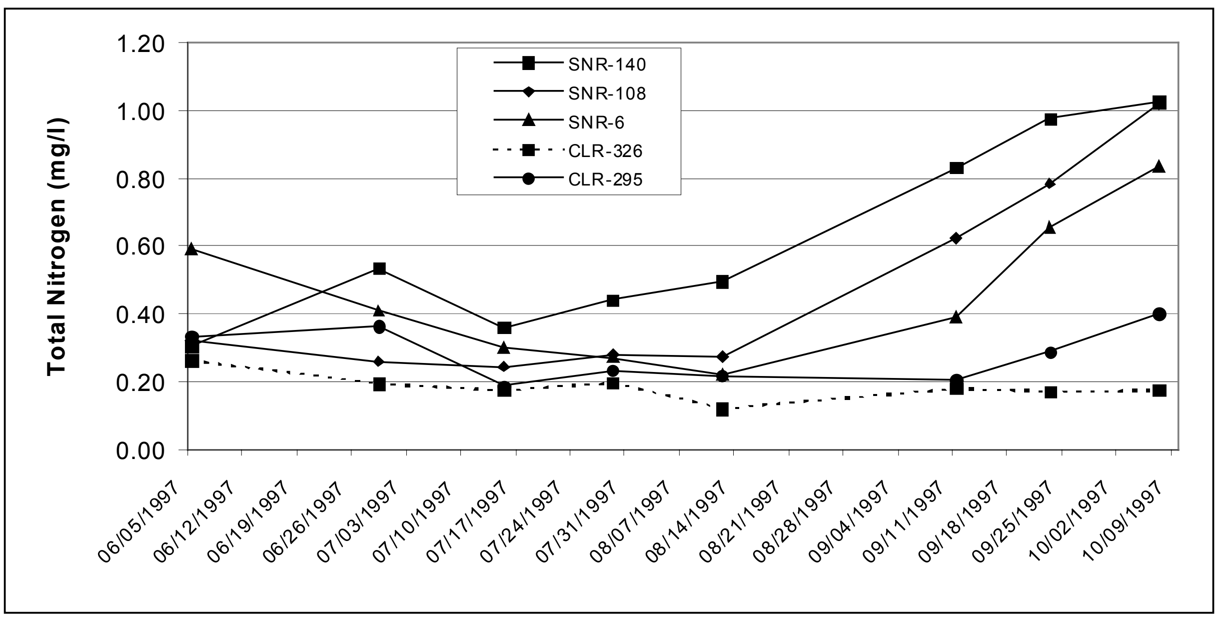


Figure 3-43. Nitrate for Selected Sampling Stations in1971 and 1995
Source: Developed by Normandeau

This late-season increase in the Snake River levels most likely caused the levels at Station CLR-295, in the McNary reservoir, to nearly double from around 0.20 to 0.40 milligram per liter, while the upstream Columbia River station remained constant at just under 0.20 milligram per liter through the fall period.

The Palouse River, again, generally had the highest levels in the study area with an overall median level of 2.15 milligrams per liter. During the spring runoff, peak levels in the Palouse River reached as high as 4.86 milligrams per liter compared to a peak level of 1 milligram per liter at SNR-140. The influence of these high levels from the Palouse River, however, seem to be localized since there was only a slight increase in the peak levels observed downstream at SNR-40 (1.35 milligrams per liter) as compared to

1.05 milligrams per liter at SNR-50. For most of the sampling period, observed levels between these two stations were essentially the same.

Phosphorus

Phosphorus is generally expressed in terms of total phosphorus and orthophosphorus. Orthophosphorus (ortho-P) 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 phosphorus. Total phosphorus consists of both the soluble fraction and that portion adsorbed to sediments or tied up with biological materials in the water column. Since phosphorus readily attaches to and travels with sediments, adsorbed or biological quantities usually represent the largest portion of total phosphorus. 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 (Wetzel, 1983).

Recent and historical data suggest that ortho-P levels in the lower Snake River tend to be highest in the spring and fall, with relatively low concentrations in the summer (Figure 3-44). The lower levels during the summer are most likely due to biological uptake by aquatic plant and algal growth. As plant growth diminishes in the fall, the phosphorus levels increase, which was most evident at the reservoir stations where algal growth is usually most abundant. In the Columbia River and the free-flowing river stations, there was little change in the ortho-P levels during the sampling season.

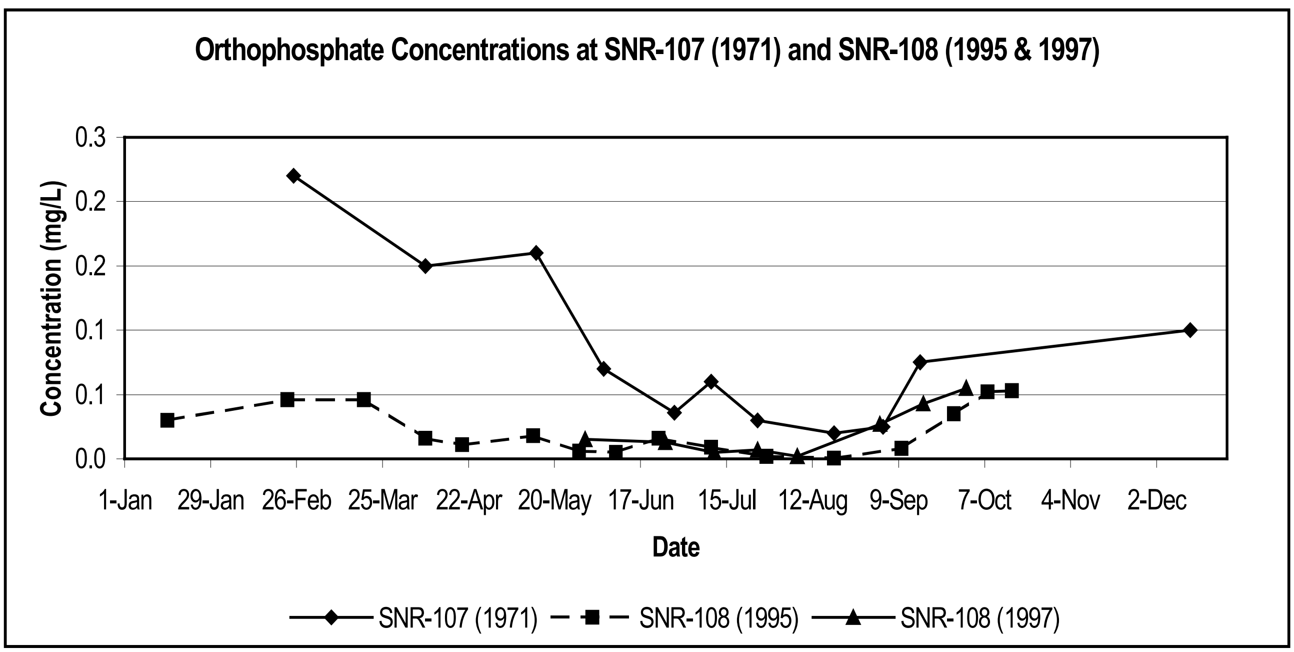


Figure 3-44. Orthophosphate Concentration at SNR-107 prior to Construction of Lower Granite Dam (1971) and SNR-108 after Dam Construction

Source: Developed by Normandeau

The Clearwater River had relatively low ortho-P levels ranging from 0.006 milligram per liter in the spring to a low of 0.001 milligram per liter in the mid-summer and up to 0.003 milligram per liter in the fall. At RM 140, upstream of the confluence with the Clearwater River, ortho-P levels generally ranged from 0.013 to 0.023 milligram per liter through the summer and from 0.054 to 0.059 milligram per liter in the fall. Throughout the lower Snake River reach, ortho-P levels through mid-August peaked at 0.018 milligram per liter and increased to 0.022 to 0.063 milligram per liter from mid-September through October. In the Columbia River, ortho-P levels were comparatively low, with most levels below 0.008 milligram per liter and a peak level of only 0.016 milligram per liter in the McNary reservoir during the fall. This comparison of data clearly indicates that ortho-P is much more available throughout the lower Snake River reach relative to other major rivers in the area.

Total phosphorus (TP) levels were also relatively low in both the Clearwater and Columbia Rivers. Median TP levels for both rivers were between 0.010 and 0.013 milligram per liter. Peak levels, which typically occurred during the spring freshet, were as high as 0.018 and 0.028 milligram per liter for the Clearwater and Columbia Rivers, respectively. Again, the highest levels in the study area were measured in the upper portions of the lower Snake reach. During the spring freshet, TP levels (water column average) throughout the lower Snake River ranged from around 0.060 to 0.11 milligram per liter (Figure 3-45).

The high TP levels during this time of year are most likely attributable to the suspended sediment contained in the peak flow period. For much of the 1997 growing season, TP levels generally ranged from 0.035 to 0.060 milligram per liter and then steadily increased in the fall. Similar concentrations were observed in 1994 and 1995, where concentrations ranged from 0.025 to 0.060 milligram per liter in the summer and then increased to around 0.09 milligram per liter in the fall.

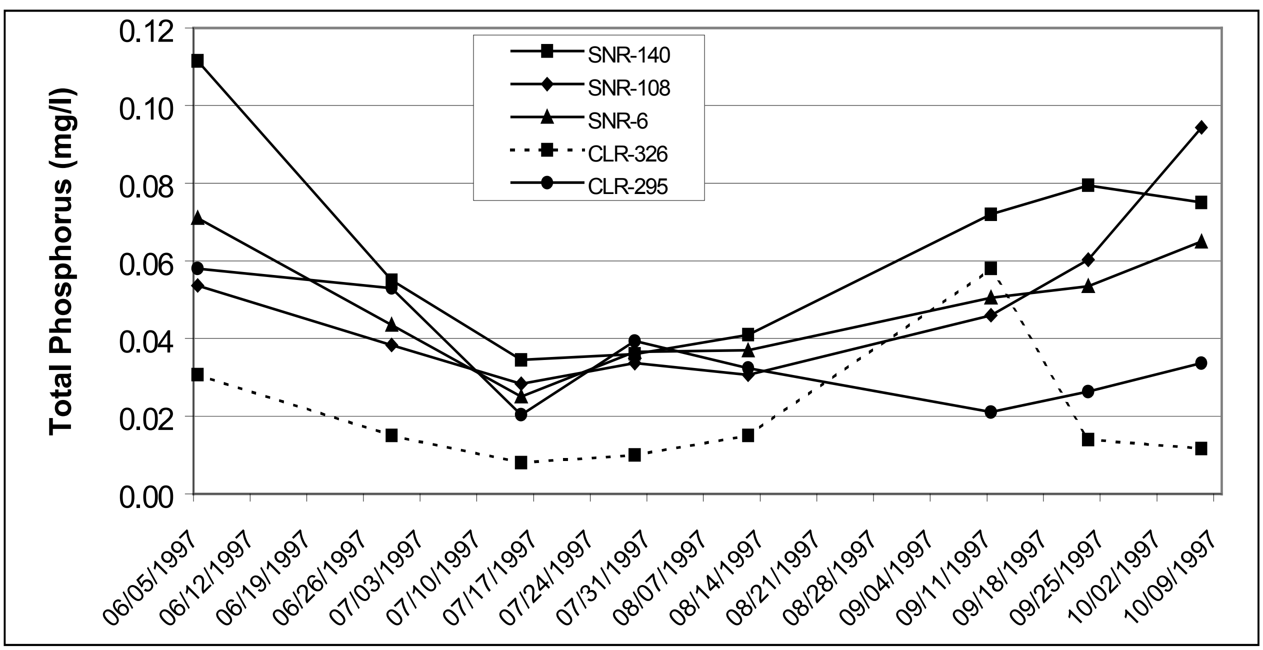


Figure 3-45. Phosphorus at Selected Sampling Stations in 1997
Source: Developed by Normandeau

In the Columbia River, the upstream station had a much lower peak TP level of 0.03 milligram per liter as compared to the downstream lower McNary reservoir (CLR-295), which had a peak TP level close to 0.06 milligram per liter. TP levels were lowest during the July and August period throughout the study area, with levels ranging between 0.02 and 0.04 milligram per liter. In September, the TP levels steadily increased, with levels at the three lower Snake River stations ranging between 0.05 and 0.10 milligram per liter.

According to the State of Washington water quality standards, total phosphorus levels above 0.020 and 0.035 milligram per liter are considered to be critical thresholds in terms of preventing excessive algal growth when ambient trophic conditions are considered to be in the lower and upper mesotrophic categories, respectively. Oligotrophic conditions represent high-quality waters with good water clarity, and low algal production and eutrophic conditions represent high nutrient levels, excessive algal growth, and poor water clarity. Mesotrophic conditions are somewhere in the middle and typically represent moderate levels of algal production, water clarity, and light transparency.

Limnological conditions in the lower Snake River impoundments have generally been considered to be in the upper mesotrophic category. Based on a review of the 1997 data, the average phosphorus levels throughout the lower Snake River appear to be in the 0.030 to 0.040 milligram per liter range during the mid-summer and slightly higher to near the 0.060 to 0.070 milligram per liter range during June and fall months. This would suggest that the average phosphorus levels in the lower Snake River for much of the entire growing season would likely be above the threshold to maintain existing conditions and prevent eutrophic conditions.

Both the Palouse and Tucannon Rivers had much higher levels throughout the 1997 sampling period. Observed total phosphorus levels ranged from 0.062 to 0.212 milligram per liter and 0.100 to 0.287 milligram per liter in the Palouse and Tucannon Rivers, respectively. However, due to the substantial difference in flow levels, these high TP levels did not appear to dramatically change measured levels in the main stem of the Snake River.

3.3 Sediment Quality

The purpose of the special sediment study for the Lower Snake River Juvenile Salmon Migration Feasibility Study (Feasibility Study) was to determine the concentration and distribution of potential contaminants residing in these sediments of the four lower Snake River dams. This study was conducted in two phases. The first phase was to collect transects of the particle-size distribution and target specific areas where finer-particle-size materials were predominant. Using these data, the Corps selected representative samples using 8-centimeter-diameter by 2-meter-long (3-inch-diameter by 7-foot-long) Balchek gravity core samplers. Approximately 800 to 1,000 pounds of weight was used on top to provide maximum penetration and recovery. Individual "Spud" sampler data (USDA, 1938) showed that the depth of sediment in the areas sampled by this study was no greater than 6 feet measured from bedrock to the surface of the sediments in 0.03-meter (1/10-foot) increments. Some samples were cut into 0.6-meter (2-foot) sections for layer analysis. Most of the cores were composited over the entire depth of sediment, (CH2M HILL, 1997; Anatek Labs, 1997).

During Phase 2 of the Feasibility Study sampling program, 94 sediment samples were collected from the lower Snake River and submitted for the laboratory analysis of selected inorganic and organic constituents. The results of the laboratory analysis for the composite or top layer samples are summarized in subsequent sections of this appendix, with detailed information regarding the number of samples above detection limits, minimum and maximum values, arithmetic and geometric means.

The results of the 1997 Feasibility Study field investigations performed by the Corps are supplemented by data collected by others within the Columbia River drainage basin. Most of these studies are linked directly to the Corps dredging authorities and projects, and these predominantly focus on the Snake and Clearwater confluence area. These include the 1990 and 1997 Corps sediment surveys for dredging and individual documents supporting 40 Code of Federal Regulations, Part 230, Section 404(b)(1) evaluations for specific dredge operations.

This section includes discussion of the sediment parameters and classes of compounds that form the basis for the selection of contaminants of concern. Data derived from the multiple sources are cited and a general summary of the results for these studies is included in this section. The purpose of this section is to present the evidence used in the analysis of alternatives in section 4 of this appendix. Included in the analysis is discussion on the weight of evidence to provide the reader with an overall feel for the degree of risk and uncertainty that should be applied to the decision process.

3.3.1 Sediment Particle Size

A total of 487 grab sediment samples were collected as part of the Phase 1 task of the special Lower Snake River Juvenile Salmon Migration Feasibility Study sediment study (CH2M HILL, 1997). Of the 487 grab samples, 356 were sieved to develop particle-size distributions. The remaining 131 samples (or 26.9 percent) were not sieved either because there was no sample recovery or because the sample consisted only of gravel and/or cobble. The average grain size distributions for the sediment samples collected from above Ice Harbor, Lower Monumental, Little Goose, and Lower Granite Locks and Dams are summarized in Table 3-9.

The mean grain size for the channel bed sediment ranges from very fine sand to silt/clay. The highest concentration of relatively coarser sediments (fine to medium sand) was found in Lake Sacajawea, above Ice Harbor Lock and Dam. The highest concentration of silt/clay-size sediments was found in Lake West, above Lower Monumental Lock and Dam. Fine-grain sediments are concentrated on the channel bottom in Lake West. The concentration of these fine-grain sediments is most likely associated with the discharge of the Palouse River into Lake West. This contribution is evidenced by the elevated TSS concentrations in the water quality samples collected from the Palouse River (Section 3.2.4.7). Soil erosion within the Palouse River drainage basin has been documented as a chronic problem due to historical land use practices (Ebbert and Roe, 1998). Recent studies have also documented that the adoption of erosion-control practices within the drainage basin has resulted in an observable decline in suspended-sediment concentrations in the Palouse River (Ebbert and Roe, 1998), and as a result, probably also into the Lower Monumental reservoir.

The Corps, Walla Walla District, sediment study of the lower Snake and Clearwater Rivers confluence (Corps, 2000) sampled 53 sites for particle size with an emphasis on depositional areas. Some mid-channel and lock approach areas from Lower Granite and Little Goose locks and dams were also included in the sampling. The average particle-size distribution for all sites were fines (silt and clay) 17.14 percent, sand 74.20 percent, and gravel 7.76 percent in the confluence samples. Particle size was very dependent on locations of the sampling sites. The lock approach sites comprised 2.5- to 15-centimeter (1- to 6-inch) cobbles exclusively. This was expected due to the velocities measured by an acoustic doppler profiler during spill events. Generally, a sample location near the confluence that was more than 75 meters from the shoreline contained less than 1 percent fines.

Table 3-9. Summary of Sieve Test Results for Sediment Samples Collected from the Lower Snake River in 1997

Sediment Size	Average Grain Size (in percent) IHR	LM	LGO	LGR	Cumulative percent IHR	LM	LGO	LGR
Gravel	2.4	2.8	1.9	0.4	2.4	2.8	1.9	0.4
Very Fine Gravel	0.1	0.6	0.7	0.3	2.5	3.4	2.6	0.7
Very Coarse Sand	0.1	1	0.7	0.5	2.6	4.4	3.3	1.2
Coarse Sand	1.1	1.1	2.8	1.7	3.7	5.5	6.1	2.9
Medium Sand	18.3	2.8	10.2	6.9	22	8.3	16.3	9.8
Fine Sand	18.3	6.7	13.1	17.1	40.3	15	29.4	26.9
Very Fine Sand	23.3	13.2	16.8	20.1	63.6	28.2	46.2	47
Silt/Clay	35.8	71.8	53.8	52.4	99.4	100	100	99.4

Notes: IHR - Ice Harbor Reservoir (Lake Sacajawea), 41 samples
LM - Lower Monumental Reservoir (Lake West), 77 samples
LGO - Little Goose Reservoir (Lake Bryan), 127 samples
LGR - Lower Granite Reservoir (Lower Granite Lake), 104 samples
Source: Developed by Normandeau

3.3.2 Organics

The sediment samples were tested for the following organic compound groups: chlorinated herbicides, dioxins, glyphosate herbicide, organochlorine pesticides, organophosphorus pesticides, semi-volatile compounds, and total petroleum hydrocarbons. A few chlorinated herbicides, organophosphorus pesticides, or semi-volatile organic compounds were detected in composite top layer sediment samples.

In other studies, organic contaminants were detected sporadically. The following sections discuss the results of the polychlorinated dibenzo dioxin/furans, glyphosate herbicide, organochlorine pesticides, polynuclear aromatic hydrocarbons (PAHs), and total petroleum hydrocarbon analyses and other historical organic chemistry data sets.

3.3.2.1 Dioxins and Furans

Tetrachlorodibenzo-p-dioxin (TCDDs) and tetrachlorinated dibenzo furans (TCDFs) are persistent toxic substances that enter the environment as unintended byproducts of several industrial processes. They represent a hazard to aquatic life and human health because of their toxicity at low levels, persistence, and bioaccumulation factors (NRCC, 1981; Eisler, 1986). The most significant sources are pulp mills, municipal waste incinerators, and fires involving PCB-contaminated oil (Palmer et al., 1988). Another potential source of deposition includes open burning of household waste in barrels (Lemieux et al., 2000). The EPA (1993) considers dioxin-like compounds to be carcinogens.

The TCDDs and TCDFs have low solubility in water—less than 1 part per billion (ppb) (Crummet and Stehl, 1973). When discharged to aquatic environments, their primary fate is sorption to the sediments and accumulation in biota (Johnson et al., 1991). Because of this solubility by lipid tissue, concentrations in fish can exceed environmental concentrations by factors as high or greater than 5,000 times (Maybee et al., 1991; EPA, 1984; Opperhuizen and Sijm, 1990). Bioconcentrations are probably greatest for species that live in contact with sediments or are part of the food webs linked to sediments (Cooke, 1987 and Kuel et al., 1987). Half-lives of TCDD and TCDF in aquatic sediments exceed 1 year and could be as high as 10 years for some congeners (Callahan et al., 1979; Eisler, 1986; CCREM, 1987).

Since the 1980s, there have been concerns about dioxin/furan contamination of the Washington portion of the Snake River. These concerns arose when 2,3,7,8- tetrachlorodibenzo-p-dioxin (2378-TCDD) was found in the effluent bleached Kraft pulp mills in the United States. Dioxin compounds also received large amounts of media attention when they were linked to "Agent Orange" and health effects suffered by Vietnam veterans and their families (Holden, 1979).

Last year, there was no 2378-TCDD detected in the Lower Granite sediment sample sites. There is considerable public interest in the potential for environmental harm caused by the release of sediments resulting from the breaching alternative. The Corps, realizing government and public concern are not equalized, continues to maintain surveillance of sediments for presence and extent of 2378-TCDD and 2,3,7,8 tetrachlorinated dibenzo furan (2378-TCDF) contamination in the Lower Granite and McNary pools through its Dredged Material Management Program study.

In 1991, in response to the dwindling salmon stocks on the Snake River, the Corps initiated a sediment quality study to attempt identification of potential sediment/contaminant factors that could be associated with mortality. This study was also conducted to determine some potential effects prior to the execution of the 1992 drawdown test of the Lower Granite pool. The 5 of 19 sediment composite samples analyzed for dioxin/furan compounds yielded 0.62 parts per trillion (ppt) 2378-TCDD and 15.5 ppt 2378-TCDF in McNary pool at approximately Columbia River RM 317 and an average of 0.43 ppt 2378-TCDD and 2.72 ppt 2378-TCDF on the Clearwater River at approximately RM 3 (Pinza et al., 1992).

In the Lewiston-Clarkston Area, the Corps operates a series of pumping plants that discharge water trapped inside the levies to the Snake and Clearwater rivers. In 1994, the Corps conducted sediment and water quality studies to determine the extent of contamination from various nonpoint sources (particularly storm water runoff). Results from dioxin/furan analysis yielded 0.68 ppt 2378-TCDD and 1.34 ppt 2378-TCDF in the Corps East Pond. An inlet stream into the east pump pond yielded 4.56 ppt 2378-TCDD and 68.6 ppt 2378-TCDF (MRI, 1994).

The special tests conducted for this study included collection of sediment samples downstream of the previous sediment samplings. Only two of the four samples yielded detection for total dioxins at 0.69 and 1 ppt (individual 2378-TCDD was undetectable) (CH2M HILL, 1997).

In 1998, prior to the confluence dredging, the Corps initiated a sediment study where nine samples were taken in the Lewiston/Clarkston confluence area near Snake River RM 139. Only two of the nine samples yielded a result with 1.3 and 1.7 ppt 2378-TCDF and no detect for 2378-TCDD (HDR, 1998).

In 1998, the Corps embarked upon its own Dredged Material Management Program study. The products resulting from the endeavor include a programmatic manual and a regionally approved sediment test framework to address methods and procedures for testing. Samples taken from approximately Columbia River RM 317 did not yield a single detection of 2378-TCDD (Corps, 1998a).

CH2M HILL conducted dioxin tests in the Lower Granite pool and Clearwater arm of the pool. Seven sites were selected and individual subsets were combined into composite sample for analysis. A Ponar sampler was used to sample the top 7.5 inches of the sediments. Results from sediments at all of the inriver sites sampled consisted of no detects and below detection limits for 2378-TCDD and 2378-TCDF. The only sample that contained a detectable level of contamination was the Corps East Pond (CH2M HILL, 1999). The concentrations of 2378-TCDD and 2378-TCDF were very low-2.8 ppt and 3.4 ppt, respectively. The East Pond receives storm runoff from multiple sources. The study was repeated in 1999 and it was discovered that the only detection was for 2378-TCDF in the amount of 2.3 ppt.

Sediment samples collected in 2000 included use of new technology to aid the Corps in detection of dioxin/furan-containing sediments. Twenty targets were punched with a 2-meter by 8-centimeter (6-foot by 3-inch) Balchek sampler for the detection of dioxins. Ten of these samples met the screen for tier 2 evaluation based on total organic carbon and percent fines. Using a cell-based assay, approximate ppt estimations for total dioxin/furan toxic equivalent measurements were taken. These tests are one-fifth of the cost ($250 versus $1,200 per sample) of a standard high-resolution gas chromatograph mass spectrometer and an excellent tool for screening to be followed up with a confirmatory examination using the EPA-approved method. Those 10 samples screened yielded detection of a dioxin/furan compound; then all 10 samples were analyzed by a gas chromatograph mass spectrometer confirmatory test and quantification of the congeners. The gas chromatograph mass spectrometer detected some of the congeners but no 2378-TCDD or 2378-TCDF.

For the last 10 years the Corps, Walla Walla District, has maintained a dioxin/furan surveillance system through the dredged material-testing program. Now with the impending regional testing and evaluation framework along with the completion of this work, many other of the environmental monitoring studies will be legitimized under a comprehensive and regionally based system. While future data analysis is underway, it appears there is a downward trend in concentration levels of dioxin/furan compounds in the Snake River and possibly the McNary pool. Only with additional analysis and testing could this be confirmed.

For purposes of estimating potential risks associated with TCDD and TCDF in the sediments, current standards should be evaluated conservatively. It is important that fisheries scientists use toxicity equivalence quotient (TEQ) values to look at the toxic equivalencies for all congeners of a particular combined class of dioxin or furan prior to any alternative or action alternative considered by this EIS. The U.S. Department of Ecology uses the standard of 0.07 ppt 2378-TCDD in fish as a state surface water quality standards (Johnson et al., 1991), and the EPA bioconcentration factor for 2378-TCDD is a factor of 5,000 (EPA, 1986a).

3.3.2.2 Glyphosate and Aminomethylphosphoric Acid

Glyphosate (N-[phosphonomethyl] glycine) is a postemergence herbicide that has found widespread agricultural and domestic use. It is sold as a terrestrial and aquatic herbicide and used in massive amounts in eastern Washington and eastern Oregon. The use of glyphosate-containing products is deemed a necessity by many eastern Washington and Oregon growers because of glyphosate's efficacy for even the most noxious pests and its cost effectiveness.

Three essential amino acids that are essential for the survival and growth of plants are affected by glyphosate. These are phenylalanine, tyrosine, and tryptophan. The herbicide action of glyphosate is probably suppression of the enzyme 5-enolpyruvyshikimate-3-phosphate synthase (Robinson et al., 1994). Glyphosate also may inhibit or repress two other enzymes, chlorismate mutase and prephrenate hydratase, involved in the synthesis of these same amino acids (EPA, 1993). Higher animals do not synthesize these essential amino acids but receive most of their needs from their diet (Metzler, 1977). A major metabolite of glyphosate is aminomethylphosphonic acid (AMPA). AMPA is much more persistent than glyphosate, with a reported range of half-life between 199 and 958 days (WHO, 1994). The AMPA found in most sediment could provide clues to the amount of impact from runoff and could also be used to determine actual environmental fate in specific watersheds. This would be highly dependent on the microbial communities in the affected sediments. There is still very little known about the long-term environmental effects of glyphosate to the local aquatic environment and further research is needed.

Glyphosate is highly absorbed on most soils with high organic content, and very little if any leaches to groundwater. Microbes are primarily responsible for the chemical breakdown of glyphosate to AMPA and other products. Because glyphosate is tightly bound to the soil, the primary means of deposition into the aquatic environment is by mobilization of fine silts and clays rather than transfer by rain or runoff. Its expected half life in the pond environment is 10 to 12 weeks (USFS, 1984). Glyphosate would likely be found in areas dominated by silt and clay as opposed to other sediments in the Snake River basin.

Although advertised as a herbicide that "can eradicate weeds and unwanted grasses effectively with a high level of environmental safety" (Cox, 1995), this class of products may not be as innocuous as once thought. Glyphosate products can kill some aquatic insects such as chironomids, beetles, and other aquatic insects considered beneficial because of their food value to game species. The acute toxicities vary from species to species but are greatly regulated by water hardness (Buhl and Faerber, 1989; Hassan, 1988). *Daphnia pulex* is killed by concentrations of roundup between 3 and 25 parts per million (ppm) (Folmer et al., 1979; Hartman and Martin, 1984; Servizi et al., 1987). Many of the reported LC50s vary greatly from researcher to researcher. Most of these effects vary due to the type of "inert ingredients" such as surfactants (Mitchell et al., 1987); water hardness (Wan et al., 1989; 1991); age and species of the test fish (Folmar et al., 1979; Wan et al., 1989.) The research concerning environmental effects of glyphosate is fragmented, incomplete (Cox, 1995), and of questionable quality (U.S. Congress 1984; U.S. Department of Justice, 1992). The overall impacts of glyphosate products to water quality are mediated by all of these mentioned variables. From the research presented, it is clear that water quality impact from a glyphosate product is site specific. A much more thorough study would be needed to quantify both short-term and long-term effects on water quality.

All top-layer sediment samples (94 total samples) were tested for glyphosate and AMPA (CH2M HILL, 1998). Glyphosate was detected in 36 percent of the samples, and AMPA was detected in 16 percent of the samples tested. The concentration of glyphosate ranged from nondetected to a maximum of 68.9 ppb (parts per billion) with an arithmetic mean of 12.52 ppb. The concentration of AMPA ranged from nondetected to a maximum of 29.3 ppb with an arithmetic mean of 7.48 ppb (Anatek, 1997). No screening criteria have been established for either glyphosate or AMPA in sediments within the Columbia River basin.

Glyphosate and AMPA were detected in sediment samples collected from each of the impoundments. The highest individual concentrations of glyphosate and AMPA were detected in samples collected from Lake Bryan (upstream of Little Goose Lock and Dam) (Anatek, 1997). The highest average reach concentration of glyphosate was found in the samples collected from Lake Sacajawea (upstream of Ice Harbor Lock and Dam, see Table 3-10).

The samples analyzed for glyphosate during this study were taken in September. Most of the glyphosate compounds are applied in the spring. The approximate time between its usual and accustomed use and the testing for this study is about 140 days. It was expected that AMPA should be higher than the glyphosate concentrations but this was not the case (Table 3-10). No sampling was conducted in the spring of 1997 because high surge levels were expected due to runoff. The intent was to determine the possibility of any long-term residues in the sediments that would potentially be resuspended during the near-natural flowing river drawdown alternative.

Table 3-10. Summary of Average Glyphosate AMPA Concentrations (μg/L) for Sediment Samples Collected during 1997 in the Lower Snake River

	Ice Harbor	Little Goose	Lower Monumental	Lower Granite	Average
Elutriate					
AMPA	ND	ND	ND	ND	ND
Glyphosate	0.58	0.69	ND	ND	0.57
Sediment					
AMPA	8.08	7.58	6.07	8.28	7.48
Glyphosate	16.80	10.42	10.60	14.85	12.52

Note: ND = Not detected; ½ the detection level is used when concentrations are less than detection level.
Source: Anatek (1997).

3.3.2.3 Organochlorine Pesticides

Several organochlorine pesticides were detected in the sediment samples collected from the lower Snake River. The organochlorine pesticide compounds detected (and their frequency of detection) included: 4,4-DDD (11); 4,4-DDE (43); 4,4-DDT (5); aldrin (3); dieldrin (4); endrin (1); heptachlor (1); and lindane (3) (Anatek 1997, see Table 3-11). The three principal organochlorine pesticide compounds detected in the sediments are related, with DDT being the parent compound and DDD-DDE being daughter products generated by the transformation of DDT in the environment (Callahan et al., 1979).

Synthetic organic chemicals containing chlorine, including carbon tetrachloride and trichloroethylene, were available commercially by 1925. Benzene hexachloride was first synthesized in 1825 and DDT in 1874; however, the insecticidal properties of these compounds were not recognized until around 1940 (Smith, 1991). The military developed and produced DDT during World War II to control mosquitoes and, thereby, the spread of malaria and other diseases. Released into civilian markets in 1945, DDT was used heavily over the next 2 decades to control agricultural and forest insects as well as disease vectors; by 1961, 1,200 formulations were available for use on 334 crops (EPA, 1992). The DDT was also used to control fishes, bats, and other wildlife. After World War II, additional organochlorine pesticides including methoxychlor, aldrin, dieldrin, and chlordane became available. These were followed in the 1950s and 1960s by endosulfan, endrin, mirex, kepone, toxaphene, and others (Smith, 1991). In addition to being highly toxic, organochlorine pesticides are relatively insoluble in water, adhere strongly to soil particles, and are resistant to physical, chemical, and biological degradation. These properties were viewed as desirable, and negative consequences from bioaccumulation and toxicity to nontarget organisms were not foreseen.

As the use of DDT declined, total organochlorine chemical use increased; in 1966, 18 million kilograms (40 million pounds) of organochlorine insecticides was used in agriculture, including 6 million kilograms (13 million pounds) of DDT and DDD, 7 million kilograms (15 million pounds) of toxaphene, and 3 million kilograms (7 million pounds) of aldrin (Eichers et al., 1971; Andrilenas, 1974). After the 1969 DDT ban, toxaphene became the most heavily used insecticide, averaging about 13 million kilograms (29 million pounds) annually through the early 1970s; most was used for insect control in cotton (Eichers et al., 1978). Cotton insects eventually developed resistance to toxaphene and usage declined through the late 1970s, well before its registration was canceled in 1983.

Table 3-11. Summary of Average Concentrations (ppb) of Organochlorine Pesticides and TPH in Sediments Collected during 1997 in the Lower Snake River

	Ice Harbor	Little Goose	Lower Granite	Lower Monumental	Average
Sediment					
4,4-DDD	ND	1.95	3.06	1.58	2.07
4,4-DDE	2.68	4.91	6.48	4.22	4.89
4,4-DDT	ND	1.64	1.72	1.56	1.62
Aldrin	0.75	0.84	0.87	0.82	0.83
Dieldrin	ND	1.74	ND	1.80	1.68
Endrin	ND	ND	ND	1.75	1.58
Lindane	ND	0.91	ND	0.90	0.85
TPH	67.63	45.86	58.25	49.15	55.41

Note: ND = Not detected; average uses ½ of detection when concentrations are less than detection level.
Source: Developed by Normandeau

Aldrin, heptachlor, chlordane, and similar compounds were used heavily to control insects in corn during the 1970s, and organochlorine chemical use in the Midwest increased accordingly (Eichers et al., 1978). As insect resistance and environmental concerns grew, use of these and other organochlorine compounds declined from 46 percent of the total used in 1971 to 29 percent in 1976 (Andrilenas, 1974; Eichers et al., 1978). At present, only endosulfan and methoxychlor are registered for agricultural use, although other organochlorine insecticides are still manufactured for export.

The ecological consequences of organochlorine pesticides were extensive, and some remain evident. Because of their insolubility in water and resistance to complete metabolic degradation, many organochlorine compounds bioaccumulate. Upon accumulation by vertebrates, DDT is metabolized to DDE, which is stable and toxic; it impairs calcium metabolism in the shell gland of adult female birds. At sufficiently high concentrations, eggshell thickness can be reduced to the extent that eggs cannot support the weight of the incubating parents. Shell breakage and death of the developing embryo can result. Susceptibility varies, but predatory birds are most vulnerable, both physiologically and because of their position at the apex of aquatic food chains (Cooke, 1973). In North America, population declines attributable to DDE-induced eggshell thinning and reduced recruitment were documented in bald eagles, ospreys, prairie falcons, merlins, double-crested cormorants, and brown pelicans (Cooke, 1973). Other contaminants and factors may also have contributed to the declines in these species, especially in the Great Lakes (Colborn, 1991). Shell thinning was also documented in other species, including peregrine falcons and herring gulls, but many factors may also have been involved in their declines (Cooke, 1973).

The use of DDT and other organochlorine insecticides in agriculture and forestry also affected aquatic organisms. In waters draining cotton-farming areas, the diversity of benthic invertebrate and fish communities was characteristically low, and the development of insecticide-resistant forage fishes led to elimination of predatory species in the 1960s. In addition to directly affecting bald eagle reproduction (Colborn, 1991), DDT spraying of United States and Canadian forests in the 1940s and 1950s to control eastern spruce budworms and western spruce budworms reduced benthic invertebrate diversity and fish populations in streams over wide areas of the Northeast (Ide, 1957; Warner and Fenderson, 1962) and the West (Adams et al., 1949; Graham, 1960; Cope, 1961).

Agricultural and forest insect control accounted for more than half of organochlorine insecticide usage in the United States during the 1960s and 1970s (Andrilenas, 1974; Eichers et al., 1978; Aspelin et al., 1992). Nevertheless, other pesticide uses, mechanisms of toxicity to birds other than eggshell thinning, and effects on other organisms were also significant. Application of chemicals to water for the control of mosquitoes, gnats, and black flies was common. Heavy applications of DDD for gnat control decimated western grebes at Clear Lake, California (Herman et al., 1969). Mosquito control using DDT led to reproductive failure of lake trout in Lake George, New York (Burdick et al., 1964), and of landlocked Atlantic salmon in Sebago Lake, Maine (Anderson and Everhart, 1966), and was suspected of reducing the survival of juvenile winter flounder in a Massachusetts estuary (Smith and Cole, 1970). During the 1950s, widespread use of DDT to control the bark beetles that transmit Dutch elm disease reduced American robin populations in many northeastern and midwestern communities (Cooper, 1991); some 80 bird species were affected by the Dutch elm disease program in Michigan alone (Wallace et al., 1961). In southern Ontario, residual DDT and dieldrin from mosquito control have been suggested as a cause of local amphibian extinctions (Russell et al., 1995).

Accumulations of newer organochlorine insecticides-such as chlordane, aldrin, dieldrin, eptachlor, and endrin-caused many wildlife kills, including endangered gray bats in Missouri (Clark, 1981; Clawson and Clark, 1989). The use of endrin to control sugarcane borers and of heptachlor to control imported red fire ants in Louisiana during the 1950s and 1960s led to repeated massive fish kills in the Atchafalaya and Lower Mississippi rivers and along the Gulf Coast (Biglane and Lafleur, 1967). The consumption by waterfowl of seeds treated with heptachlor and other organochlorine pesticides caused frequent kills in agricultural areas. Dieldrin is suspected of contributing to the decline of the Great Lakes population of bald eagles (Colborn, 1991). As presently known, however, the overall extent of ecological injury attributable to these newer organochlorine insecticides has been less than that of DDT.

It was expected that no organochlorine pesticide would be present in the sediments tested during the special sediment study of 1997 for the Feasibility Study. In response to the dramatic decline in use of the organochlorine pesticides, other researchers reported sharp drops in the delectability of these compounds (Mineau and Peakall, 1987; Prouty and Bunck, 1986; Bunck et al., 1987; Baumann and Whittle, 1988; Schmitt et al., 1990; Wiemeyer et al., 1993; Weseloh et al., 1994; Mora, 1995) and therefore no contamination was expected because of the length of time since the use of these products. This study was considered to be a benchmark of 20 years after phase-out of these compounds. However, to the contrary, organochlorine pesticide residues were detected in the sediments of all the four lower Snake River reservoirs (Table 3-11).

The predominant organochlorine compound detected was DDE, which ranged in average concentration from 2.68 in Ice Harbor to 6.48 in the Lower Granite reach, with an arithmetic mean concentration of 4.89 ppb. DDD was detected in 11 sediment samples, with an average maximum concentration of 6.48 ppb in Lower Granite reach and an arithmetic mean of 2.07 ppb. DDT was detected in only five samples, with a mean arithmetic concentration of 1.62 ppb.

Total DDT (DDD, DDE, and DDT) concentrations ranged from nondetect to 32.8 ppb with an average concentration of 8.23 ppb (Table 3-11). The highest mean reach concentration for total DDT was 11.3 ppb for Lower Granite Lake. The average reach concentration of total DDT decreased steadily from Lower Granite Lake down to 5.7 ppb as recorded in Lake Sacajawea. The maximum and average total DDT concentrations in the lower Snake River sediments exceed the guidance levels set forth in "Puget Sound Dredged Disposal Analysis Guidance Manual: Data Quality Evaluation for Proposed Dredged Material Disposal Projects" (PTI, 1989) or recommended screening concentration (6.9 ppb), but are lower than the bioaccumulation trigger concentration of 50 ppb as established in the Portland District

Dredged Material Evaluation Framework (DMEF) (Corps, 1998b). Concentration levels above the screening level prompt biological testing to ascertain health risks to aquatic organisms using the DMEF (Corps 1998b).

The pesticides aldrin, dieldrin, endrin, heptachlor, and lindane were all detected in five or fewer of the 1994 dredge material sediment evaluation samples. The concentration of aldrin ranged from nondetect to 3.5 ppb, dieldrin from nondetect to 8 ppb, endrin from nondetect to 9.4 ppb, heptachlor from nondetect to 4.9 ppb, and lindane from nondetect to 5.5 ppb. The maximum concentrations of aldrin, dieldrin, heptachlor, and lindane in the Snake River sediment are lower than their screening level concentration of 10 ppb. No screening level has been established for endrin in the DMEF (Corps, 1998b).

Since the early 1980s, the Walla Walla District monitored sediment prior to dredge operations for a suite of organic compounds. Organochlorine pesticides were at the top of the list of constituents for which to test. In the sediment analysis studies for 1984 and 1985 for interim dredging (Corps, 1986; 1987), the Corps sampled sediments between the Port of Lewiston and the confluence of the Snake and Clearwater River. In seven sample sites 4,4-DDD ranged from less than 0.3 to 3 ppb and 4,4-DDE ranged from less than 0.3 to 4.8 ppb. In 1988, during the Interim Flood Control Dredging at Lower Granite Lock and Dam, the Walla Walla District found 4,4-DDT (6 ppb) at a site in the Clearwater Snake River Confluence and at 7 and 21 ppb at two areas in an in-water disposal site at Snake River RM 120 (Corps, 1987).

The Walla Walla District study (Sediment Sampling of Proposed Dredge Sites in the Confluence of the Snake and Clearwater Rivers, Pinza et al., [1992]) tested 19 sites for chlorinated pesticides at port areas on the lower Snake and Columbia rivers. The compound 4,4-DDT ranged from 0.6 to 1.8 ppb, which was below laboratory detection limits. The DDT metabolite 4,4-DDD ranged from 0.4 to 4.8 ppb and DDT metabolite 4,4-DDE ranged from 0.6 to 16 ppb. Other organochlorine pesticides were found at below laboratory detection limits of 0.2 ppb.

The Walla Walla District 1996 Sediment Study for the Confluence Dredging of the lower Snake and Clearwater Rivers for the Port of Lewiston and Port of Clarkston (Corps, 1996) reported low levels of DDT. The compound 4,4-DDT at two sites in the Confluence ranged from 3 to 6 ppb, 4,4-DDD in five sites at the Confluence ranged from 2 to 4 ppb, 4,4-DDE in five sites at the Confluence ranged from 3 to 11 ppb.

The USGS collected and analyzed bed sediments from the Snake River upstream of this study area (Clark and Maret, 1998). The only organochlorine compound detected in all of the bed sediment samples analyzed by the USGS was DDE at concentrations ranging from 1 to 11 ppb. These concentrations are similar to those reported for the sediment samples analyzed for this investigation. Reports of previous investigations performed on the Lower Columbia River (Bi-State Study and Portland Corps 1997 Survey) also document that pesticides are typically detected only at low concentrations (Corps, 1998b). Although detection of these compounds is at a low level, it is expected that this contaminant would be readily mobilized in the finer sediments during the dam breaching alternative. Because of these compound well-documented impacts to water quality, further monitoring will be required in the future until the pesticide residues are completely out of the system.

3.3.2.4 Total Petroleum Hydrocarbons and Oil and Grease

The lower Snake River area is not heavily industrialized, and therefore huge pollution problems from oil spills and heavy industrial pollution were not expected to be as common as in the major northwest port areas of Puget Sound and Portland. At certain portions of the river there are ports and grain terminals that have heavy use during harvest season. These areas would be expected to have concentrations of residues from fueling operations and bilge wash. Some of the sediments have TOC levels combined with sulfur concentrations that interfere with traditional TPH identification tests. In some cases, because of the richness of interfering substances associated with the sediments, it is preferable to conduct a traditional oil and grease test because it could be more reliable and free from interference. In preparation for the 1998/1999 dredge material analysis, a modified version of EPA Method 418.1 was used for the analysis of the sediments to determine the concentration of petroleum products. Use of this analytical method provides an indication only of the amount of petroleum material in the sediments but does not quantitatively identify the specific type of petroleum material present. Again, a significant percentage of the samples were inconclusive because of combined interference from sulfur and TOC.

For the 1997 Feasibility Study, TPHs were analyzed in all four lower Snake River reservoirs. The concentration of TPH ranged from nondetect to 256 ppm (LM 1-2) with an arithmetic mean of 55.41 ppm (Anatek 1997). Along the lower Snake River, the average concentration of TPH generally increases in the downstream direction with the highest average reach concentration (62.13 ppm) found in Lake Sacajawea. No screening levels were established for TPH under the Portland District's Dredged Material Evaluation Framework (Corps, 1998b).

Although sporadic, at least to some degree, the following were analyzed prior to dredge operations: TPH; fats, oils, and greases (FOG); and traditional oil and grease. Unfortunately, there was little continuity in the testing and analysis, so trends are impossible to detect. Crecelius and Gurtisen (1985) reported oil and grease concentrations from sediment sites near Clarkston, Washington, on the Snake River ranging from 62 to 222 ppm. The Walla Walla District reported oil and grease in the sediments ranging from 38 to 1,096 ppm. Only two of these samples were greater than 500 ppm and these were both detected at disposal site on Lower Granite pool at Snake River RM 120. Pinza et al. (1992) reported oil and grease concentrations ranging from 12.62 to 208.70 ppm for mid Columbia and Snake River sites. Total petroleum hydrocarbons ranged from 12.20 to 96.27 ppm.

Most recently, the Walla Walla District conducted a sediment study at the confluence of the lower Snake and Clearwater rivers (Corps, 2000). The researchers sampled 38 sites for oil and grease and found concentrations ranging from 134 to 770 ppm. Only three of the sites had oil and grease concentrations greater than 400 ppm. There are insufficient data to conclude that TPH or oil and grease could pose problems. Additional hindrances to such a conclusion are changes in the testing program for these parameters and the complexity of the matrix with all of its interference from other constituents.

3.3.2.5 Polynuclear Aromatic Hydrocarbons

PAHs represent the largest class of suspected carcinogens and represent a threat to aquatic life. The PAHs typically found in dredge material in this region are most likely discharged by petroleum-fueled internal combustion engines. These compounds are found in the engine's emission as a byproduct of incomplete combustion. Another source of PAHs is the burning of coal. Coal is associated with 84 percent of the PAHs in the United Kingdom, where a majority of the power production is coal rather than hydropower (Van Metre et al., 2000).

Many PAHs present in dust and soil are known carcinogens or mutagens, and adverse health effects have been linked to exposure to these compounds. Humans can be exposed to PAH by inhaling contaminated air, by ingesting contaminated food, and by nondietary ingestion of contaminated dust or soil. Dermal contact or ingestion of such dust or soil could be more important exposure pathways in the case of children than adults because of their play activities. An interesting case study of effects on aquatic life can be found in the Black River study where Baumann et al. (1982) and Baumann and Harshbarger (1995) looked at hepatic cancer in brown bullheads residing in an impacted stream. They reported that a quarter of the sampled fish had cancer, half had neoplasm or altered hepatocytes, and less than a quarter of the fish had normal liver cells.

The first recorded sample and analysis effort for PAHs was conducted in the Port of Lewiston area in 1985. Crecelius and Gurtisen (1985) did the first serious study of PAHs in the Snake River system prior to a confluence dredging analysis. In their analysis of sediment core samples from the Port of Clarkston area, the total PAH concentration ranged from 77 ppb to 865 ppb. Because no freshwater criteria are available, the most conservative Puget Sound (Barrick et al., 1988) estuary apparent no-effect (AET) for potential inwater disposal limit criterion is used because it is the most conservative analysis tool available to date. Using the Puget Sound AET guideline, low-weight PAH ranged from 16 to 58 ppb compared to the limit of 5,200 ppb. High-weight PAH ranged from 0 to 111 ppb compared to the limit of 12,000 ppb. This suggested that there was low probability of risk associated with suction dredging at this time. However, it does suggest future efforts should require some regular monitoring for PAH contamination.

Crecelius and Cotter (1986) revisited the Lewiston area locations and found only trace amounts of low-molecular-weight PAH compounds. High-molecular-weight PAH compounds ranged from 54 ppb to 818 ppb. During this study fluoranthene was the predominant detected PAH. This compound is usually associated with other similar-weight compounds; it was unusual to detect flouranthene with little or none of the other high-molecular-weight PAHs.

The Clearwater and Snake River confluence sediments were again looked at prior to the 1987 dredging evaluation. During this investigation, no low-molecular-weight PAH compounds were detected (Corps, 1987). The PAH compounds present were predominantly pyrene and perylene. Perylene is commonly found in sediments containing substantial amounts of decaying material and was not an EPA priority pollutant in 1987. In this study, flouranthene was present but in small amounts relative to pyrene, which was the most commonly found PAH. The highest concentration of low-molecular PAH was 1,544 ppb in the disposal area. This would still allow in water disposal for total high-molecular-weight PAHs if the Puget Sound limits were in effect for fresh water at that time.

Pinza et al. (1992) conducted the next significant dredge material study again in the Snake and Clearwater confluence area. This study analyzed the 10 most common PAH compounds expected to be found in this area based on industrial and regional land use practices. Composite samples were taken from the proposed disposal site, Port of Wilma, Port of Clarkston sampling stations SRP 24 and 25, Port of Clarkston sampling stations SRP 26 and 27, Port of Lewiston sampling stations SRP 28 and 29, Port of Lewiston sampling stations SRP 30, 31, and 32, Port of Lewiston sampling stations SRP 33 and 34; and the Port of Almota. The calculated results for low-molecular-weight PAHs derived from the reported individual species were: 12.4 ppb, less than detection limit, 13.7 ppb, 10.9 ppb, less than detection limit, 15.3 ppb, less than detection limit, and less than detection limit, respectively. The calculated results for the high-molecular-weight PAHs were: 34.4 ppb, less than detection limit, 46 ppb, 15.8 ppb, less than detection limit, 25.7 ppb, 211 ppb, and less than detection limit, respectively. This study suggested PAHs were relatively low in environmental concentration and substantially less than found in the previous studies.

During the interim between the 1992 study and the very recent dredge material studies, there were no additional investigations of PAH compounds available from public accessible sources. The best data available in 1998 and 1999 are the Potlatch Corporation reports (CH2M HILL, 1998a: and 1999). CH2M HILL (1998) reported that the Lower Granite pool and Clearwater arm samples generally showed low-molecular-weight PAHs to be less than 10 ppb and high-molecular-weight PAHs to be less than 50 ppb. The exception was the Corps East Pond. The East Pond results reported 300 ppb low-molecular-weight PAHs and 492 ppb high-molecular-weight PAHs. This is borderline for the Puget Sound protocols for low-molecular-weight compounds. It is noteworthy that the East Pond sample detected every PAH compound that was analyzed. The 1999 sampling (CH2M HILL, 1999) reported slightly lower total concentrations of

PAH compound than the previous year. The East Pond still contained all species of PAHs tested, with 157 ppb and 446 ppb levels for low- and high-molecular-weight compounds, respectively.

Just prior to the completion of this document, the Corps evaluated sediments for the proposed fiscal year 2001 confluence dredging in the Lower Granite pool. Results from this investigation found phenanthrene, flouranthene, and benzo (a) pyrene in a single sample from the Clearwater at about RM 3. The calculated concentration of high-molecular-weight PAHs was 161.4 ppb (below the Puget Sound protocols). No low-molecular-weight compounds were detected.

Over a period of 15 years when the Corps dredging teams tested the compounds in the Lower Granite confluence area, there was a steady decrease of PAHs in the sediments of this area. Several factors may have a relationship to this trend:

- Reduction in automobile emissions, phasing out of tetraethyl lead fuels, better catalytic converters, better fuel economy in all classes of vehicles, and reduction of rail traffic in the area.
- Tightening of EPA air quality standards for stack emissions, phasing out of coal and bunker oil for commercial or residential heating, and a dramatic increased use of emission-free electric heat.
- Significant reduction of PAHs in the NPDES-regulated discharges from local industries and publicly owned wastewater treatment facilities.

The concentrations and distributions of PAH compounds are adequately documented in the Lower Granite pool. Almost no data exist on PAH distributions and concentrations in the Little Goose, Lower Monumental, and Ice Harbor pools. There are probably some PAH data for the Columbia River describing PAH distribution in the McNary pool but they were not available to the Corps or public entities. There is also a complete lack of knowledge of PAH compound distribution in the other three lower Snake River pools. Based on land use and volume of commercial traffic, it is rather dubious to expect a serious PAH contamination problem in the other three lower Snake River reservoirs. There is no heavy industrialization in the Lower Snake and mid-Columbia River Dredge Management Areas; and from the data collected over the last 15 years, apparent trends, and reduction of emissions overall, PAH compounds do not pose a concern compared to today's standard. However, specific dredged areas will require continued monitoring for PAHs because these areas contain the bulk of the PAH contaminants detected.

3.3.3 Metals

Each of the 94 sediment samples was analyzed for a suite of 18 metals (inorganic). The metals analyzed included: arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, molybdenum, nickel, selenium, silver, strontium, thallium, vanadium, and zinc. Of the 18 metals analyzed only cadmium, mercury, silver, and strontium were not detected in all 94 samples. Cadmium was detected in only two samples, mercury in 37 samples, silver was not detected in any of the samples, and strontium was detected in only four samples (CH2M HILL, 1998; Anatek, 1997).

The metal consistently found in the highest concentrations was manganese. This metal is commonly detected in river sediments due to its high relative abundance in the natural environment.

Concentrations of manganese in individual sediment samples collected from the lower Snake River during this investigation ranged from 250 ppm to 1,044 ppm with an average concentration of 430 ppm (Anatek 1997). In comparison, the concentration of manganese in sediment samples collected upstream of the study area by the USGS (Clark and Maret, 1998) ranged from 370 ppm to 1,000 ppm with an average concentration of 564 ppm.

No consistent trends in sediment metal concentrations were observed going downstream from Lower Granite Lake to Lake Sacajawea (Table 3-12). When compared with the results obtained by the USGS (Clark and Maret, 1998) in their investigation of the Snake River upstream of the study area several trends do become apparent. In the USGS investigation, bed sediments were collected and analyzed for a broad range of trace elements. Upstream concentrations of arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc (USGS) were lower than downstream concentrations (this investigation).

Concentration values for metals in sediments are also available for the Lower Columbia River drainage basin (Bi-State Study and 1997 Corps Survey). Of the reported values for the metals arsenic, cadmium, copper, lead, mercury, nickel, silver, and zinc in these previous investigations, only the concentrations of arsenic, manganese, and lead were found to be slightly higher for the samples collected from the lower Snake River during this investigation.

Table 3-12. Summary of Mean Metal Concentrations for Sediment Samples Collected during Phase 2 (1997) in the Lower Snake River

Metal (mg/kg)	Ice Harbor	Lower Monumental	Little Goose	Lower Granite
Arsenic	6.3	3.9	6	5.2
Barium	170.6	157.2	192.7	180.8
Beryllium	0.6	0.6	0.7	0.7
Cadmium	ND	ND	ND	0.1
Chromium	20.2	17.7	22.4	23
Cobalt	10.9	8.2	11.1	12
Copper	20.8	16.8	24.8	29.8
Lead	10.5	8.8	12.6	12.9
Manganese	510.1	384.6	475.2	408.9
Mercury	0.1	0.1	0.1	0.1
Molybdenum	0.3	0.2	0.2	0.3
Nickel	14.2	12.4	15.6	16.6
Selenium	1.6	1.4	1.3	1.5
Silver	ND	ND	ND	ND
Strontium	0.1	0.1	ND	0.1
Thallium	0.2	0.2	0.2	0.2
Vanadium	45.1	37.9	47.2	60.9
Zinc	52.5	45	57.3	61.4

Notes: ND=not detected
All concentrations in mg/kg (ppm)
Ice Harbor Lock and Dam-Lake Sacajawea
Lower Monumental Lock and Dam-Lake West
Little Goose Lock and Dam-Lake Bryan
Lower Granite Lock and Dam-Lower Granite Lake
Source: Developed by Normandeau

Crecelius et al. (1985) conducted a comprehensive evaluation of the dredge materials prior to the confluence dredging of 1986. Metals concentrations of copper, lead, zinc, and cadmium were very similar to levels found during the sediment examination for the Feasibility Study (Table 3-12) when compared to the geometric mean. The outliers were mercury and chromium. Mercury levels were lower and ranged from 0.015 to 0.049 milligrams per kilograms (mg/kg). Chromium was somewhat higher and ranged from 26 to 43 ppm. The highest readings compared to themean background levels (San Juan, 1994; Table 3-13) resulted in chromium at 3.4 times higher and mercury at 7 times higher than the eastern Washington median value. The highest mercury level in this study was twice as high as the maximum eastern Washington State natural background levels (San Juan, 1994).

Table 3-13. Comparison of Metal Concentrations in Eastern Washington (Minimum, Mean, and Maximum) to the State Wide Mean. The table was adapted using data from the San Juan (1994) report.

	Metals (ppm)			
	Median	**Minimum**	**Maximum**	**Statewide mean**
Aluminum	14,800.000	6,140.000	29,000.000	19,575.000
Arsenic	2.530	0.500	7.190	2.920
Beryllium	0.305	0.230	0.875	0.670
Cadmium	N/A	N/A	N/A	0.490
Chromium	12.600	5.000	71.300	18.420
Copper	14.700	9.100	53.000	17.070
Iron	21,300.000	10,400.000	30,000.000	22,033.000
Lead	6.400	4.200	11.700	7.900
Manganese	345.000	223.000	652.000	509.580
Mercury	0.007	0.004	0.025	0.020
Nickel	11.700	6.400	34.100	16.430
Selenium	N/A	N/A	N/A	0.525
Silver	N/A	N/A	N/A	0.037
Tin	N/A	N/A	N/A	4.000
Zinc	41.00	26.300	82.300	51.120

N/A = not applicable
Source: Developed by the Corps

The Walla Walla District 1988 Interim Flood Control Dredging (Corps, 1987) study reported concentrations of sediment metals as follows: arsenic 2.6 to 12.6 ppm, copper 17 to 48 ppm, lead 13 to 27 ppm, mercury 0.018 to 0.186 ppm, and zinc 77 to 138 ppm. Cadmium ranged from 0.075 to 1.02 ppm, which is below Puget Sound Screening level of 5.1 ppm. Maximum metal concentrations from this study compared to mean background levels (San Juan, 1994) were: 3.3 times higher for zinc; 26.5 times higher for mercury; 5.6 times higher for copper; 16 times higher for arsenic; and 4.2 times higher for lead.

The 1992 sediment sampling of the dredge sites on the Snake and Clearwater Rivers (Pinza et al., 1992) reported concentrations as follows: arsenic from 1.11 to 9.46 ppm; cadmium from 0.2 to 1.6 ppm; chromium from 6.6 to 23.4 ppm; copper from 6.9 to 38.8 ppm; lead from 2.5 to 20.8 ppm and mercury from 0.06 to 0.20 ppm. Concentrations of zinc ranged from 26 to 78.7 ppm with the exception of one site that reported a concentration of 277 ppm.

For the 1996/1997 Confluence Dredging in the lower Snake and Clearwater rivers, sediments were sampled and tested for Resource Conservation and Recovery Act metals (Corps, 1996). In this study the sediment concentrations ranged as follows: aluminum 12,200 to 20,300 ppm; arsenic 1.25 to 4.36 ppm; barium 135 to 234 ppm; beryllium 0.5 to 0.71 ppm; chromium 12.2 to 18.2 ppm; cobalt 9.97 to 13.4 ppm;

copper 17.3 to 34.9 ppm; lead 6.79 to 10.9 ppm; manganese 259 to 580 ppm; molybdenum 0.29 to 1.35 ppm; nickel 10.3 to 13.5 ppm; selenium 1.29 to 2.17 ppm; thallium 0.15 to 0.19 ppm; vanadium 52.1 to 68.7 ppm; and zinc 38.4 to 69.8 ppm. The aluminum detected in this study was above the mean background levels (San Juan, 1994) in a significant number of the samples. In this study, comparison of the highest concentrations to the mean background level for eastern Washington were as follows: copper was about twice as high, arsenic was about twice as high, lead was one and a half times higher, but mercury was not detected in any of these samples.

In their reported values for the metals arsenic, cadmium, copper, lead, mercury, nickel, silver, and zinc, only the concentrations of arsenic and lead were found to be slightly higher for the samples collected from the lower Snake River during this investigation. The authors suggested that the major source of metals except arsenic and lead is from the lower Snake River basin.

Sediments were tested in the Lower Granite pool and the Clearwater River in depositional areas upstream and downstream from the Potlatch Plant outfall pipe. Sediment metal concentrations for both years ranged from: arsenic 1.5 to 13 ppm, cadmium 0.08 to 0.56 ppm, chromium 8.7 to 39 ppm, copper 13 to 55 ppm, lead 4.2 to 34 ppm, nicke19 to 18 ppm, selenium 1 to 2 ppm, and zinc 32 to 249 ppm. In the 1999 report, aluminum ranged from a low of 6,623 ppm to a high of 29,367 ppm. The highest concentration of aluminum was found in the Corps East Pond. The level in the East Pond was about twice the background levels (San Juan, 1994). Lead and chromium were also found above background levels, with the highest number found in the East Pond. The highest copper detection is slightly higher than the highest reading reported in the eastern Washington background samples. In all of the samples from the Potlatch 1999 study, mercury was a nondetected constituent.

The Walla Walla District June 2000 sediment study in the lower Snake and Clearwater rivers tested for metals in 32 sample sites. Results for this study were as follows: antimony all below detection limits; aluminum 232 to 7,885 ppm; barium 2.2 to 108 ppm; beryllium all nondetects; calcium 12.77 to 37.87 ppm; cadmium 0.122 to 1.058 ppm; chromium 1.20 to 9.13 ppm; cobalt 1.099 to 9.573 ppm; copper 2.22 to 44.33 ppm; iron 1.242 to 15,529 ppm; magnesium 9.123 to 405 ppm; manganese 10.97 to 4,009 ppm; molybdenum all nondetects; nickel 1.921 to 9.478 ppm; potassium 6.529 to 2,023 ppm; sodium 5.623 to 253 ppm; vanadium 1.292 to 65.22 ppm; zinc 1.090 to 41.56 ppm; mercury all nondetects; and lead 1.109 to 8.353 ppm. In these samples, chromium and aluminum were detected in significantly lower quantities than in previous studies. These parameters were also well below the expected average background levels. On the other hand, manganese was detected at chronic levels with the highest sample at 11 times more than background levels (San Juan, 1994). Cadmium was twice as high as the state median background level. The maximum copper concentration was two and a half times higher than the eastern Washington background. There are no published state background levels for vanadium. But there is a median datum published for the Spokane River basin, and maximum lower Snake River vanadium is more than twice the median in the Spokane River basin.

Studies of metals conducted in the 1980s contained significant levels of metals-well above the 1994 background levels. For the most part, metals detected in the current studies, the 1997 Feasibility Study, and current evaluations agreed with each other and were within the range of the expected background levels. There were some notable exceptions and these will be discussed further.

The most notable of these differences is manganese. Manganese concentrations appear to be higher in Snake River sediments than in Clearwater River sediments. Manganese concentrations were highly variable but each successive year of testing yielded a higher maximum concentration. The high manganese concentrations occurred in results from several laboratories, suggesting that a procedural error is unlikely. Some analyses were by atomic absorption spectrometry while others were conducted by inductively coupled plasma mass spectrometry, suggesting a method error was unlikely. At this time there is no explanation for this occurrence. Fractional isotope analysis could provide clues by determining what species of manganese salts and proportion to geologic material is present.

Lead and arsenic appear in quantities above background level (San Juan, 1994) in previous studies. Currently some of the highest levels are present in the Corps East Pond in Lewiston. Lead and arsenic were also used as paint pigments around the turn of the 20th century (Scott, 1887). Lead carbonate was also used as the primary white paint pigment and surface primer until the EPA banned it in 1978 (U.S. Department of Housing and Urban Development, 1990). Sparse population in this region suggests that this source, like tetraethyl lead (Nriagy, 1990), would not be sufficient to contribute greatly to the totals found in sediments.

Lead and arsenic were components of the chemical insecticide first manufactured around 1867 under the name Paris green. Paris green was first used to kill rats in the Paris sewers and later was widely used in orchards prior to World War II. Paris green proved to be an effective and persistent biocide. Other inorganic pesticides, including copper sulfate, lead, arsenic, sodium arsenate, and sodium dichromate, became widely available in the 19th and early 20th centuries (Clarkson, 1991; Stevens and Sumner, 1991). The organo-metallic compounds (organic chemicals containing arsenic, mercury, tin, and others) followed these, most of which remained in use until after World War II; a few organo-arsenicals are still used (Clarkson, 1991).

Since there are few industrial sources of pollution in the area, the source most likely responsible for above background levels for lead and arsenic is past agricultural practices in areas historically containing orchards. The Otis Orchards suburb of Lewiston, Idaho, was once teeming with orchard-grown crops. Old pre-impoundment USGS 7.5-minute quad maps show orchards where there are now reservoir waters.

Aluminum levels are highly variable and have been found in orders of magnitude lower than background (San Juan, 1994) and levels as high as twice background. The higher levels appear to be found predominantly in areas with higher percentage of fine sediments. Aluminum has had a checkered past. It was first considered a toxic agent 50 years ago but its toxic properties were dismissed about 15 years ago (Hewitt et al., 1990). It is again under scientific scrutiny as a possible toxic agent in the etiology of Alzheimer's disease, Guamiam amyotropic lateral sclerosis, and Parkinsonism-dementia (Hewitt et al., 1990).

Aluminum toxicity in the aquatic environment is mediated by pH (increases of acidity effect toxicity) (Verbost et al., 1992). Aluminum is acutely toxic to fish in acidic waters (Exley et al., 1991). Insufficient data are available on aluminum distribution or potential sources for elevated levels. Based on research of the scientific literature, the only potential anthropogenic source with a possible fit is a pesticide called aluminum phosphide sold under the trade names Phostoxin and Weevilcide. It is used to control rodents as a fumigant or bait at crop transport, storage, and processing facilities, ship-holds, or railcars (Royal Society of Chemistry, 1991; EPA, 1992). Whatever the case, aluminum is considered to be a potential problem for water quality in this area and will be examined in further dredge material activities.

Copper is an essential nutrient and is required for the proper functioning of many important enzyme systems (Linder and Hazegh-Azam, 1996). In eastern Washington, the most prevalent use of copper next to wire and currency is the algaecide copper sulfate. Copper sulfate, under various formulations and trade names, is the most widely used chemical for the control of algae in farm ponds (MacKay, 2000). It is cheap, effective, and can be used on ponds that provide animals with drinking water. Copper sulfate is also used as a fugalcide for a wide variety of fruit products (Old Bridge Chemicals, 2000). Copper sulfate is very toxic to fish. Its toxicity to fish varies from species to species and the chemical characteristics of the water they reside in (Pimentel, 1971). Fish eggs are more resistant than young fish fry to the toxic effects of copper sulfate (Gangstad, 1986). Copper sulfate significantly reduces the populations of aquatic invertebrates (EPA, 1986b).

Metals pose a significant contamination problem. In this area of eastern Washington and eastern Oregon, the primary impacts of metals would be from past mining and agricultural practices. Very little if any manufacturing has occurred here in the last 100 years. One interesting aspect of the metals concentrations found on the lower Snake River is that they can appear as a substantial contaminant (tenfold over background average) one year and prove to be well below background level in subsequent years. Then, the metal of concern may be found at a different location in quantities well above background levels at a site where it was at nondetect levels a few years earlier. Any future endeavors in these pools will require a thorough and comprehensive examination for metal contamination of the sediments. This would include further dredging and the dam breaching alternative.

3.3.4 Sediment Nutrients

For the Lower Snake River Feasibility Sediment Study 84 of the sediment samples were also analyzed for a number of chemical parameters, designated as the nutrient group (although not all of the parameters are true nutrients). The sediments were analyzed for: ammonia, total Kjeldahl nitrogen (TKN), nitrogen as nitrate/nitrite, total organic nitrogen, total organic matter, pH, phosphorus bicarbonate, and sulfate. The mean reach concentrations for each of the nutrient group parameters are summarized in Table 3-14. No screening levels have been established under the DMEF (Corps, 1998b) for nutrients, and comparison with water quality standards is not appropriate.

Table 3-14. Summary of Mean Nutrient Concentrations for Sediment Samples Collected (1997) in the Lower Snake River

Parameter	Ice Harbor	Lower Monumental	Little Goose	Lower Granite
Ammonia	81.3	59.6	64.3	75.7
Total Kjeldahl Nitrogen	1,317.1	1,146.1	1,344.1	1,746.5
Nitrate/Nitrite	0.7	0.6	0.7	1.4
Total Organic Nitrogen	1,235.7	1,086.7	1,280	1,671.3
Total Organic Matter (percent)	2.5	2.2	3.3	5.2
Phosphorus Bicarbonate	37.7	38.2	35	34.1
Sulfate	7.7	8.4	10.5	17.9
pH (standard units)	6.9	6.9	7.1	6.8

Notes: All results in mg/kg unless otherwise noted
Ice Harbor Lock and Dam – Lake Sacajawea
Lower Monumental Lock and Dam – Lake West
Little Goose Lock and Dam – Lake Bryan
Lower Granite Lock and Dam – Lower Granite Lake
Source: Developed by Normandeau

3.3.5 Elutriate Fraction

For each of the sediment samples, an ambient pH elutriate was prepared and analyzed for organophosphorus pesticides, organochlorine pesticides, metals, and nutrients, glyphosate, and AMPA. TPH and dioxin were not tested in the ambient pH elutriates. The purpose of the elutriate tests was to evaluate potential impacts on surface water quality from the resuspension of channel sediment. The elutriate tests were used to determine which inorganic or organic constituents would preferentially partition by dissolution into the water and to determine their resulting aqueous concentration. The elutriate concentrations (maximum values) were then compared with applicable surface water quality standards to identify the Chemicals of Concern (CoC). The results of the laboratory analyses for the ambient pH elutriates, which are summarized in Tables 3-15 and 3-16, are presented in Anatek (1997). Results include the number of samples analyzed, the number of samples above detection limits, the minimum value and maximum value detected, the arithmetic and geometric mean, and the standard deviation for each parameter analyzed.

Table 3-15. Summary of Mean Metal Concentrations for Ambient pH Elutriate Samples Collected (1997) of the Lower Snake River Project

Metal (µg/L)	Ice Harbor	Lower Monumental	Little Goose	Lower Granite
Arsenic	3.9	2.6	2.2	1.8
Barium	243.6	197.5	140.9	83.3
Beryllium	ND	ND	ND	ND
Cadmium	ND	ND	0.1	ND
Chromium	0.6	0.8	0.4	0.6
Cobalt	0.5	1.2	0.4	0.5
Copper	2.9	3.2	3.2	4
Lead	ND	0.1	0.1	0.1
Manganese	861.5	1,432.1	799.9	504.4
Mercury	ND	0.1	0.1	0.1
Molybdenum	3	3.5	3.8	2.2
Nickel	2.8	4.1	0.7	0.9
Selenium	2.3	1.2	0.3	0.3
Silver	ND	ND	ND	ND
Strontium	0.4	0.3	0.3	0.2
Thallium	ND	ND	ND	ND
Vanadium	2.1	1.2	1.8	1.5
Zinc	37.7	17.8	16.9	12.9

Notes: ND=not detected
Ice Harbor Lock and Dam – Lake Sacajawea
Lower Monumental Lock and Dam – Lake West
Little Goose Lock and Dam – Lake Bryan
Lower Granite Lock and Dam – Lower Granite Lake
Source: Developed by Normandeau

Table 3-16. Summary of Mean Nutrient Concentrations for Ambient pH Elutriate Samples Collected during Phase 2 (1997) in the Lower Snake River

Parameter (milligrams per liter)	Ice Harbor	Lower Monumental	Little Goose	Lower Granite
Ammonia	3.6	2.5	2.6	3.6
Total Kjeldahl Nitrogen	8.8	5.7	4.1	6.2
Nitrate/Nitrite	0.2	0.2	0.3	0.4
Phosphate	0.1	0.1	0.1	0.1
Sulfate	19.6	17.9	26.9	29.7

Note: Ice Harbor Lock and Dam – Lake Sacajawea
Lower Monumental Dam – Lake West
Little Goose Lock and Dam – Lake Bryan
Lower Granite Lock and Dam – Lower Granite Lake
Source: Developed by Normandeau

3.3.5.1 Organophosphorus Pesticides

The ambient pH elutriates were tested for the presence of organophosphorus pesticides, which as a group consist of 25 different organic compounds. The only organophosphorus pesticide detected was ethyl parathion, in one sample (Little Goose 8-4), at a concentration of 1.0 ppb (µg/L). Although identified in the one elutriate sample, ethyl parathion was not detected in any of the sediment samples. Parathion is a regulated substance in fresh waters in the states of Oregon and Washington with a maximum allowable concentration of 0.013 ppb (chronic).

3.3.5.2 Organochlorine Pesticides

No organochlorine pesticides were detected in any of the ambient pH elutriate samples. The organochlorine pesticides DDT (and its metabolites), aldrin, dieldrin, endrin, heptachlor and lindane had been detected in several of the sediment samples tested. The results of the elutriate tests suggest that although these compounds are present in the sediments they do not readily partition into water.

3.3.5.3 Glyphosate

Glyphosate was detected in only 2 of the 94 ambient pH elutriate samples, while AMPA was not detected. Glyphosate was detected at a concentration of 0.69 µg/L in a sample collected from Lake Bryan and at a concentration of 0.58 µg/L in a sample collected from Lake Sacajawea (Table 3-10). In comparison, the maximum contaminant level established for glyphosate by the EPA in drinking water is 700 µg/L, well above the concentrations detected in the two elutriate analyses.

3.3.5.4 Metals

Each of the 94 ambient pH elutriates were tested for the same suite of metals that were analyzed on their corresponding sediments. The results of the individual samples are summarized in a table included in Normandeau (1999b). For the 18 metals analyzed only beryllium, silver, and thallium were not detected in the elutriate samples. Of these metals only silver was not detected in the original sediment samples.

The mean metal concentrations for the ambient pH elutriates are summarized by river reach in Table 3-15. The predominant metals detected include barium and manganese. The average concentration of barium, by river reach, in the ambient pH elutriates increases from 83.3 ppb for the samples collected from Lower Granite Lake to 243.6 ppb for the sediment samples collected from Lake Sacajawea. Although a corresponding trend in the concentration of barium in the sediment samples was not observed, it was one of the predominant metals detected. Its relatively high concentration in the ambient pH elutriates is most likely the result of its concentration in the sediments and its relatively high solubility in water (Hem, 1989).

The predominant metal identified in the ambient pH elutriates was manganese (Table 3-11). The average concentration of manganese, by river reach, in the ambient pH elutriates ranged from 504 ppb for the samples collected from Lower Granite Lake to 1,432 ppb for the samples collected from Lake West. In general, the trend in manganese concentrations in the ambient pH elutriate samples increases with distance downstream. As observed with barium, there does not appear to be a clear relationship between the concentration of manganese in the sediment samples and in the ambient pH elutriates.

The maximum metal concentrations detected in the ambient pH elutriates (Anatek, 1997) were also compared with the recommended surface water quality standards of the State of Oregon Department of Ecology, the United Nations (agricultural water quality goals), EPA, and Ecology to identify any chemicals of concern. The maximum concentration of four metals: arsenic, copper, manganese, and mercury-were found to exceed their applicable water quality standards.

Because these metals also occur naturally in the environment, their concentrations were compared with representative background values to determine if they represent a concern. The results of the ambient pH elutriate tests were compared with historical water quality data collected by the USGS from the Snake River near Anatone, Washington. The maximum detected concentrations of arsenic, copper, and mercury were found to be less than their average background concentrations and as a result were not considered to be of concern.

3.3.5.5 Nutrients

The ambient pH elutriate samples were also analyzed for the following nutrients: ammonia, nitrate/nitrite, phosphate, sulfate, and Total Kjeldahl Nitrogen (TKN) (Cascade Analytical, 1997). The mean concentration of each of these nutrients for the four reaches along the lower Snake River are summarized in Table 3-16.

The dominant form of nitrogen found in the elutriate samples was ammonia, which was also the dominant form of nitrogen identified in the sediment samples. The dominance of ammonia may reflect the limited oxygen environment of the channel bed sediments as a result of the decomposition of organic material. The consumption of oxygen by the decay of organic material would lead to the reduction of nitrate/nitrite to amonia, thus limiting their concentrations in both the sediment and elutriate samples.

Concentrations of ammonia in sediment elutriate and in ambient river are summarized in Figure 3-46. These data indicate that erosion and suspension of sediments can substantially elevate ammonia in the water column above ambient levels. Although elevated ammonia levels are expected to be transient, they nevertheless could affect aquatic life.

Total ammonia in fresh water exists as two chemical species; un-ionized ammonia (NH_3) and ammonium ion (NH_4^{+1}). Toxicity is primarily attributed to the un-ionized ammonia. In fresh water, the concentration of the un-ionized form is a function of temperature and pH. EPA (1999) provides a detailed discussion of the dependence of ammonia toxicity on temperature and pH. Because un-ionized ammonia is difficult to measure directly, its acute[1] and chronic[2] effects can be expressed in terms of the total ammonia concentrations calculated for site-specific values of temperature and pH. In addition to dependence on pH and temperature, EPA (1999) has shown that salmonids and early life stages of aquatic organisms are especially sensitive to ammonia. A listing of critical criteria continuous concentration (CCC) values for salmonids and early life stages at various pH and temperature conditions is provided in Table 3-17.

Potential ammonia toxicity associated with the resuspension of sediments is dependent on seasonal conditions of pH and temperature as shown in Figure 3-46. When temperatures are 14°C (57.2°F) or less, pH in the lower Snake River ranges from 7.3 to 7.6 and corresponding CCC values for ammonia range from 4.0 to 5.1 mg/L.[3] Under these conditions, the average ammonia levels predicted by sediment elutriate measurements are below the critical CCC levels in all four reaches of the lower Snake River. When temperatures exceed 14°C (57.2°F), pH in the lower Snake River ranges widely from 6.4 to 8.9 and CCC values for ammonia range from 0.5 to 4.8 mg/L. Thus, warmer water and higher pH values result in average predicted ammonia concentrations that would frequently exceed the critical CCC value in the various reach of the lower Snake River.

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[1] Expressed as the criteria maximum concentration, which is a one-hour average acute limit that protects aquatic life from short-term exposure to relatively high concentrations.
[2] Expressed as a criteria continuous concentration, which is a four-day average chronic limit that provides protection of aquatic life and its uses.
[3] Values are expressed to three significant figures to minimize rounding errors.

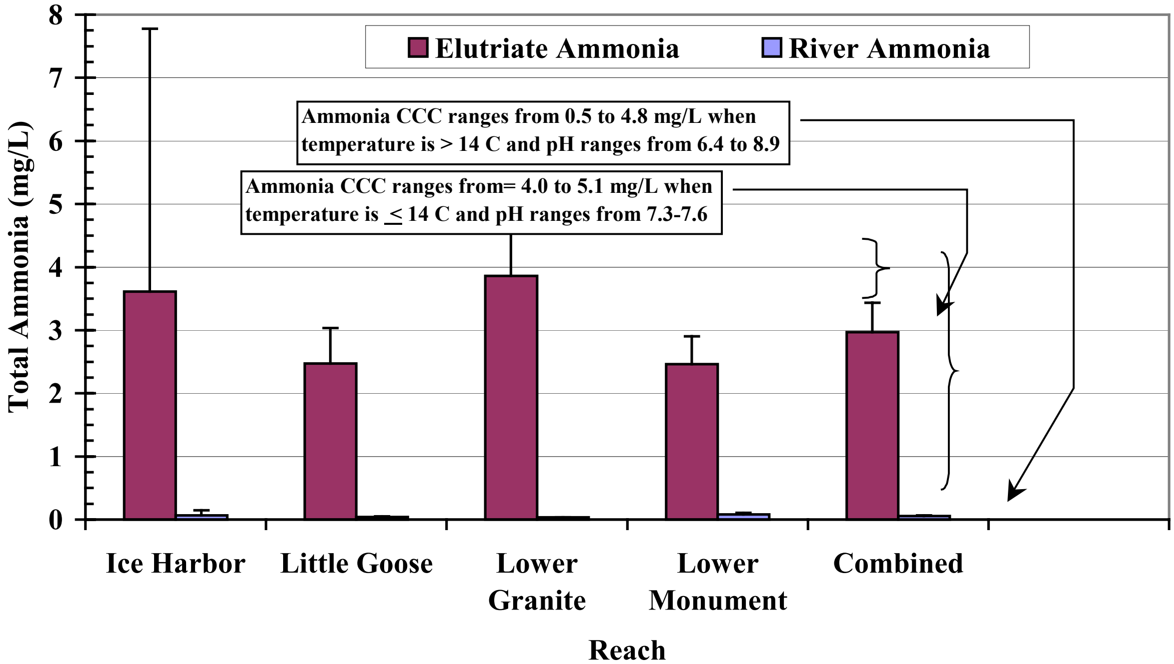

Figure 3-46. Mean Concentrations and 95 Percent Confidence Limits of In-river Water and In-sediment Elutriate at Ambient pH

Table 3-17. Critical Values for Total Ammonia CCC Values That are Protective of Salmonids and Sensitive Life Stages

	pH										
Temperature (°C)	**6.5**	**6.6**	**6.7**	**6.8**	**6.9**	**7**	**7.1**	**7.2**	**7.3**	**7.4**	**7.5**
1	6.67	6.57	6.44	6.29	6.12	5.91	5.67	5.39	5.08	4.73	4.36
14[1/]	6.67	6.57	6.44	6.29	6.12	5.91	5.67	5.39	5.08	4.73	4.36
15	6.46	6.36	6.25	6.10	5.93	5.73	5.49	5.22	4.92	4.59	4.23
16	6.06	5.97	5.86	5.72	5.56	5.37	5.15	4.90	4.61	4.30	3.97
17	5.68	5.59	5.49	5.36	5.21	5.04	4.83	4.59	4.33	4.03	3.72
18	5.33	5.25	5.15	5.03	4.89	4.72	4.53	4.31	4.06	3.78	3.49
19	4.99	4.92	4.83	4.72	4.58	4.43	4.25	4.04	3.80	3.55	3.27
20	4.68	4.61	4.52	4.42	4.30	4.15	3.98	3.78	3.57	3.32	3.06
21	4.39	4.32	4.24	4.14	4.03	3.89	3.73	3.55	3.34	3.12	2.87
22	4.12	4.05	3.98	3.89	3.78	3.65	3.50	3.33	3.13	2.92	2.69
23	3.86	3.80	3.73	3.64	3.54	3.42	3.28	3.12	2.94	2.74	2.53

[1/] When temperature is ≤ 14°C, the method of calculating CCC values yields identical results for each pH concentration regardless of temperature.

These comparisons indicate that seasonally dependent conditions of ambient pH and temperature and their influence on potential toxicity of ammonia will need to be factored into environmental management decisions. For example, the upper 95 percent confidence limit[4] for total ammonia in sediment elutriate for the Lower Granite reach is 4.9 mg/L. Hence, the critical value (i.e., the CCC) for ammonia should be higher than 4.9 mg/L to be reasonably certain that ammonia toxicity does not occur to salmonids or sensitive life stages as a result of sediment resuspension. Inspection of Table 3-17 indicates that a critical CCC value of 4.9 mg/L can be obtained when temperatures are less than or equal to 16°C and pH is less than or equal to 7.3. This critical value could also be obtained at higher temperatures (e.g., 19°C) but only if ambient pH was in the range of 6.5 to 6.6.

3.3.6 Additional Sediment Constituents for which There are No Data

The provisional information compiled to make this list of compounds was derived from U.S. Department of Agriculture and USGS sources. Geographical Information Systems maps were used to determine which compounds were used in the specific drainage areas. The compounds listed below (Table 3-18) were added to the list based on heavy and moderate use rated against the USGS compiled national average (USGS estimated annual agricultural usage). The terms "heavy or moderate use" are somewhat subjective since they were based on color codes rather than geometric means when compared to the national average range bands used in the USGS Pesticide National Synthesis Project. This list is by no means quantitative or inclusive but it is the first attempt to define a material test regime based on the newer and more commonly used chemicals of the region. Table 3-18 lists the heavy and moderate use of compounds in the lower Snake River basin.

3.4 Primary Productivity/Food Web Complex

Biological productivity data also were collected throughout the study area as part of the recent sampling efforts. These biological productivity data were collected at both impounded and free-flowing reaches within the study area to compare differences between the two types of aquatic environments and to evaluate whether these data may be useful in predicting changes in biological productivity under the dam breaching and flow augmentation alternatives. Aquatic plant growth, particularly algae, which convert sunlight into energy, represents the primary producer in aquatic systems. Aquatic plants generally fall into three major categories: phytoplankton, ABA, and macrophytes. Phytoplankton refers to free-floating or suspended algae in the water column. ABA refers to unicellular and filamentous forms of algae that attach to rocks and other hard substrate in water depths where sunlight penetrates to the bottom (i.e., photic zone). Both ABA and phytoplankton represent the base of the food chain and are an important food source for zooplankton, benthic animals (i.e., crayfish, amphipods, oligochaetes), aquatic insects, and benthivorous fishes. Chlorophyll *a* is often used as an indirect measure of phytoplankton and ABA biomass, because it is generally highly correlated with algal biomass. Macrophytes consist of the larger rooted plants that grow in shallow water, typically up to 2 meters in depth, along the shorelines of lakes and backwater areas. Although macrophytes are less important as a food source, they provide important shelter areas for insects as well as fish. Additionally, macrophytes help stabilize shorelines by reducing flow velocities and they also recycle nutrients through plant uptake.

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[4] The 95 percent upper confidence limit is a statistical estimate of how high the average concentration could be with repeated sampling. By definition there is only a 5 percent probability that the true mean would be higher than that indicated by the upper 95 percent confidence limit.

Table 3-18. Compounds with Heavy and Moderate Use in Lower Snake River Basin, Mid-Lower Columbia Reach, Eastern Oregon, and Idaho (page 1 of 3)

Heavy Use in the Lower Snake River Basin			
1,3-D	2,4-DB	Benomyl	Bromoxynil
Chloropicrin	Chlorothalonil	Chlorpyrifos	Chlorsulfuron
Clopyralid	Copper Sulfate	Cytokinins	DCPA
Diazinon	Dichlobenil	Dicloflop	Dicofol
Difenzoquat	Dimethoate	Dimethomorph	Diquat
Disulfoton	Diuron	EPTC	Endothall
Endosulfan	Ethalfluralin	Ethion	Ethoprop
Fenarimol	Gibberellic Acid	Imidacloprid	Iprodione
MCPB	Malthion	Mancozeb	Metalaxyl
Metam Sodium	Methamidophos	Methoxychlor	Metiram
Metribuzin	Metasulfuron	Mevinphos	Myclobutanil
NAA	NAD	Oxamyl	Oxyfluorfen
Oxythioquinox	Phorate	Phosmet	Pronamide
Propamocarb	Propargite	Streptomycin	Terbacil
Thiabendazole	Thifensulfuron	Thiophanate methyl	Triadimefon
Triasulfuron	Tribenuron	Triphenyltin HYD	Vinclozolin
Ziram			
Moderate Use in the Lower Snake River Basin			
Alachlor	Aldicarb	Bentazon	Chloramben
DCNA	Dinocap	Dodine	Esfenvalerate
Ethyl Parathion	Formatanate HCL	Hexazinone	Imazethapyr
Maleic Hydrazide	Maneb	Metaldehyde	Methidathion
Methyl Parathion	Naled	Norflurazon	Oxydemeton methyl
Oxytetracycline	Paraquat	Permethrin	Sethoxydim
Simazine	Triallate	Trifluralin	Triforine
Vernolate			
Heavy Use in Mid-Lower Columbia River Reach			
Abemectin	Amitraz	Azinphos Methyl Benomyl	
Bifenthrin	Bromoxynil	Captan	Carbaryl
Chlorpyrifos	Copper Sulfate	Cytokinins	DCNA
DCPA	Diazinon	Dichlobenil	Dicloflop
Difenzoquat	Dimethoate	Dimethomorph	Dinocap
Diquat	Disulfoton	Diuron	Dodine
EPTC	Endosulfan	Ethalfluralin	Ethion
Ethoprop	Fenarimol	Formatanate HCL	Gibberellic Acid
Heavy Use in Mid-Lower Columbia River Reach (continued)			
Imidacloprid	Iprodione	MCPB	MCPA
Malathion	Mancozeb	Maneb	Metalaxyl
Metam Sodium	Methamidophos	Methidathion	Methoxychlor
Methyl Parathion	Methomyl	Metiram	Metribuzin
Mevinphos	Myclobutanil	NAA	NAD
Oryzalin	Oxamyl	Oxyfluorfen	Oxytetracycline
Oxythioquinox	Permethrin	Phorate	Phosmet
Propamocarb	Propargite	Streptomycin	Sulfur
Terbacil	Thiabendazole	Thifensulfuron	Thiophanate methyl
Triadimefon	Triasulfuron	Tribenuron	Triphenyltin HYD
Vinclozolin	Ziram		

Table 3-18. Compounds with Heavy and Moderate Use in Lower Snake River Basin, Mid-Lower Columbia Reach, Eastern Oregon, and Idaho (page 2 of 3)

Moderate Use in the Mid-Lower Columbia Reach			
Acephate	Atrazine	Bentazon	Chlorothalonil
Chlorsulfuron	Clopyralid	Cyanazine	Cyfluthrin
Cypermethrin	Dicofol	Esfenvalerate	Ethephon
Ethyl Parathion	Fenarimol	Hexazinone	Linuron
Maleic Hydrazide	Metaldehyde	Metolachlor	Metribuzin
Naled	Napropamide	Norflurazon	Oxydemeton methyl
Paraquat	Pendimethalin	Pronamide	Simazine
Trifluralin			
Heavy Use in Eastern Oregon			
Benomyl	Bensulide	Chlorsulfuron	Clopyralid
Copper Sulfate	Cycloate	Cytokinins	DCPA
Dicloflop	Difenzoquat	Dimethomorph	Disulfoton
Diuron	Endothall	Gibberellic Acid	Iprodione
MCPB	MCPA	Mancozeb	Metalaxyl
Metam Sodium	Oxamyl	Oxydemeton methyl	Phorate
Propamocarb	Propargite	Propiconazole	Streptomycin
Sulfur	Terbacil	Thifensulfuron	Thiophanate methyl
Triadimefon	Triclopyr	Tribenuron	Ziram
Moderate Use in Eastern Oregon			
Aldicarb	Atrizine	Azinphos Methyl Carbaryl	
Chlorpyrifos	Cymoxanil	Cypermethrin	Desmedipham
Diazinon	Dichlobenil	Dicofol	Dimethoate
Dinocap	Diquat	Dodine	EPTC
Endosulfan	Ethofumesate	Ethoprop	Ethyl Parathion
Fenarimol	Fluazifop	Fonofos	Gibberellic Acid
Imidacloprid	Lamdacyhalothrin	Linuron	Malathion
Maleic Hydrazide	Maneb	Metaldehyde	Methamidophos
Methidathion	Methoxychlor	Metribuzin	Mevinphos
Muclobutanil	NAA	Naled	Oryzalin
Oxyfluorfen	Pendimethalin	Permethrin	Phosmet
Pronamide	Propachlor	Pyrazon	Sethoxydim
Simazine	Thiabendazole	Triasulfuron	Trichlorfon
Trifluralin	Triforine	Triphenyltin HYD	Vinclozolin
Heavy Use in Idaho			
1,3-D	Bensulide	Bifenthrin	Bromoxynil
Chloropicrin	Chlorothalonil	Copper Sulfate	Cycloate
Dicloflop	Dicofol	Difenzoquat	Diuron
Ethalfluralin	Ethion	Iprodione	MCPA
Mancozeb	Metalaxyl	Metam Sodium	Metribuzin
Oryzalin	Oxydemeton methyl	Phorate	Sulfur
Terbacil	Thifensulfuron	Tribenuron	Vinclozolin

Table 3-18. Compounds with Heavy and Moderate Use in Lower Snake River Basin, Mid-Lower Columbia Reach, Eastern Oregon, and Idaho (page 3 of 3)

Moderate Use in Idaho

2,4-DB	Azinphos Methyl	Bentazon	Chlorpyrifos
Chlorsulfuron	Clopyralid	Cyanazine	DCPA
Desmedipham	Diazinon	Diquat	Disulfoton
Dodine	EPTC	Endosulfan	Esfenvalerate
Ethofumesate	Ethyl Parathion	Hexazinone	Imazamethabenz
MCPB	Malathion	Maleic Hydrazide	Methamidophos
NAA	Naled	Norflurazon	Pendimethalin
Permethrin	Phosmet	Picloram	Pronamide
Propargite	Pyrazon	Sethoxydim	Simazine
Thiophanate methyl	Trifluralin	Tribenuron	Triphenyltin HYD
Ziram			

Note: The list of pesticides in Table 3-17 was compiled by applying USGS color code in the USGS Pesticide Annual Use Maps (USGS, 1998) as a criterion. The colors range from warm to cold hues with warm being the heaviest use to blue being the slightest use. White corresponds to no use, no data, or no estimated use. Maps of annual pesticide use have been compiled for 208 compounds used in U.S. crop production. The maps are based on pesticide use rates compiled by the National Center for Food and Agricultural Policy (NCFAP) from pesticide use information collected by state and federal agencies over a four year period (1990 to 1993 and 1995), and on crop acreage data obtained from the 1992 Census of Agriculture. The NCFAP database contains state-based estimates of pesticide use rates for 208 compounds and 87 crops. For each of the compounds, NCFAP has developed two use coefficients, the percent of acres treated for 87 specific crops and the pounds of an active ingredient applied annually to each acre of that crop. State use coefficients for 200 of these compounds are contained in "Pesticide Use in U.S. Crop Production—National Data Report" and information for eight additional compounds (acetochlor, cymoxanil, dimethanamid, dimethomorph, flumetsulam, imidacloprid, propamocarb, and tebufenozide) were supplied by written communication. Pesticide National Synthesis has applied the NCFAP state-based pesticide use coefficients to county-level crop acreages obtained from the 1992 Census of Agriculture to produce maps showing the distribution of average annual pesticide use. County-level estimates of pesticide use were derived for over 18,000 possible compound and crop combinations found within the United States. For each of the crops that NCFAP surveyed within a state, a county acreage was obtained from the 1992 Census of Agriculture. These county crop acreages ages were then combined with the state use coefficients developed by NCFAP (percent of a crop's acres treated in a state and the average annual application rate of the active ingredient per treated acre) to calculate county-level pesticide usage by compound and by crop. Maps of average annual pesticide use by compound show the estimated pounds of a compound applied per square mile in a county (sum of pounds of a compound applied to all crop acres treated/total county area). In order to make the maps comparable, the same five intervals representing the pounds of an active ingredient applied/square mile have been used for all compounds.
Data Source: USGS, 1998.
USGS Source; Corps, Walla Walla District

Other biological productivity data collected included the sampling of zooplankton and macroinvertebrates. Zooplankton are tiny, floating animals that feed principally on algae and are an important food source for larger aquatic organisms such as snails and small fish (Wetzel, 1983). Data for macroinvertebrates were collected at a limited number of sites throughout the study area.

The existing food web in the lower Snake River is driven by phytoplankton primary producers, with small contributions from ABA (Figure 3-47). Zooplankton compose the primary herbivore population; they, in turn, are consumed by planktivorous or plankton-eating fish. Aquatic insects are of lesser importance and are a food source for bottom-feeding fish. Piscivores, organisms that eat fish, form the top of the food web.

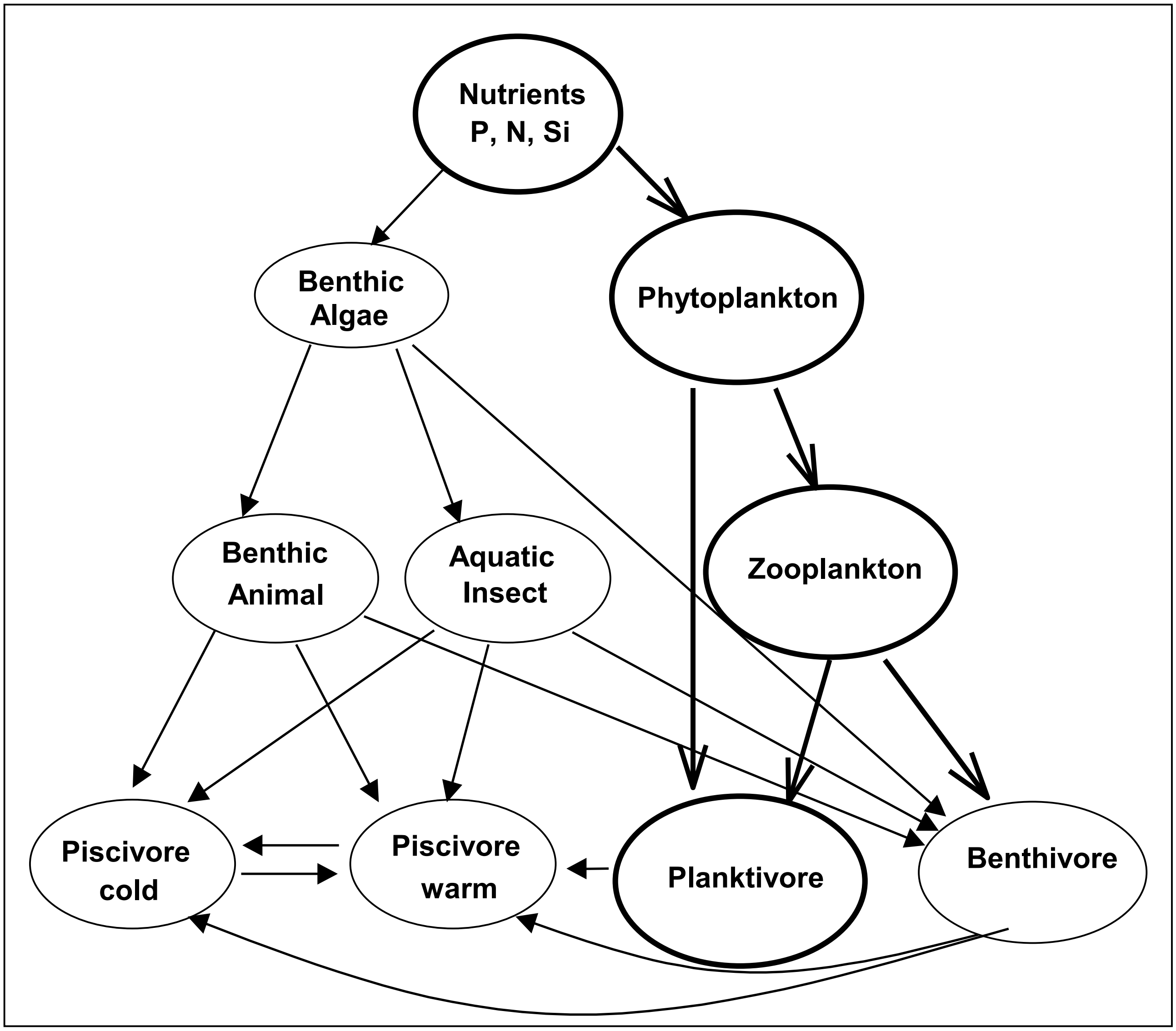


Figure 3-47. Generalized Food Web, Normalized Snake River
Source: Developed by Normandeau

3.4.1 Chlorophyll *a*

Chlorophyll *a* concentrations measured from samples collected at various sites during the growing season between 1994 and 1997 did not display any clear patterns of increasing or decreasing (see Table 3-18). However, data collected during the 1997 growing season were highly variable both temporally and spatially. Although not universal for all stations, the highest seasonal chlorophyll *a* levels were generally observed in June and were associated with an abundance of diatoms. Figure 3-48 illustrates the seasonal changes in chlorophyll *a* levels for selected stations within the study area. There appear to be no distinct differences in the chlorophyll *a* levels between the Snake and Columbia river systems. Both the upstream Snake River station (SNR-148) and the McNary reservoir station (CLR-295) had similar peak levels in June, which is in contrast to most other parameters.

Previous research suggests that average chlorophyll *a* levels above 5.0 and 14.5 µg/L are indicative of mesotrophic and eutrophic conditions, respectively. Chlorophyll *a* levels will typically range between 3.0

and 11.0 μg/L and 3.0 and 78.0 μg/L for mesotrophic and eutrophic conditions, respectively (Wetzel, 1983). Based on the data presented in Table 3-18, concentrations of chlorophyll *a* generally are between the criteria for mesotrophic and eutrophic, with an average concentration between 3.8 and 11.4 μg/L, and an upper confidence interval (95 percent) of 18.9 μg/L for the period between 1994 and 1997.

Based on estimated median concentrations for 1997, Station SNR-108, in the Lower Granite reservoir, had the highest median chlorophyll *a* level of 8.74 μg/L, and a mean concentration of 8.1 μg/L. In the Snake River, there was a general progressive decline in levels moving downstream with the seasonal median level for Station SNR-18 in the Ice Harbor reservoir at 3.2 μg/L (and a mean concentration of 5.6 μg/L). The opposite was true in the Columbia River, where the median concentrations appeared to increase downstream. The median concentration at the upstream station (CLR-397) was 6.72 μg/L and gradually increased to 8.01 μg/L at Station CLR-295 in the McNary reservoir.

3.4.2 Phytoplankton

Phytoplankton are the most important primary producer in the lower Snake River. The foundation of the food web, they transform light and nutrients into energy for herbivores such as zooplankton, which in turn support higher trophic levels. Phytoplankton grow best in low-velocity waters with warm temperatures and high nutrient availability, particularly phosphorus. Phytoplankton growth is generally limited in stream or riverine systems, which have much greater flow velocities. In evaluating phytoplankton data, a relative increase in species diversity or richness under similar habitat conditions is often considered a positive indication of improving ambient water quality conditions. In contrast, the dominance of certain robust species, such as some species of blue-green algae, can often be indicative of poor water quality conditions. To evaluate the importance of phytoplankton as a food source, the volume or quantity of algae available for consumption is often the most critical parameter to be considered. For this reason, phytoplankton data are typically expressed in terms of overall biovolume (i.e., μm^3/mL) or population densities (i.e., cells/mL) as well as species composition.

Assemblages throughout much of the study area were quite similar through 1997, showing a peak in overall density during the last week of June and early July, followed by a decline through mid-to-late summer and a secondary peak in autumn. The lower Snake River reach generally had the highest peak densities, ranging from 1,303 cells/mL at SNR-108 to 2,842 cells/mL at Station SNR-6. The corresponding biovolumes for these peak densities are 1,133,792 and 1,749,869 μm^3/mL for Stations SNR-108 and SNR-6, respectively. Peak algal densities for the Columbia River stations ranged from 1,516 to 1,832 cells/mL with corresponding biovolumes of 699,846 to 879,791 μm^3/mL at Stations CLR-369 and CLR-295, respectively. The Clearwater River at Station CLW-11 had a relatively low peak density of 749 cells/mL and a biovolume of 321,077 μm^3/mL during the same time period.

Table 3-19. Average and 95 Percent Confidence Intervals for Growing Season Chlorophyll *a* Concentrations (μg/L) in the Surface Water at Selected Sampling Sites and Years

Site	1994		1995		1996		1997	
	Avg	CI	Avg	CI	Avg	CI	Avg	CI
SNR-18		3.1		1.5		3.7		3.1
	7.8		3.8		8.7		5.6	
		12.6		6.1		13.6		8.2
SNR-83		4.0		(2.0)		4.2		5.8
	6.2		8.5		9.1		7.9	
		8.4		18.9		13.9		10.0
SNR-108		2.4		2.9		8.0		6.3
	6.0		7.8		11.4		8.1	
		9.6		12.7		14.9		9.9
SNR-118		1.7		3.0				5.2
	7.7		7.0		ND	ND	6.8	
		13.8		11.1				8.4
SNR-129		4.7		4.6		6.6		5.3
	9.7		6.0		8.7		8.6	
		14.7		7.5		10.8		12.0

Note: ND = not detected
Source: Developed by Normandeau

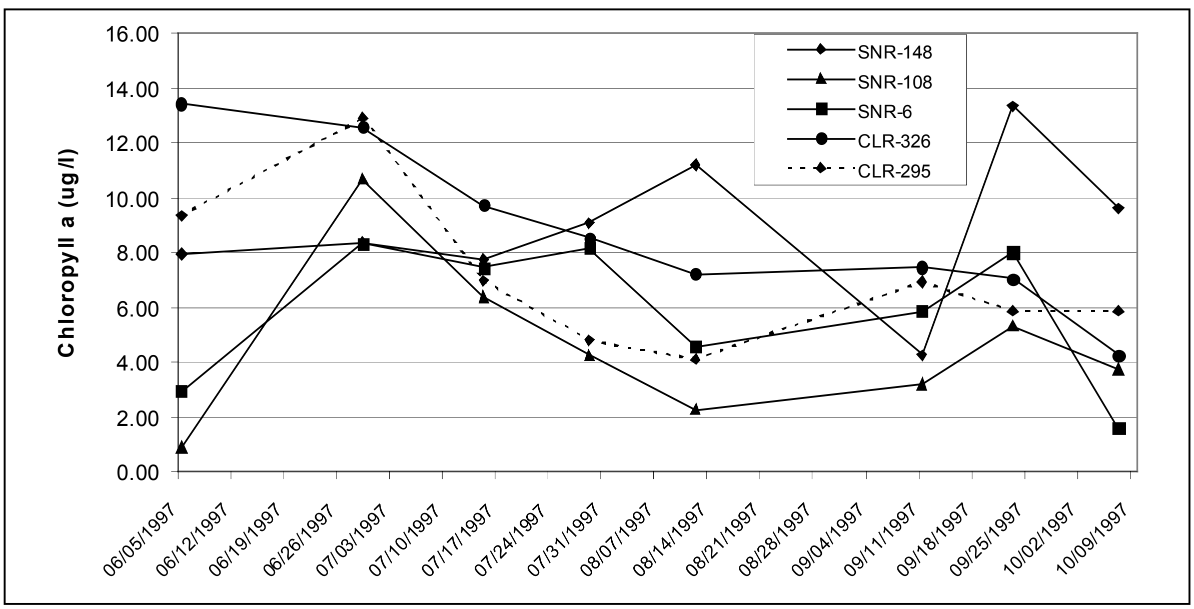


Figure 3-48. Chlorophyll *a* for Selected Sampling Stations in 1997
Source: Developed by Normandeau

There were few differences in the number and types of phytoplankton observed at the impounded pool sites above dams and transitional sites below dams within the lower Snake River system. For most of the study area, diatoms (Bacillariophyta) were typically dominant throughout much of the season, but especially during the peak flow period. At this time, diatoms typically accounted for more than 90 percent of phytoplankton biovolumes. The cryptophytes (*Rhodamonas minuta* and *R. m. nanoplanctica)* became dominant or co-dominant (by numerical density) at most sites in the lower 50 miles of the Snake River during the second half of the season. However, because of their small size, they composed a relatively small fraction of assemblage biovolume. Phytoplankton blooms (dominated by the genus *Aphanizomenom* and *Anabaena*) do occur in the lower Snake River reservoirs. These blooms are typically brief, lasting only a few weeks, but significant in their total community dominance during that time period and potential subsequent impacts on oxygen concentrations and invertebrate food supply. There have been documented occurrences of surface scum resulting from these taxa. Research has noted much littoral detrital accumulation from senescing planktonic algae blooms during the later summer that deposits on attached benthic algal communities. This senescing planktonic algae likely provides a significant late-summer nutrient input to the attached benthic algal communities, as well as a direct food source for littoral benthic macroinvertebrates.

Other commonly observed taxa within the lower Snake River reservoirs include the diatoms *Melosira islandica* (18.3 percent of the total collection), *Cyclotella meneghiniona* (11.7 percent) and *Fragillaria crotenensis* (11.3 percent), and the cryptophytes *Rhodomonas minuta* (7.3 percent) and its variant *R. m. nanoplanctica* (14.4 percent). Few other taxa exceeded 2 percent of the total collection except the diatom species *Asterionella formosa* (4.5 percent) and *Melosira granulata* (3.7 percent), diatoms of the genera *Diatoma* (5.3 percent) and *Synedra* (3.4 percent), the green algal genus *Scenedesmus* (2.3 percent), and the blue-green *Anabaena* spp. (4.3 percent).

Similarly, there was very little difference in the phytoplankton assemblage at the upstream, free-flowing Snake River station (SNR-148) as compared to the impounded Snake River locations. The free-flowing Snake River assemblage was essentially similar to those downstream in the Lower Granite reservoir, except for having higher densities of diatoms early in the year and lower densities of cryptophytes on most occasions. These minor differences may be due in part to the higher-than-normal flows that occurred in 1997, which reduces the hydraulic residence time and limits algae growth in the impoundments.

The Clearwater River phytoplankton assemblage differed somewhat from Snake and Columbia River assemblages in that blue-green algae *Oscillatoria* and *Anabaena* dominated throughout much of the sampling season. Blue-green algae peaked in abundance from mid-July to mid-September. During this time period, blue-green algae accounted for 28.5 to 85 percent of the cell densities and 29 to 48 percent of the sample biovolume. Outside of this period, the blue-green algae accounted for less than 5 percent of the total biovolume.

3.4.3 Attached Benthic Algae

ABA are a secondary source of primary productivity in the lower Snake River. As algae that are attached to rocks and other hard substrate, they provide a food source for benthic organisms such as aquatic insect larvae, amphipods, and oligochaetes. The 1997 empirical data on ABA were based primarily on measurements of chlorophyll *a* concentrations in samples collected from tile and mylar subtrates placed in the field for a 14-day incubation period. Mean concentrations (mg/m^2) of five "species" of photosynthetic pigments (evaluated from tile substrates) were reported including chlorophyll *a* (mono- and trichromatic), *b*, and *c*, and phaeophytin.

The upstream station (SNR-148) had consistently high values of chlorophyll *a* throughout the season, ranging from 29.06 to 93.6 mg/m^2, with the highest value occurring in October. Only the downstream station in the Ice Harbor reservoir (SNR-18) had chlorophyll *a* values that were higher, which were frequently above 100 mg/m^2 from July through early September. In the Lower Granite reservoir (SNR-118), the ABA chlorophyll *a* values ranged from 23.04 to 73.35 mg/m^2, which are generally lower than those recorded at the upstream station SNR-148.

Trichromatic chlorophyll *a* levels (the measure of chlorophyll used in the 1975 and 1976) EPA surveys of Falter and Ringe (1974) measured in the high-flow year 1997 at the free-flowing SNR-148 were in the 30 to 100 mg/m^2 range at a 1.5-meter (4.9-foot) depth. In the low-flow 1998, the range was 60 to 110 mg/m^2 at a 1.5-meter (4.9-foot) depth. The ABA trichromatic chlorophyll *a* levels obtained in 1975 and 1976 at this site were 10 to 20 mg/m^2. There was essentially no overlap between the ranges of 1976 and 1997-98. The earlier data are from glass-slide incubations while the later data are from a combination of natural rock, tile, and a mylar substrate. Even though substrates were different, these ABA data over the 24-year time spread are probably one of the better indicators available of increasing productivity of the Snake River coming into the project area over this time period.

The mean biomass, as measured by the ash-free, oven dry weight (AFODW), for the ABA samples collected in 1997 follows a similar pattern with the Ice Harbor reservoir station (SNR-18) having highest biomass of 10.94 to 37.09 g/m^2. The AFODW for the Lower Granite reservoir samples (SNR-118) ranged from 9.09 to 25.25 g/m^2. Samples from the upstream Lower Snake station (SNR-148) had ADOFWs ranging between 4.39 and 15.17 g/m^2. Historical data indicate that ABA ash-free biomass in 1976 averaged 1.64 g/m^2 at SNR-148. In contrast, the results from 1997, when samples were collected from a comparable depth and time period and a nonsilt-collecting mylar substrate, averaged 6.65 g/m^2, and in 1998 7.95 g/m^2. The different measure of ABA ash-free biomass further suggests that productivity in the lower Snake River is increasing.

AFODW samples collected from the upper the McNary reservoir (CLR-326) had AFODWs ranging from 2.26 to 30.27 g/m^2 with the highest level occurring later in the season toward the end of September. Samples collected in the free-flowing Hanford section (CLR-369) had relatively low biomass values, with AFODWs for most samples below 6.0 g/m^2 and a seasonal range of 0.64 to 14.09 g/m^2. The McNary reservoir and the lower Snake River reservoirs apparently produce a considerable amount of ABA biomass along the littoral and shoreline areas. However, much of the system has accumulated fine sediments, which limit the amount of ABA and epilithic periphyton. This finding may prove interesting in evaluating the proposed Dam Breaching alternative, because ABA is generally more prolific in riverine conditions rather than in a reservoir environment. As discussed earlier, deep-water releases from the Dworshak Dam were started in 1994 in conjunction with releases for fish flow augmentation to help lower downstream temperatures in the lower Snake River in July and August.

3.4.4 Primary Productivity

Primary productivity is a measure of the amount of carbon per unit time produced by all aquatic plants. As primary producers form the base of the food chain, the level of primary productivity ultimately dictates the productivity of the entire ecosystem. The number of primary productivity data points available from the mid-1970s are limited. SNR-18 in 1976 was the most complete, while the other stations typically had at most one or two determinations per year, and thus were not used to construct Tables 3-19 and 3-20. The general pattern was for greater pelagic primary productivity in 1994 when discharge was low and hydrologic residence times were maximized. Productivity decreased in 1995 when discharge increased, with what appears to be an exception at Central Ferry (SNR-83). The reason for the apparent increase in productivity was one sample day in late August when the rate was unusually high, thus skewing the distribution.

3.4.5 Zooplankton

Zooplankton are an important source of food for plankton-eating fish, which in turn are consumed by other fish. Zooplankton assemblages are also expressed in terms of total biomass, population densities, or species composition. Species composition is usually determined first through enumeration and identification of the various organisms in a sample. Total biomass is then calculated through established length/width relationships for each species type. Zooplankton data were analyzed with the same techniques described for phytoplankton. Time series graphs of density estimates were plotted for each location using totals for three major taxonomic groups: rotifers, copepods, and cladocerans. Throughout the season, the highest densities of zooplankton were generally observed in the lower Snake River reservoir sites and in the two McNary reservoir sites (CLR-306 and CLR-295). The transition sites in the upper McNary reservoir (CLR-326) and the Lower Granite reservoir (SNR-118 and SNR-129) generally had lower and somewhat more variable densities. The same was true for the free-flowing Snake (SNR-140 and SNR-148) and Clearwater River sites (CLR-11). However, densities were quite high early in the season at the upstream riverine Snake River site (SNR-148), but dropped sharply thereafter. Densities and taxonomic composition through time in the free-flowing Hanford Reach (CLR-369) were quite similar to what was found farther upstream in the Priest Rapids reservoir (CLR-397).

Table 3-20. Average and 95 Percent Confidence Intervals for Growing Season Primary Productivity Rates (mgC/m^3/hr) at 1 meter for Selected Sampling Sites and Years

	1975		1976		1977		1994		1995		1996		1997	
Site	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI
SNR-				-0.4				23.3		11.0				16.0
18	ND	ND	14.1	28.6	ND	ND	44.8	66.4	14.9	18.8	ND	ND	23.8	31.6
SNR-								5.9		-15.3				14.4
83	ND	ND	ND	ND	ND	ND	35.6	65.3	62.3	139.8	ND	ND	27.9	41.5
SNR-								25.2		19.1				
108	ND	ND	ND	ND	ND	ND	77.1	128.9	42.9	66.8	ND	ND	ND	ND
SNR-								20.4		< 1.0				21.2
118	ND	ND	ND	ND	ND	ND	77.2	134.1	20.4	41.3	ND	ND	23.7	26.2
SNR-								25.2		12.5				
129	ND	ND	ND	ND	ND	ND	67.4	109.6	23.6	34.8	ND	ND	ND	ND

Note: ND=not detected
Source: Developed by Normandeau

Table 3-21. Average and 95 Percent Confidence Intervals for Depth-Weighted Growing Season Primary Productivity Rates (mgC/m^3/hr) for Selected Sampling Sites and Years

	1975		1976		1977		1994		1995		1996		1997	
Site	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI	Avg	CI
SNR-								16.2		7.1				9.7
18	ND	ND	ND	ND	ND	ND	29.9	43.5	9.6	12.0	ND	ND	15.7	21.7
SNR-								9.2		-5.1				10.1
83	ND	ND	ND	ND	ND	ND	25.3	41.4	34.0	73.0	ND	ND	17.3	24.6
SNR-								22.7		12.2				
108	ND	ND	ND	ND	ND	ND	49.2	75.8	27.1	42.1	ND	ND	ND	ND
SNR-								9.3		0.0				12.4
118	ND	ND	ND	ND	ND	ND	45.9	82.4	14.0	28.5	ND	ND	14.4	16.4
SNR-								17.4		8.2				
129	ND	ND	ND	ND	ND	ND	39.5	61.6	14.5	20.8	ND	ND	ND	ND

Note: ND=not detected
Source: Developed by Normandeau

Over the entire study area, the 1997 zooplankton assemblage was composed of 30 nominal taxa, distributed fairly equally among members of the phylum Rotifera and two major groups of microcrustacea, Copepoda and Cladocera. This total included at least nine species of cladocerans, four species of copepods, and 11 species of rotifers. At most locations, rotifers were most abundant early in the season, then tapered off in density later in the sampling season. However, the number of different species (i.e., taxa richness) at most locations varied considerably between sampling events. Over the entire season, total zooplankton richness seems to depend mostly on the reach type, with reservoir sites supporting the most taxa and riverine sites supporting the fewest. Reservoir and transition sites usually supported between 7 and 16 individual taxa, while riverine sites on the Snake and Clearwater rivers supported no more than eight taxa at any given time. The Hanford Reach, CLR-369, had the greatest variability with anywhere between 1 and 11 taxa identified depending on the sampling event.

In decreasing order of system-wide abundance, taxa that occurred at a mean density of more than or equal to 0.1 individuals/L (averaged over all sites and times) included the cladoceran *Daphnia retrocurva*, cyclopoid copepods, the copepod *Diacyclops thomasi*, the cladoceran *Bosmina longirostris*, copepod nauplii, and the rotifer *Keratella cochlearis*.

3.4.6 Benthic Macroinvertebrates

Benthic mac-roinvertebrates are an important link in the food web between primary producers and secondary consumers such as bottom-feeding fish and large invertebrates, which consume benthic macroinvertebrates. The most recent benthic macroinvertebrate data were collected in 1994 and 1995 from three locations in the Lower Granite reservoir on the Snake River. The three locations sampled include the Offield site in the lower portion of the Lower Granite reservoir, an artificial shoal-dredge disposal area in mid-reservoir called Centennial Island, and the Silcott Island site, a large island/backwater complex located a few miles downstream of Lewiston-Clarkston within the upper third of the reservoir. Sampling methods involved use of an aerial sampler that enabled a spatial density measurement of the number of organisms per square meter of bottom area.

Samples were collected at approximately monthly intervals from March 1994 through October 1995. Separate grabs were taken at depths of 3, 9, and 18 meters (10, 30, and 59 feet) from each location. A total of 42 nominal taxa of benthic macroinvertebrates were collected in 1994 and 1995 at the three different locations. Generally, a greater variety of taxa were observed at the shallower depths (3 and 9 meters [10 and 30 feet]). Some organisms were identified to species, but many taxa were lumped into broad taxonomic categories (e.g., Bivalvia). Oligochaete worms were numerically dominant at all three stations in both years, but were particularly abundant at Silcott and Centennial islands. Chironomids were second in abundance at all three sites, and actually exceeded densities of oligochaetes on a few occasions at the Offield site. These two groups comprised 82 to 97 percent of the total collection from each station. Bivalve mollusks comprised nearly 12 percent of the collection at Offield, but made up less than 2 percent of the collection elsewhere.

4. Alternative Analysis

4.1 Overview of Available Data Sources

This section describes the various alternatives being evaluated to improve juvenile salmon migration and the potential water quality impacts that may result from these alternatives. The discussion of potential impacts focuses primarily on those parameters that may have some effect on anadromous fish and/or those parameters that are likely to be most affected by the proposed dam breaching options. The additional flow augmentation and the dam breaching alternative obviously have the potential to change flow and other hydrologic conditions and, thus, water quality. In addition, the proposed TDG improvements and changes in the Corps spill regime could also affect water quality, particularly dissolved gas saturation levels. The potential impacts associated with these alternatives are evaluated for both short-term or transition-period construction effects and long-term operational effects. The major system improvements pathway focuses on optimizing the fish bypass collection and transportation systems as well as measures to reduce TDG.

4.1.1 Productivity Modeling

Water quality data were collected from 1994 through 1997 in the lower Snake River system (Normandeau, 1999a) to establish a baseline of information on the existing reservoir system and to assist in the evaluation of changes in biology and water chemistry associated with proposed dam breaching (Normandeau, 1999a). Parameters were selected in order to support a modeling effort, and included physical (transparency, conductivity, and temperature), chemical (nutrients, DO, anions and cations, pH, and alkalinity), and biological (chlorophyll *a*, biochemical oxygen demand, primary productivity, and zooplankton biomass) parameters. All field measurements and laboratory analyses were conducted according to a standard operating procedures manual developed as a part of an overall quality assurance program (Normandeau, 1999a). The quality assurance program included both field and laboratory audits. Empirical data collected under this program are given greater weight than modeling results in the analysis contained herein.

The objective of the WQRRS modeling effort was to simulate total primary and secondary productivity as an indication of the food sources available for higher trophic levels, which include salmon and steelhead. Predictive modeling of the standing crop of individual species was not the intent of the modeling. The currently available data for the food web of the lower Snake River are insufficient to support such an effort for the entire system. Some of the trophic levels as simulated were split into functional groups based on similar feeding and physiological characteristics. The food web presented in Figure 3-46 presents the general representation of the system used by the model. When the food web is represented with simplified ecological relationships, the contribution of salmon and steelhead to the sum of the productivity in the fish community is very small.

Numerous models could lend insight into the potential future biological productivity of the lower Snake River. Among those models considered were HEC-5Q, RBM-10, WASPS, CE-QUAL-W2, and WQRRS. All of the models considered (including the selected model) had limitations with respect to application to the lower Snake River and the essential components detailed above. The WQRRS was selected as the model of choice because of its ability to simulate the biological components well, its flexibility with regard to modeling hydrodynamics, and its ability to adequately model temperature.

Data collected from 1994 through 1997 were used in conjunction with historic data where needed, in a modeling effort which used the WQRRS model to simulate future changes in biological productivity as well as temperature and water quality under the proposed near-natural condition (Normandeau, 1999a). Few data describe the limnology and biological productivity of the lower Snake River system during the period prior to the closure of the four hydroelectric and navigation dams in the 1960s and 1970s. Those data that are available have limited utility in predicting water quality in the future if the dams were breached and the river was returned to its near-natural state. Water quality conditions in the watershed have changed markedly over the intervening years because of changing irrigation withdrawals, timber harvest practices, and wastewater treatment improvements, and as a result, water quality prior to impoundment would not likely approximate water quality to be expected in the near-natural river. Water quality modeling is the only practical way to predict what water quality can be expected if the dams on the lower Snake River were breached and to compare alternatives for managing the near-natural river system.

The WQRRS model of the drawdown alternative was built and calibrated using bathymetric and hydraulics data from 1934 and temperature data from the 1950s prior to damming. The model assumes that the post-breaching river would have physical characteristics similar to the river in 1934. The hydraulic computations for the model were completed using the Modified Puls method, which provides a stable solution even with high water velocities and is relatively simple compared with the other available methods. Then this model was applied to calendar years 1994, 1995, and 1997 using the actual hydrologic, meteorologic, and inflow water quality data from those years as input. Data from 1994 were used to represent a dry year, 1995 to represent an average year, and 1997 to represent a wet year. Based on mean annual flows measured in the Clearwater River at Spalding and the Snake River at Anatone, 1997 ranked as the highest flow year of record, 1994 ranked near the lowest 10 percent of flows, and 1995 was slightly above normal. Water year 1997 was abnormally wet all year. Water year 1994 was dry during the May-June snowmelt period and during the August-October low-flow period. Water year 1995 followed the historical mean monthly flow pattern quite well except in August and September when flows were more normal. Model predictions of primary and secondary productivity from the 1997 simulation were then compared to actual field data from that year, where data were available. The existing system with the dams in place was not modeled during this effort.

For several reasons, WQRRS modeling results are best used on a relative basis for comparing flow and hydraulic and nutrient loading scenarios rather than as a definitive predictive model. First, the biological data available to calibrate and verify the model were limited. Second, the base-case scenario for the modeling assumes a steady-state system some years after breaching and after the sediments behind each dam have been either redistributed or stabilized. The model assumed that the stream channel remaining after the redistribution of sediments would be similar to that present in the system in 1934, the last pre-dam year when extensive bathymetric data were collected on the system.

4.1.2 Temperature Modeling

The WQRRS modeling effort included near-natural river temperature simulation for the years that biological productivity simulations were completed (1994, 1995, and 1997). Figures 4-1 and 4-2 show predicted temperatures at RM 110.5 and RM 15.94, which are located in the forebays 3 to 5 miles upstream from Lower Granite and Ice Harbor dams, respectively. However, these temperature predictions were not the primary objective of the study and were included to provide a picture of the seasonal patterns of temperatures that govern many of the biological processes that contribute to total primary and secondary productivity. The temperature predictions are nonetheless useful in providing a

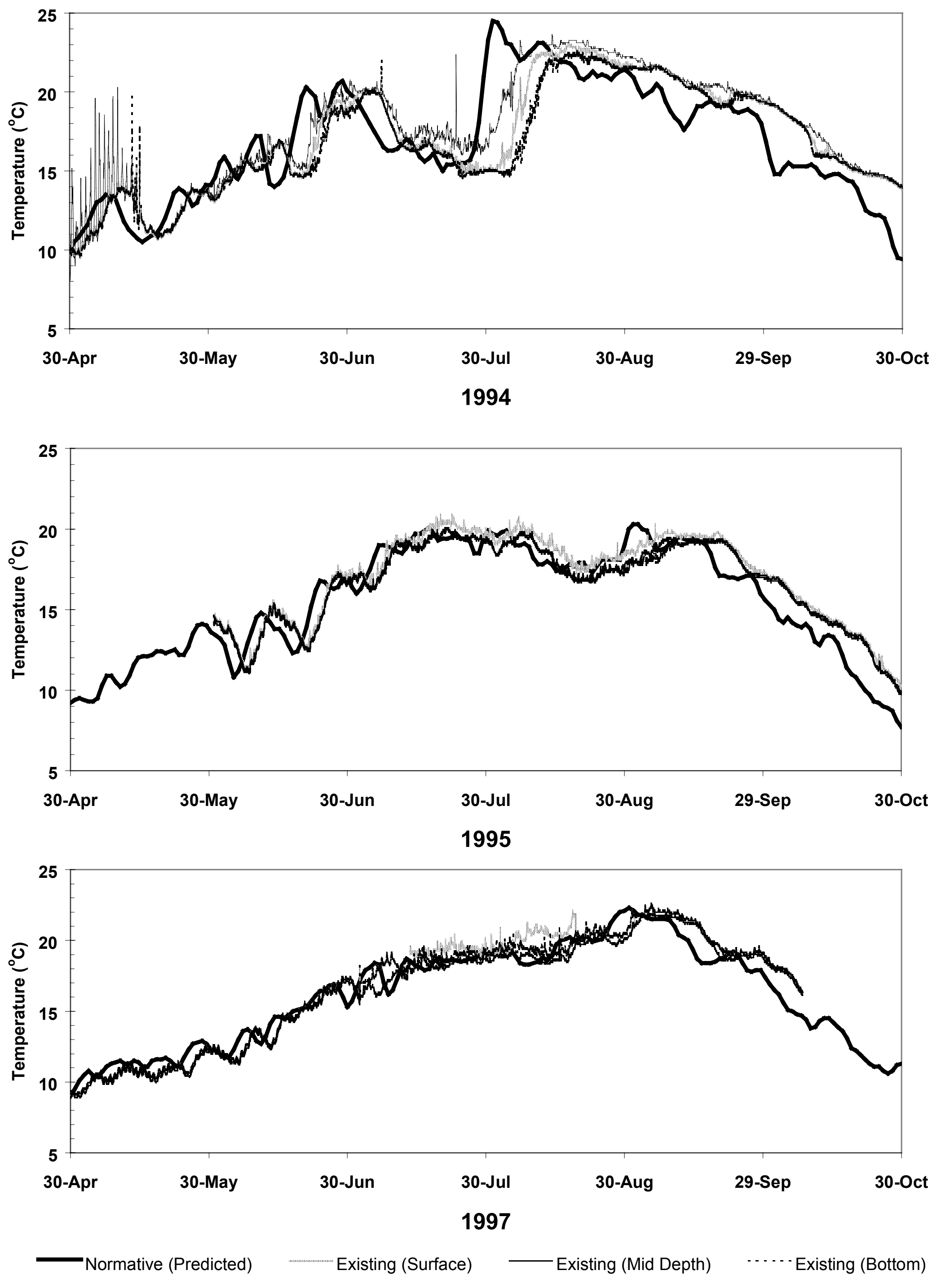


Figure 4-1. 1994, 1995, 1997 Near-natural River Temperatures Predicted by WQRRS Compared with Existing Water Temperatures (Normandeau, 1999a) RM 110.5

Source: Developed by Normandeau

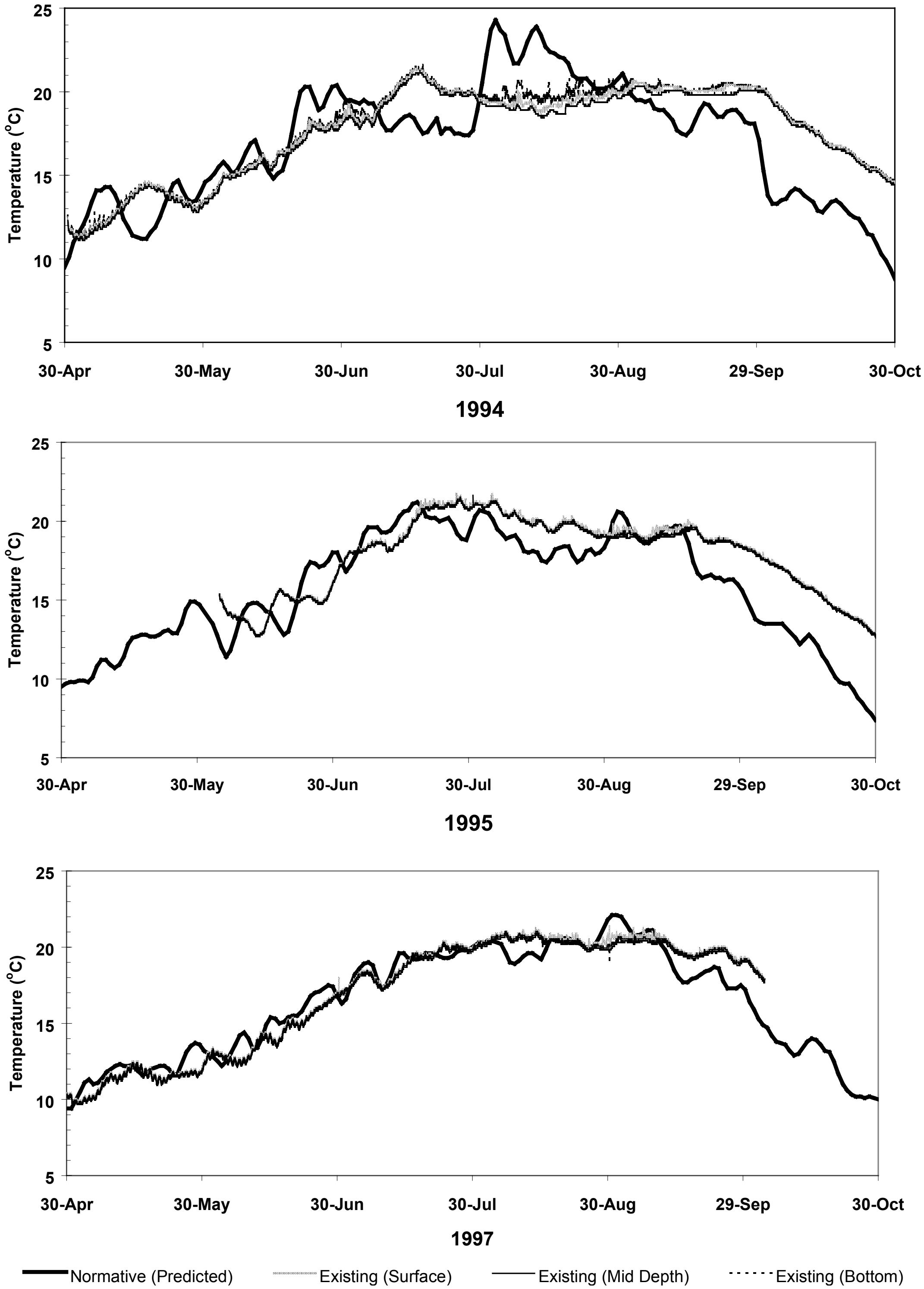


Figure 4-2. 1994, 1995, 1997 Near-natural River Temperatures Predicted by WQRRS Compared with Existing Water Temperatures (Normandeau, 1999a) at RM 15.94

Source: Developed by Normandeau

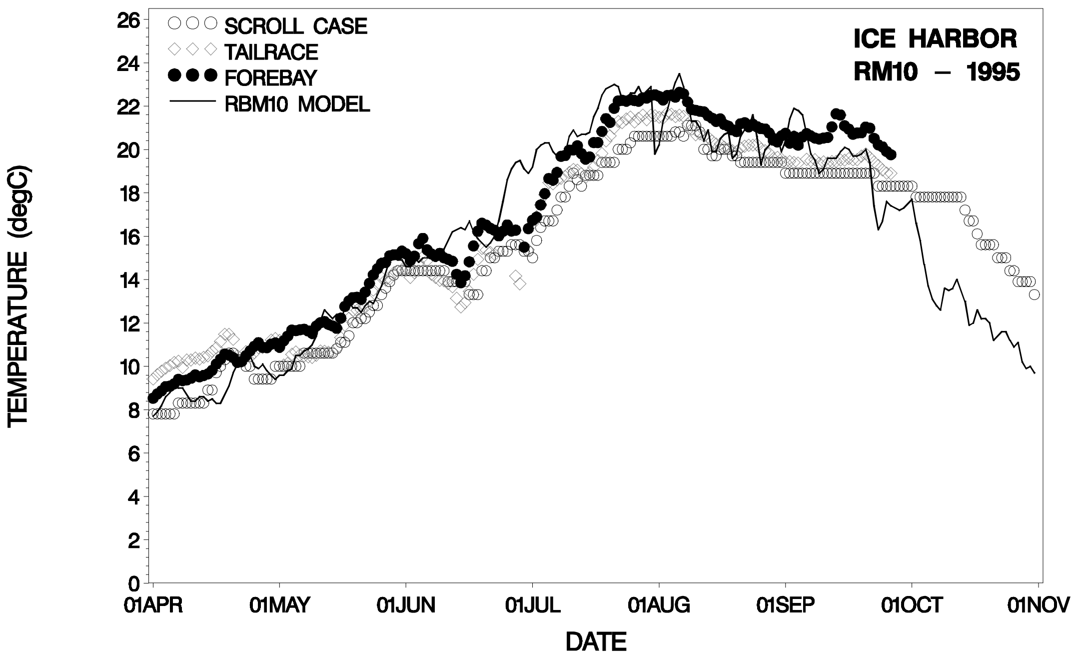
comparison of the seasonal temperature regime among 3 years with very different hydrometeorologic conditions.

The EPA has recently developed a temperature model of the Columbia System to support a total maximum daily load (TMDL) for temperature as required under Section 303(d) of the Clean Water Act (Yearsley, 1999). EPA's model (RBM-10) utilizes a thermal budget approach similar to the full heat budget method used in the WQRRS modeling effort referenced above, but uses a mixed Eularian-Lagrangian solution method. A comparison of RBM-10 results with monitoring results is presented in Figure 4-3.

The EPA cooperated with the Corps and provided its temperature modeling expertise and resources to assist the Corps in evaluating the effects of the dams and impoundments on lower Snake River temperature. In this endeavor EPA used its RBM-10 model calibrated with USFWS 1950s temperature data from Sacajawea and meteorological data for its "without dam" simulations and used the Corps' TDGMS tailrace temperature data as the benchmark for its "with dams" predictions.

Four primary scenarios were examined for the lower Snake River portion of the system-with and without dams and with and without Dworshak augmentation. Results from this modeling effort at RMs 107 and 10 are presented in Figures 4-4 through 4-7. The years chosen for presentation span a range of hydrometeorologic conditions prior to the augmentation of the lower Snake River with water from Dworshak (1980, 1984, and 1988) and after the augmentation began (1994, 1995, and 1997). The latter group of years are the same years that were simulated during the WQRRS modeling effort discussed above. 1984 and 1997 have been characterized as wet years. 1988 and 1994 have been characterized as dry years. 1980 and 1995 have been characterized as average years. These results are discussed further in subsequent sections. A modeling study completed by the Pacific Northwest Laboratory (Perkins and Richmond, 1999) used MASS1 (Modular Aquatic Simulation System 1D), a one-dimensional hydrodynamic and water quality model to predict water temperatures on the lower Snake River for the period 1960 through 1995. This spans a long period of time, including several years prior to the construction of the dams, and several years with all four dams in place. This study can be viewed in its entirety at http://www.nww.usace.army.mil.

Predictions of future water quality conditions are based on the results of water quality and productivity modeling using data collected mainly from 1994 through 1997, as well as two other recent temperature modeling studies (Perkins and Richmond, 1999; Yearsley, 1999). Table 4-1 is a summary of productivity and temperature model runs that were available for this study and analysis. A comparison of the three temperature models used is presented in matrix form in Table 4-2. A comparison of the relative strengths and weaknesses of each model as applied to the lower Snake River is presented in Table 4-3.



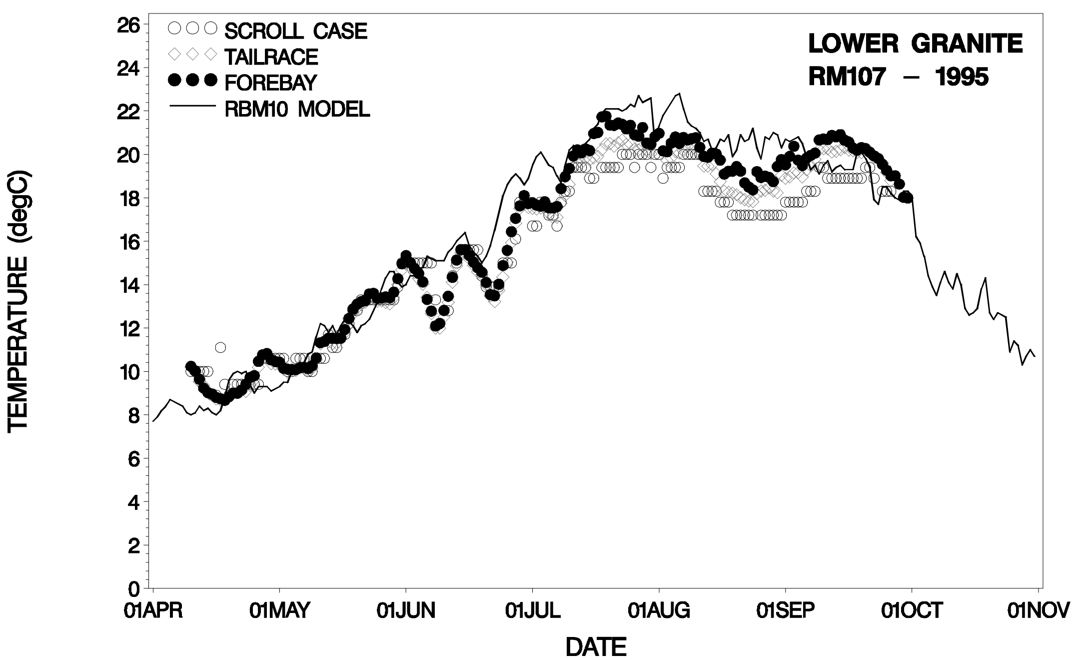


Figure 4-3. 1995 RBM10 Temperature Modeling Results and 1995 Temperature Monitoring Data
Source: Developed by Normandeau

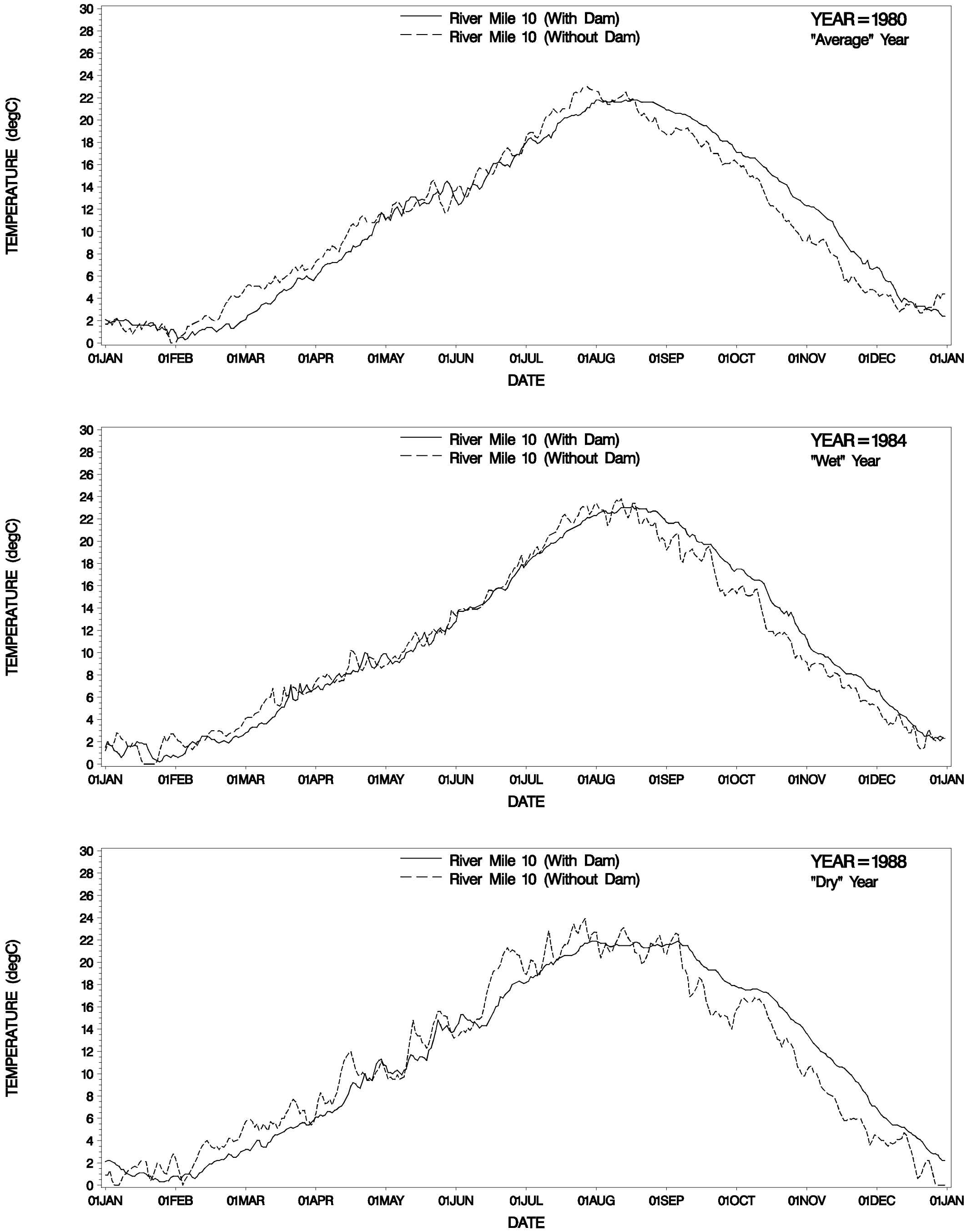
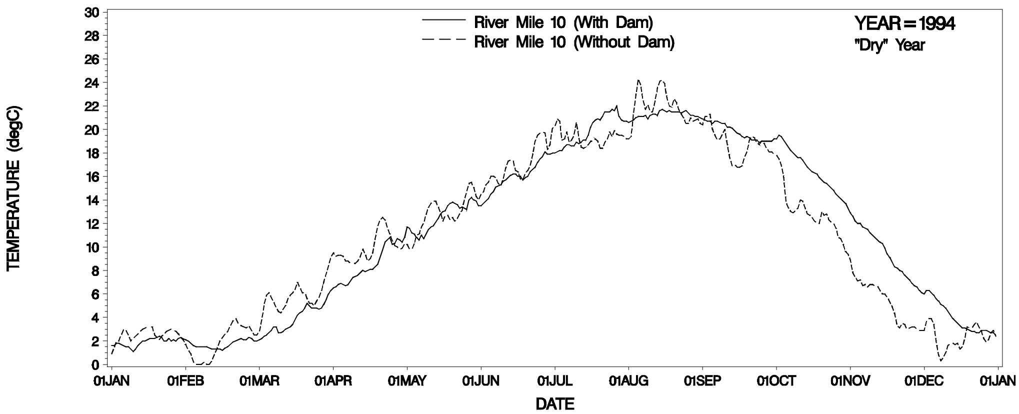
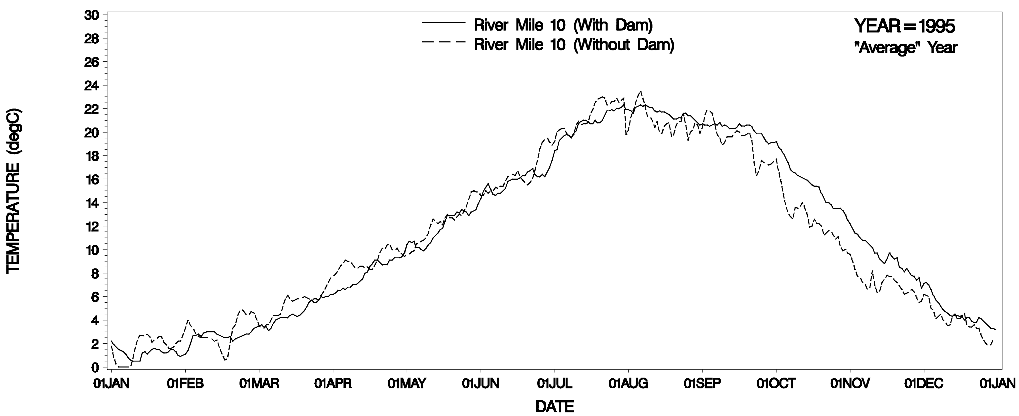


Figure 4-4. Temperatures Predicted by RBM10 at RM 10 with and without Lower Snake River Dams for Years Prior to Dworshak Flow Augmentation

Source: Developed by Normandeau





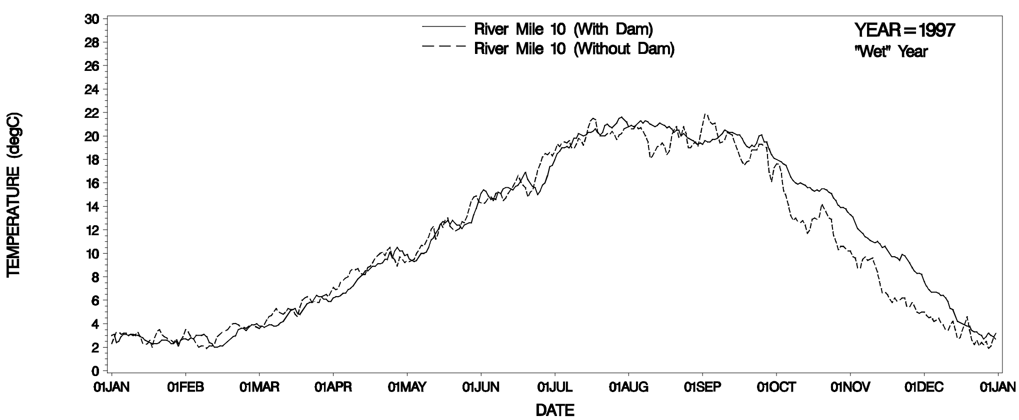


Figure 4-5. Temperatures Predicted by RBM10 at RM 10 with and without Lower Snake River Dams for Years Since Dworshak Flow Augmentation

Source: Developed by the Corps

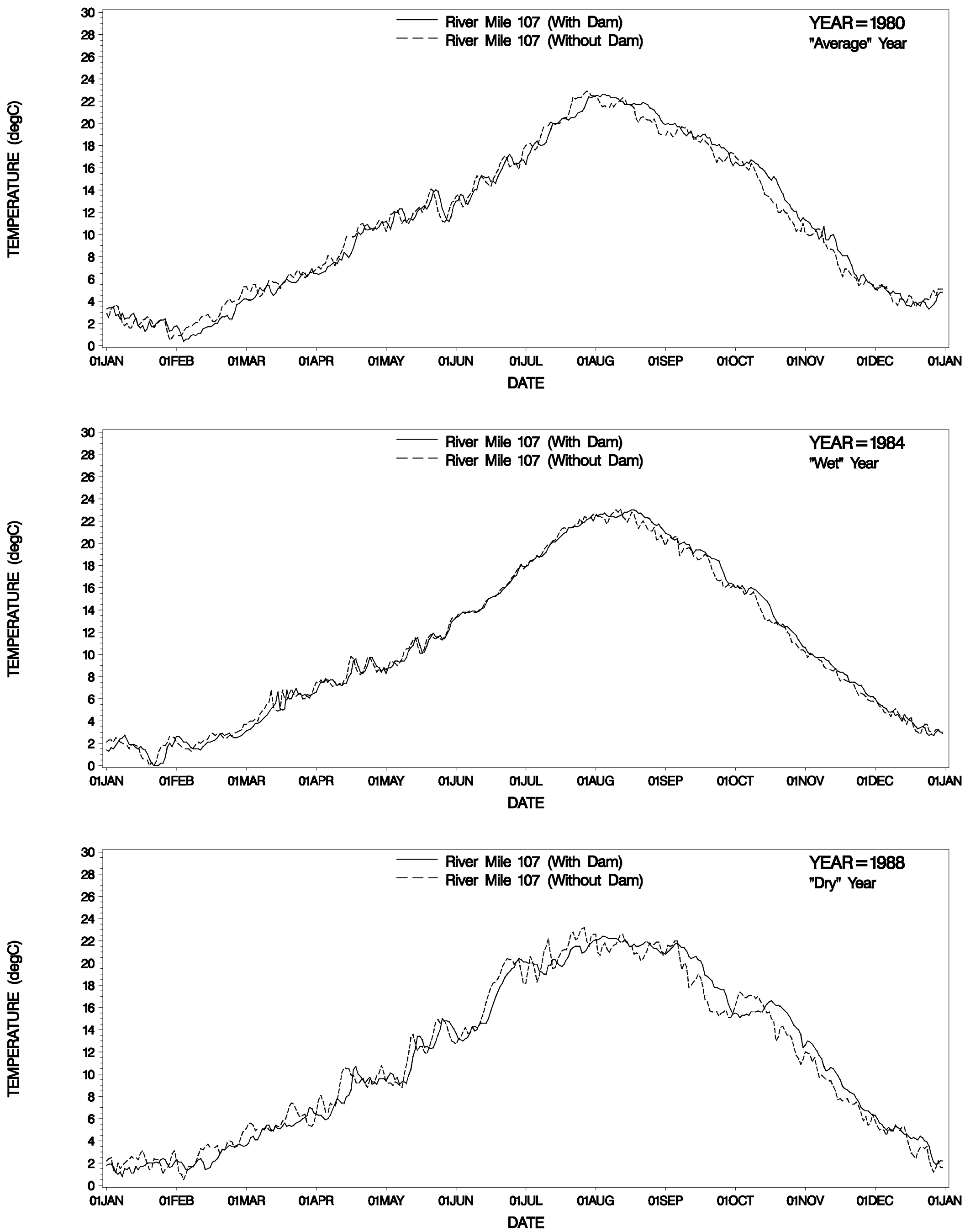


Figure 4-6. Temperatures Predicted by RBM10 at RM 107 with and without Lower Snake River Dams for Years Prior to Dworshak Flow Augmentation

Source: Developed by Normandeau

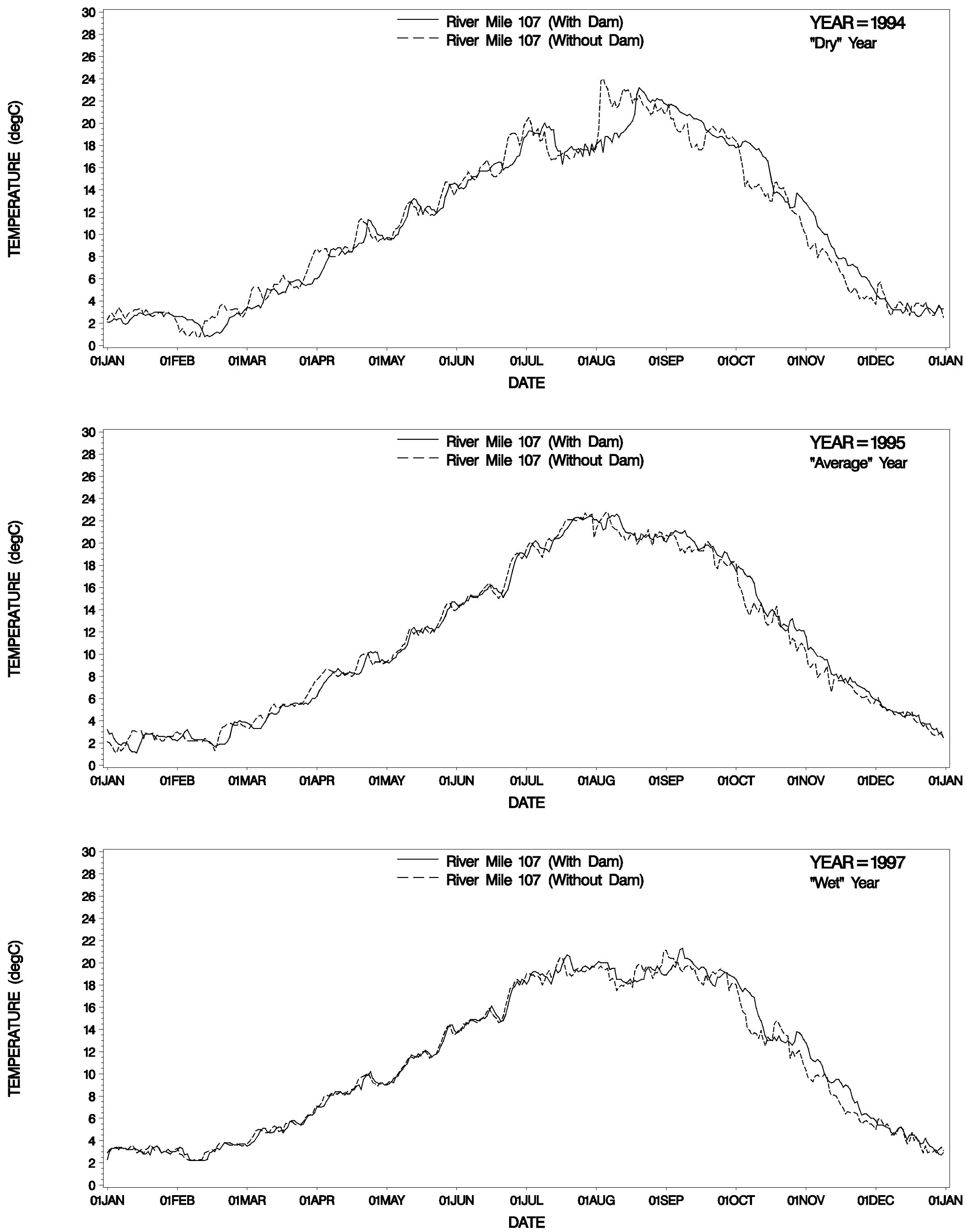


Figure 4-7. Temperatures Predicted by RBM10 at RM 107 with and without Lower Snake River Dams for Years Since Dworshak Flow Augmentation

Source: Developed by Normandeau

Table 4-1. Temperature and Productivity Modeling Simulations That Were Available for This Analysis

MODEL RUNS COMPLETED TEMPERATURE MODELING		
	With Dams	**Without Dams**
Temperature with Dworshak Augmentation	RBM-10 MASS-2	WQRRS RBM-10 MASS-2
Temperature without Dworshak Augmentation	RBM-10 MASS-2	RBM-10 MASS-2

BIOLOGICAL PRODUCTIVITY MODELING		
	With Dams	**Without Dams**
Biological Productivity with Dworshak Augmentation	None	WQRRS
Biological Productivity without Dworshak Augmentation	None	None

Source: Developed by Normandeau

Table 4-2. Comparison Matrix of Temperature Models as Applied to the Lower Snake River

	WQRRS	RBM10	MASS1
1-D (longitudinal) simulation of rivers	Yes	Yes	Yes
1-D (vertical) simulation of reservoirs	Yes	Not in Reverse Particle Tracking Version	No
Unsteady flow simulation	Yes	Yes	Yes
Hydrodynamic equations	St. Venant, Muskingum, or Modified Puls	Plug flow advection, no dispersion	St. Venant
Channel representation	Cross sections	Specified implicitly by depth and velocity functions	Cross sections
Depth and velocity	Directly from solution of hydrodynamics	Depth = a * Flow^b Velocity = c * Flow^d	Directly from solution of hydrodynamics
Surface heat exchange method	Heat budget or equilibrium temperature	Heat budget	Heat budget
Bottom heat exchange	Yes	No	No
Tributary and point source inflows	Yes	Yes	Yes
Previous applications	Many throughout United States	Several in Pacific NW	
Independent peer review	Reviewed and tested by multiple organizations	Documented EPA Peer Review Process-1999	Not verified prior to Snake River application
User documentation	Yes	In Final Draft-Tetra Tech	

Source: Developed by Normandeau

Table 4-3. Strengths and Weaknesses of Temperature Models Applied to the Lower Snake River

Relative Strengths of Each Model for Application to Lower Snake River

WQRRS:

1. Can simulate existing condition as well as near-natural condition.
2. Has been widely used and tested.
3. Well documented for users other than the model developers.
4. Bottom heat exchange is included in the module for simulating streams.
5. Rigorous hydrodynamic solution including use of actual cross sections.

RBM10:

1. Can simulate existing condition as well as near-natural condition.
2. Has been used previously in the Pacific Northwest region.
3. Apparently produces less numerical dispersion than other models (according to unpublished EPA memo).
4. Can simulate diel temperature oscillation.

MASS1:

1. Rigorous hydrodynamic solution including use of actual cross sections.
2. Surface heat exchange is same formulation as MASS2, which has been extensively verified (according to MASS1 report).
3. Can simulate diel temperature oscillation.

Relative Weaknesses of Each Model for Application to Lower Snake River

WQRRS:

1. Limitations on number of cross sections required dividing the river into multiple models with output from one model as input to the next model downstream.
2. Modified Puls routing requires use of HEC-2 or other model to develop table of storage vs. outflow values.
3. Cannot simulate diel temperature oscillation.

RBM10:

1. Need to use HEC-2 or other model to estimate coefficients for relationships between flow and depth/velocity.
2. Bottom heat exchange is apparently not included in the model.

MASS1:

1. Bottom heat exchange is apparently not included in the model.

Source: Developed by Normandeau

4.1.3 Sediment Contaminants

To assess the potential impacts from sediment transport associated with each of the alternatives under consideration, a study of sediment contaminants was conducted. The results of this study were summarized in previous sections of this report and were used to determine the impacts of resuspended contaminants on water quality, as discussed in later sections. Sediments were collected in areas where fine sediments are likely to accumulate following drawdown and analyzed for potential contaminants. Elutriate analyses were also conducted to ascertain likely transport into the overlying water column. The CoC were selected based on comparisons with known standards, and concentrations were examined at points of compliance, where exceedances might increase risks to biota and human health.

Empirical data recorded during the 1992 experimental Lower Granite Reservoir Drawdown Study also provide relevant information that can be used to extrapolate and predict potential changes resulting from the proposed dam breaching condition (Corps 1992). Although the proposed drawdown of all four reservoirs would be on a much larger scale than that conducted in the 1992 study, the information is most useful in describing potential water quality conditions during the short-term transition phase. During the 1992 study, the reservoir was drawn down 10 meters (33 feet), which is much less than the approximately 30 meters (100 feet) of drawdown currently being proposed. Much of the relevant empirical data collected pertains to changes in flow velocities, sediment transport, and turbidity levels as the Lower Granite reservoir was being drawn down.

4.2 Discussion of Potential Water Quality Impacts

4.2.1 Water Temperature

Water temperature and increases in dissolved gas saturation levels represent two of the principal water quality concerns related to hydropower dam operations. Impoundments tend to change the timing and rate of water heating and cooling such that the seasonal rise and decline of water temperatures may differ from the inherent natural riverine patterns. Previous studies indicate that peak temperatures occur later in the summer and the autumn cooling period becomes more prolonged due to the heat attenuation in the impoundments compared to the free-flowing river (BPA, 1995; EPA and NMFS, 1971). These changes affect aquatic species that have adapted to the natural seasonal temperature cycles, especially native fish. The Corps and other agencies have made considerable efforts and expenditures to mitigate existing periodic problems associated with elevated temperatures during critical flow conditions, primarily through flow augmentation. Existing flow augmentation efforts from deep storage reservoirs where selective deep-water outlets are available (i.e., Dworshak Dam) have been shown to cause some reductions in water temperatures downstream (Bennett et al., 1997 and Karr et al., 1997). The temperature reduction effect produced by the Dworshak Dam flow releases appears to be most beneficial during low-flow years.

The effect of increased flow augmentation on water temperature also depends on the temperature of the source waters used for augmentation. The deep cooler waters released from Dworshak Dam definitely provide some benefit; however, this dam has limited additional storage capacity to provide more flow beyond the existing flow augmentation volumes. The upper and middle Snake River Dams do not have selective withdrawal facilities to release the deep, cooler temperature waters. Without selective withdrawals, water released from the upstream Hells Canyon Complex Dams would likely reach ambient temperatures by the time it reaches the lower Snake River reach approximately 270 kilometers (160 miles) downstream.

Temperature differences between the existing conditions and the dam breaching alternative (near-natural system) were evaluated in two ways. First, measured data from 1994, 1995, and 1997 were compared with measured data from 1956 through 1958 at both Central Ferry (RM 83.2) and Sacajawea near RM zero. Although the study years represent the spectrum of dry through wet conditions and the period 1956 through 1958 represents more average conditions; the comparison clearly shows some distinct differences. The existing impounded system also tended to warm more slowly in the spring and cool slower in the fall due to the larger volume of water and larger heat capacity of the impoundments compared to the near-natural system. Also consistent with information previously described by Bennett et al. (1997), maximum temperatures during the summer months of July through August will probably be approximately 2 to 5°C (3.6 to 9°F) higher under the near-natural system, approaching 26 to 27°C (79 to 81°F). This difference is due largely to the cooling effects of Dworshak augmentation.

Then secondly, the WQRRS near natural river simulated temperature data were compared to the measured Lower Snake River Reservoir temperature data for periods of similar hydrometeorological conditions (Normandeau, 1999a). Measured temperatures in the existing system were found to be similar in magnitude to predicted temperatures except that temperatures in the existing system lagged those predicted for the drawdown. The time lag increased progressively and river flows decreased, apparently related to increased travel times and volumes.

Lowering the four impoundments to natural river elevations would produce a dramatic change in the volume and the heat storage capacity of open water in the lower Snake River. With less open water area and shallower depths, water temperatures over the long term would likely warm up faster early in the season but also cool down faster in early fall. Recent model results predict a more dramatic cool-down under certain hydrometeorologic conditions (Normandeau, 1999a). Figure 4-1 illustrates the predicted temperatures at Lower Granite Lock and Dam (RM 110.5) under the proposed dam breaching alternative in comparison to the observed temperatures for each of the 3 flow years. Using the same meteorological and hydrological data recorded during the sampling period as model input, water temperatures under the near-natural river system were predicted to drop to 15°C (59°F) 15 days earlier than under existing conditions. During a high-flow year, such as in 1997, the difference between the predicted date for water temperatures to drop to 15°C (59°F) and the observed date under existing conditions was closer to 5 days. Thus, the temperature benefit of the dam breaching alternative would occur during low-flow years.

The WQRRS temperature modeling results also suggest that water temperatures during low-flow years in the lower Snake River could reach higher summer peaks under the natural river conditions than under the existing impounded river conditions. Under wet and average hydrometeorologic conditions, peak summer temperatures are projected to be similar to those observed for the existing system. The WQRRS modeling results for 1994, 1995, and 1997 are presented graphically in Figures 5.5-1 through 5.5-12 in Normandeau (1999a) (http://www.nww.usace.army.mil).

The EPA temperature modeling effort found a greater likelihood that, without Dworshak augmentation, water temperatures would more frequently exceed the benchmark 20°C (68°F) in the lower Snake River with the dams in place than with the dams removed. The EPA temperature model (RBM-10) was calibrated and simulations were generated for a broad array of hydrometeorological conditions with and without Dworshak augmentation and with and without the dams in place. The frequency of exceedance of 20°C (68°F) has been reduced in years since flow augmentation from Dworshak (1994, 1995, and

1997). The impact of these releases from Dworshak are greater in the upstream reservoirs. The effect of the Dworshak releases diminishes in the downstream impoundments. A comparison of the simulated frequency and magnitude with which water temperatures are predicted to exceed 20°C (68°F) in the existing and near-natural system is presented in Table 4-4. Throughout the length of the lower Snake River the predicted frequency with which water temperatures are expected to exceed 20°C (68°F) is greater under the existing system than the near-natural system. The difference in predicted exceedance frequency between the existing and near-natural systems increases from upstream (RM 107) to downstream (RM-10). 1998 and 1994 are considered "dry" years. 1980 and 1995 are considered "average" years. 1984 and 1997 are considered "wet" years.

The Pacific Northwest Laboratory study (Perkins and Richmond, 1999) found that the primary difference between the current and natural river conditions scenarios is that the reservoirs decrease the water temperature variability. This is illustrated in Figures 4-8 through 4-11. Simulated temperature variability is similar at the beginning and end of the simulations (April and October), but variability is much greater in the near-natural scenario during the peak of the growing season (June through September). The reservoirs also create a thermal inertia effect that tends to keep water cooler later in the spring and warmer later into the fall compared to the near-natural river condition. This is similar to the results predicted by the WQRRS model (Normandeau, 1999a). However, due to the uncertainties in the simulation model, the authors conclude that the results showed only small differences between the current and natural river temperature regimes.

As discussed in the preceding pages, three different models have been used to predict temperature in the lower Snake River: 1) WQRRS for unimpounded conditions; 2) EPA model for impounded and unimpounded conditions; and 3) MASS2 for impounded and unimpounded conditions. While the three models are all credible in their logic and fundamental temperature computation, it is worthy of note that they differ somewhat in the way they were calibrated and the way they output their simulations. The WQRRS and RBM-10 are very similar on all counts except that RBM-10 uses a more recently developed numeric scheme and is more responsive to short-term changes in temperature. Both were calibrated on 1950s USFWS water temperature data collected near the mouth of the river and USGS data collected upriver at Central Ferry. Both predict temperature for each day (multiple times for RBM-10) through the year and present an average temperature for each day. The MASS2 calculates the temperature multiple times for each day, then selects the temperature for 3 p.m., which is assumed to be the maximum, and outputs the number of days the temperature exceeds several different levels. In this analysis the Corps and EPA cooperated to ensure accurate calibration of their RBM-10 model, and the Corps relied very heavily on EPA temperature model simulations.

The 1950s temperature data used in the calibration of WQRRS and RBM-10 are of unknown quality. The source of the data was an article on the effects of temperature on fish disease published by USFWS and University of Washington researchers (Ordal and Pacha, 1963). Although the methodology for temperature data collection was not described in the publication, it was assumed that researchers of this caliber would publish only data of the highest quality.

Table 4-4. Comparison of Number of Days Temperature Is Expected to Exceed 20 ^{o}C (68°F) Benchmark and the Magnitude of Exceedances. Based on RBM-10 Model Simulations

Flow Conditions		Dworshak Augmentation	Location	Days Temp. greater than 20ºC	Average Above 20ºC
SNAKE RIVER WITHOUT DAMS					
1984	High Flows	No	RM 10	56	2.0º
		No	RM 107	55	1.7º
		Not Influenced	RM 168*	59	1.3º
1980	Average Flows	No	RM 10	46	1.6º
		No	RM 107	44	1.4º
		Not Influenced	RM 168*	44	1.4º
1988	Low Flows	No	RM 10	69	1.6º
		No	RM 107	67	1.4º
		Not Influenced	RM 168*	70	1.3º
1997	High Flows	Yes	RM 10	34	0.7º
		Yes	RM 107	8	0.6º
1995	Average Flows	Yes	RM 10	55	1.5º
		Yes	RM 107	59	1.2º
1994	Low Flows	Yes	RM 10	38	1.8º
		Yes	RM 107	35	1.8º
SNAKE RIVER WITH DAMS					
Flow Conditions		**Dworshak Augmentation**	**Location**	**Days Temp. greater than 20ºC**	**Average Above 20ºC**
1984	High Flows	No	RM 10	61	2.0º
		No	RM 107	57	1.8º
		Not Influenced	RM 168*	59	1.3º
1980	Average Flows	No	RM 10	57	1.2º
		No	RM 107	46	1.6º
		Not Influenced	RM 168*	44	1.4º
1988	Low Flows	No	RM 10	64	1.3º
		No	RM 107	67	1.3º
		Not Influenced	RM 168*	70	1.3º
1997	High Flows	Yes	RM 10	52	0.7º
		Yes	RM 107	11	0.5º
1995	Average Flows	Yes	RM 10	75	1.2º
		Yes	RM 107	64	1.2º
1994	Low Flows	Yes	RM 10	60	1.1º
		Yes	RM 107	31	1.5º

* Anatone Site
Source: Developed by Normandeau

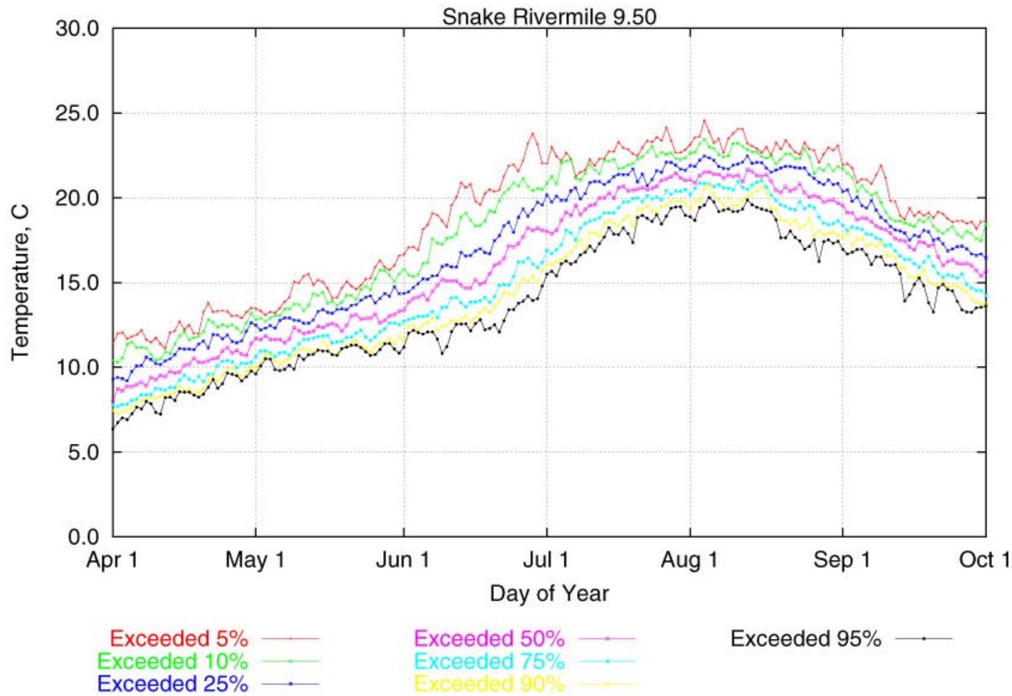


Figure 4-8. The MASS 2 Simulated Temperatures for Lower Snake River with Near-Natural River Conditions at RM 10

Source: Developed by Normandeau. Adapted by Perkins and Richmond, 1999

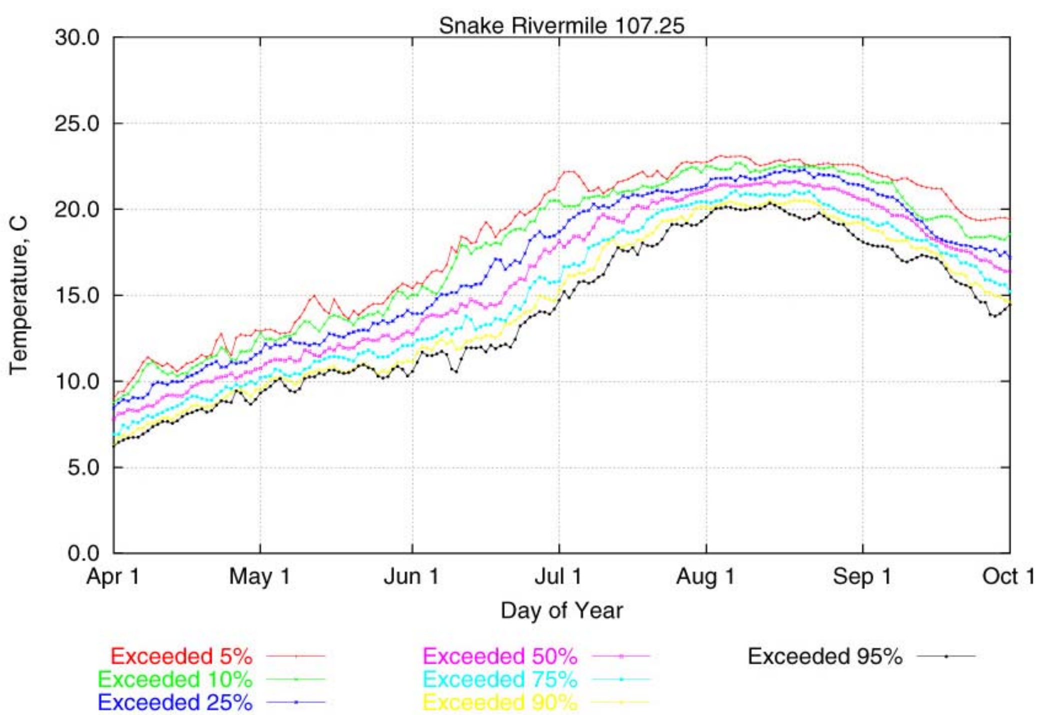


Figure 4-9. The MASS 2 Simulated Temperatures for Lower Snake River with Reservoirs in Place at RM 107

Source: Developed by Normandeau. Adapted by Perkins and Richmond, 1999

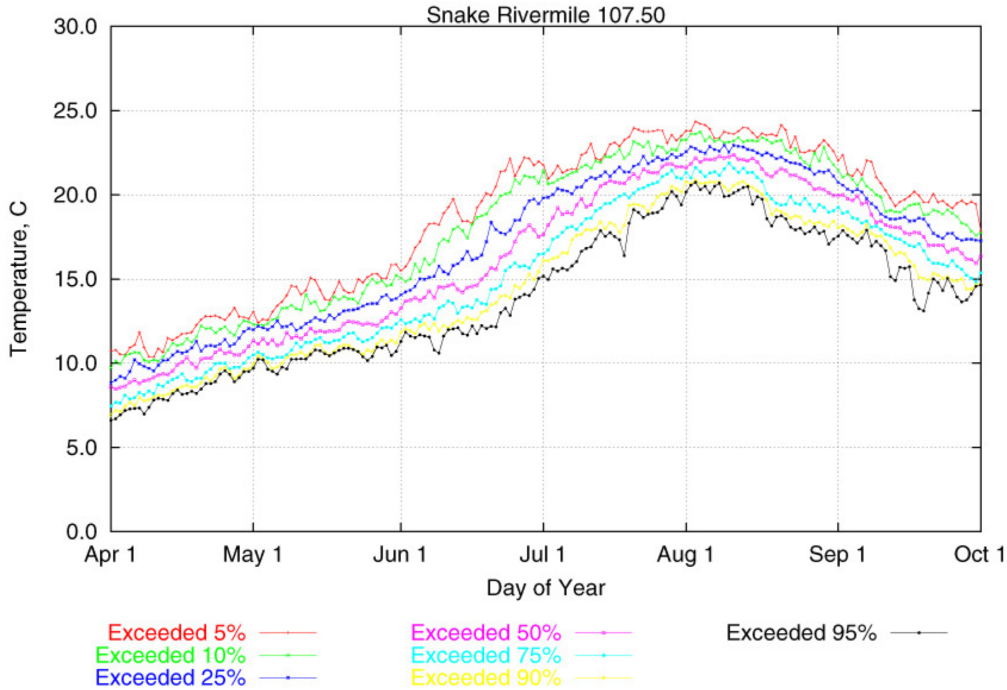


Figure 4-10. The MASS 2 Simulated Temperatures for Lower Snake River with Near-Natural River Conditions at RM 107

Source: Developed by Normandeau. Adapted from Perkins and Richmond, 1999

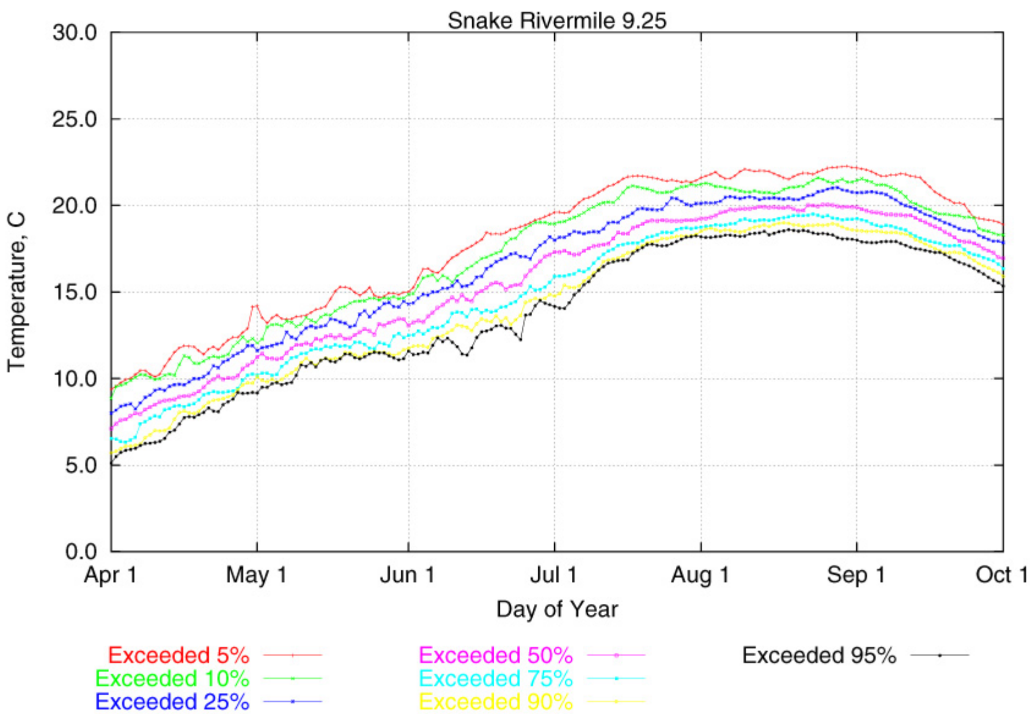


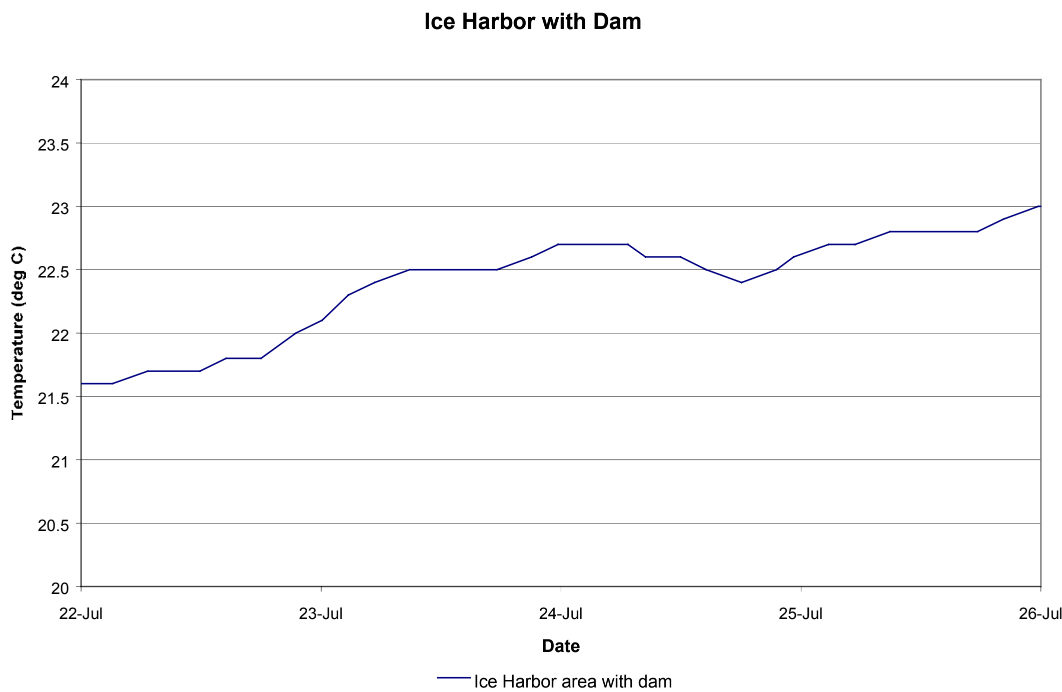
Figure 4-11. The MASS 2 Simulated Temperatures for Lower Snake River with Reservoirs in Place at RM 10

Source: Developed by Normandeau. Adapted from Perkins and Richmond, 1999

As shown by the scroll-case temperature data reported earlier, there is considerable variability in the maximum temperature reached each year and in the duration of exceedance above the 68ºF benchmark. Water temperature in the reservoirs is a function primarily of the temperature of water entering from each of the annual flow volumes of Salmon River, the Clearwater River, and the Hells Canyon reach of the lower Snake River. The Lower Granite reservoir warms up first as the warm summer low flows replace cooler water from the spring, and it cools off first as cool water from the fall displaces the warmer summer water. Depending upon the source, inflow volume of water, and seasonal ambient air temperature, cool water releases from the Dworshak reservoir can reduce the maximum temperature by a few degrees and the duration of exceedance starting with the Lower Granite reservoir and diminishing as the water flows downstream through the Ice Harbor reservoir.

Battelle Pacific Northwest Laboratories has just recently completed the initial phase of a study of the effects of hyporheic (ground water) flows on river water temperature in an area of basalt geology that supports fall chinook spawning. Preliminary results indicate very large temperature differences between ground water and surface water. These differences diminish spatially over fairly small scales (tens of meters). During the spawning and incubation period (fall/winter/early spring) in some of the study areas, ground water is 8 to 10ºC (14.4 to 18°F) warmer than the surface water. Over most of the study area the difference is 3 to 5ºC (5.4 to 9°F). That warm water upwelling has significant implications for emergence timing, and probably plays a role in redd site selection by spawning adults. Because ground water temperatures remain fairly constant year round, the difference in temperature reverses during late summer when groundwater can be 6 to 8ºC (10.8 to 14.4°F) cooler than the surface water. In general, temperature differences would almost always be apparent at the interface of ground water and surface water, but the magnitude of the influence depends heavily on the vertical hydraulic gradient of the hyporheic water-which helps quantify the degree of upwelling (or downwelling) in an area. The vertical hydraulic gradient is heavily controlled by channel morphology and the substrate composition/character, as well as the stage and discharge of river water (Tim Hanrahan, Battelle Pacific Northwest Laboratories, personal communication). Deep water in reservoirs decreases the amount of upwelling of ground water due to its vertical hydraulic gradient. In addition, the anoxic conditions that develop near the bottom of reservoirs in the summer months make many ground water upwellings inaccessible to salmon and steelhead. These ground water upwelling areas could be extremely important as temperature refugia for salmonids during the hot summer period when average surface water temperatures are at or near lethal levels.

The RBM-10 model was used to predict the natural daily oscillations in surface water temperature, which is also important in terms of the effects of high temperatures on salmonids. An example of daily temperature fluctuations in the warmest part of a low flow year is shown in Figure 4-12. It can be seen from this simulation that temperature can be expected to vary about 1 to 2°C (1.8 to 3.6°F) within a 24-hour period with maximum temperature occurring about midday and minimum temperature occurring during the night. RBM-10 temperature simulations can be compared to temperature measured at the scroll case or at the TDGMS tailwater station by looking at Table 4-4, which is based on simulation temperature data and Tables 3-3, 3-4, and 3-5, which are based on temperature measured at the scroll case or at the TDGMS tailwater station. From this comparison, it can be seen that the difference in days exceeding the 20 degree benchmark range from 0 to 27.



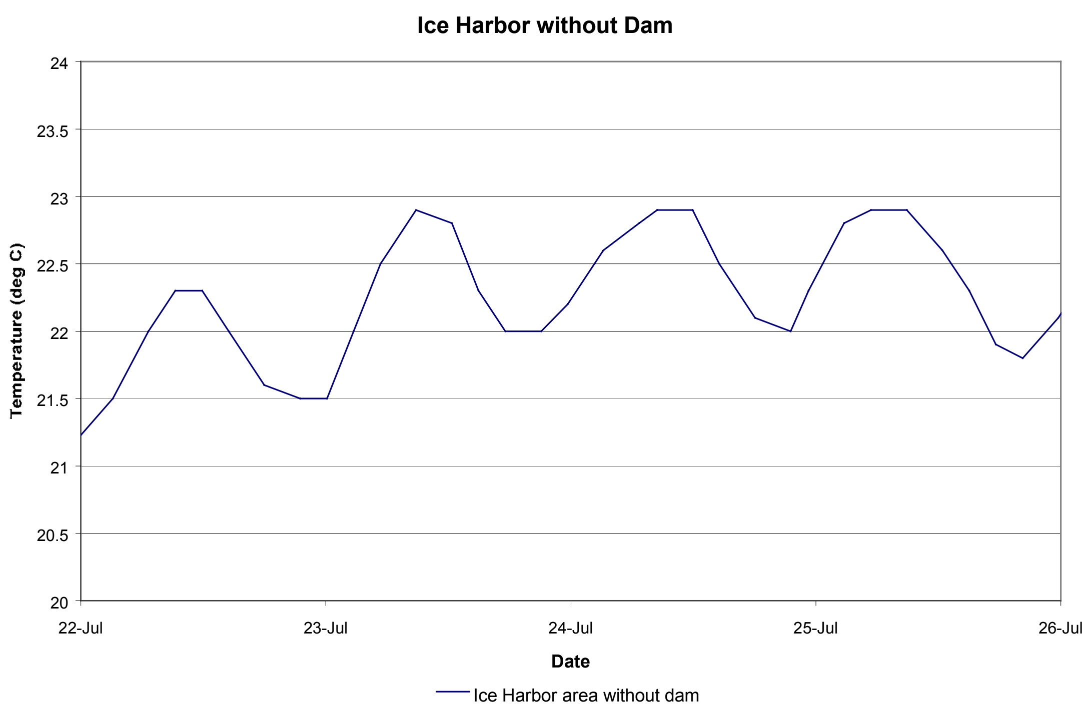


Figure 4-12. Diel Variation in Temperature of the Lower Snake River at Ice Harbor Dam (RM 10) with Dams in Place and without Dams

Source: Developed by the Corps

Based on these methods and models, temperature-related effects on dam breaching are summarized as follows:

(a) Temperature would warm up a few days earlier in the spring and cool down about 2 weeks earlier in the late summer. The effects of the lag in warming in the spring on ESA-listed fish is probably inconsequential. The effects of the lag in cooling in late summer probably has an effect on the timing of adult in-migration; however, the precise effect is not understood well enough to be described.

(b) Dworshak augmentation has lowered the mid-summer temperatures in the lower Snake River. This has been a definite benefit to fall chinook salmon reared in the Snake River. It is most beneficial when augmentation occurs in July and August. Fall chinook salmon reared in the Clearwater River have actually experienced a change in their life history as a result of Dworshak augmentation. Many of these fish outmigrate a year later than they originally did, which may have some effect on overall production although it is impossible to quantify at this time. If Dworshak augmentation for temperature control is deemed necessary, it would be more effective in lowering temperature with the dams removed.

(c) Diel temperature variations would be much greater with temperatures being 1 to 2°C (1.8 to 3.6°F) cooler at night. The effect of this daily oscillation in temperature is very important to the well being of fall chinook rearing in the Snake River, as well as adult immigrants of the other ESA-listed stocks. Salmonids are coldwater species that generally cannot tolerate temperatures much above the low to mid 70s. It is recognized, however-although difficult to quantify-salmonids can tolerate otherwise lethal temperatures if they have a periodic reprieve from the high temperatures. This phenomenon is actually best manifested by spring chinook that rear in tributary streams and survive temperature fluctuations much greater than those occurring in the lower Snake River. In the lower Snake River, fall chinook would suffer less temperature stress if the temperature fluctuates more on a daily basis, which means lower temperatures during the night.

(d) With removal of the dams more ground water seeps would be available as temperature refugia for migrating adults of chinook and steelhead and juvenile fall chinook. In the impounded river there are probably fewer ground water upwellings, and most upwellings that do exist are not available during the critical hot part of the year because the dissolved oxygen level near the bottom is too low to support fish.

(e) Without Dworshak augmentation, average temperature would probably be almost the same for the impounded river and the free-flowing river. Using hydrometeorological data from the 15-year period, 1975 through 1990, the RBM-10 model predicts that the average temperature during the June through August period would be 18.9°C (66.0°F) at RM 107 (Lower Granite Dam) with the dams in place and 19.1°C (66.4°F) without the dams and 18.8°C (65.8°F) at RM 10 (Ice Harbor Dam) with the dams in place and 19.4°C (66.9°F) without the dams.

Short-term changes would be expected above and below dams as they are breached. Temperatures would equilibrate rapidly as the velocity of flow increases in the reservoir areas as they return to the natural river level. Upstream releases could still be used to moderate temperatures.

4.2.2 Dissolved Oxygen

Dams and reservoirs each have an effect on dissolved oxygen concentrations in an impounded river system such as the lower Snake River. As discussed previously, the process of spilling at dams increases the concentration of dissolved atmospheric gases, including oxygen, in the river and under certain conditions, supersaturation can result. The effects of dams on oxygen concentration is covered in Section 3.2.4.3, Total Dissolved Gas Superstaturation and the effects of the various alternatives on levels of spill at the lower Snake River dams will be discussed in Section 4.3. The present discussion will cover the effects of reservoirs on dissolved oxygen. Alternatives 1, 2, and 3 would maintain the four lower Snake River reservoirs, while Alternative 4—Dam Breaching would return the river to a free flowing condition.

Free-flowing rivers rarely experience dissolved oxygen levels that are low enough to be a cause for concern. This is because flowing water is continually circulating and re-aerating because the entire volume of the river is in continual contact with the atmosphere. As a general rule, when rivers are impounded lower, dissolved oxygen concentrations occur in certain areas of the reservoir at certain times of the year. The worst cases of oxygen depletion in reservoirs occur in large storage reservoirs with long retention times located in warm, fertile regions. In temperate climates oxygen depletion in storage reservoirs is usually a problem only during the late summer months. It occurs when reservoirs have little or no circulation and, hence, little or no atmospheric reaeration at the water/air interface and oxygen is consumed faster than it is produced. Oxygen is produced by photosynthesis near the surface and it is consumed by the biological respiration of organisms in the water column and on the bottom (e.g. microbial decomposers), as well as chemical oxidation in the sediments. With no atmospheric re-aeration, the finite amount of oxygen dissolved in the water of the reservoir is first depleted near the bottom in the headwater section of the reservoir where organic content of the sediments is high. As the season progresses the area of oxygen depletion gradually moves downstream and further up into the water column.

4.2.2.1 Alternative 1—Existing Condition

The lower Snake River dams and reservoirs are run-of-the-river which, in terms of oxygen depletion, means they are somewhere between a free-flowing river and a storage reservoir. When the Snake River flows are high (either naturally or because of augmentation), they are more similar to a free-flowing river, but when flows are low, they are more similar to non-circulating reservoirs. Oxygen depletion would be expected to occur near the bottom of upstream reaches during the summer when flows in the Snake River are at their lowest level. The discussion in Section 3.2.4.2 indicates that dissolved oxygen concentrations in the free-flowing river immediately upstream of the Lower Granite pool (USGS gaging station at RM 167.2) averages 8 to 9 mg/L, with its lowest level of the year during July and August. Dissolved oxygen levels below 8 mg/L have been observed throughout the water column at downstream sampling stations; however, oxygen depletion generally follows the expected scenario.

4.2.2.2 Alternative 4—Dam Breaching

Breaching of the dams on the lower Snake River would result in a return of the river to a somewhat free-flowing state, which would result in dissolved oxygen concentrations similar to those shown in Figure 3-36. Dissolved oxygen concentrations would normally remain above 8 mg/L throughout the year.

4.3 Total Dissolved Gas Saturation

A potential negative aspect of increased flow augmentation is that more water may need to be released in the dam spillways, which can increase dissolved gas saturation levels. Water that contains high TDG, hence high TGP can be harmful to fish. The TDG saturation is directly related to the amount and duration of spills. The Corps has instituted a program over the years to reduce gas entrainment in the tailwater by regulating spills based on measured TDG in the river above and below each dam. However, these measures have limited effectiveness if dissolved gas levels contributed from upstream are already elevated. The TDG saturation generally becomes progressively worse downstream as the waters pass through several spillways.

For the major system improvement alternative, potential structural changes have been evaluated to change TDG conditions. For the Major System Improvement alternative, potential structural changes have been evaluated to TDG conditions.

4.3.1 Alternative 1—Existing Conditions

This alternative represents a continuation of the current system operations as they have been implemented since the issuance of the 1995 Biological Opinion by NMFS, including flow augmentation up to 527 million cubic meters (427 thousand acre-feet). The addition of end bay deflectors at Lower Monumental and Little Goose Locks and Dams is assumed for this alternative. Modified deflectors at Lower Monumental, Little Goose, and Lower Granite Dams are also assumed as part of this base case.

Spill to 120 percent TDG as defined in the 1995 and 1998 Biological Opinion would be executed. Forced spill would likely be similar to 1996 through 1998 operations. Spill caps could remain at current kcfs or be increased as TDG production is reduced due to spillway improvements. The increases in spill discharge to attain 120 percent TDG are estimated to be from 45 kcfs to 68 kcfs at Lower Granite Lock and Dam, from 48 kcfs to 68 kcfs at Little Goose Lock and Dam, and from 43 kcfs to 68 kcfs at Lower Monumental Lock and Dam. The gas abatement improvements used with current voluntary spill discharges would result in TDG levels of 112 to 115 percent.

The TDG monitoring results from 1998 depict existing conditions in the tailwaters of the four dams on the lower Snake River (Corps, 1999a). Transect data collected at the TDGMS located 1.1 kilometers (0.65 mile) downstream of the Lower Granite spillway indicate TDG can exceed 110 percent with spill discharges ranging from near zero to 30 kcfs and 120 percent TDG with spill flows of near 60 kcfs. Figure 4-13 illustrates the average TDG concentration generated by discharge from the existing spillway structure, assuming a uniform distribution of spill through all eight bays. It should be noted that special spill operations for the ongoing surface collection study significantly reduce the discharge through spillbays 1 and 2. As a result, TDG detected by the fixed monitoring station below Lower Granite Lock and Dam may detect higher concentrations than that expected from a uniform spill distribution.

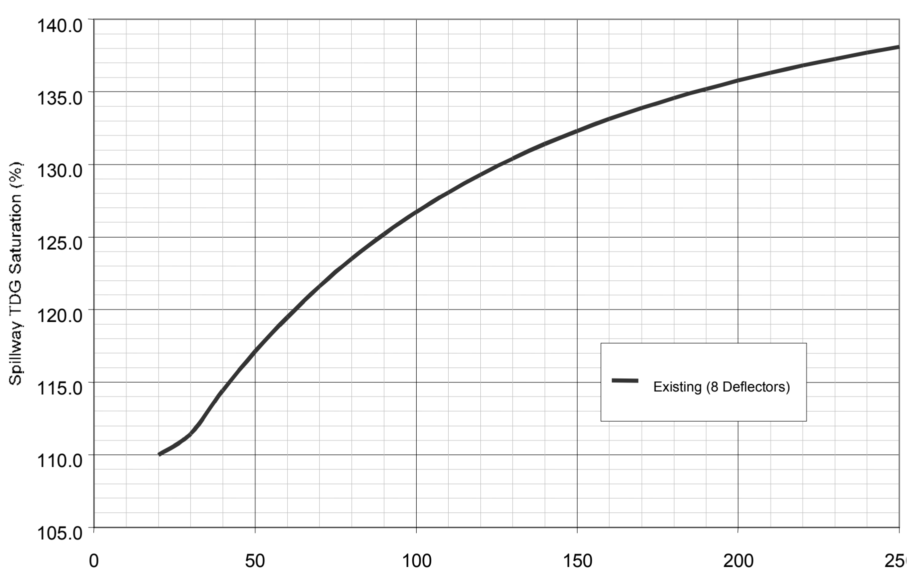


Figure 4-13. Lower Granite Lock and Dam 1998 TDG and Total Spillway Flow
Source: Developed by the Corps

The TDG data collected at the TDGMS located 1.4 kilometers (0.85 mile) downstream of the Little Goose spillway indicate TDG can exceed 110 percent with spill releases from near zero to 20 kcfs. When spilling uniformly through the six deflected bays, spill volumes of 45 to 50 kcfs would generate near 120 percent TDG. Increased spill would cause exceedance of 110 percent benchmark.

The adult fish passage spill pattern requires in excess of 25 percent of the total spill volume through the two outside nondeflected spillbays, allowing TDG to exceed 120 percent with as little as 25 kcfs. Figure 4-14 compares the average TDG concentration generated by daytime spill releases to the average concentration generated by nighttime releases. During the daylight hours the spill is distributed across the spillway according to the adult fish passage spill pattern. At night spill flows are released uniformly through the six deflected spillbays.

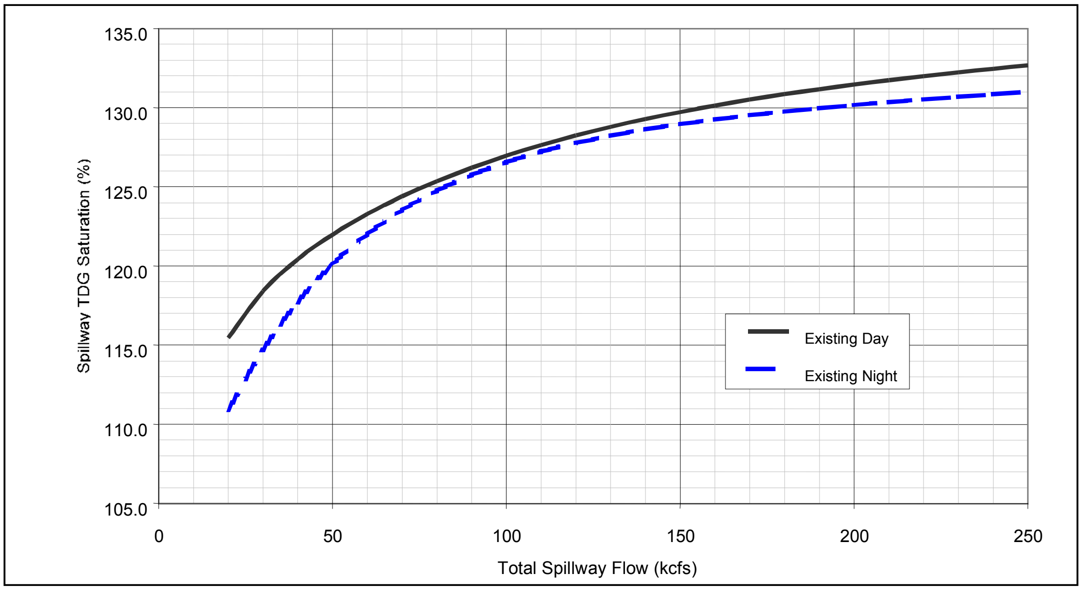


Figure 4-14. 1998 Little Goose Lock and Dam TDG and Total Spillway Flow (day versus night)
Source: Developed by the Corps

The TDG data collected at the TDGMS located approximately 1.3 kilometers (0.83 mile) downstream of the Lower Monumental spillway indicate TDG can exceed 110 percent with spill releases from near zero to 16 kcfs. When spilling uniformly through the six deflected spillbays, spill volumes of 40 kcfs can be reached before exceeding 120 percent TDG. The daytime or adult fish passage spill pattern requires in excess of 25 percent of the total spill volume through the two outside nondeflected spillbays, allowing TDG to exceed 120 percent with as little as 20 kcfs. Figure 4-15 was developed from near-field test results of the Lower Monumental spillway. This figure compares the TDG concentration generated by daytime and nighttime spillway releases. The TDG data are collected at the TDGMS located approximately 5.8 kilometers (3.6 miles) downstream of the Ice Harbor spillway. The data indicate that TDG of 110 percent can be exceeded with spill releases from near zero to 20 kcfs. Spill volumes have reached as high as 90 kcfs before exceeding 120 percent TDG. With construction of the two new deflectors in spillbays 1 and 10, the 120 percent TDG may be expected to range from 90 to 100 kcfs. Figure 4-16 compares the existing 8-deflector (1998) performance to the expected 10-deflector (1999) performance. These plots have been generated from TDG measurements at the FMS below Ice Harbor Lock and Dam.

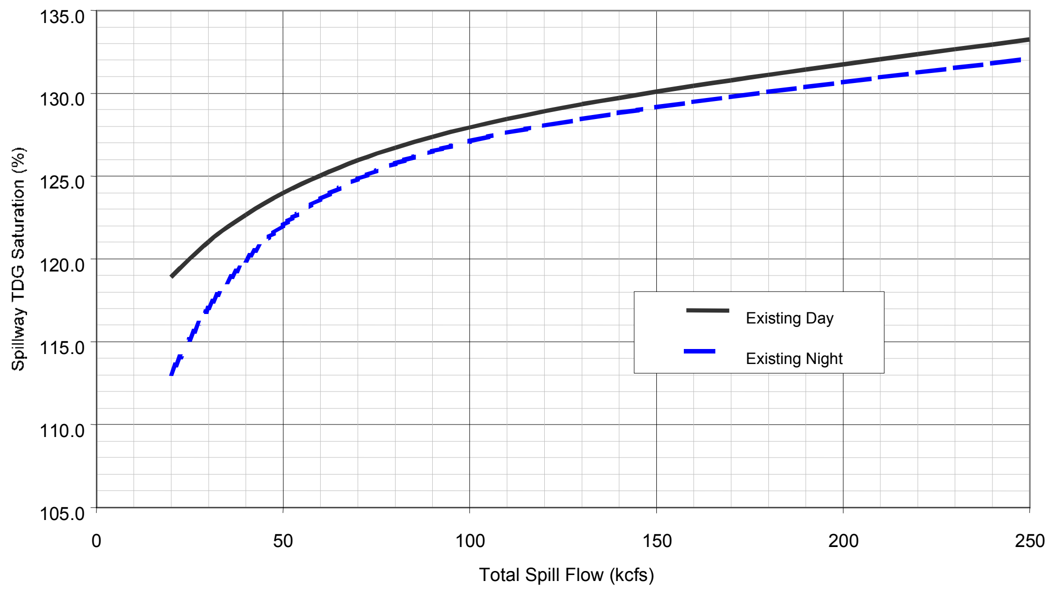
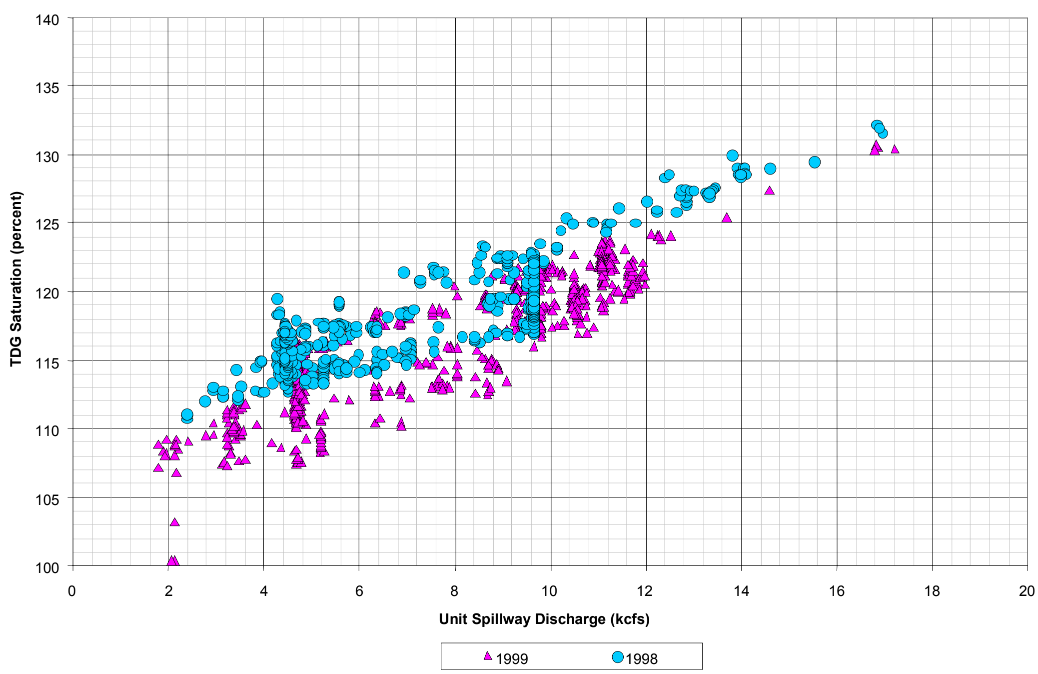


Figure 4-15. Lower Monumental Lock and Dam 1998 TDG and Total Spill Flow (day versus night)
Source: Developed by the Corps



Source: Corps, Walla Walla District
Note: 8 of 10 deflectors installed in 1998, 10 of 10 deflectors installed in 1999

Figure 4-16. The TDG Measured Below Ice Harbor Lock and Dam, 1998 and 1999
Source: Developed by the Corps

Figure 4-17 illustrates 1998 spill season total discharge, spill discharge, and resulting downstream TDG at Ice Harbor Lock and Dam. The TDG response closely tracked spill discharge, even when averaged over 6 hours. Plots for the TDG stations in the rest of the lower Snake River Dam tailwaters display similar characteristics.

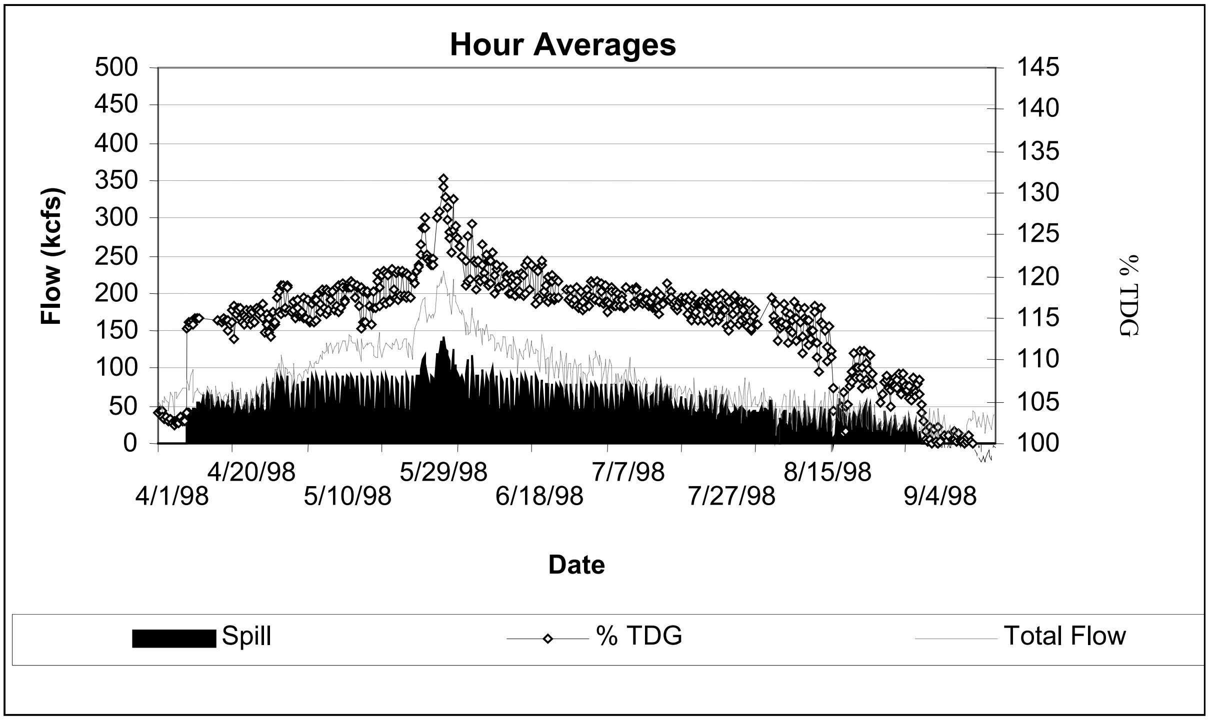


Figure 4-17. Ice Harbor Lock and Dam 1998 Total Flow, Spill and Percent TDG at Downstream Fixed Monitoring Station (IDSW)-6

Source: Developed by the Corps

4.3.2 Alternative 2—Maximum Transport of Juvenile Salmon

The addition of end bay deflectors at Lower Monumental and Little Goose Dams is assumed for this alternative. Voluntary spill remains only for noncollected smolts at Ice Harbor Lock and Dam under this alternative, and the voluntary spill discharge cap of only 110 percent of TDG would be expected.

Forced spill would likely be similar to 1996 through 1998 operations. Supersaturation of TDG during involuntary spill conditions could still exceed 130 to 140 percent in tailwaters for several weeks system-wide. Production of TDG supersaturation at Lower Monumental and Little Goose Locks and Dams would be reduced somewhat due to additional end bay deflectors. Because voluntary spill is eliminated at these facilities, increases in spill caps are not pertinent.

4.3.3 Alternative 3—Major System Improvements

End bay deflectors at Lower Monumental and Little Goose Dams would be added for these alternatives. The major fish passage improvements proposed under this alternative would result in a small spill discharge resulting from dewatering of the Surface Bypass Collector over a spillbay. This, however, would lead to only small increases in TDG loading to the system. A proportional increase in this source would occur as river flows decrease and TDG levels would have to be limited to 110 percent. Production of TDG supersaturation at Lower Monumental and Little Goose Dams would be reduced somewhat due to additional end bay deflectors. Supersaturation of TDG during involuntary spill conditions could still exceed 130 to 140 percent in tailwaters for several weeks system-wide.

The engineering options being studied to reduce dissolved gas include:

- Powerhouse/spillway separation wall
- Additional spillbays.

For more information on the engineering options, refer to Annex C of Appendix E, Existing Systems/Major System Improvements Engineering.

A cutoff wall (separation wall) constructed between the powerhouse and spillway would prevent powerhouse flow from becoming entrained and aerated within the spillway's stilling basin. This would allow mixing of powerhouse and spillway releases to occur downstream of the highly aerated spillway releases. This mixing in the downstream channel as opposed to the stilling basin would allow dilution of higher TDG spillway water with lower TDG powerhouse water, which is normally at lower forebay TDG levels. In addition to the gas reduction benefits of the flow separation wall, the wall would prevent juvenile fish passed through the turbines from being drawn into the spillway. This condition has been observed at McNary during the 1999 turbine survival studies. The separation wall would streamline powerhouse flow and improve current flow patterns below the juvenile fishway out-falls and would reduce or eliminate large eddies that might otherwise delay juvenile fish egress from both powerhouse and spillway tailrace regions. Two concept level designs were developed based upon models of Lower Granite and Ice Harbor Dams. Both designs include two 75-foot-long concrete monolithic structures that are post tensioned. The first design concept utilizes sheet pile to construct the wall forms and fills the form with mass tremie concrete. The second design concept utilizes pre-cast concrete cells set in place and then filled with tremie concrete. The design and construction of a divider wall at either of the lower Snake River projects could take 3 to 4 years.

The walls could be added with any of the surface bypass collector (SBC) types included in this appendix. However, more study is required to determine if the separator walls would be an appropriate addition to the dams. Because of this uncertainty, the walls were not included in any of the Major System Improvement options described herein.

Adding more spillbays at each dam would reduce the generation of TDG by reducing the unit spill discharge requirements and necessary stilling basin depths. Unlike conventional spillways designed to pass and adequately dissipate the energy of flow for the Spillway Design Flood, the additional spillbays could be designed for much less spill. The spillway would be designed specifically to reduce the saturation of TDG for normal or voluntary spill flows, while improving the spill passage efficiency and survival of juvenile fish.

Additional spillbays could be constructed in place of the earthen non-overflow embankments of the lower Snake River dams.

4.3.4 Alternative 4—Dam Breaching

Under this alternative, there would be essentially no more hydraulic head at the four lower Snake River dams and, therefore, no spill. However, due to formation of plunge pools during resetting of quasi-equilibrated fluvial geomorphology of channel and water temperature dynamics, geographically localized TDG above 110 percent is possible infrequently and for short durations. Maximum salmonid production requires active management of increasing flow magnitude and frequency according to historical hydrograph shaping. This could increase the frequency of increased spill from Hells Canyon and Dworshak Dams in large pulses to lower Columbia River Dams, which would likely exceed the 110 percent saturation benchmark.

4.4 Sediment Movement and Related Downstream Impacts

Changes in sediment movement are affected by flow rate and water depth. Flow rate and water depth in turn affect erosion and downstream sediment movement and increase suspended sediments. All the above processes impact water quality and primary productivity.

The quantity of material suspended in the water can be measured as total suspended solids (TSS), which is merely a weight per unit volume measure, or more commonly, as turbidity, which is a measure of light reflection and refraction, which is expressed in nephelometric turbity units (NTU). Under normal conditions, turbidity is the parameter of choice due to the ease with which it can be measured versus the more time consuming filtering, drying, and weighing of TSS. However, when sediment loads are heavy enough that light transmission is impaired, it is mandatory that TSS be used to measure them. Unfortunately, there is no easy, generic conversion from NTUs to TSS due to the varying composition and density of the material suspended in the sample.

4.4.1 Alternative 1—Existing Conditions

Under existing conditions, the total annual sediment influx to the lower Snake River system has been estimated to be around 2.3 million cubic meters (3 million cubic yards). Much of this sediment is contributed from the middle and upper reaches of the lower Snake River. However, a significant portion may be contributed from several of the smaller tributaries, especially the Palouse River. Although flow contributions from the Palouse River are relatively minor and generally account for less than 1 percent of the total flow, peak suspended sediment concentrations have been reported to be more than 10 times greater than those measured in the lower Snake River. Based on the 1997 data, suspended sediment concentrations throughout the lower Snake River were generally below 25 milligrams per liter, with peak concentrations ranging between 50 and 75 milligrams per liter. The peak concentrations in the Palouse River were well above 1,000 milligrams per liter and averaged around 200 milligrams per liter for the sampling season.

As mentioned earlier, since the construction of the four dams, turbidity levels and suspended concentrations in the lower Snake River have generally been viewed as being considerably reduced from pre-impoundment conditions due to the slower flow velocities in each of the impoundments, allowing for greater settling. In general, the net effect was more likely reflected in a reduction in suspended sediment concentrations rather than the turbidity levels. Turbidity data collected from 1954 to 1957, prior to dam construction, indicate that turbidity levels typically ranged from 10 to 20 NTUs in the lower Snake River and higher during peak flow periods (BPA, 1995). During the 1997 sampling season, except for some occasional peak levels of 15 to 20 NTUs during the spring freshet, turbidity levels throughout the lower Snake River were typically below 10 NTUs. Because monthly flow conditions during 1997 generally were well above normal, the observed suspended sediment and turbidity levels would also be expected to be above normal. In general, the existing turbidity levels and suspended sediment concentrations are considered to be below levels that may be harmful for fisheries and other aquatic life. However, periods of elevated turbidity levels above 10 NTUs, typically during the spring freshet, can temporarily have an adverse effect or require additional treatment for downstream potable and irrigation water supply users.

4.4.2 Alternative 2—Maximum Transport of Juvenile Salmon

Flow operations under the Maximum Transport of Juvenile Salmon alternative would be the same as Alternative 1, and, therefore, no significant short-term or long-term changes in sediment erosion and movement would be expected.

4.4.3 Alternatives 3—Major System Improvements

Similar to Alternative 2, significant changes in sediment erosion and movement would not be expected with any of the four options included in the Major System Improvements alternative.

4.4.4 Alternative 4—Dam Breaching

To understand and explain differences in the water quality and biological production of the Existing Conditions and Dam Breaching alternatives, some understanding of the difference in physical characteristics expected between the two is necessary. As part of the Columbia River Salmon Mitigation Analysis System Configuration Study Phase (BPA, 1995), the Corps developed time of travel curves as a function of river flow for the study reach for various alternatives using HEC-2 model simulations. A comparison of the curves shows that the time of travel for the Dam Breaching alternative is about one order of magnitude less than for the existing conditions. At a typical summer low flow of about 25 kcfs, travel time from the Clearwater River to Columbia River for the existing conditions is about 35 days compared to about 2.5 days for the Dam Breaching alternative. At a typical May/June flow of 120 kcfs, travel time would be on the order of 1 day for the Dam Breaching alternative and 7 days for the Existing Conditions alternative. Therefore, breaching the dams would be expected to have very rapid flushing compared to the existing conditions.

Water depths and available habitat also would be expected to vary significantly between the existing conditions and the dam breaching alternative. Flow depths currently remain relatively constant throughout the year and range from about 6 meters (20 feet) in the tailwater areas to over 30 meters (100 feet) at the dams. In contrast, flow depths would vary seasonally with flow with the dam breaching alternative. During a typical spring runoff period (120 kcfs), average flow depth over a cross section would be on the order of 7.6 meters (25 feet) compared to 4.6 meters (15 feet) during a typical summer flow condition. Differences in surface area or average width would not be as drastic. The total surface area for the existing system is currently approximately 13,451 hectares (33,236 acres) compared to 7,877 hectares (19,464 acres) expected for the Dam Breaching alternative. Reduced volume would affect concentrations of water quality.

Over time, the bottom substrate would likely change following dam breaching. The existing reservoirs contain embedded fine sediments that have accumulated over a 40-year period. It is anticipated that extremely high river flows would be necessary to resuspend fine sediments and remove interstitial materials. A geomorphological study that evaluates the river flows and substrate changes following dam breaching (Hanrahan et al., 1999) indicates that with sufficient flows over time the substrate would consist primarily of bedrock and boulders in fast-moving sections and cobble in slow-moving sections.

One of the primary water quality concerns associated with this alternative relates to the potential for considerable increases in suspended sediment concentrations and turbidity levels as the accumulated sediment behind these dams becomes resuspended and moves downstream. The potential increases would reach levels during the initial drawdown period that would adversely affect aquatic biota and other beneficial uses. The increased turbidity can adversely affect both primary food production (i.e., phytoplankton and attached benthic algae growth) and fish feeding efficiency. In addition, depending on the magnitude of the TSS concentrations, impairments to other biological functions such as respiration (i.e., gill clogging) and reproduction are possible. Recent sediment volume estimates developed by the Hydrology Branch of the Corps, Walla Walla District, indicate that approximately 76 to 155 million cubic meters (99 to 203 million cubic yards) of sediment has accumulated behind the four lower Snake River dams. Roughly 65 percent of the total accumulation is in the Lower Granite pool simply because it has been the upstream reservoir for 27 of the 40 impounded years. The other 35 percent is distributed between the other three reservoirs such that approximately 80 percent of the accumulated sediment is in two upstream reservoirs and 20 percent is in the two downstream pools. Approximately 50 percent of this previously deposited sediment would be expected to erode and move downstream within the first few years following dam breaching, particularly during peak flow periods (Corps, 1998c). Most of this eroded material would be expected to settle out downstream in the McNary reservoir. The McNary reservoir is generally considered to have comparatively lower-flow velocities than those in the lower Snake River impoundments, mainly because it is nearly twice the size of the largest reservoir in the lower Snake River. More-recent analyses indicate that sediment would be expected to accumulate mainly on the eastern shore of the McNary reservoir between the Snake River Confluence and Wallula Gap. Smaller areas of deposition would be anticipated on the opposite shoreline, on the western shore immediately downstream of Wallula Gap, and just upstream of McNary Dam (Appendix F, Hydrology-Hydraulics and Sedimentation, Corps [1999b]).

The actual amount of sediment transported by the Snake River prior to impoundment is unknown. It can only be assumed that the suspended sediment load was highest during peak flows associated with the spring snow melt and experienced considerable sporadic variation as a result of naturally occurring events such as fires or land slides in the catchment area. With the advent of agriculture, silviculture, road building, and other land-disturbing activities throughout the drainage, the amount of sediment entering the system as a result of normal erosion has increased dramatically. Under present conditions the normal sediment transport regime of the Snake River drainage is highly manipulated by anthropogenic activities throughout the basin. Sediment from the middle Snake, Salmon River, and Clearwater River drainages, or about half of the entire Snake River catchment, accumulates in the Lower Granite pool. Dams upstream on the Snake River store sediment from the rest of the Snake River drainage.

The particle size of sediment entering the Lower Granite reservoir ranges from less than 0.002 millimeter (0.00008 inch) to 128 millimeters (5.04 inches) in diameter with about 95 percent of the material being medium sand (0.25 to 0.5 millimeter in diameter) or smaller (Appendix F). It is assumed that the particle size distribution presented in Appendix F is reasonably representative of the material that is deposited in the four lower Snake reservoirs, although it is probable that some fraction of the smallest sized material remains suspended and washes on through the system.

Alternative 4 calls for the two upstream dams to be breached the first year and the two downstream dams to be breached the following year. With initial breaching, presumably in the late summer low-flow period, velocities would immediately increase to the point that the accumulations of sediment in the Lower Granite and Little Goose reservoirs would be mobilized first at the upper end of the reservoirs and then progressing downstream as the energy gradient permits. Based on observations in the Corps' 1992 experimental drawdown test, it is believed that very large amounts of material would be transported downstream to the Lower Monumental pool, where most of it would probably settle out. Much of smaller material would be transported on through the two remaining dams and into the McNary reservoir, where all but the very finest clay fraction would settle out. Although it would remain high, the suspended sediment load would possibly decline somewhat within a few weeks/months following initial breaching.

However, as flows increase in the October/November time frame, the suspended sediment concentration would increase dramatically. Very heavy sediment loads as a result of breaching the two upstream dams would probably be present throughout the following year until the following August, when the two downstream dams would be breached. The timing of sediment movement from breaching Lower Monumental and Ice Harbor Dams would be about the same as described for the upstream dams.

The actual concentrations of suspended sediment that would be experienced as a result of breaching the lower Snake River dams are unknown. The only pertinent empirical data available were collected during the Corps' 1992 experimental drawdown test when 2,000 milligrams per liter TSS was recorded in the upper end of the Lower Granite pool. Judging from observations at the site when the drawdown experiment was conducted, this concentration is low compared to concentrations that would probably be experienced with dam breaching.

Total suspended sediment concentration observed in 1992 is consistent with concentrations predicted by the HEC5-Q numeric model, which was used in a previous study conducted in 1994 to predict total suspended solid concentrations if the lower Snake Dams were breached. The HEC5-Q predicted concentrations of 3,600 milligrams per liter at Lower Granite Lock and Dam, 6,000 milligrams per liter at Little Goose Lock and Dam, 7,000 milligrams per liter at Lower Monumental Lock and Dam, and 9,000 milligrams per liter at Ice Harbor Lock and Dam (Sediment Transport Report, Columbia River System Operation Review).

Predictions of post-drawdown water quality focus on TSS, which does not have associated water quality standards, rather than turbidity, which is used in state water quality classification (Table 3-1). The predicted TSS levels indicate that turbidity standards would be violated at an unknown frequency.

The toxic effect of extreme suspended sediment loads on aquatic life has been researched and documented but is not very well understood. The 25 milligrams per liter TSS threshold has potential sublethal effects for adult salmonids and lethal effects for juveniles (Newcombe and Jensen, 1996). Adult salmon had reduced feeding activity after four hours of exposure to 25 milligrams per liter TSS (Phillips, 1970). However, Newcombe and Jensen (1996), in their review article of TSS effects on fish, saw no evidence of ill effects of 25 milligrams per liter TSS on adult salmonids. They theorized that TSS at 20 milligrams per liter would show sublethal effects at four months' exposure. Arctic grayling salmon showed an avoidance response after 24 hours of exposure at 20 milligrams per liter (Birtwell et al., 1984). Larvae had a 5.7 percent mortality at 24 hours of exposure at 25 milligrams per liter (Newcombe and Jensen, 1996). Juvenile coho salmon showed decreased feeding rates at 1-hour exposure at 25 milligrams per liter (Noggle, 1978), and increased physiological stress at 12 hours of exposure at 53.5 milligrams per liter. Newcombe and Jensen (1996) predicted that a 2-week exposure of juvenile salmonids to 20 milligrams per liter TSS would cause severe effects. At these concentrations, TSS has caused increased mortality of juvenile chinook, egg, fry, smolt, and under-yearling coho, and juvenile and under-yearling sockeye (Newcombe and Jensen, 1996). Direct mortality is a result of impaired gas exchange at the gills due to reduced gill/water interface and actually results from asphyxia more so than from gill abrasion. There seems to be considerable variation in the effect of different types of sediment particles on fish. For example, sharp and pointed volcanic ash is more damaging than rounded-off, fine sand particles. There also seems to be some variation in the ability of fish stocks with different genetic backgrounds to withstand the effects of high sediment loads; for example, stocks that evolved in river systems with high loads of glacial flour are not as affected by high loads of suspended sediment as other stocks.

Perhaps the most extensive quantitative assessment of the effect of suspended sediment on fish was done by Newcombe and Jensen in 1996. From their work, it was possible to identify the following as benchmarks for the effects that we anticipate from elevated suspended sediment loads resulting from Alternative 4:

- 110 milligrams per liter silt—adult salmon cease upstream migration
- 3,000 milligrams per liter silt for 1 day—physiological stress in adult and juvenile salmon
- 3,000 milligrams per liter silt for 49 days—greater than 60 percent mortality in adult and juvenile salmon
- 4,300 milligrams per liter silt/very fine sand for less than 57 hours—80 to 100 percent mortality in salmonids.

Figure 4-18 graphically shows when extreme levels of suspended sediment would be present in the downstream reaches of the lower Snake River if the dams were breached. Keep in mind that it is impossible to accurately predict the concentrations of suspended sediment that would occur but it is safe to assume that concentrations would be extremely high.

Figures 4-19 and 4-20 show the presence of adults and juveniles of the ESA-listed stocks as recorded at counting stations located at the dams. It is important to note when interpreting this figure that, while spring/summer chinook salmon, steelhead and sockeye salmon use the lower Snake River as a migratory route only, fall chinook salmon also spawn and rear in the mainstem lower Snake River (Aquatic Ecology Appendix).

From Figures 4-19 and 4-20 it is evident that juvenile spring/summer chinook and steelhead that outmigrate with the spring freshet (February through April) and fall chinook that outmigrate a few weeks later (April through September) but are present in the system year round, would all be exposed to extreme suspended sediment concentrations in the 238 kilometers (148 miles) of the lower Snake River. The time required for outmigrating juveniles to traverse the lower river reach after breaching would be 2 to 3 days.

Adults trying to migrate through the lower Snake River to their natal streams to spawn would also encounter extreme sediment loads. These fish would probably not enter the lower Snake River but stray to non-natal streams to spawn unless mitigation measures such as trap and haul are implemented. Adults would probably hold below the mouth of the lower Snake River, where they would remain for as long as a few weeks and then either fall back to a downstream spawning site or move upstream to the Hanford Reach to spawn. Direct mortality of adults would probably be insignificant. Passage of adults of all ESA-listed stocks through the lower Snake River would be highly improbable for a minimum of 5 to 6 years following initial dam breaching. The relative impact of suspended sediment anticipated for each of the different life history stages of the lower Snake River stocks is shown in Table 4-5.

Table 4-5. Relative Impact of High Suspended Sediment Loads on Life Stages of ESA-Listed Snake River Stocks

	Adults	Juveniles	Spawning Success	Juvenile Rearing
Fall Chinook Salmon	Low	High	High	high
Spring/Summer Chinook Salmon	Low	High	High	NA
Sockeye Salmon	Low	High	High	NA
Steelhead Trout	Low	High	High	NA

Source: Developed by the Corps

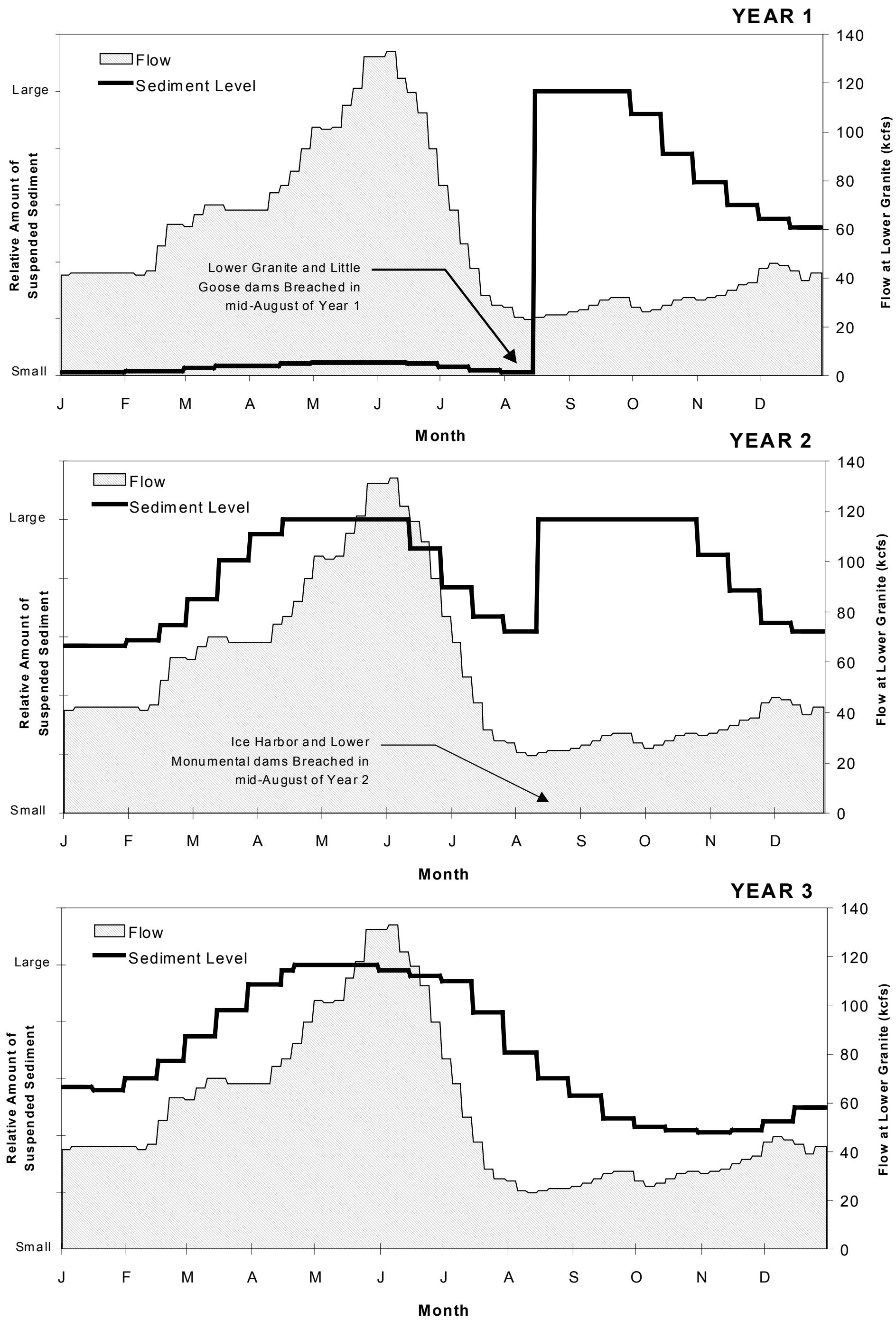


Figure 4-18. Estimated Timing of Sediment Transport Resulting from Breaching of the Lower Snake River Dams

Source: Developed by the Corps (data) and Normandeau (graphics)

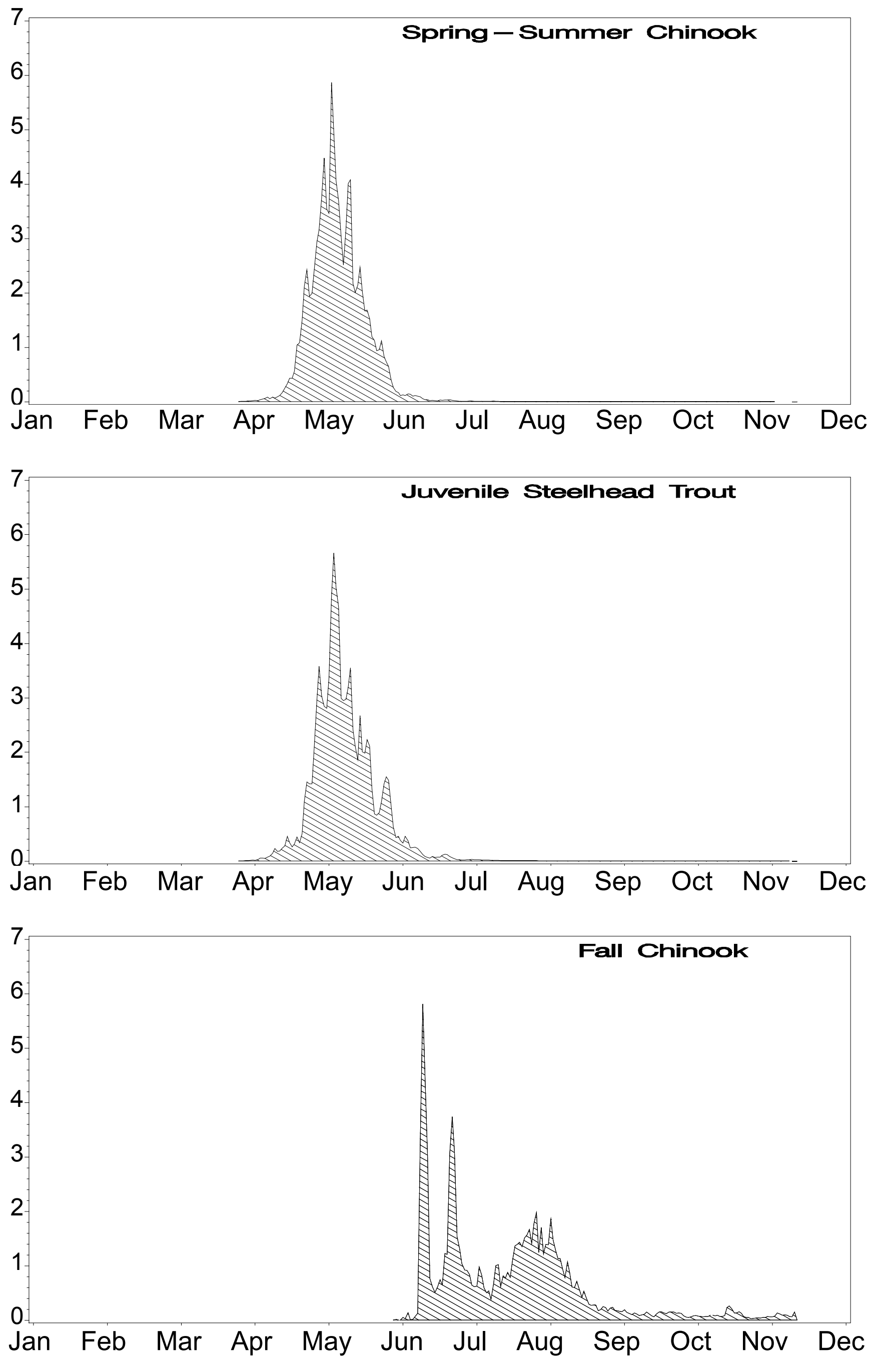


Figure 4-19. Timing of Outmigration of Spring/Summer Chinook, Steelhead and Fall Chinook, Expressed in Percent of Run Passing Lower Granite Lock and Dam per Day

Source: Developed by the Corps (data) and Normandeau (graphics)

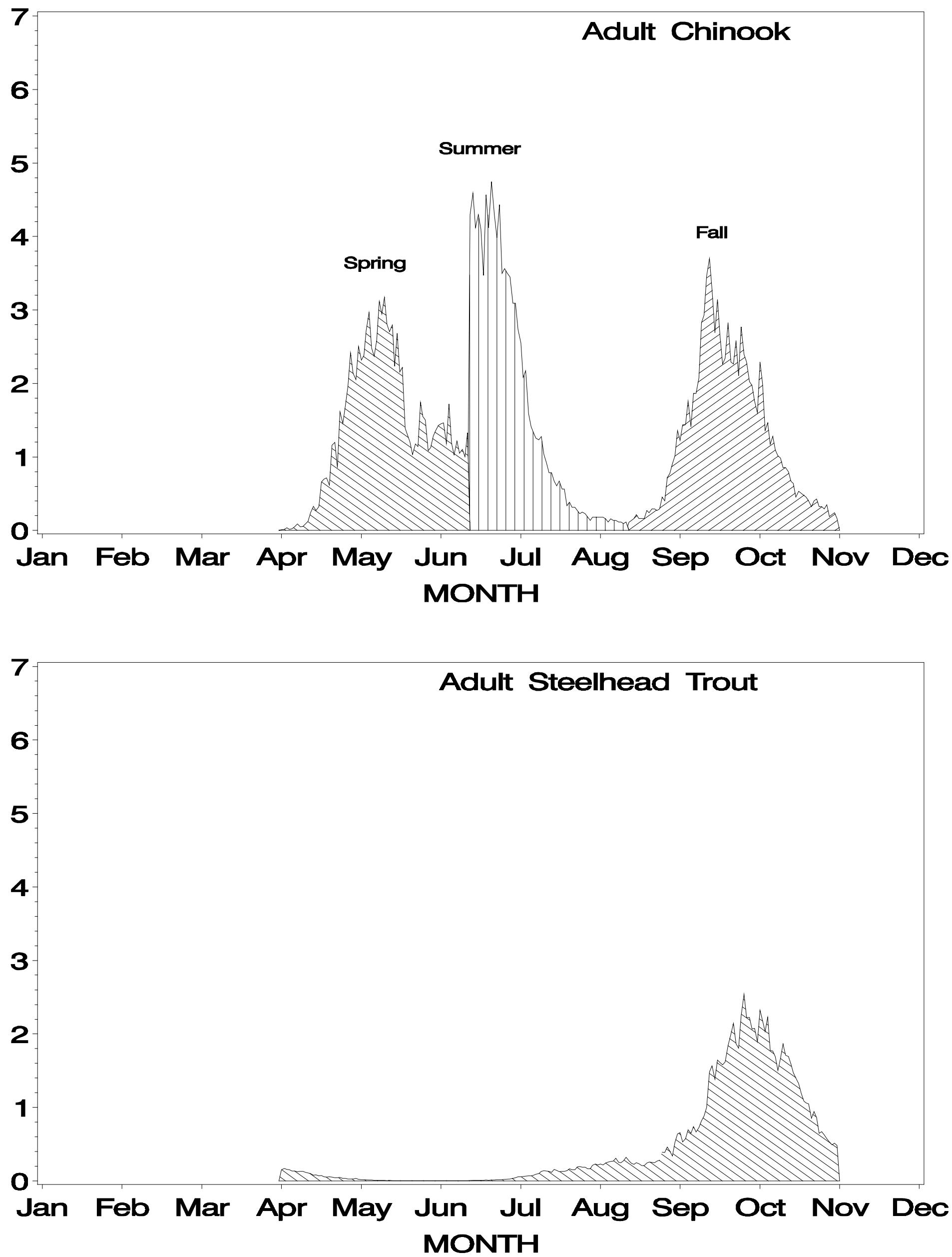


Figure 4-20. Timing of Inmigration of Adult Chinook Salmon and Steelhead, Expressed in Percent of Run Passing Ice Harbor Lock and Dam per Day

Source: Developed by the Corps (data) and Normandeau (graphics)

Sediment movement processes that affect morphological change generally occur at bankfull flow-it is anticipated that bankfull flows [95,600 cfs or 2,707 cubic meters per second would be exceeded an average of 14 percent of the time or between 0 and 121 days per year. Extreme concentrations of suspended sediment would exist for at least 2 years following breaching but would gradually decrease through the next 5 to 10 years (Appendix H). It is impossible to anticipate how much suspended sediment concentrations would decrease annually since unpredictable weather patterns (i.e., peak flows) are the controlling factor. A series of wet, high peak runoff years would accelerate sediment flushing while a series of dry years would result in lower rates of flushing. Eventually, perhaps after 2 to 3 runoff cycles, extremely high concentrations of fine sediment would be experienced only when peak runoff flows exceed the previous post-drawdown high. Material larger than gravel, the majority of which is in the upper end of the Lower Granite pool, would, in the course of decades, move downstream as bed load and out of the Snake River.

The deposition of eroded sediment downstream represents another principal concern, mostly from a physical habitat standpoint rather than a water quality concern. Because the flow velocities in Lake Wallula (the McNary reservoir) are considered to be lower than that observed in the lower Snake River reservoirs, most of this eroded sediment is expected to drop out in the upper portions of the McNary reservoir near the mouth of the lower Snake River. The coarsest sediment would settle out first in the vicinity of the Ice Harbor Dam, and the finer-grained sediment would progressively settle out farther downstream. The very fine sediments that do not deposit in Lake Wallula would continue to be conveyed downstream of McNary Dam, with their ultimate destination likely being the Columbia River Estuary or the Pacific Ocean. Probable areas of deposition in Lake Wallula are shown in Appendix F, Figures 20-1 through 20-5.

The large amounts of sediment transported out of the lower Snake River would have a detrimental impact on the biota of the Columbia River in McNary pool in the area between the confluence and Wallula Gap until the near-natural lower Snake River returns to a state of equilibrium. The direct effect of high sediment loads on the biota would generally be a result of: a) high levels of turbidity that impact planktonic filter feeders and the associated food web; and b) large amounts of sediment that settle out and cover up the benthic community and its associated food web. The associated food web is described in Normandeau (1999a). During the period of extreme sediment movement, the food web would be rendered essentially nonfunctional until the rate of sedimentation decreases to the point that recolonization can occur. The disruption of the food web would cause sublethal effects at the higher trophic levels that include the ESA-listed lower Snake River stocks.

It is difficult to estimate the volumes and locations in which the various-sized particles that make up the accumulated sediment would be redistributed downstream. The previous SOR modeling indicated that most of this sediment would be redeposited in the upper end of the McNary pool between RM 320 and 325 on the Columbia River and RM 0 and 10 on the lower Snake River. The maximum accumulation rate was estimated to be approximately 230 kilograms per meter squared (BPA, 1995).

Assuming one-third of the McNary pool area represents the primary deposition zone, the depth of new sediment could be as much as 1.3 meters (4.2 feet). This potential depth of material is not likely to present navigation problems since most of the McNary reservoir is greater than 20 meters (66 feet) deep. However, this could present problems with existing water withdrawal intakes, including those used for drinking water supply. In addition, redeposited sediment would likely cover large areas of benthic habitat, which, in turn, could cause a major short-term disruption in the primary productivity and food supply for benthivores and other bottom feeders.

The area where sedimentation would occur in the McNary pool is presently utilized as rearing habitat by fall chinook salmon. It is anticipated that sedimentation would be of sufficient magnitude that it would result in a change in the morphology of the area although the nature of the change is unknown. It is likely that a pattern of disruption of fall chinook rearing would follow the pattern of sedimentation and that suitable areas for fall chinook rearing would develop after a few years.

The release of chemical compounds currently attached to bottom sediments represents another major concern to downstream beneficial uses. Potential risks include human health, agricultural (livestock and crops), and ecological risks. Human health risks result from direct contact with water and sediment. Ecological risks include acute and chronic effects on organisms (from contact with water and sediment) and food web effects as contaminants bioaccumulate in higher trophic levels. Additional risk occurs when humans ingest food where contaminants have bioaccumulated. Sediment and water quality criteria have been developed that represent concentrations of CoC in water and sediment that are protective of aquatic life and human health. Maximum concentrations of chemicals are compared to the appropriate minimum criteria to estimate the human, agricultural, and ecological risk. Several levels of uncertainty surround this type of assessment. First, concentrations of chemical constituents show high variability in time and space. Section 3.3, which summarizes results from other studies, reflects the range in concentrations that can be expected in the lower Snake River and environs. This evaluation relies on 1997 sediment and elutriate data collected by the Corps and thus represents a snapshot of risk in time and space. Second, risk cannot be quantified for a number of chemicals because there are no screening criteria. Sediment and elutriate chemicals with no criteria include a number of metals, nutrients, many pesticides and herbicides, total petroleum hydrocarbons, and glyphosate (Normandeau, 1999b).

The previous SOR study modeled the potential for increases in DDT, lead, and ammonia in the water column. Moderate increases in the percentage of time exceeding certain concentration thresholds were predicted for the natural river drawdown scenario as compared to existing conditions. The SOR results were updated with the Corps' 1997 sediment and elutriate test results as well as the Corps' revised sediment volume estimates (Normandeau, 1999b, Appendix B). The results of the analyses of sediment samples collected from the lower Snake River and their ambient pH elutriates were compared with state and Federal guidelines for surface water and sediment quality to identify any CoC. Three parameters exceeded their respective sediment quality criteria. The first and most significant CoC was ammonia. Concentrations ranged from 60 ppm to 81 ppm in all four reservoirs and elutriate values averaged 3 ppm. Comparison to the 1999 Federal water criteria most of the year, the un-ionized portion of ammonia would exceed chronic and acute criteria. This is based primarily on temperature and ambient pH levels in the reservoir (EPA, 1999). The other two were total DDT (using the Corps 1998 Lower Columbia River Dredged Material Evaluation Handbook) and dioxin TEQ, based on the New York Department of Conservation (NYDEC) 1998 Technical Guidance for Screening Contaminated Sediments. Four metals and one organic compound exceeded applicable water quality criteria in the elutriate testing: arsenic (Oregon 1998 Water Quality Criteria [ODEQ]); copper (EPA National Recommended Water Quality Criteria); mercury and manganese (ODEQ); and ethyl parathion (ODEQ and Washington State Freshwater Sediment Quality Values [Ecology]). Of these, four CoC were selected based on the level and frequency of occurrence and comparison to relevant background levels: dioxin TEQ (sediment), total DDT (sediment), and manganese (water) and ammonia (water and sediment).

Points of compliance, locations where CoC exceed the common sediment or water quality criteria, were evaluated in the areas where exceeded. The criteria reflect the protection of piscivorous wildlife (total DDT and dioxin TEQ), human health (DDT from bioaccumulation and manganese from drinking water and eating fish), invertebrate communities (total DDT in sediments), and agricultural uses (manganese in irrigation water).

Dioxin and DDT were detected only in sediment, so the potential areas of concern would include those areas where these organic compounds were detected and where they would be redeposited after the natural river drawdown alternative is implemented. Dioxin was found only in samples collected from Lower Granite Lake downstream from Clarkston-Lewiston, while total DDT was detected in the sediments behind each of the dams. Following the implementation of natural river drawdown alternative it is expected that the resuspended sediments would eventually be deposited behind the McNary Dam.

Manganese was found to partition from the sediment and into water (elutriate) at concentrations which exceeded the State of Oregon surface water quality standard of 50 ppb. Thus, the quality of surface water and its suitability for use may be compromised. To evaluate the potential impact of the release of manganese on the lower Snake River water users, the number and type of users were inventoried. Identified water users in the study area include at least 23 withdrawals on the lower Snake River for irrigation use, a pulp and paper mill located near Burbank, Washington, and the Port of Hermiston, Oregon, withdrawal on the Columbia River, which is used for public water supply. For the purposes of this investigation, five points of compliance for manganese were selected including the Port of Hermiston municipal intake and the reaches 90 meters (300 feet) below each of the dams.

The potential impact of dioxin TEQ, total DDT, and manganese was evaluated by revising the hydrologic simulations performed as part of the SOR (BPA et al., 1995) for the Columbia River. The revisions to the SOR HEC-5Q simulations took into consideration the differences in the alternative evaluated, different partition coefficients, and new estimates of sediment accumulation and different CoC concentration data. In the SOR simulations, ammonia, DDT, and lead were identified as the CoC. Based on the results of the analytical testing and their screening with applicable standards, ammonia and lead were replaced by dioxin TEQ and manganese. The results of the revisions to the SOR HEC-5Q model indicate that the peak suspended sediment concentration would be 9,000 milligrams per liter, or twice the original estimate. The estimated volume of sediment accumulation in the McNary reservoir is 18.4 million cubic meters (24.1 million cubic yards), or 3.4 times greater than the original estimates. Although the concentration and volume of sediment is expected to be higher than originally estimated, the concentrations of total DDT and dioxin TEQ were not found to exceed their respective sediment quality criteria at any of the points of compliance.

The increase in suspended sediment concentrations and the partitioning of manganese from the sediment into the water column would result in the degradation of water quality below acceptable limits. The estimated range of manganese concentrations (394 ppb to 1,328 ppb) exceeds both the limit of 50 ppb for the protection of human health (State of Oregon) and 200 ppb for agricultural use (United Nations) at below each of the four dams. Manganese concentrations are expected to be within standards at the Port of Hermiston (Normandeau, 1999b).

Manganese is usually not considered a human or environmental health risk. The EPA criteria are based upon its effects on the aesthetic properties of water for domestic use. The presence of manganese at concentrations higher than 150 μg/L imparts an undesirable taste and browns laundry (EPA, 1976). Manganese is usually found in salts and other compounds, but is not found naturally as a metal. The concentration of manganese ions in nature is rarely higher than 1 milligram per liter, and most freshwater organisms tolerate concentrations between 1.5 and 1,000 milligrams per liter (EPA, 1976). Similar to other metals, the solubility of manganese is affected by pH and DO, which tends to cause manganese to precipitate. The AFS (1979) reported studies indicating that several freshwater fish species survived exposures to concentrations of manganese up to 2,700 milligrams per liter. Permanganates, a family of manganese-containing compounds, are reported to have a 96-hour LD50 (lethal dose that results in 50 percent mortality in 96 hours of exposure) of 16 milligrams per liter for young rainbow trout (AFS, 1979). Manganese sulfate has also been cited as deleterious to rainbow trout eggs at concentrations of 0.37 to 4.0 milligrams per liter, but had no effect on fry (AFS, 1979). Overall, it appears that the presence of manganese at 3,680 μg/L from the elutriate screening test is not a direct concern to salmonids. However, it is indicative of one or more sources of manganese compounds entering the system and should be a concern relative to domestic water supplies.

Total DDT concentration is the sum of 4,4-DDT and its two metabolites 4,4-DDE and 4,4-DDD. Current Washington State criteria are 1.1 μg/L for instantaneous concentrations and .001 μg/L under chronic (24-hour-average) conditions. The LD50s for chinook salmon and coho salmon have been measured at 12 μg/L and 14 μg/L, respectively. The DDT and its metabolites affect salmonids at both acute and sub-lethal levels (EPA, 1999). Sub-lethal effects to salmonids (not necessarily Pacific salmon) include inhibition of Na^+-K^+ ATPase (an enzyme important to immune response), reduced light discrimination, altered temperature selection, changes in avoidance behavior, and lateral line hypersensitivity. The DDT can bioaccumulate in tissues; consequently, tissue burdens may increase for larger predatory fish, birds, and mammals that utilize forage species exposed to DDT. Model results suggest that aquatic DDT levels could nearly match the chronic criteria level at Lower Granite Lock and Dam during year 1, but should be far below the acute toxic levels. Consequently, the results suggest there is a potential for sub-lethal effects during the first year of drawdown at Lower Granite Lock and Dam.

Dioxin is highly toxic to a variety of species (EPA, 1998). In addition, it is highly stable, resistant to leaching and biodegradation, and only slightly soluble in water. As a result of its relative insolubility in water, dioxins affect salmonids primarily through bioaccumulation from forage species and through direct exposure to sediments to which dioxin is bound. Dioxin affects the immune response in organisms and can cause cancer and liver damage (EPA, 1998). In addition, Walker et al. (1991) demonstrated high sensitivity of salmonids during the egg and alevin life stages. They reported LD50 levels of 65 ppt for lake trout eggs, which compared to LD50 levels of 300 ppt for rainbow trout alevins. Lake fry trout experienced fluid accumulations in their yolk sacs and subcutaneous hemorrhages. Existing sediment concentrations (1.0 ppt maximum in the four samples collected) are well below these levels. Currently there are no freshwater dissolved dioxin (2378-TCDD congener) criteria recommended by EPA, but a criterion of 1.3×10^{-8} μg/L is recommended for human consumption of water and organisms (Federal Register 63, No. 237, December 10, 1998). However, NYDEC recommends a TEQ for the 19 dioxin congeners of 0.0002 milligram per kilogram (NYDEC, 1998). Peak model predictions for dioxin levels are not expected to exceed the NYDEC criteria under any of the modeled scenarios.

Ammonia itself is not as toxic as is the un-ionized portion (Goldman and Horne, 1983). The portion of un-ionized ammonia is variable and subject to change based on environmental conditions. Because of these variabilities, any determination of ammonia in the Snake River system would require a full analysis of water temperature and pH to fully understand the toxicity implications of sediment resuspension in the study area. Disregarding the simple chemical process would potentially result in extreme biological consequences to all species present in the system. The EPA (1986a) recommends the un-ionized portion of ammonia not exceed 7 μg/L for the protection of aquatic life. Trussell (1972) studied the relationship of un-ionized ammonia to pH and temperature. He discovered with an increase of temperature and pH there is a direct increase of the un-ionized portion of ammonia. As an example: The elutriate results in

1997 were quantified at 3,600 μg/L. In August when these samples were taken the temperature was about 20°C (68°F) and pH was about 7.5. According to Trussell's findings, the un-ionized portion of ammonia could be about 4 percent. This equates to the un-ionized portion of ammonia being about 144 μg/L. This is 20 times the EPA's (1999) recommended water quality criteria for the protection of aquatic life.

A review of sediment chemistry studies (Section 3.3) indicates that the 1997 results (Normandeau, 1999b) are generally representative of conditions in the lower Snake River and, thus, confirm updated modeling results for those CoC that have sediment/water quality criteria. The ecological, agricultural, and human health risks of resuspension of chemicals without criteria remain undefined.

While there are many unknowns associated with resuspension of chemicals and contaminants (concentrations, timing, high flow dilution, or low flow concentration, etc.), risks associated with this alternative are considered to be manageable. If this alternative is selected as the preferred alternative, additional testing is recommended to develop containment procedures necessary to minimize resuspension impacts. Also, there should be a monitoring program to assess conditions during and after dam breaching. Initial cost estimates for the additional testing and monitoring indicate that these costs could be approximately $9.6 million.

Sediment transport would also release other sediment-bound nutrients into the water column, increasing their availability to primary producers. As a result, primary productivity could increase, depending on temperature, dissolved water depth, light, and season. Ammonia, the predominant nitrogen compound, would be resuspended. Un-ionized ammonia (NH_3) is toxic to fish but the ionized form of ammonia (NH_4^+) is not. However, the concentration of un-ionized ammonia could increase under increased temperature and pH to levels that exceed the EPA (1999) water quality criteria for protection of aquatic life.

4.4.5 Primary Productivity

4.4.5.1 Alternative 1—Existing Conditions

Primary productivity throughout the lower Snake River system can be categorized as mesotrophic to eutrophic based on both chlorophyll *a* concentrations and phytoplankton densities. Seasonal peaks in chlorophyll *a* and phytoplankton density occurred in June, due to diatoms, and September, due to blue-green algae. Spatially, algal densities were highest upstream and declined downstream. Phytoplankton was the primary source of primary productivity, peaking in June/July and again in autumn. Diatoms predominated, and species composition did not vary between riverine and impounded areas. Low velocities, warm water, and large surface water areas produced favorable conditions to algal production. The reservoirs would be classified as borderline between mesotrophic and eutrophic. According to Dr. Steve Juul, Washington Water Resources Research Institute (personal communication), it is unlikely that the lower Snake River reservoirs would ever become truly eutrophic (overabundant macrophytes, anoxia due to shading caused by phytoplankton, extensive mats of floating aquatic plants, etc.). This is based primarily on a long-term database dating from the mid 1970s which indicates that the major indicators of trophic state have not changed in the 25+ years since sampling began. This being a run-of-the-river series of reservoirs subjected to high velocities with each spring freshet, it is likely that the high flow keeps fine organic material from accumulating thus preventing accumulation of nutrients.

4.4.5.2 Alternative 2—Maximum Transport of Juvenile Salmon

Alternative 2 is similar to the Existing Conditions alternative, with the exception of the elimination of spills and maximization of transport. As factors affecting primary productivity such as temperature and velocity would not change with this alternative, primary productivity would also not be expected to change over the short-term or long-term time frames.

4.4.5.3 Alternative 3—Major System Improvements

Alternative 3 has no differences in spill or flow augmentation, so would not change the physico-chemical environment. Therefore, there should be no short- or long-term changes in the type or level of primary productivity compared to existing conditions. Increased spill would result in exceedance of the 110 percent saturation benchmark.

4.4.5.4 Alternative 4—Dam Breaching

Primary productivity in the existing conditions is based primarily on phytoplankton, a result of the deep, slow-moving waters with fine sediment substrate. With sufficient flow, the accumulated fine material would be moved downstream. It is estimated that flows up to 200,000 cfs would be necessary to remove embedded sediments and return the substrate to its original sand, cobble, and bedrock (Hanrahan et al., 1999). A return to riverine conditions can allow the development of attached benthic algae and periphyton, which should replace phytoplankton as the dominant primary producers. Riparian conditions along the shoreline can develop, adding shade to shoreline waters and allowing input of allochthonous material, which would add additional organic material to the system.

The effects of the Dam Breaching alternative on primary productivity can be estimated from modeling results as well as examining current data from free-flowing river areas. Comparison of the predicted total biomass of primary producers with measured biomass of primary producers from 1997 indicates that on a per unit length basis, primary productivity in the lower Snake River would likely be substantially higher under the Dam Breaching alternative than under the existing impounded system. In addition, the bulk of the primary productivity would be shifted from the phytoplankton to the attached benthic algae component of the food chain. Total primary productivity is predicted to be higher under the Dam Breaching alternative than in the existing impounded river. The predicted elevated algae production is a function of shallower water depths, increased water velocities, warmer water temperatures, and associated scour.

The biological data on primary producers for impounded and free-flowing river areas are very limited. Most of the available data are from the 1997 field effort. Diatoms dominate phytoplankton in the lower Snake River. Phytoplankton concentrations in the free-flowing section had a mean biovolume of 578,980 μm^3/mL, which was within the range of values observed in the impounded section, 323,697 to 738,062 μm^3/mL. However, river volume per unit of river length would be greatly reduced for the Dam Breaching alternative relative to the existing impounded river. As a result, the overall contribution of phytoplankton to system productivity is anticipated to be small.

Attached benthic algae would account for the majority of the total primary productivity for the natural Dam Breaching alternative. The most extensive data for attached benthic algae were collected in 1997. This survey showed the free-flowing site to be more productive than the impounded sites with respect to chlorophyll *a* accumulations. On a dry weight basis, attached benthic algae at the free-flowing site averaged 24.02 mg/m^2 at a depth of 0.75 meters. The other sites in the impounded section had dry weight values of 15.34 to 55.85 mg/m^2. There should be more substrate available for the growth of attached benthic algae following dam breaching than in the existing impounded river because shallower water depths would allow sunlight to reach more of the river bottom, resulting in increased growth of attached benthic algae.

Primary productivity would undergo short-term changes during and after the 2-year dam breaching period. Hanrahan et al. (1999) estimated that sediments would be transported from the Lower Monumental reservoir within 5 to 10 years, so the transition period could be more than 10 years, depending on the river location. Sediment transport would increase suspended sediments and turbidity and, following deposition, alter river topography. Primary productivity would likely decrease with decreased light transmission. Benthic colonization of new substrate may take several seasons to reach full productivity. Therefore, there may be a period of reduced primary production as primary productivity from phytoplankton is reduced but attached benthic algae have not yet fully colonized new substrate. When the near-natural river reaches equilibrium, primary production is expected to be higher per length of river than when it was impounded.

4.4.6 Secondary Productivity

4.4.6.1 Alternative 1—Existing Conditions

Zooplankton are the primary herbivores in the lower Snake reservoirs and, as such, form the base of the secondary production in the reservoirs. They consume phytoplankton, which are the primary source of energy in the impounded Snake River. Zooplankton are an important source of food for plankton-eating fish, which are the prey base for other reservoir fishes. The zooplankton assemblage is composed of about 30 taxa distributed between Rotifers, Cladocera, and Copepods. Taxa that were most abundant included the cladoceran *Daphnia retrocurva,* cyclopoid copepods, the copepod *Diacyclops thomasi,* the cladoceran *Bosmina longirostris*, copepod nauplii, and the rotifer *Keratella cochlearis.*

Benthic macroinvertebrates are an important link in the food web between primary producers and the higher trophic levels. The most common benthic macroinvertebrate taxa, which included both aquatic insects and the taxa, collected in the Lower Granite, Little Goose, and Lower Monumental reservoirs from November 1993 to September 1995 were Oligochaetes, Amphipods (primarily Corrophiidae), Nematodes, Dipterans (primarily chironomids) and Pelecypoda (primarily mussels). In the hard substrate, Diptera (again primarily chironomids), Tricoptera (primarily caddis flies) and amphipods (both Gammaridae and Corophiidae) were the most common taxa. The insect larvae would provide a food source for salmonids.

4.4.6.2 Alternative 2—Maximum Transport of Juvenile Salmon

Alternative 2 is similar to existing conditions, with the exception of the elimination of voluntary spills and maximization of transport. As factors affecting secondary productivity such as primary productivity, temperature, and velocity would not change with this alternative, secondary productivity would also not be expected to change.

4.4.6.3 Alternative 3—Major System Improvements

The pathway Alternatives 2 and 3 differ only in terms of fish transport would not be expected to change any parameters affecting biota. There should be no short- or long-term changes in the species assemblage or density of herbivores.

4.4.6.4 Alternative 4—Dam Breaching

The Dam Breaching alternative would be expected to expose large littoral areas. It is estimated that riverine habitat following dam breaching would compose approximately 39 percent of the available aquatic habitat in reservoirs (NAI and Bennett, 1999). Most of the habitat (90 percent) would be swift-flowing (greater than 2.0 feet per second) areas less than 14 feet deep during moderate summer flows. Most of the reduced velocity habitats would occur in narrow bands along channel edges. This would have a dramatic change on secondary producers.

The effects of the Dam Breaching alternative on secondary producers can be estimated from modeling results as well as examining current data from free-flowing river areas. Few data exist on the composition of the aquatic insect or benthic animal community prior to construction of the dams. One study was conducted in 1973 upstream of the project area in a free-flowing section of the lower Snake River in Hells Canyon (Brusven et al., 1973). In the absence of any better information, the community observed during that study was assumed to be representative of what could be expected in the lower Snake River after breaching of the dams. The aquatic insect portion of this community would likely still have a high proportion of chironomids as is exhibited in the existing impounded system. Tricoptera would remain moderately abundant while Ephemeroptera (mayflies) and Lepidoptera would represent the balance of the community. Secondary productivity is expected to increase in response to increased primary productivity. Organisms that feed on attached benthic algae, such as aquatic insects, would experience the greatest increase.

The shift in secondary productivity from zooplankton to benthic species that feed on attached benthic algae, such as aquatic insect larvae, would ultimately result in a shift to benthic feeding fish. Increased secondary productivity would result in increased biomass of higher trophic levels, including those feeding on aquatic insects, as well as piscivores (salmonids, smallmouth bass, northern pikeminnow, and catfish). The change from a lacustrine to riverine system would be of particular benefit to fall chinook salmon, which rear in the lower Snake River and rely upon aquatic insects as a food source.

A transition period would be likely where the secondary producer community would in flux. As primary producers change from planktonic to benthic, with possible reductions during the transition, secondary producers would also be affected. Secondary production could be severely reduced in the first 3 to 6 years following initial breaching. After the river reaches a state of equilibrium, both the Snake River proper and the areas of deposition in the McNary pool would recolonize rapidly.

4.4.7 Food Web

4.4.7.1 Alternative 1—Existing Conditions

The current food web in the lower Snake River is driven by phytoplankton primary productivity, with small contributions from attached benthic algae (Figure 3-46). Zooplankton is the primary herbivores, which are consumed by planktivorous fish. Aquatic insects are of lesser importance, and consumed by bottom-feeding fish. At the top of the food web are piscivores.

4.4.7.2 Alternative 2—Maximum Transport of Juvenile Salmon

Alternative 2 is similar to existing conditions, with the exception of the elimination of voluntary spills and maximization of transport. As factors affecting the food web such as primary productivity, secondary productivity, temperature, and velocity would not change with this alternative, the food web would not be expected to change either in the short or long term.

4.4.7.3 Alternative 3—Major System Improvements

Major System Improvements would be not expected to change primary or secondary producers. Therefore, no short- or long-term changes in the food web would be expected to result from these alternatives.

4.4.7.4 Alternative 4—Dam Breaching

Recent biological productivity modeling suggests that upstream areas in the lower Snake River would experience a major shift in the primary productivity components and potentially an overall net gain in primary productivity under the Dam Breaching alternative (Figure 4-21). Results from the biologic model indicate that benthic algae throughout the lower Snake River would dominate the primary productivity following dam breaching. Phytoplankton productivity would be of relatively lesser importance. Total primary productivity under the Dam Breaching alternative could increase in comparison to the existing impounded river. After equilibrium is reached, the fish community would likely consist primarily of benthivores rather than the mix of planktivores and benthivores in the existing system. Benthic algae, aquatic insects, and benthivorous fish production are predicted to be highest during dry years. The elevated benthic algae production is a function of increased light penetration, warmer water temperatures, shallower water depths, decreased water velocities, and associated scour. The elevated aquatic insect and warmwater benthivorous fish production is a function of increased benthic algae production. The greatest changes in secondary productivity can be expected to occur in those trophic levels that feed directly on the attached benthic algae, with fewer effects in higher trophic levels. The top piscivores would likely remain northern pikeminnow and smallmouth bass, but the transfer of the bulk of the energy from primary productivity would be through the pathway that includes attached benthic algae, macrophytes, aquatic insects, benthic animals, and benthivorous fishes. Phytoplankton and zooplankton would become minor components of the food web. Salmonids would benefit from the change from a reservoir-based system to a riverine system. The secondary producers would change to mainly aquatic insect larvae, which was the primary food source for rearing fall chinook in the Snake River prior to impoundment. In addition, increased flow and reduction in fine sediments would provide the best habitat for salmonids. The shift in habitat and food sources would be especially beneficial to fall chinook, which spawn and rear in the lower Snake River and prefer low-velocity sandy habitat less than 6 meters (20 feet) deep (Bennett et al., 1993).

A transition period could be expected to last 3 to 6 years, where planktonic production is low, but benthic production is not yet fully developed. Planktivorous species would have little food available, and production for benthic species would not be fully developed.

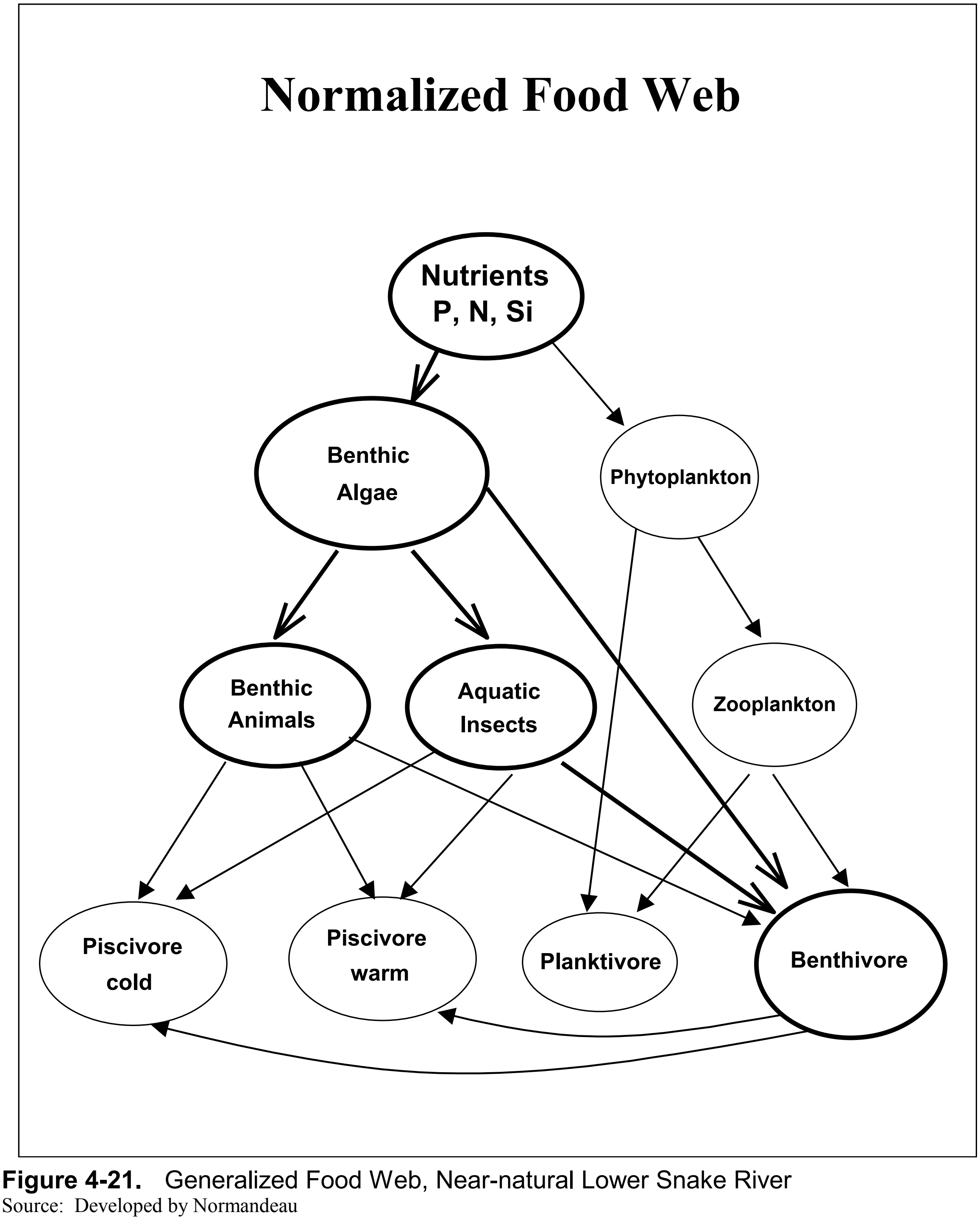


Figure 4-21. Generalized Food Web, Near-natural Lower Snake River
Source: Developed by Normandeau

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6. Glossary

Aesthetics: Of or pertaining to the sense of the beautiful.

Allochthonous: Pertaining to substances (usually organic carbon) produced outside of and flowing into an aquatic or wetland ecosystem.

Alluvium: A general term for all detrital material deposited or in transit by streams.

Anadromous fish: Fish, such as salmon or steelhead trout, that hatch in fresh water, migrate to and mature in the ocean, and return to fresh water as adults to spawn.

Anion: A negatively charged ion.

Anthropogenic: Changes made by human activity.

Augmentation: Increased river flows above levels that would occur under normal operation by releasing more water from storage reservoirs.

Basaltic: Having a texture of a hard, dense, dark volcanic rock.

Behavioral guidance structure (BGS): Long, steel, floating structure designed to simulate the natural shoreline and guide fish toward the surface bypass collection system by taking advantage of their natural tendency to follow the shore.

Benthic community: Aquatic organisms and plants that live on the bottom of lakes or rivers, such as algae, insects, worms, snails, and crayfish. Benthic plants and organisms contribute significantly to the diets of many reservoir fish species.

Benthivore: An organism that consumes bottom-dwelling organisms.

Biomass: The total mass of living tissues (plant and animal).

Biota: The animal and plant life of a particular region considered as a total ecological entity.

Biovolume: Volume of an organism available for consumption.

Bivalve: A mollusk such as mussel or clam, having a shell consisting of two hinged parts.

Bulkhead channel: Channel through which fish are carried upward through the turbines via a bulkhead slot if they are not diverted by turbine intake screens.

Bypass channel: Fish diverted from turbine passage are directed through a bypass channel to a holding area for release or loading onto juvenile fish transportation barges or trucks.

Cation: A positively charged ion.

Chironomid: An insect, midge, which has a benthic larval stage.

Chlorophyll *a*: A green plant pigment necessary for plants to produce carbon from sunlight.

Collection channel: Holding area within the powerhouse that fish enter after exiting the bulkhead slot.

Dam breaching: In the context of this FR/EIS, dam breaching involves removal of the earthen embankment section at Lower Granite and Little Goose Locks and Dams, and formation of a channel around Lower Monumental and Ice Harbor Locks and Dams.

DDT: An organochlorine pesticide compound.

Detritus: Dead plant material that is in the process of microbial decomposition (adjective: detrital).

Diatom: A unicellular or colonial algae (aquatic plant) having siliceous walls.

Dissolved gas supersaturation: Caused when water passing through a dam's spillway carries trapped air deep into the waters of the plunge pool, increasing pressure and causing the air to dissolve into the water. Deep in the pool, the water is "supersaturated" with dissolved gas compared to the conditions at the water's surface.

Drawdown: In the context of this document, drawdown means returning the lower Snake River to a near-natural, more free-flowing condition via dam breaching.

Elutriate: Type of water sample created by mixing sediment and water.

Endangered species: A native species found by the Secretary of the Interior to be threatened with extinction.

Epithelial: Having membranous tissue, usually in a single layer, and forming the covering of most internal surfaces, organs and the outer surface of an animal body.

Eutrophic: A body of water in which the increase of mineral and organic nutrients reduces DO, producing an environment that favors plant over animal life.

Exopthalmia: Having an abnormal protrusion of the eyeball.

Fauna: Animals collectively, especially the animals of a particular region or time.

Fecal Coliform Bacteria: A group of organisms belonging to the coliform group and whose presence denotes recent fecal pollution from warm-blooded animals.

Federal Columbia River Power System (FCRPS): Official term for the 14 Federal dams on the Columbia and Snake Rivers.

Feral: Existing in a wild or untamed state.

Fish collection/handling facility: Holding area where juvenile salmon and steelhead are separated from adult fish and debris by a separator and then passed to holding ponds or raceways until they are loaded onto juvenile fish transportation barges or trucks.

Fish guidance efficiency (FGE): Percent of juvenile salmon and steelhead diverted away from the turbines by submersed screens or other structures.

Fish passage efficiency (FPE): Portion of all juvenile salmon and steelhead passing a facility that do not pass through the turbines.

Flora: Plants collectively; especially the plants of a particular region or time.

Flow augmentation: Increasing river flows above levels that would occur under normal operation by releasing more water from storage reservoirs upstream.

Fluvial: Formed or produced by the action of flowing water.

Foraging habitat: Areas where wildlife search for food.

Forebay: The area of water directly upstream of a dam.

Freshet: A sudden overflow of a stream resulting from a heavy rain or thaw.

Gas bubble disease or trauma (GBD or GBT): Condition caused when dissolved gas in supersaturated water comes out of solution and equilibrates with atmospheric conditions, forming bubbles within the tissues of aquatic organisms. This condition can kill or harm fish.

Geometric mean: Average of $\log_{10}$ (original value +1).

Geomorphology: The systematic examination of landforms and their interpretation as records of geologic history.

Hydrology: The science dealing with the continuous water cycle of evapotranspiration, precipitation, and runoff.

Impoundment: Accumulated water in a reservoir.

Inundation: The covering of pre-existing land and structures by water.

Irrigation: Artificial application of water to usually dry land for agricultural use.

Juvenile fish transportation system: System of barges and trucks used to transport juvenile salmon and steelhead from the Lower Snake River or McNary Dam to below Bonneville Dam for release back into the river; alternative to in-river migration.

Lacustrine: Of or pertaining to a lake.

Larva/larvae: An early life stage of an animal.

Limnology: The study of the physical, chemical and biological aspects of lakes.

Littoral zone: The shore area along a body of water, usually a lake, down to the depth of 10 meters

Lower Snake River Hydropower Project: The four hydropower facilities operated by the Corps on the Lower Snake River: Lower Granite, Little Goose, Lower Monumental, and Ice Harbor Locks and Dams.

Macroinvertebrate: Organism without a backbone generally measuring more than 0.5–1 millimeters in size.

Macrophytes: large, vascular aquatic plants that grow in shallow water along the shorelines of lakes or in the slow-moving reaches of rivers.

Megawatt (MW): One million watts, a measure of electrical power or generating capacity. A megawatt will typically serve about 1,000 people. The Dalles Dam produces an average of about 1,000 megawatts.

Metamorphic: Rock that has been greatly altered from its previous condition through combined action of heat and pressure.

Minimum operating pool (MOP): The bottom 1 foot of the operating range for each reservoir. The reservoirs normally have a 3- to 5-foot operating range.

Mitigation: To moderate or compensate for an impact or effect.

Mollusk: Any member of phylum Mollusca, largely marine invertebrates.

Navigation: Method of transporting commodities via waterways; usually refers to transportation on regulated waterways via a system of dams and locks.

Outwash: Coarse-textured materials left by streams of melt water flowing from receding glaciers.

Passage model: Mathematical simulation of the effect of downstream passage (through eight Federal mainstem hydro projects) on the survival of juvenile salmonids.

Pelagic food sources: Food sources for aquatic organisms that live in the water column.

pH: An index of the hydrogen ion concentration in water, measured on a scale of 0 to 14. A value of 7 indicates a neutral condition, values less than 7 indicate acidic conditions, and values greater than 7 indicate alkaline conditions.

Physiographic province: A geographic region.

Phytoplankton: Drifting plants such as microscopic algae that nourish themselves from the energy of the sun; they are at the base of the food chain and provide a food source for bacteria, water molds, and zooplankton.

Piscivorous: Feeding on fishes.

Planktivorous: Feeding on planktonic organisms.

Pumping stations: Facilities that draw water through intake screens in the reservoir and pump the water uphill to corresponding distribution systems for irrigation and other purposes.

Recovery: The process by which the ecosystem is restored so it can support self-sustaining and self-regulating populations of ESA-listed species as persistent members of the native biotic community. This process results in improvement in the status of a species to the point at which listing is no longer appropriate under the ESA.

Reservoir fluctuation area: Area between the minimum and maximum pool levels of a reservoir which includes the littoral, wave-action, and inundation zones.

Resident fish: Fish species that reside in fresh water throughout their lifecycle.

Riparian: Ecosystem that lies adjacent to streams or rivers and is influenced by the stream and its associated groundwater.

Rip-rap: A permanent, erosion-resistant groundcover constructed of large, loose, angular or subangular rounded stone.

Run-of-river: This describes hydropower facilities that do not have storage or the associated flood control capacity; run-of-river facilities essentially pass through as much water as they have coming in, either through the turbines or over the spillways.

Salmonid: Of or belonging to the family Salmonidae, which includes salmon, trout, and whitefishes.

Scour: Cleared, removed by water.

Scouring: Concentrated erosive action, especially be stream or river water, as on the outside curve of a bend.

Senescing: Aging, growing old.

Smolt: A young salmon at the stage at which it migrates from fresh water to the sea.

Spawning: The reproductive process for aquatic organisms which involves producing or depositing eggs or discharging sperm.

Spill: Water released through the dam spillways, rather than through the turbines. Involuntary spill occurs when reservoirs are full and flows exceed the capacity of the powerhouse or power output needs. Voluntary spill is one method used to pass juvenile fish without danger of turbine passage.

Spillway flow deflectors (flip lips): Structures that limit the plunge depth of water over the dam spillway, producing a less forceful, more horizontal spill. These structures reduce the amount of dissolved gas trapped in the spilled water.

Stilling Basin: A concrete-lined pool below the dam where water dissipates energy prior to flowing downstream.

Substrate: Substances used by organisms for growth in a liquid medium; surface area of solids or soils used by organisms to attach.

Surface bypass collector (SBC) system: System designed to divert fish at the surface before they have to dive and encounter the existing turbine intake screens. The SBCs direct the juvenile fish into the forebay, where they are passed downstream either through the dam spillway or via the juvenile fish transportation system of barges and trucks.

Surface erosion: Movement of soil particles down or across a slope, as a result to gravity and a moving medium such as rain or wind. The transport of sediment depends on the steepness of the slope, the texture and cohesion of the soil particles, the activity of rainsplash, sheetwash, gullying, dry ravel processes, and the presence of buffers.

Surficial deposits: Unconsolidated alluvial, residual, or glacial deposits overlying bedrock or occurring on or near the surface of the earth.

Tailrace: The canal or channel that carries water away from a dam.

Tailwater: The water surface immediately downstream from a dam.

Talus: Accumulated fragments of rock and soil at the foot of cliffs or steep slopes.

Taxon/Taxa: Any level of classification, as genus, species, *etc*.

Terracing: Creation of a relatively level bench or step-like surface, breaking the continuity of a slope.

Thermocline: A density gradient due to changing temperatures within a water body.

Threatened species: A native species likely to become endangered within the foreseeable future.

Total suspended solids (TSS): The portion of the sediment load suspended in the water column. The grain size of suspended sediment is usually less than one millimeter in diameter (clays and silts). High TSS concentrations can adversely affect primary food production and fish feeding efficiency. Extremely high TSS concentrations can impair other biological functions such as respiration and reproduction.

Transect: A line on the ground along where sample plots or points are established for data collection.

Trichopteran: An insect, caddisfly, which has a benthic larval stage.

Trophic level: Position in the food chain determined by the number of energy-transfer steps to that level.

Turbidity: An indicator of the amount of sediment suspended in water. It refers to the amount of light scattered or absorbed by a fluid. In streams or rivers, turbidity is affected by suspended particles of silts and clays, and also by organic compounds like plankton and microorganisms. Turbidity is measured in nephelometric turbidity units.

Turbine intakes: Water intakes for each generating unit at a hydropower facility.

Turbine intake screens: Standard-length traveling fish screens or extended-length submerged bar screens that are lowered into the turbine bulkhead slots to divert fish from the turbine intake.

Watershed: The area draining into a river, river system, or body of water.

Wetland: An ecosystem in which groundwater saturates the surface layer of soil during a portion of the growing season, often in the absence of surface water. This water remains at or near the surface of the soil layer long enough to induce the development of characteristic vegetative, physical, and chemical conditions.

Yearling: Salmon less than 1 year old.

Zooplankton: Tiny, floating animals that provide a food source for larger aquatic organisms such as snails and small fish.